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Evolutionary success and failure of wildlife conservancy programs*

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Abstract:

This paper develops an evolutionary bio-economic model for hunting, farming and tourism (non-consumptive and safari hunting) to study the determinants of the prosperity of conservancy programs. The model is inspired in the Conservancy program of Namibia, despite it is of more general applicability to other contexts. We explore the relevance of the design attributes of conservancy programs in their prosperity in the long-run as well as the relevance of variables of the context of application highlighted in empirical literature. In addition, we explore the welfare implications of conservancies for local communities and its compatibility with conservation objectives. We discuss the results of the conservancy model with respect to the benchmark of open access and of compensation policies for agricultural losses out of wildlife.

Keywords: Bioeconomic modeling; Community-based management; Wildlife.

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Evolutionary success and failure of wildlife conservancy programs

1. INTRODUCTION

Wildlife Management has been based historically in a strategy of 'Fences and Fines'. That is to say, on the establishment of strongly restricted uses in natural protected areas which entail very often the displacement of rural communities. The result of such strategy is that wildlife becomes a nuisance for locals: herbivores compete for pasture with livestock, predators kill livestock and endanger the life of members of the communities. In short, local communities bear the cost of wildlife protection but do not get any of the benefits. In sub-Saharan Africa, this comes often times with a lack of resources by park agencies to enforce protection, resulting in a *de facto* open access situation (Gibson and Marks, 1995). An observable result has been persistently high levels of poaching which threaten the objectives of preservation of wildlife (Weaver and Skyer, 2005).

Governments and conservation NGOs have implemented compensation policies in countries such as Zambia, Tanzania, Kenya, Botswana, and Zimbabwe aiming to mitigate the incentives of farmers to kill wildlife by giving them money, seeds or livestock to cover a portion of the losses imposed by wildlife (Barrett and Arcese, 1995; Bulte and Rondeau, 2007; Gibson and Marks, 1995; Rondeau and Bulte, 2007). Thus, they amount to an agricultural price subsidy, fostering agricultural expansion and consequently damaging wildlife in those contexts where the wildlife population is sensitive to habitat destruction (Rondeau and Bulte, 2007).

Alternatively, the underlying premise in community-based programs¹ that we analyze in this paper is that communities can benefit from wildlife management when they receive sufficient authority and control over wildlife resources (Bandyopadhyay et al., 2004). Previous theoretical literature analyzing community-based management is quite pessimistic on its capacity to make compatible conservation objectives and the wellbeing of local communities (Barrett and Arcese, 1998a; Gibson and Marks, 1995). Features of economic and biological

¹ See the special issues in Environmental Conservation vol. 37 issues 01 and 03 for a broad perspective into community-based natural resource management.

systems leading to this result include continued poaching due to free-ride within communities (Gibson and Marks, 1995);, as well as the public good attributes of both habitat and wildlife (Barrett and Arcese, 1998b). Yet, some cases such as Namibia have been described as an empirical success of the usefulness of community-based management in terms of coupling conservation objectives and wellbeing of local communities (Barnes et al., 2002; Naidoo et al., 2011a; Naidoo et al., 2011c; Weaver and Skyer, 2005).

Conservancy programs emerged in Namibia in 1996 after the Ministry of Environment and Tourism passed legislation that devolved conditional user rights over natural resources, including wildlife, to communities by registering as communal-area conservancies (Barnes et al., 2002; Naidoo et al., 2011a). The objective was to promote and integrate wildlife conservation and related tourism development in the welfare of communal-area residents (Weaver and Skyer, 2005). In doing so it was expected to alleviate the severe poaching pressures that wildlife in the country was suffering and which contributed to historically low populations of big African mammals in the country until the mid 90s (Gibson and Marks, 1995; Gibson, 2001; Naidoo et al., 2011a).

According to empirical literature the program has been successful both in engaging communities and improving wildlife conservation; 39% of all communal lands in Namibia have been registered as conservancies (Naidoo et al., 2011a) resulting in more than 220,000 people included in the program with the creation of 52 communal conservancies covering 120.000 Km² of prime wildlife habitat (WWF, 2011). Concurrently, the stocks of wildlife shifted to a recovery trend (Naidoo et al., 2011a; Naidoo et al., 2011b). This is often attributed to conservancies enabling communities to benefit from wildlife (Bandyopadhyay et al., 2004; Barnes et al., 2002; Weaver and Skyer, 2005).

This empirical observation is consistent with previous theoretical work supporting that rather than compensation policies, it might be more sensible to develop alternative employment opportunities to agriculture and hunting that supply ecological services as a byproduct, such as ecotourism, to induce incentives for conservation (Rondeau and Bulte, 2007). In Namibia, donors and joint ventures with private companies have played a relevant role in the development

of conservancies (Naidoo et al., 2011b). The former often bore many of the initial capital and input costs, including wildlife repopulation and the latter brought earnings out of economic activities dependent on wildlife (Barnes et al., 2002; Weaver and Skyer, 2005). Donors expect these policies to be temporary, having the objective to make conservancy program financially viable in the long-run (Barnes et al., 2002).

