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Habitat Conservation, Wildlife Extraction and Agricultural Expansion*

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Proposed running head: wildlife, habitat and agriculture

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Habitat Conservation, Wildlife Extraction and Agricultural Expansion

ABSTRACT: We develop a model of open access wildlife exploitation, habitat conservation and agricultural expansion, which is consistent with rural communities at the fringe of natural habitats in areas such as sub-Saharan Africa. Farmers have the option of either hunting for wildlife or growing crops. The opportunity cost of each activity is the return to the other (economic interdependence), and habitat conversion affects the returns to both activities (ecological interdependence). We show how different patterns of conservation and agricultural expansion may emerge, and that greater conservation may be consistent with higher incomes. We also show that wildlife stocks under open access may be greater than wildlife stocks under a common property (or social planner's) regime.

Keywords: Competing land-use, conservation, multiple equilibria, trade measures

1. INTRODUCTION

Wildlife is threatened by many factors, of which habitat conversion to other uses (such as agriculture and urban development) and human (over) exploitation are best known and most notorious. According to the World Conservation Monitoring Center [42], of 486 documented extinctions since 1600, 80 have resulted from hunting and 98 have resulted from habitat destruction. Over-exploitation, usually combined with habitat destruction, is believed to threaten about one-third of the world's endangered mammals and birds.

Interestingly, wildlife exploitation and habitat conversion choices differ significantly across societies so that the worldwide distribution of “nature” varies considerably. Some countries host the bulk of nature and biodiversity whereas others have converted their natural assets in the past or are currently doing so [42]. Consider recent land use developments in Kenya and Southern Africa as an example. Child [10] concludes “Kenya’s wildlife is in trouble” whereas wildlife “thrives” in Southern Africa. In less than 20 years, private South African land allocated to wildlife has increased from 1 to 8 million hectares (compared to 2.8 million hectares controlled by the National Parks Board). In Kenya, during the same period, wildlife was excluded from large areas of its traditional habitat to minimize competition for forage with livestock and to protect and promote the expansion of arable agriculture [25]. As a result, Kenyan herbivore populations have declined by 40-60% since 1977 while South African wildlife populations have increased. A number of factors contribute to these differences in conservation choices, including differences in population pressures [35], property rights and use rights for land and wildlife resources [9], and fiscal policies – particularly those promoting agricultural expansion [28]. However, conservation choices differ even in societies with similar property rights regimes and agricultural expansion policies. It is important to understand why this may occur, particularly in less developed regions where property rights are ill defined and conversion of ‘nature’ is an important issue.

South African experiences suggest that wildlife exploitation and conservation efforts do not necessarily conflict. Indeed, allowing (local) people to utilize wildlife as a renewable resource may trigger incentives to carefully manage wildlife as a valuable asset and to allocate scarce land to its survival. So-called integrated conservation and development projects (ICDPs) are based on this notion and are financially supported by hundreds of millions of dollars by organizations such as the World Bank [3]. ICDPs are increasingly promoted to encourage communities on the periphery of natural habitat to sustainably harvest wildlife, possibly in return for alternate sources of income or sustenance. Barrett and Arcese [3,4] critically assess the potential long-run benefits of such programs, concluding that the design of ICDPs over the past decade leaves much to be desired.

This paper contributes to the understanding of development policies and wildlife conservation by explicitly modeling the role of habitat and spillover benefits. One objective is to investigate why, under conditions of ill-defined property rights for habitat and wildlife resources, some societies preserve wildlife at the expense of agricultural expansion, while others expand agricultural lands at the expense of conservation. We find that multiple equilibria may lead to an outcome with little habitat and wildlife or, in contrast, with little agricultural conversion and abundant wildlife stocks.¹ In particular, we find conservation choices may be driven by spillover effects that are somewhat analogous to those of the ‘big push’ literature in development economics (e.g., [22]). We identify situations in which a ‘little pull’ can encourage conservation *and* increase economic well being by creating backward *and* forward linkages in the wildlife sector that promote economic expansion. This can be interpreted as theoretical support for ICDPs and for the efforts of organizations such as Africa Resources Trust to promote wildlife utilization by local people.

Another objective is to investigate the effects of conservation and agricultural expansion policies when property rights remain ill-defined. For example, trade measures to

promote wildlife conservation may achieve the opposite result, depending on the particular steady state that the system is in. We also find that ill-conceived domestic policies to expand agriculture may make everyone in the economy worse off. Accordingly, conservation and development policies should be jointly considered and implemented with great care.

Finally, we evaluate the often-heard claim that lack of property rights is a major cause of excessive resource depletion. We compare wildlife stocks under conditions of open access and cooperative common property management, finding that open access stock levels may exceed socially optimal levels. Thus, it is too simple to equate ill-defined property rights with over-exploitation. We are not familiar with existing literature arriving at this conclusion.

2. EXISTING LITERATURE ON WILDLIFE MANAGEMENT

Given the obvious relation between natural habitat availability and the potential for wildlife exploitation, it makes sense to capture both in a single model. However, the early focus of bioeconomic models (e.g., [11]) was on marine resources, so the bioeconomic literature did not initially focus much attention on the opportunity cost of conservation, or the potential earnings from alternative uses of the resource base. (It was tacitly assumed that there are no profitable alternative uses for oceans and lakes.) Alternatively, an important, non-bioeconomic literature (e.g., [16], [19], [2]) addressed the economic tradeoffs associated with preserving versus converting natural environments, where the benefits of preservation may consist of amenity values. More recently, the bioeconomic– and the preservation-versus-conversion frameworks have been extended to analyze decisions that jointly impact natural environments and the species contained therein (e.g., [36], [38], [33]) or other ecological services that may even feedback into the economy (e.g., [13]).²

Four types of models dominate the current literature on wildlife harvesting, and we briefly describe each. First, there is a class of models in which individuals (poachers) hunt

wildlife under open access conditions, with entry and exit typically assumed proportional to profit. Poachers are possibly impeded by a wildlife agency that aims to protect the species. An agency may invest in patrolling because of tourism-related benefits that the species provides, or because of nonuse values associated with conservation. Such models have been used, for example, to study poaching of elephants [7] and conservation of rhinos [6].