Rigorous empirical evaluations of the impact of most types of conservation interventions are strikingly scarce (see Ferraro and Pattanayak, 2006 for a discussion and; Naidoo et al., 2011a; Naidoo et al., 2011c for some empirical evaluations). Even scarcer are theoretical developments that try to analyze and compare different conservation policies and strategies. In this paper we develop a bio-economic model of conservancies to discuss the determinants of success and failure cases of implementation. In addition, we explore the welfare implications of conservancies for local communities as well as its compatibility with conservation objectives. We discuss the results of the conservancy model with respect to the benchmark of open access and of compensation policies for agricultural losses out of wildlife. Our model builds on the work of Bulte and Rondeau (2007) in that we jointly consider two threats to wildlife and three alternative labor opportunities. The wildlife population suffers not only from hunting activities but also from habitat conversion to agricultural land. Also agents in our model have three alternative economic activities, including hunting and agriculture, but in our case the third activity is based on tourism (we abstract from defensive hunting explored in Bulte and Rondeau, 2007). In our model, consistent with the Namibian experience, the conservancy program promotes tourism activities dependent on wildlife that include both ecotourism (i.e., non-consumptive tourism) and Safari hunting. As opposed to Winkler (2011), where tourism is a harmless activity, in our model both ecotourism and safaris affect the carrying capacity of the population of wildlife through land occupation. Additionally, safaris negatively affect wildlife through the intake of animals. In this way, a first contribution with respect to previous literature is that we incorporate non-innocuous tourism activities to the set of feasible activities of communities. A second contribution with respect to previous studies (Bulte and Rondeau, 2007; Bulte and Horan, 2003; Rondeau and Bulte, 2007; Winkler, 2011) is that

we model *bounded rational communities*. We do so by using evolutionary game theory. This methodology is particularly useful for studying spread of new institutions (Friedman, 1991; 1998; Mailath, 1998) and it has been previously used in varied topics in environmental economics (e.g. Noailly et al., 2003; Osés-Eraso and Viladrich-Grau, 2007; Rege, 2004; Sethi and Somanathan, 1996).

We start by presenting a simplified model where communities can choose between farming and hunting activities. This model allows us to analyze the existence and stability of equilibria in the open access scenario as well as to discuss the implications of compensation policies for damages of wildlife on farming activities. Our results for this simple model are consistent with findings in Bulte and Rondeau (2007). Next, we extend the model to incorporate the possibility that communities become members of a conservancy program promoting ecotourism and safari hunting. We focus on the conditions under which the conservancy program can prosper in the long-run. We first provide a formal analysis of existence and stability of equilibria where the conservancy program has membership. Next, we explore the relevance of variables of the context of implementation of conservancy programs that according to previous empirical literature influence the prosperity of conservancies, namely population density, agricultural productivity, the intensity of wildlife damage over agricultural activities and restocking policies. Moreover, we address the conservation achievements and welfare effects of conservancy programs. Finally, section 6 concludes.

2. THE MODEL FOR AGRICULTURE AND HUNTING

2.1. *Time constraint and population dynamics.*

We consider the existence of a group of N communities of agropastoralists that live in the ecosystem of a specific population of wildlife of stock K . By considering *communities* instead of individuals within a single community, as is habitual in the literature, our model is better

suitable to address community-based management, which implies membership at the community level.²

Each community faces a time constraint whereby the total available productive time is divided between agricultural and hunting labor. Therefore, we consider that there are only two options (strategies) available to communities for the allocation of effort: specializing in agriculture (farmer communities) or in hunting (hunter communities). Formally, the time constraints for each type of community are $a_F + h_F = a_H + h_H = 1$, where $a_i \in [0,1]$ for $i=F, H$ is the labor devoted to agriculture by farmers and hunters and $h_i \in [0,1]$ is equivalently, hunting labor by each type of community. Farmers devote more labor to agriculture than to hunting, that is, $a_F > h_F$. The inverse is true for hunters, $a_H < h_H$. Given this, it can be easily verified that farmers devote more time to agriculture than hunters and vice versa, i.e. $a_F > a_H$ and $h_F < h_H$.

Agricultural production is obtained by using labor and land in fixed proportions and is defined by $a_i(A - \alpha K)$. In addition to agricultural labor, a_i , production depends on the agricultural labor productivity in the absence of wildlife, A , and a negative effect of wildlife on agricultural activities, αK . The latter results, for example, from damage by wildlife to crops and to water points (as documented in Barnes et al., 2002; and modeled in Bulte and Rondeau, 2007). When normalizing the price for agricultural product to 1, agricultural production also defines agricultural revenues.