A second class of models involves wildlife competing with livestock for forage (or otherwise interacting through spreading of diseases, for example), reducing returns to local livestock producers (“pastoralists”). Wildlife may spill over from adjacent natural parks or otherwise, and may be under the control of a management agency. Social inefficiencies arise when externalities associated with wildlife conservation are not corrected (see [33]).

Third are models in which a “single agent” (e.g., the government, a private party, or even a group of local people) manages a wildlife resource with a fixed habitat base. These are conventional bioeconomic models [11]. Recent work in this area has focused significantly on issues associated with species conservation and extinction, and with the possibility of multiple equilibria in the presence of stock-dependent benefits and nonconvexities [23].

Finally, there are models in which a single agent balances the benefits of exploiting nature or converting habitat and practicing agriculture instead. For example, Swanson [38] studies the trade-offs associated with elephant conservation. Further, Skonhøft and Solstad [34] study the trade-off between grazing cattle and harvesting wildlife. Finally, Swallow [36] provides a theoretical foundation when conversion of habitat is irreversible. Since the focus is on a single decision-maker (with well-defined property rights), optimal control models are typically applied to find intertemporally optimal allocations.

The model we employ is somewhat unique in that it combines the first and fourth categories of models. We focus on a (large) group of local people who have to choose between agriculture (or cropping) and wildlife harvesting (hunting). Due to ignorance or

social and cultural factors (group size and heterogeneity, etc.), however, socially optimal outcomes are not achieved and open access management prevails, both for habitat and wildlife. Many environmental resources are unregulated common property in developing countries [9].³ The paper extends current theory by combining the following features:

- Habitat and wildlife are both open access resources as property rights are not defined (enforced). Modeling habitat as an open access resource is particularly realistic in less developed countries where record-keeping of deeds and surveys may be poor, enforcement is often weak due to corruption and under-funded and under-trained staff, and where local people living at the extensive margin exhibit “complete alienation from or disinterest in” wildlife conservation (parks are often considered a legacy of the colonial era, see [39]);
- Land and wildlife are biologically interconnected. Wildlife carrying capacities are reduced as habitat is converted to agricultural land and increased as agricultural lands revert back to nature.
- Land and wildlife are economically interconnected. The opportunity cost of time spent growing crops are the foregone returns from harvesting wildlife (and vice versa), and therefore *endogenous* to the model. This assumption differs from previous open access models where the cost of hunting effort is considered exogenous (e.g., [43], [12]).

3. A SIMPLE MODEL OF WILDLIFE AND HABITAT DYNAMICS

Consider a group of people who earn a living by hunting wildlife or growing crops (or alternatively raise livestock).⁴ Crops are perishable and are traded in local (regional) markets. Wildlife commodities, on the other hand, may be traded locally and internationally, depending on the nature of the good. We ignore capital and industrialization, focusing instead on the options of households close to the extensive margin. The model set-up is consistent with the

options of agricultural communities at the fringe of nature reserves, and our findings are therefore related to IDCs such as the CAMPFIRE program in Zimbabwe.

Property rights for both animals and land are ill-defined (or not enforced), resulting in open access exploitation. The incentives for behavior are at the level of the individual (“what is the best response for me?”) and we assume that individuals earn the average product of their labor, so that total income equals gross revenues (see [29]). For simplicity, behavior is modeled by aggregating over all individuals. The community has access to a land base (L) and time endowment for labor (T) and we (initially) assume both are constant. The time endowment is proportional to the population in the community, which is consistent with observations on rural communities in developing countries. For example, Dasgupta [14, p.226] notes that “village communities in poor countries are often self-contained enclaves of production”, where “migration within spatially proximate villages is not so common”. Indeed, there is evidence that wages are uniform within villages but typically different between villages [14]. We relax the assumption of a fixed T below to explore the consequences of migration into or out of the area.⁵

Land is homogenous and suitable for agriculture (A) or wildlife habitat (H):⁶

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

Available time is spent hunting, E , or for cropping available agricultural land, W :

$$(2) \quad T = E + W.$$

Hunting is characterized by the well-known Schaefer production function:

$$(3) \quad h = qxE$$

where h is the quantity of wildlife harvested, q is a catchability coefficient and x is the available wildlife stock. In reality, hunters will chase a number of species but for analytical ease and without loss we focus on the single species case. Harvesting revenues are simply pqx , where p is the exogenous price of harvests. This set up is consistent with the case of

some bird species, ivory, hides and rhino horn being exported. (In an appendix, available upon request, we analyze an alternative case where meat is traded locally and demand is downward sloping.) Although CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) bans international trade in certain wildlife commodities, including ivory and rhino horn, we do not distinguish between legal and illegal harvesting (poaching) as we ignore enforcement issues (see [21]).⁷ The average return to labor in hunting is proportional to the wildlife stock:

$$(4) \quad \pi_H = pqxE/E = pqx.$$

As in conventional open access models, people allocate their labor to the use where the (average) return is greatest. Define π_A as average revenue of income from agriculture. Thus, the short- and medium-run dynamics of hunting effort are characterized by:

$$(5) \quad dE/dt = \eta[\pi_H - \pi_A]$$

where η is an adjustment parameter, measuring the speed with which hunting effort is expanded (contracted) if hunting is a relatively profitable (unprofitable) activity. Hence, if the average return to labor in cropping exceeds the return to labor in hunting, some hunters will convert habitat into agricultural land. If the average returns to hunting effort exceed the returns to cropping labor, farmers will spend less time in agricultural activities, reducing cropland (which immediately return to natural habitat), and spend more time hunting. A simplifying assumption is that converting habitat to agricultural land, or vice versa, is costless. The assumption that conversion is reversible (i.e., abandoned agricultural land immediately reverts to habitat) is essential for what follows, and not unrealistic in many cases. In Sub-Saharan Africa, for example, significant areas of agricultural lands have been converted into “game ranches” that support wildlife. In other settings, reversal may still be possible but less

immediate. For example, a Central African rain forest system is more complex than an East African grazing system and will take longer to re-establish.

Define the agricultural crop production function by $C(W, A)$, which is increasing in both arguments and is quasi-concave. Total revenues are given by $B(C)C$, where B is the downward sloping inverse demand for C . Average revenue, or income, is denoted by $\pi_A = B(C)C/W$. Given labor W , A can be chosen to maximize income. The first order condition is (with individuals taking B as given)

$$(6) \quad d\pi_A/dA = (B/W) dC/dA = 0.$$

The solution to (6) yields $A(W)$ ($A' > 0$). For simplicity, assume C is quadratic with land and labor both being necessary inputs, i.e., $C = c_1A^2 + c_2W^2 + c_3AW$, with $c_1 < 0$, $c_2 < 0$ and $c_3 > 0$. Then $A = (1/\alpha)W$, where $\alpha = -2 c_1/c_3 > 0$. The relation $W = \alpha A$ is used to eliminate W in the function $C(A)$ and in equation (2). Given that T is fixed and equation (2) holds, then from (6) and (2) we have $dW/dt = -dE/dt$. Because $A = (1/\alpha)W$, A must evolve according to:

$$(7) \quad dA/dt = -(1/\alpha)dE/dt = \phi[\pi_A - \pi_H],$$

where $\phi = \alpha\eta > 0$ is an adjustment parameter.

Demand for agricultural crops, $B(C)$, is downward sloping as crops are sold on “thin” local or regional markets. Assuming iso-elastic demand and defining $-\epsilon$ as the inverse of the elasticity of demand for the agricultural commodity, gross revenues of cropping are:

$$(8) \quad R = B(C)C = [\Phi A^2]^{-\epsilon} \Phi A^2 = \phi A^\beta, \quad \text{where } \Phi = (c_1 + c_2\alpha^2 + c_3\alpha).$$

The revenue function is strictly concave (that is, $\beta < 1$) if $\epsilon > 1/2$, or alternatively, if the elasticity of demand is smaller than 2. This is consistent with empirical results from a wide set of developing countries as summarized by Dasgupta [14]. We therefore assume $\beta < 1$. Given this specification, the average return to labor in agriculture is:

$$(9) \quad \pi_A = \phi A^\beta / W = \phi A^\beta / \alpha A = (\phi / \alpha) A^{\beta-1}.$$

To complete the model, consider how the wildlife species grows over time. Consistent with earlier work by others (e.g., [38], [33]), the carrying capacity is determined by available habitat. We assume a simple specification for replenishment $G(X) = \rho X(K-X)$, where ρ is a parameter, X is the animal population and K is the carrying capacity (measured in animals). Recall that H is the carrying capacity in terms of land area. Define D as the maximum density of X per unit of land area. Then G may be written as $G(X) = \rho D^2 (X/D)(K/D - X/D)$. Defining $x = X/D$, $H = K/D$, and $\gamma = \rho D^2$, the species growth function can be written as:

$$(10) \quad G(x) = \gamma x(H-x),$$

The growth specification is unchanged by this rescaling and is thus well-behaved. Hence, the critical assumptions of population growth are maintained: (i) $G(0) = G(H) = 0$ and $G(x) > 0$ for $0 < x < H$, (ii) $G''(x) < 0$, and (iii) $G'(x) > 0$ for $x < H/2$ and $G'(x) < 0$ for $x > H/2$ (i.e., G is concave and single-peaked at an intermediate density).⁸ Rescaling species abundance implies that the Schaefer production function in (3) has also been rescaled. Conventionally, the Schaefer production function is defined as $h = \nu XE$, where ν is a catchability coefficient. In the model, this is rewritten as $h = \nu XE = (\nu D)(X/D)E = qxE$. Using (1), (3), (10), and the relation $W = \alpha A$, the dynamics of wildlife are:

$$(11) \quad dx/dt = G(x) - h = \gamma x(L-A-x) - qx(T-\alpha A).$$

The pair of equations (7) and (11) describe a dynamic system whose behavior over time may be analyzed. Note the similarity with the well-known Gordon-Schaefer open access model, where individuals' decisions to enter or exit a fishery (rather than convert or abandon a plot of land) depend on the profitability of fishing (e.g., [43]). Equilibrium profits are zero in the Gordon-Schaefer model; they have been dissipated by excessive entry. Entry is restricted in our model as T is exogenous, but equilibrium profits are zero. In the steady state, returns to

labor in both activities are equal by condition (7), and hence profits are zero if the opportunity cost of labor is accounted for. The model extends the Gordon-Schaefer specification because of the ecological and economic interactions mentioned in the introductory section. To analyze the model, assume a steady state exists, substitute (4) and (9) into (7) and set the right-hand-sides of (7) and (11) equal to zero. This yields the following set of conditions that should simultaneously hold for the steady state:

$$(12) \quad dA/dt = 0 \rightarrow A = (p\alpha qx/\phi)^{1/(\beta-1)}, \text{ and}$$

$$(13) \quad dx/dt = 0 \rightarrow A = [L - x - (qT/\gamma)] / [1 - (q\alpha/\gamma)].$$

(Conversely, $x=0$ also implies $dx/dt=0$). The slope of the isocline defined by (12) is:

$$(14) \quad dA/dx = \frac{p\alpha q}{\phi(\beta-1)} \left(\frac{xp\alpha q}{\phi} \right)^{(2-\beta)/(\beta-1)} < 0.$$

This slope is negative because, with diminishing returns to A and constant returns to x , an increase in x requires a decrease in A for average returns for the two activities to remain balanced, as is required along the isocline (see the appendix, available upon request, for a model with downward sloping demand for the wildlife commodity).

The slope of the isocline defined by (13) is:

$$(15) \quad dA/dx = -\frac{1}{1 - (q\alpha/\gamma)}.$$

The sign of dA/dx in (15) is ambiguous. It depends on the relationship between the parameter α , representing how much labor is required to operate each unit of the agricultural land base, and the parameter coefficient γ/q . The coefficient γ/q represents how much labor is required to search through each unit of “excess habitat” (defined $H-x$) when attempting to harvest in the steady state.⁹ This result occurs for two reasons. First, by reducing available habitat for wildlife, agricultural expansion will slow down growth of wildlife and depress future stocks (the ‘habitat effect’). Second, and in contrast, cultivating the expanding agricultural fields

implies that less labor is available for hunting animals, thus supporting thicker wildlife stocks (the ‘hunting effect’). Expanding the agricultural land base triggers counter-acting effects on wildlife via its impacts on habitat and hunting effort. When $\alpha < (\gamma/q)$, the habitat effect dominates the hunting effect and the $dx/dt=0$ isocline slopes downward. The agricultural sector absorbs little labor from the wildlife sector when α takes a low value, so that expanding the agricultural area has only a modest effect on hunting intensity. If, on the other hand, $\alpha > (\gamma/q)$, then the hunting effect dominates the habitat effect and $dx/dA > 0$.