Communities can also hunt wildlife to obtain revenues from it, amounting to $h_i p_g q K$, where h_i is the labor applied to hunting by each type of community, q is a catchability coefficient of game, $q K$ is the hunting labor productivity, and p_g is the price the local

² This necessarily abstracts from strategic interactions between group members. Theoretically community members who do not obtain benefits from the conservancy can have incentives to free-ride on the conservation efforts of others, hindering the environmental achievements of integrated conservation and development programs (Gibson and Marks, 1995) and some authors defend the empirical relevance of this phenomenon (Barrett and Arcese, 1995). Yet, in a survey including over 1000 households in conservancy communities in Namibia the World Bank does not find a significant difference in the benefits derived from the conservancy for households in a community involved in the conservancy with respect to non-involved (Bandyopadhyay et al., 2004). According to the authors of the report this result should not be interpreted as the benefits being small but rather as benefits from the conservancy being more evenly distributed between participants and non-participants than initially expected.

communities obtain from game (e.g. meat and hunting products).³ Hunting drives indirectly an additional benefit insofar by reducing K it reduces wildlife damage on agriculture.

Combining revenues from agriculture and hunting we define revenues for farmer and hunter communities in equation 1.

$$\left. \begin{aligned} \pi_F &= \alpha_F A - \alpha K + h_F p_g q K \\ \pi_H &= \alpha_H A - \alpha K + h_H p_g q K \end{aligned} \right\} (1)$$

The stock of the natural resource, K , affects revenues for each strategy directly. Hunting productivity is increasing in the stock of wildlife whereas agricultural revenues are decreasing in the stock of wildlife. We will further assume that overall farmers' (hunters') revenues negatively (positively) depend on the stock of wildlife, which is guaranteed by the following condition: $\frac{\alpha_H}{1-\alpha_H} < \frac{p_g q}{\alpha} < \frac{\alpha_F}{1-\alpha_F}$. Additionally, strategy selection by each community affects revenues indirectly, though the different impact of each strategy on the wildlife stock as described in section 2.2.

Revenues are not the only determinant of the livelihood of rural communities (WWF, 2006). However, we simplify assuming that the choice of strategy is only determined by the comparison of revenues. Specifically, revenue differentials exert evolutionary pressures on the composition of the group of communities, encouraging the strategies with higher earnings. Through imitation, learning, or other forms of cultural evolution (Friedman, 1991; 1998) communities respond to differences in revenues by modifying their strategies. This behavioral pattern has some inertia, which according to Friedman (1998) can derive from adjustment costs, information imperfections, or bounded rationality. In the following we apply the replicator dynamics (see Mailath, 1998 for a discussion of the relationship between Nash equilibria and evolutionary stable strategies), which is the most widely used deterministic selection criteria (Safarzyńska and van den Bergh, 2010). The evolution over time for the frequency of type i is (Hofbauer and Sigmund, 1998) $\partial s_i / \partial t = s_i(\pi_i - \pi)$, where π is the average revenue in the group as a whole. Then, a community following the high-earning strategy will stick to its

³ As we normalize agricultural price to 1, the other prices in the model represent relative prices to agriculture.

strategy whereas if it is following the low-earning strategy it will change to the other strategy with some inertia defined by the relative spread of this strategy in the population. We impose the habitual simplifying assumption that habitat conversion from agriculture to habitat-supporting land for wildlife, or vice versa, is costless and immediate (Bulte and Rondeau, 2007). Because shares of the composition of the group of communities must equal 1, the system can be reduced to its specification for farmers, and the dynamics of hunters can be determined residually. Then, the dynamics of farmers are $\partial s_F / \partial t = s_F(1 - s_F)(\pi_F - \pi_H)$. After including equation 1 in this expression, the dynamics of farming strategy selection become:

$$\partial s_F / \partial t = s_F(1 - s_F)(a_F - a_H) A - \alpha + p_g q K \quad (2)$$

2.2. Land constraint and wildlife stocks dynamics.

Following previous literature (Bulte and Rondeau, 2007; Bulte and Horan, 2003; Rondeau and Bulte, 2007; Schulz and Skonhofs, 1996; Swanson, 1994) we further assume a quadratic replenishment function of wildlife where the carrying capacity K is determined by available habitat. Specifically, replenishment takes the form $F(K) = rK(K - K)$, where r is a growth parameter (as defined in previous literature above). Agricultural damage to wildlife occurs through habitat destruction, affecting the carrying capacity of the ecosystem, K , which depends on the (homogeneous) total available land L , land occupied by agriculture, L_A , and the substitutability between agricultural land and natural habitat, w_A , so that $K = L - w_A L_A$. Given the fixed proportions technology for agricultural production, the extent of damaged habitat from agriculture is defined by the agricultural labor exercised by the group of communities, so that $\lambda_A L_A = N(a_F s_F + a_H(1 - s_F))$, where λ_A is a parameter of the fixed proportions agricultural production function that determines the agricultural labor required per hectare of agricultural land and the right hand side defines the aggregate agricultural labor from all communities.

Hunting activities also exercise damage on the population of wildlife by the intake of animals. The severity of hunting over the dynamics of wildlife will depend on the number of communities in the region, N , the hunting productivity, qK , the share of communities

undertaking high hunting labor s_H , and the hunting effort of farmers and hunters, h_F and h_H respectively. This results in hunting damage $qKN (h_H s_H + h_F(1 - s_H))$.

Combining the replenishment function, the agricultural damage, and the hunting damage, the dynamics of the resource are specified in equation 3.

$$\frac{\partial K}{\partial t} = rK(L - K) - rNK \frac{w_A}{\lambda_A} (a_H + s_F(a_F - a_H)) - qNK (1 - a_F) + s_H(a_F - a_H) \quad (3)$$

2.3 Open Access equilibria

Equations 2 and 3 constitute a system of two non-linear differential equations that define a set of stable and unstable steady state equilibria (fixed points), which are defined as vectors (s_F, K) that are constant through time. An equilibrium is locally stable when the system tends to return back to this equilibrium after a small departure from it in whatever direction. Otherwise, it is unstable. Only stable equilibria will be considered as reasonable long-run outcome of the system. We label the homogeneous equilibria composed only by farmers or hunters, as F and H respectively. The FH equilibrium is the heterogeneous equilibrium where the two types of communities coexist. The stocks of wildlife corresponding to each of the defined equilibria are K_F^* , K_H^* and K_{FH}^* ⁴.

Homogeneous equilibria always exist. This is because, when there exists only one type of community, its revenues necessarily coincide with average revenues and, consequently (according to equation 2) the population shares are constant. The two homogeneous equilibria define the feasible region for the stock of wildlife. This is, feasible levels of K are between K_F^* and K_H^* whatever is the composition of the set of communities. The existence of a heterogeneous equilibrium requires that both types of communities earn the same revenues. According to equations 1.1. and 1.2, this occurs for the level of wildlife $K_{FH}^\# = A (\alpha + p_g q)$. An heterogeneous equilibrium exists if and only if $K_{FH}^\#$ is in the feasible region. Comparing 1.1 and 1.2 it can be verified that hunters earn higher revenues than farmers when the stock of wildlife in a given situation is higher than $K_{FH}^\#$ and the inverse is true for values of K below

⁴ Section II in Appendix 1 present proofs of existence and stability of equilibria.

$K_{FH}^\#$. This is because the stock of wildlife K has a positive influence on hunters' revenues and a negative one on farmers'. When K_{FH}^* does not exist, either F or H can be stable. If $K_{FH}^\#$ is higher (lower) than both K_F^* and K_H^* , the only stable equilibrium is that where all communities become farmers (hunters).

When a heterogeneous equilibrium exists, its stability depends on the relative harm of the two alternative economic activities on wildlife. When hunting damage of wildlife is higher than damage of agriculture (see figure 1.i) it occurs that $K_H^* < K_F^*$. In this case, the heterogeneous equilibrium is the only stable equilibrium. The proof is straightforward by exploration of equations 1.1 and 1.2. An exogenous marginal increase in the share of farmers from the heterogeneous equilibrium increases the stock of wildlife as a result of the lower hunting human pressure. This wildlife improvement sets revenues for farmers below revenues for hunters, inducing a return back to the equilibrium FH. A similar reasoning can be applied for a marginal increase in the share of hunters to show that there is also convergence back to the heterogeneous equilibrium.

A very different behavior emerges if the environmental impact of agriculture is higher than that of hunting, as in figure 1.ii. In this case, $K_F^* < K_H^*$ and a marginal exogenous increase in the share of farmers reduces, instead of increases, the wildlife stock. As this reduction sets revenues for farmers above those of hunters, it triggers further increases in the share of farmers to the point where all communities are farmers. Similarly, a marginal increase in the share of hunters increases the stock of wildlife and stimulates further shifts to the hunting strategy until all communities are hunters. Therefore, in this case the heterogeneous equilibrium, if it exists, is unstable and both homogeneous equilibria are simultaneously stable.

INSERT FIGURE 1 ABOUT HERE

2.4 Discussion on compensation policies

As it stands, the model can be used to analyze the effects of a policy where communities are compensated for the damage on crops or livestock by wildlife. We just need to introduce a parameter such as ν in the revenue functions in 1.1 and 1.2 that reduces wildlife damage on

agriculture revenues, that is, $\pi_H = a_H A - \alpha K(1 - v) + h_H p_g q K$ and $\pi_F = a_F A - \alpha K(1 - v) + h_F p_g q K$. This would modify the wildlife stock for which revenues of the two strategies are equal, becoming $K_{FH}^{##} = A (\alpha(1 - v) + p_H q)$, whereas K_F^* and K_H^* would remain unaltered.

Consequently, compensation policies will only have an effect on the system when the heterogeneous equilibrium exists. Furthermore, in a similar way as in Rondeau and Bulte (2007) and Bulte and Rondeau (2007), the consequences on wildlife of the compensation policy would depend on the relative environmental damage of hunting and agriculture.