A priori it is unclear whether α will be greater or smaller than γ/q , and “back-of-the-envelope” calculations indicate that both cases may occur. Our particular interest in what follows is for the case in which $\alpha < (\gamma/q)$. For example, assume people in Eastern Africa have to choose between elephant harvesting and cattle herding. Using available data on elephant harvesting, density, and growth [7], and cattle stocking rates [17], we find that $\alpha < (\gamma/q)$ when farmers are able to manage a herd of about 30-40 cows.

4. MULTIPLE EQUILIBRIA

In this section we analyze the steady states defined by (12) and (13). The phase diagram for the system is presented in Figure 1 for one set of parameters for which $\alpha < \gamma/q$. To the right of the $dA/dt=0$ isocline, hunting is more profitable than cropping and some farmers will abandon their fields which convert back to habitat. The opposite occurs to the left of the $dA/dt=0$ isocline. Above the $dx/dt=0$ isocline, the reduction in habitat reduces resource productivity and, even though hunting effort is reduced, harvest levels exceed growth (recall $\alpha < \gamma/q$). The opposite occurs below the $dx/dt=0$ isocline.

Figure 1 illustrates two interior steady states, λ and μ , although in general this is only one possibility. Other possibilities include one equilibrium (when the isoclines are tangent) or

no equilibrium (when the $dA/dt=0$ isocline is everywhere to the northeast of the $dx/dt=0$ isocline). The maximum number of interior equilibria is two because the $dA/dt=0$ isocline is strictly convex when $\beta < 1$ and the $dx/dt=0$ isocline is linear for all parameter combinations. The system may end up at either λ or μ , depending on the initial conditions. The λ steady state is referred to as the ‘nature scenario’ because more time is spent harvesting animals in this steady state relative to the μ steady state, which we refer to as the ‘agricultural scenario’. In addition to these interior equilibria, the system may end up in a corner solution where all habitat has been converted to agricultural land and no animals remain ($A=L, x=H=0$).

<Insert Figure 1 about here>

Steady state λ is a stable equilibrium (a stable, improper node) in which habitat and wildlife are abundant and agricultural expansion has been modest. Due to the abundance of wildlife and its excellent potential for replenishment, most people earn an income by sustainably harvesting animals. In contrast, steady state μ is a saddle point, consistent with the outcome that less habitat and fewer animals remain, and people predominantly earn an income by cropping. The saddle path leading to μ is given by SS. Since the number of equilibria depends on the parameter values chosen, it is not possible to analytically determine the stability of potential equilibria for all parameter combinations. However, the phase plane can be used to determine the robustness of our stability results [20].¹⁰ For the case of two equilibria, the μ equilibria will always be a saddle while the λ equilibria is either a stable proper node or a stable improper node. This is evident from the phase arrows in the different isosectors in Figure 1 as well as the slopes of the isoclines. The central and southeast isosectors are terminal isosectors. Starting in the southwest isosector to the right of saddle path SS, movement must be in the direction of λ or into the central isosector. Starting in the

northeast isosector to the right of SS, movement must be in the direction of λ or into the southeast isosector. For a different curvature of the $dA/dt=0$ isocline, movement in the southwest, southeast, central, and northeast isosectors (to the right of SS) may only be towards λ . Finally, a saddle-node bifurcation will result when the two isoclines are tangent.

Movement towards any of the steady states depends on the initial stocks of wildlife and agricultural land, providing an additional explanation why some societies preserve wildlife at the expense of agricultural land while others expand agricultural lands at the expense of wildlife conservation. For instance, steady state μ can only be attained if the initial stock levels are on the saddle path SS. Otherwise, the system will move either towards steady state λ or wildlife extinction. Extinction will result for initial stock combinations to the left of the SS saddle path.¹¹ The reason for falling back to this corner solution is that hunting cannot compete with cropping at low levels of animal abundance. Ongoing habitat conversion undermines the potential for the species to recover (recall that the habitat effect dominates the hunting effect), and all habitat is eventually converted to agricultural land.¹²

5. ECONOMIC LINKAGES AND STEADY STATE COMPARISONS

Income and total revenues are larger in the λ steady state than in the μ steady state. Average revenues from agriculture are larger if the land base is small (due to a concave revenue function for cropping), and therefore agricultural income (average return to labor) in λ strictly dominates income in μ . Because people are indifferent between both activities in any steady state, it must hold that everybody is better off in the nature scenario. (Since $x(\lambda) > x(\mu)$, it is also clear that $pqx(\lambda) > pqx(\mu)$.) Accordingly, local people at steady state μ would be better off by moving to steady state λ (also benefiting conservationists because habitat and wildlife stocks are larger in the λ steady state). Why then may the less desirable steady state μ

persist? While everybody would gain in the long run if some farmers leave their fields and start hunting, an individual farmer has no incentive to do so. While the instantaneous effect of abandoning a field is to increase agricultural income (since $d\pi_A/dA < 0$), there is no immediate effect on hunting income (i.e., there is no instantaneous increase in x and so $\pi_H = pqx$ is unaffected) and thus his/her own income is unchanged. Accordingly, at equilibrium μ , an individual farmer is indifferent to agricultural production or wildlife hunting and will therefore stay in agriculture. The mechanic open access nature of the model militates against foresighted acts and society is locked up (trapped) in an undesirable steady state (see Berck and Perloff [5] for a more complex model based on rational expectations).

What happens if one farmer would switch to hunting in return for the promise of future gains? The current model predicts a snowball effect or, more technically, a positive feedback effect due to *forward and backward linkages* (driven by technological externalities). Investing in habitat implies providing an input into producing x , which in turn is an input (or an intermediate good) in producing harvests. Producing x increases everyone else's productivity in the harvesting sector, thus providing an incentive for others to enter the harvesting sector and use x as an input in the harvesting process. This represents a forward linkage: the development of an input x stimulates others to also use the input, effectively expanding the output sector. But in order to harvest, the person must reduce the time spent in agriculture. He will remove some of his land from cropping, at the same time investing in habitat for wildlife. This represents the backward linkage: an expanding output sector encourages investment in raw inputs – habitat in this case. The investment in habitat and divestment of agricultural land increases the absolute returns to both activities, with the returns to harvesting increasing more after the initial investment than those from agriculture due to the forward and backward linkages.

The discussion may be contrasted with existing theory on industrialization and the existence of “surplus labor” in the agricultural sector. Murphy et al. [22] provide one seminal model, based on increasing returns to scale in modern manufacturing combined with an elastic supply of labor, where a “big push” towards industrialization may result if several firms adopt modern production techniques simultaneously.¹³ The current model requires only a “little pull” back to nature, although it should be noted that the required pull is larger if it takes longer for agricultural lands to revert to their natural state. After the initial (possibly small) investment, it is in the interest of others to switch too. This discussion reveals that outside intervention may be desirable. Through direct regulation or provision of financial incentives, the open access stalemate may be dissolved so that everybody gains.

6. POPULATION GROWTH, HABITAT AND WILDLIFE

Communities at the periphery of natural habitat in developing countries typically do not face perfect markets for all factors and commodities. It is well documented that transaction costs (e.g, search costs and transport costs) drive a wedge between purchase and sale prices and may indeed render some transactions uneconomic altogether [30]. The model developed above deals with the case of a “missing labor market,” which is clearly an extreme case of market failure, albeit not unrealistic for the type of community that we consider ([8], [15], [14], [44]). In this section we explore what happens if we allow “some” migration, for example in response to income differentials between adjacent villages. In an appendix (available upon request) we consider the (unrealistic) case of perfectly elastic supply of labor.

We established above that if the local community is on a trajectory towards steady state λ , per capita income increases. Assuming that rising incomes attract some extra labor (dT) from neighboring regions, the implications are apparent from considering the effect on the $dx/dt=0$ isocline (13) (the $dA/dt=0$ isocline is unaffected), i.e.:

$$(16) \quad dx/dT = (q/\gamma)/[q\alpha/\gamma - 1].$$

An inflow of labor shifts the isocline downward (for $\alpha < \gamma/q$). Two outcomes are possible. First, the qualitative nature of the analysis is unaffected for a sufficiently small immigration response, although the inflow of labor will shift the steady states λ and μ (reducing per capita income in the λ equilibrium, perhaps discouraging further immigration). Second, for a sufficiently large labor inflow, the isoclines will no longer intersect and an interior solution will not materialize. If emigration is sluggish in responsive to declining incomes, the system will eventually settle at the unattractive corner solution ($L=A, x=H=0$).

7. COMPARATIVE STATICS

We now consider the comparative statics of the above-mentioned steady states, and the potential implications of policy-related efforts to alter the parameters of the system. As mentioned above, there may be scope for promoting sustainable exploitation by implementing well-targeted policies in this second best world. The main results are summarized in Table I.

<Insert Table I about here>

The results for λ and μ in Table I are of opposite sign, or ambiguous for the cases involving q and γ . Indeed, the comparative statics are not of a consistent sign for any row in Table I, demonstrating the inherent difficulty in predicting system responses to policies that would impact one or more parameter values. Accordingly, policies to promote wildlife conservation (e.g., trade sanctions that reduce p or, in the case of poaching, enforcement policies that reduce q) or agricultural expansion should be implemented with care. Depending on the situation, changing key parameters may promote or discourage conservation, and may enhance or decrease welfare. Thus, prior to implementing new policies, investment in research could

help to identify what ecological-economic state exists to indicate what outcomes may be expected. For the sake of brevity, we limit our discussion to the case of changing p , focusing on the multiple equilibrium case (i.e., $\alpha < \gamma/q$). However, similar intuition applies to policies that reduce q , or to agricultural expansion policies, such as increasing ϕ (akin to increasing agricultural prices) or reducing α (akin to technical change that increases the agricultural land base society can manage).

From Table I, the effect of a policy that reduces p depends on the steady state the system is in. Previous work has also found that resource pricing policies may both promote and discourage conservation, depending on the relative strength of the price change on the incentive to (i) harvest and (ii) allocate more land as habitat to support species' growth (e.g., [7], [31], [38]). But the intuition behind the current result is different. Decreasing p causes the $dA/dt=0$ isocline to shift up (see 12). Consider the λ steady state first. Reducing the price of the wildlife good makes hunters worse off and, to restore equilibrium and make people indifferent between cropping and hunting, the returns to cropping should also decline. This requires an increase in agricultural production (convert some habitat), so that stocks will (i) diminish due to less habitat (the habitat effect) but (ii) increase since hunting effort is decreased as more labor is required for the larger agricultural land base (the hunting effect). Since $\alpha < \gamma/q$, the habitat effect dominates and equilibrium wildlife stocks (and exploitation) are reduced. This is clearly an unintended outcome of the (trade) policy.

In contrast, a decrease in p increases the μ steady state habitat and resource stocks. This may seem good news for CITES enthusiasts, but unfortunately the story is not that clear. While nature conservationists may admittedly prefer the "new" steady state to the "old" one, the system may never actually reach this new steady state. Starting at the old μ steady state, a decrease in p places society in the northwest region of Figure 1 (in which $dA/dt > 0$ and

$dx/dt < 0$) leading to the corner outcome $H=x=0$.¹⁴ Hence, when $\alpha < \gamma/q$, wildlife conservation is never promoted by trade sanctions. By the same logic, it is easy to show that, starting at the μ steady state, a small increase in p moves society on a path that converges to the λ steady state. Thus, a small increase in the price of wildlife harvests could increase welfare for all by shifting production to the wildlife sector. Clearly, from this perspective, subsidizing wildlife harvesting makes more sense than taxing it.

8. OPEN ACCESS AND COMMON PROPERTY COMPARED

Until now we have assumed that exploitation of wildlife and habitat takes place under conditions of open access. While this may be realistic for some regions, it is well known that cooperation among users (or regulated common property management) may potentially result in outcomes that are close to the social optimum (e.g., [1], [24]). If certain conditions are satisfied—possibly related to group size, homogeneity and trust—society could decide jointly on management and realize a welfare gain. Such cooperation is most likely to succeed at the local level (e.g., for a single village). It has been taken as an article of faith by environmental and development economists alike that such cooperation will promote sustainable resource use. The reason is of course that cooperation erodes the “grab and run” mentality that is rewarded under open access. In this section we address this issue by comparing the *in situ* wildlife stocks under open access and perfect cooperation.