It can be easily verified that as the amount of compensation increases, the higher is the stock of wildlife in the heterogeneous equilibrium (when it exists), $\partial K_{FH}^* / \partial v > 0$. Then, if the heterogeneous equilibrium is the only stable equilibrium, an implementation/increase in compensation has a positive effect on wildlife. However, if the heterogeneous equilibrium is not stable and both homogeneous equilibria are stable, the implementation/increase in compensation increases the basin of attraction of the all-farmers equilibrium to the detriment of all-hunters equilibrium. As a result, for a broader range of initial conditions the system leads to a path of convergence to the low-wildlife stock equilibrium F where all communities are farmers. These results are also in line with findings in Rondeau and Bulte (2007).

3. THE MODEL FOR CONSERVANCIES

The model with conservancies offers communities three strategies; two of them are specializing in agriculture or hunting, as described in section 2. The third one implies registering as a conservancy. As it is the case in many experiences in Southern Africa (such as in Namibia) this entails promoting new economic activities dependent on wildlife; specifically, tourism activities that include both ecotourism (i.e., non-consumptive tourism) and Safari hunting (as described in Barnes et al., 2002; Naidoo et al., 2011a; Naidoo et al., 2011b; Weaver and Skyer, 2005). We incorporate these activities in the conservancy model and at the same time, we allow for conservancies to maintain agricultural and local hunting activities as it is habitual in

implementation of conservancies (Barnes et al., 2002; Naidoo et al., 2011b; Naidoo et al., 2011c; Roe et al., 2001).

Since tourism development needs productive resources that must be diverted from other economic activities (as presented in Barnes et al., 2002), tourism reduces the pressures of communities over wildlife from agriculture and hunting. This is formalized in a time constraint which imposes $a_c + h_c + T = 1$ where T , a_c , and h_c are the labor devoted to tourism, agriculture and hunting activities by a conservancy. Given the time constraint in the open access model, a conservancy devotes less time to agriculture and hunting, jointly considered, than hunters or farmer communities. For simplicity we assume that, when a farmer or a hunting community becomes a conservancy, the main activity shrinks (agriculture and hunting, respectively) while its secondary activity does not substantially change (hunting and agriculture, respectively). This implies $a_c = a_H$ and $h_c = h_F$.

As empirically proved by Naidoo et al. (2011a; 2011c) for the case of Namibian conservancies, income from ecotourism and safari hunting positively depends on wildlife. We incorporate this evidence into the model by considering that tourism activities result in a revenue $T\gamma P_T(K)$, where γ denotes the physical productivity of tourism in terms of number of visitors per unit of labor, and $P_T(K)$ is the price of the tourism product that is assumed to depend linearly on the size of the stock of wildlife, $P_T(K) = \varepsilon K$. Thus, ε is a parameter of the price function accounting for the sensitivity of tourism price with respect to wildlife conditions.⁵

Given these assumptions, equation 4 presents the profit function for the conservancy strategy.

$$\pi_c = a_c A - \alpha K + h_c p_g q K + T\gamma \varepsilon K \quad (4)$$

Substituting $a_c = a_H$ and $h_c = h_F$ and the time constraints, we can express profits of conservancies in terms of the parameters of agricultural effort of farmer and hunter communities, that is, $\pi_c = a_H A - \alpha K + (1 - a_F) p_g q K + (a_F - a_H) \gamma \varepsilon K$.

⁵ For simplification we assume a common price for ecotourism and safari hunting.

Describing again the evolution over time of the different strategies for the communities sharing the same population of wildlife as $\partial s_i / \partial t = s_i(\pi_i - \pi)$ but now for $i=F, H, C$ and considering that the average profit of the group of communities, π , is composed of three strategies, the dynamics of the share of communities are described by equations 5 and 6.

$$\partial s_F / \partial t = s_F(a_F - a_H) (1 - s_F) (A - \alpha + p_g q K) - s_C (\gamma \varepsilon - p_g q K) \quad (5)$$

$$\partial s_C / \partial t = s_C(a_F - a_H) (1 - s_C) (\varepsilon \gamma - p_g q K) - s_F (A - \alpha + p_g q K) \quad (6)$$

We allow both types of tourism activities to impose environmental damages as a result of land occupation and intake of animals. Similarly to agricultural pressures, land occupation of both non-consumptive tourism and safari hunting affects the carrying capacity of the population of wildlife. The carrying capacity is now negatively affected not only by the land occupied by agriculture, L_A , but also by the land occupied by tourism activities, L_T . This results in $K = L - w_A L_A - w_T L_T$, where w_T measures the substitutability between natural habitat and tourism land. Given a fixed proportions technology for tourism production, total labor devoted to tourism, TNs_C , defines the amount of land occupied by tourism activities, that is, $L_T = \frac{TNs_C}{\lambda_T}$, where λ_T is a parameter of the fixed proportions tourism production function that defines the amount of tourism labor required per unit of land devoted to tourism activities.

In addition, safari hunting negatively affects the population of wildlife through the intake of animals. This damage depends on the total tourism labor developed by the group of communities (TNs_C), the physical productivity of labor γ that will define the resulting number of visitors, the proportion of these visitors who engage in safari hunting while at the destination θ , and the catchability of game in Safari hunting q' . The catchability parameter of safari hunting is defined by the more restrictive of two factors: the Safari hunting technology, determining the number of animals than can possibly be killed by unit of labor in safari hunting; and the institutional context, which through licenses and hunting restrictions limits the number of animals

that can be killed if the community is to remain in the conservancy program.⁶ Resulting total damage by Safari hunting is $TN s_C \gamma \theta q' K$.

When adding to the agricultural and hunting damage by local communities (second and third terms of the right hand side of equation 7) the habitat destruction of tourism occupation and hunting damage of safaris (last term of the right hand side of equation 7), the dynamic of the natural resource is described in equation 7, again in terms of parameters of agricultural effort:

$$\frac{\partial K}{\partial t} = rK \left(L - K \right) - rNK \frac{w_A}{\lambda_A} \left(a_H + s_F a_F - a_H \right) - qNK \left(1 - a_F + s_H a_F - a_H \right) - NK \left(a_F - a_H \right) s_C r \frac{w_T}{\lambda_T} + \gamma \theta q' \quad (7)$$

Comparing equations 3 and 7, it can be seen that when a community becomes a conservancy, human pressure on the natural resources changes, with consequences on the dynamics of wildlife. Therefore, additionally, that is, a sensible effect of the conservation scheme on the ecosystem services generated (Naidoo et al., 2011a; Wunder, 2007), is an assumption of the model.

Depending on the parameter values conservancies enhance or are detrimental to the population of wildlife, and consequently we envisage two alternative possible scenarios. First, it may be the case that the set of incentives and restrictions associated to a conservancy program makes conservancies more environmentally-friendly than farmer and hunter communities. We will refer to this conservancy program as *ecologically strongly-designed* (ESD). Empirical observations show that ESD conservancy programs are those that set and monitor conservation objectives (see Stuart-Hill et al., 2005 for the case of Namibia); that convince communities to consider wildlife as an economic asset that is worthwhile to preserve for future revenues; or that achieve to boost a sense of pride and ownership towards wildlife among community members resulting in a desire to leave wildlife as a bequest for future generations (Naidoo et al., 2011a; Weaver and Skyer, 2005). Formally, a conservancy program is ecologically strongly designed

⁶ Arguably, institutions for hunting control in conservancies can be effectively monitored and enforced within the community through local legislation enforcement, social norms and peer-pressure. These can emerge from the devolution of wildlife property rights to the community (Infield, 2001; Naidoo et al., 2011b; Suich, 2010; Weaver and Skyer, 2005).

when a reallocation of a unit of labor from either agricultural or hunting activities to tourism activities reduces total damage on wildlife. This occurs when conditions 8 and 9 are simultaneously met:

$$\frac{rw_T}{\lambda_T} + \gamma\theta q' < \frac{rw_A}{\lambda_A} \quad (8)$$

$$\frac{rw_T}{\lambda_T} + \gamma\theta q' < q \quad (9)$$

Expression 8 warrants that a conservancy has lower environmental impact than a farmer community. It results from the comparison of the environmental impact of tourism activities developed by a conservancy community, $K a_F - a_H r \frac{w_T}{\lambda_T} + \gamma\theta q'$ and the difference in agricultural environmental impact between a farmer and a conservancy community, $rK \frac{w_A}{\lambda_A} a_F - a_H$ (the impact on wildlife of hunting activities by locals is the same in both types of communities). For this condition to be met, the negative impact on wildlife habitat of ecotourism and safari hunting activities should be lower than the agricultural impact $\frac{rw_T}{\lambda_T} < \frac{rw_A}{\lambda_A}$. This would be favoured if tourism land is more compatible with wildlife than agricultural land $w_T < w_A$ and/or if tourism activities are less land-intensive than agricultural production $\lambda_T > \lambda_A$. Moreover, the share of tourist that practice safari hunting $\gamma\theta$ and the catchability of game in safari hunting (q') should not be too large.

As to expression 9, it implies that a conservancy has lower pressure on wildlife than a hunting community. Since both types of communities produce the same environmental impact from agriculture, the comparison of environmental impact lays on hunting and tourism impact. A hunting community damages wildlife through hunting by locals to a larger extend measured by $qK a_F - a_H$ and a conservancy causes an environmental impact through tourism equal to $K a_F - a_H r \frac{w_T}{\lambda_T} + \gamma\theta q'$. For expression 9 to be met, labor devoted to tourism related activities should imply killing less animals through safari hunting than the same labor devoted to direct hunting $\gamma\theta q' < q$. This usually requires a system of hunting permits and quotas effectively enforced by the conservancy that guarantees a value for θ that is low enough

(Naidoo et al., 2011a; Naidoo et al., 2011b; WWF, 2006). The lower number of animals killed by safari should be relevant enough to more than compensate the habitat destruction brought by tourism activities $\frac{r_{WT}}{\lambda_T}$, which is lacking in hunting activities.