Assume society maximizes aggregate surplus, defined as the present value of total revenue, where r is the discount rate:

$$(17) \quad \text{Max}_A \quad V = \int_0^{\infty} [\phi A^\beta + pqx(T - \alpha A)] e^{-rt} dt,$$

subject to (11) and the boundary conditions $x \leq H$ and $\alpha A \leq T$. The modified current value Hamiltonian is $H_c = \phi A^\beta + pqx(T - \alpha A) + \psi [\gamma x(L - A - x) - qx(T - \alpha A)] + \Lambda_1(H - x) + \Lambda_2(T - \alpha A)$,

where ψ is the co-state variable and Λ_1 and Λ_2 are Lagrangian multipliers associated with the boundary conditions. Along with (11), the following conditions are necessary for an optimal, interior solution:

$$(18) \quad \psi = \frac{\phi\beta A^{\beta-1} - pqx\alpha}{x(\gamma - q\alpha)}, \text{ and}$$

$$(19) \quad \frac{d\psi}{dt} = r\psi - pq(T - \alpha A) - \psi\gamma(L - A - x) + \psi\gamma x + \psi q(T - \alpha A).$$

Differentiate equation (18) with respect to time and equate this expression with equation (19). The result is an expression for agricultural expansion that, along with equation (11), describes how the system optimally moves over time:

$$(20) \quad \frac{dA}{dt} = \frac{A}{\beta-1} (\gamma(L - A - x) - q(T - \alpha A)) + \frac{x(\gamma - q\alpha)}{\phi\beta(\beta-1)A^{\beta-2}} ((r - \gamma(L - A - x) + \gamma x + q(T - \alpha A))\psi - pq(T - \alpha A))$$

Unlike the case for the open access model, the slope of the $dA/dt=0$ isocline is mathematically cumbersome to present and its sign is analytically ambiguous. Since our focus in this section is on comparing the optimal steady state solution with the open access results, we do not present the phase planes nor do we describe the features of the optimal solution in great detail. However, we do note that the $dx/dt=0$ isocline is the same as in the case of open access.

Assuming an interior solution exists, the steady state wildlife stock x^* and habitat A^* are implicit in the following pair of equations:¹⁵

$$(21) \quad x^* = L - A^* - (q/\gamma)[T - \alpha A^*], \text{ and}$$

$$(22) \quad r = -\gamma x^* + \frac{pq(T - \alpha A^*)x^*(\gamma - q\alpha)}{\phi\beta A^{*\beta-1} - pqx^*\alpha}.$$

Analytical comparison of the wildlife stock under common property management to the stock emerging under open access is difficult due to the many non-linearities involved in the

steady state equations. In conventional open access models, the wildlife stock is smaller under open access as over-harvesting results from a lack of property rights. In the present case, however, we argue that wildlife stocks must be larger under open access when $\alpha < \gamma/q$.

Since the $dx/dt=0$ isocline is the same in both the open access and common property models, the steady states for each management regime must lie somewhere on this locus. The common property steady state must lie to the left of μ when $\alpha < \gamma/q$. Suppose the common property steady state lies somewhere between λ and μ . If x and A are such that society is initially at this steady state, then it would be optimal to stay there. However, given the discussion of the open access equilibrium, we know that this cannot be. Instead, there must exist a path that moves society towards λ and increases welfare for all *at each point in time* (we know this since λ is a sink in the open access case). Hence, the present value of social welfare is also increased by moving towards λ , and so a steady state that lies between λ and μ would not be optimal. A similar argument can be made for steady states to the right of λ . However, given that the steady state conditions for a social optimum differ from those for open access, we know that λ will not be a steady state for the planner's problem. Instead, the optimal steady state must lie to the left of μ . Thus, both open access equilibria will exhibit larger wildlife stocks (and less agricultural land) than that of the social optimum. Indeed, using the same parameters as in Figure 1 (and assuming $r = 10\%$), the optimal wildlife stock x^* is about 10,000 animals. This stock is significantly smaller than both the saddle point equilibrium μ (about 30,000) and the stable equilibrium λ (about 44,000) in Figure 1.

The rather unexpected result that the absence of property rights (or cooperation) promotes conservation of wildlife (when $\alpha < \gamma/q$) is explained by two opposing externalities relative to the common property regime. First, individual users fail to take the scarcity value of wildlife into account, which results in excessive harvesting and, therefore, too little

cropping (habitat conversion). Second, individuals do not consider the effect of habitat conversion (and subsequent cropping) on the (average) profitability of agriculture, nor do they incorporate the effects of conversion on the ability of wildlife stocks to replenish. The result of this second effect is excessive habitat conversion and too little wildlife harvesting. We find that the first effect dominates when $\alpha < \gamma/q$, so that too much effort is devoted to hunting and too little land is developed relative to the optimum. It is not possible to unambiguously rank the open access and optimal management equilibria when $\alpha > \gamma/q$ due to the nonlinearities and different dynamic features involved in this case.

Of course, the present value of aggregate income is greater under common property—the social planner can always mimic the open access outcome and do equally well. This implies that per capita income in cropping and hunting may well be different (income differentials are not eroded by excessive entry, as in the open access case), and that inequality in society increases without redistributive transfers. This suggests a subtle trade off between aggregate income, income distribution and, possibly, environmental effects.

9. CONCLUSIONS, DISCUSSION AND POLICY IMPLICATIONS

Using a fairly simple model, we have shown how different patterns of conservation and agricultural expansion might emerge in less developed countries. Specifically, such differences may reflect the multiple equilibria that might arise when property rights are not well defined and the opportunity cost of harvesting is endogenous.

Contrary to conventional wisdom, we obtain three primary results. First, equilibria with greater conservation and less agriculture may be Pareto superior to equilibria in which agriculture plays a more dominant role. A society ‘stuck’ in an agricultural equilibrium may need a ‘little pull’ to start it on a path to a higher income, nature equilibrium. We find that ICDPs (e.g., the CAMPFIRE program in Zimbabwe) and policies aimed at promoting wildlife

utilization, for example by relaxing restrictions on the use of harvested animals such as currently exist in East Africa, may enhance welfare and wildlife if they are implemented in the context of imperfect markets for factors and commodities (also see [10]).¹⁶ Second, conservation of wildlife and habitat may be greater under open access than under common property management (or management by a social planner). This result sheds new light on the common call for “privatization” of common resources (see [1]). Third, adopting second-best conservation policies when property rights remain ill-defined, such as reducing the price of wildlife commodities (as commonly attempted in the context of CITES), may have ambiguous effects on wildlife abundance and welfare. Similar ambiguities are associated with the effect of policies aimed at changing technology and agricultural prices. These results indicate that implementing well-intended policies, possibly successful in other regions, without considering the underlying system could be detrimental for both income and wildlife.

Our results are based on the assumptions of imperfect labor markets and perfectly elastic demand for wildlife commodities. The main insights spill over to the cases where migration is “sufficiently costly” and where demand for wildlife commodities is “sufficiently elastic” (that is, has a finite but greater than unity elasticity). A perfectly elastic labor supply would rule out the possibility of multiple equilibria under open access and would resolve many of the ambiguities surrounding the current analysis. As the transactions costs of trading labor among communities are reduced, alternative routes to income growth and wildlife conservation are opened up. In the short- and medium run, however, given the current institutional context, economists may find it worthwhile to more closely scrutinize the ICDP approach to development.

Relaxing the assumption of “sufficiently elastic” demand for wildlife commodities does not destroy the multiple equilibria property of the analysis. The qualitative nature of the results may be affected, however. For example, if the elasticity of demand is smaller than

unity, a stable steady state with high incomes may exist in which habitat is abundant (agricultural land is scarce) and wildlife stocks are depleted. Depletion in this case is due to price effects: small steady state harvest levels imply high commodity prices. The results will differ for a rural community that is completely isolated from the rest of the world. A CGE approach would be appropriate to study such situations.

Finally, while the discussion of spillover benefits was cast in terms of the relation between habitat and the replenishment potential of wildlife, the same intuition applies to the case of game ranching and tourism (including trophy hunting). While game ranching typically takes place on (partially) fenced in private land (so that there are no ‘physical externalities’ in terms of animals spilling over to other land), it is clear that there are other externalities with similar effects. For example, tourists are expected to have a “wilderness experience” when they visit a (private) game ranch, irrespective of whether their main objective is hunting or game watching. For individual game ranches it is hard to keep up a wilderness image when the ranch is surrounded by farmland, hence the perception of nature (and the success of the ranch) will be affected by land use of neighboring landowners (do they raise livestock or keep game too?). Therefore, starting a fenced-in game ranch may yield spillover benefits and “snowball dynamics”, similar as in the open access model developed above, and we may expect most results of the hunting model to hold for the tourism case too.

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Table I: Comparative statics for various key parameters

	$\alpha < \gamma/q$				$\alpha > \gamma/q$	
	steady state $\hat{\lambda}$		steady state μ		dx	dA
	dx	dA	dx	dA		
d ϕ	-	+	+	-	+	+
d p	+	-	-	+	-	-
d L	+	-	-	+	+	-
d T	-	+	+	-	-	+
d α	+	-	-	+	-	\pm
d q	\pm	\pm	\pm	\pm	\pm	\pm
d γ	\pm	\pm	\pm	\pm	\pm	\pm

Note: Derivations of the comparative statics are available upon request.

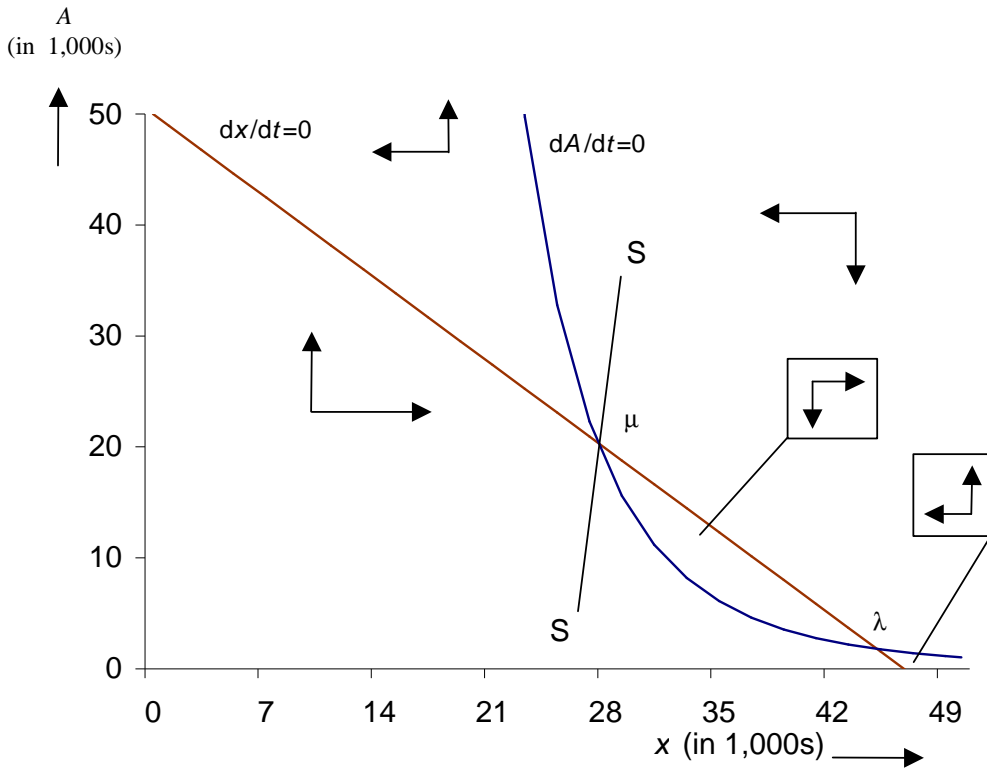


Figure 1: Open access wildlife harvesting and habitat conversion: Interior solutions ($p=10, q=0.000025, L=50,000, T=10,000, \alpha=0.2, \beta=0.8, \gamma=0.00007, \varphi=10$).