Yet, tourism is not always an environmentally friendly activity and in some cases has brought large environmental impacts (as documented, for instance by Liu et al., 2001). Therefore, *ecologically weakly-designed* (EWD) conservancy programs where conservancies bring larger environmental impacts than hunter and farmer communities are not impossible.⁷ This might occur when there exist failures in the institutional design of the conservancy program combined with short-term objectives. Given weak institutions, the desire to obtain quick and large revenues may induce mass-low-quality tourism (too large γ and/or too low λ_T) and/or setting too big hunting quotas (too large θ). Empirically, Kellert et al. (2000) find that a bias of conservancies towards socioeconomic goals endangers the achievement of biodiversity conservation objectives, being the community-based natural resource management programs developed in Nepal and Kenya two examples. Formally, a conservancy program is EWD when a reallocation of a unit of labor from either agricultural or hunting activities to tourism activities increases total damage on wildlife. This happens when both expressions 8 and 9 are not satisfied. In this case, paradoxically, the conservancy program goes against conservation goals.

4. WHEN DO CONSERVANCY PROGRAMS PROSPER?

We now turn to discussing different factors and conditions that determine the degree of adoption of a conservancy program. We present a formal analysis of existence and stability of equilibria. Afterwards, we analyze the role of population density, agricultural productivity, damage from wildlife and restocking policies on a conservancy program adoption.

4.1. Formal analysis.

⁷ For illustrative purposes we focus on extreme cases where either tourism environmental pressures are lower than those of hunting and agricultural activities (ESD conservancies) or on those cases where tourism environmental pressures are the highest of all economic activities (EWD conservancies). Other intermediate situations where tourism environmental pressures are between those of hunting and agricultural activities are also possible.

Equations 5, 6 and 7 constitute a system of three non-linear differential equations that define a set of stable and unstable dynamic equilibria and trajectories. An steady state equilibrium is now defined as a vector (s_C, s_F, K) that is constant through time. In addition to the farmers and hunters homogeneous equilibrium defined above (F and H respectively) it now becomes feasible a homogeneous equilibrium composed by conservancies, C. Furthermore, FC, FH, CH and FCH equilibria are heterogeneous equilibria where two or three different types of communities, coexist. The stocks of wildlife corresponding to each of the defined equilibria are $K_F^*, K_H^*, K_C^*, K_{FC}^*, K_{FH}^*, K_{CH}^*, K_{FCH}^*$.

We will consider that a conservancy program prospers when at least a certain proportion of communities belong to the conservancy program in the long-run. For this to occur it is necessary that at least one equilibrium with conservancies (either C, FC, CH or FCH) exist and is stable. Moreover, if any of these stable equilibria coexist with other stable equilibria without conservancies, it is also necessary that the vector (s_C, s_F, K) of an initial situation of the system is in the basin of attraction of the equilibrium with conservancies.

Similarly to the analysis of the model without conservancies, we can define $K_{FC}^\#, K_{CH}^\#$ and $K_{FCH}^\#$ as respectively the level of wildlife for which revenues of farming and conservancies, conservancies and hunting or the three strategies coincide. Propositions 1-3 present the conditions for existence and stability of equilibria where at least some communities are conservancies. Appendix 1 presents the proof for propositions and corollaries as well as the existence and stability conditions for all equilibria in the system.

Proposition 1. *Equilibrium C always exist and it is stable if and only if $P_g q < \varepsilon \gamma$ and $K_C^* > K_{FC}^\#$. The stock of wildlife when all communities belong to the conservancy, K_C^* , is $\max \{ L - 1 -$*

$$\frac{N}{L} \frac{1}{r} r \frac{w_A}{\lambda_A} a_H + q (1 - a_F) + a_F - a_H - r \frac{w_T}{\lambda_T} + \gamma \theta q' \quad , 0 \ .$$

Condition $P_g q < \gamma \varepsilon$ implies that tourism revenues per unit of effort and unit of wildlife is larger than unitary revenues from hunting. This condition defines that a change of strategy from hunter to conservancy increases revenues for any level of K and composition of the

population of communities. For this to happen a large enough number of visitors should be willing to visit the conservancy area (γ large enough) and should be willing to pay a large enough price for enjoying wildlife (ε large enough). These two combined factors are a measure of the commercial success of conservancies ($\gamma\varepsilon$). Additionally, it is easy to show that condition $K_C^* > K_{FC}^\#$ implies that revenues for conservancies are larger than revenues for farmer communities in equilibrium C.⁸