ENDNOTES

¹ Others have found that multiple equilibria may characterize extraction of resources without well defined property rights, but this is typically in the context of models with few players acting strategically (e.g., see the oligopoly paper by Reinganum and Stokey [27]). Also, see Kremer and Morcom [18] for a general model of open access including storage.

² The literature cited deals with conversion of habitat into alternative uses. In a related but somewhat different vein, there is also an economic literature concerned with the management of environmental quality (e.g., within a habitat) and its impacts on species productivity (e.g., [40], [41]).

³ The assumption that wildlife is not “owned” by anyone, or is subject to socially wasteful open access exploitation is a key feature of many models of wildlife exploitation and poaching (e.g., [7]). While analytically convenient, it is not obvious that such models are necessarily the most apt description of reality in all cases. For example, Baland and Platteau [1] discuss under what conditions coordination and cooperation among local users may result in socially optimal management (see also [24]).

⁴ Fenced-in livestock directly reduces available habitat for wildlife. Free roaming livestock competes with some wildlife species (e.g., gnu) for forage, and thus also reduces the carrying capacity for supporting wildlife. Ranching thus leaves the ecosystem productivity below “natural” for some wild species.

⁵ Labor supply may also be affected by population growth or endogenous choice of leisure [8].

⁶ The assumption of homogeneous land is obviously a simplifying one that helps us to take a first step into looking at the issues presented here. In reality, some land will be better suited for agricultural purposes and some land will be better suited for wildlife habitat, and the suitability of land for different uses is not necessarily positively correlated. This would obviously affect the incentives associated with placing particular parcels of land into different uses, and is likely to affect the differences between the open access case and the social optimum. See Swallow [37] for a model of irreversible habitat conversion, who shows that it may be optimal to develop low grade land first if land quality for different uses is positively correlated. Also, see Crocker and Tschirhart [13] for a model with economy-ecosystem interactions that permits identification of which ecosystem services and organisms may be efficiently used and which can be efficiently maintained.

⁷ In principle, we could include enforcement in the model by either reducing the catchability coefficient (as suggested by one referee) or by increasing the cost of hunting (see [7]).

⁸ Of course, the linearity of the relation between K and H is a limiting assumption about the system’s ecology. Also note that there are many alternative specifications for the growth function, including logistic growth, the Ricker growth function and the Beverton-Holt growth function. While the qualitative results in the main text may be reproduced for these and other specifications, it is also feasible to find alternative outcomes, possibly considerably more complex due to additional nonlinearities that may arise in more complex models.

⁹ The term excess habitat may require some explanation. Habitat and x have been defined in units of land area. Thus, excess habitat refers to the total habitat area not currently occupied by the stock. In the steady state, $E=\gamma/q$ ($H-x$). Thus, it is clear that more effort is required to harvest the less dense is the stock, or the larger is the excess habitat that must be combed before a unit of the stock is found and harvested.

¹⁰ Stability can also be determined for particular parameter combinations to confirm the results of the diagrammatic analysis. The eigenvalues of the linearized differential system determine stability. For the present case and for a number of other parameter combinations we experimented with (resulting in two equilibria), we found two (distinct) negative eigenvalues associated with the λ equilibrium, indicating a stable, improper node.

¹¹ The nature of the saddle implies that there is only one path to μ and, in the present open access situation, μ is only achieved when the initial stock and habitat levels lie on this path. It is unclear how the results may change if we relaxed the assumptions of costless and immediate habitat conversion and restoration, and a formal analysis of a model with set-up costs is beyond the scope of this paper. However, at least intuitively, sufficiently large restoration and conversion costs (and time lags) would increase the opportunity costs of habitat restoration and conversion, and hence vertical movements in the phase plane. The result may be that μ becomes locally stable for a range of initial values. We would like to thank Chuck Mason for pointing this out.

¹² It is interesting to compare the results with the case where $\alpha>\gamma/q$ (hunting effect dominating habitat effect), in which case the $dx/dt=0$ isocline slopes upward. This case results in a single interior steady state which, based on analysis of the phase plane, may be a focus or a center as is common in open access models (e.g., [12]). *Ceteris paribus*, $\alpha>\gamma/q$ is likely to occur for slow growing species (i.e., γ is small), as is the case for many large mammals. Although bioeconomic models have shown that hunting slow-growing (and not fast-growing) species to extinction may be optimal (e.g., [11]), we find the opposite result may hold here. Assuming the focus is stable, then in the case of slow-growing species there is no isosector directing the system to extinction – unlike the case for fast-growing species in our model. Thus, we find that extinction may be *less likely* for slow-growing species.

¹³ Due to “backward linkages” in industrial production (where each good produced in the modern sector enlarges the market for all other goods), welfare gains are realized if the economy is pushed to the modern equilibrium, but the economy may also be stuck at the traditional equilibrium. The latter may occur if the fixed costs associated with switching to the new equilibrium is not earned back by firms.

¹⁴ Starting at the original μ steady state (which must be to the northwest of the new μ steady state in Figure 1), a decrease in p promotes conversion of habitat to agricultural land (from (7)) due to the relative decrease in returns from harvesting. In turn, starting from an equilibrium in which $dx/dt=0$, equation (11) indicates that the increase in A will reduce the wildlife stock. The result is absolute habitat conversion and extinction. An important distinction exists between the fragile saddle equilibrium in the present model and saddle equilibria that typically arise in optimal control models (planner’s problem). In optimal control models, a shift in the steady state is accompanied by an adjustment in harvests that places society on the separatrix. In contrast, there is no planner or public authority in this framework to make the needed adjustment; society is left to operate on its own.

¹⁵ We are not ruling out multiple solutions.

¹⁶ Zimbabwe’s ‘Communal Areas Management Programme for Indigenous Resources’ (CAMPFIRE) much acclaimed success in promoting rural development and wildlife conservation has been subject of debate. While the international community has channelled millions of dollars in the program, Patel [26] recently concluded that there is little evidence to show that CAMPFIRE actually contributed to their stated objectives. Patel argues that CAMPFIRE is not the “community-based, community-directed” program that it claims to be, but rather a scheme that has been exploited by the white-dominated private wildlife industry to protect their interests. While some of the results of CAMPFIRE are consistent with the outcomes of our model (e.g., displacement of agricultural workers to favor wildlife), there is evidence of “forced evictions” and “coerced resettlement of households.” The distributional issue that has apparently been created is not analyzed in our model.