Proposition 2. *Heterogeneous equilibrium FC exists if and only if $K_{FC}^\#$ is strictly between K_F^* and K_C^* . When equilibrium FC exists, then $K_{FC}^* = K_{FC}^\# = \frac{A}{\alpha + \varepsilon\gamma}$. For an ESD program, the equilibrium cannot be stable. For an EWD conservancy program, a necessary condition for equilibrium FC to be stable is $P_g q < \varepsilon\gamma$.*

Equilibrium FC cannot be stable in ESD programs, since its stability requires $\frac{r w_T}{\lambda_T} + \gamma\theta q' > \frac{r w_A}{\lambda_A}$, which is contrary to the definition of ESD conservancy in condition 8. As in proposition 1, the $P_g q < \varepsilon\gamma$ condition for stability in EWD conservancy programs implies that the conservancies always obtain higher revenues than the hunter communities.

Proposition 3. *Heterogeneous equilibrium CH exists if and only if $P_H q = \varepsilon\gamma$ and $K_{CH}^\#$ is strictly between K_C^* and K_H^* . Heterogeneous equilibrium FCH exists if and only if $P_H q = \varepsilon\gamma$ and $K_{FCH}^\#$ is strictly between the smaller and larger of K_F^* , K_C^* and K_H^* .*

The conditions for the existence of equilibria CH and FCH entail a cutting-edge condition. The probability for this condition to be met in real settings is so small that in the following we will assume that $P_H q \neq \varepsilon\gamma$ and, therefore, equilibria CH and FCH are not relevant for the rest of the analysis.

From propositions 1-3 we derive corollary 1 on the coexistence of multiple equilibria with conservancy communities.

⁸ This condition is satisfied for ESD conservancy programs when $K_{FC}^\# < K_F^* < K_C^*$ or $K_F^* < K_{FC}^\# < K_C^*$; and for EWD conservancy programs when $K_{FC}^\# < K_C^* < K_F^*$.

Corollary 1. *For ESD programs, homogeneous equilibrium C is the only possible stable equilibrium with conservancies. For EWD either equilibrium C or FC is stable.*

Corollary 1 implies that there cannot be multiple simultaneous equilibria with conservancies. When conservancy programs are ESD, the only possible stable equilibrium with conservancies is that where all communities living in the ecosystem join to the conservancy program. On the contrary, when the conservancy program is EWD and $P_H q < \epsilon \gamma$ - which is a requisite for conservancies to prosper - C or FC can be stable, although not simultaneously. For EWD programs, equilibrium C is stable if equilibrium FC does not exist and if $K_C^* > K_{FC}^\#$. When equilibrium FC exists for a EWD program, it is stable given that $P_H q < \epsilon \gamma$ holds.

Despite conservancy equilibria cannot coexist, it is possible for equilibria with and without conservancies to coexist as presented in corollary 2.

Corollary 2. *Only for ESD programs an equilibrium with conservancies can coexist with an equilibrium without conservancies; equilibrium C and F can coexist.*

Equilibria C and F can be simultaneously stable (see appendix 1 for existence and stability conditions of F) but equilibria C and H cannot. The conditions for simultaneous stability of F and C require that a change of strategy from hunter to conservancy always increases revenues ($P_H q < \epsilon \gamma$). Furthermore, switching from farmers to conservancies need to reduce environmental damage ($K_F^* < K_{FC}^* < K_C^*$). The latter defines that F and C can be simultaneously stable only for ESD conservancies. It also entails that it is necessary for equilibria C and F to be simultaneously stable for ESD conservancy programs that equilibrium FC exists. In these circumstances, equilibrium FC is unstable as it is a saddle point. For the system to converge to equilibrium FC, the initial situation must lay on the stable arm of equilibrium FC. Since the probability for this to happen is very small, we rule out this possibility in further discussion on the behavior of the system. Yet, the stable arm of equilibrium FC is highly relevant insofar it separates the basin of attraction of equilibria C and F.

When a stable equilibrium with conservancies coexists with a stable equilibrium without conservancies, initial conditions will determine the long-run prosperity of the conservancy program. Specifically, when both equilibria C and F are stable, the conservancy program is viable for sufficiently high levels of the stock of wildlife (specifically, higher than K_{FC}^*). However, for low enough levels of the stock (lower than K_{FC}^*), revenues for farmers are larger than revenues for conservancies and therefore the system will end up in equilibrium F. The relevance of this finding is further explored in section 4.5.

Figure 2 represents a situation where equilibria C and F are simultaneously stable for a simplified system where hunting communities are omitted⁹. When in this simplified graphical representation the initial values of (s_c, K) belongs to the basin of attraction of equilibrium C (white area) the system will converge to the all-conservancy equilibrium and when the initial values are in the basin of attraction of F (grey area) the system converges to the all-farmers equilibrium.

INSERT FIGURE 2 ABOUT HERE

4.2. *Conservancies in highly populated areas*

Several authors suggest that a conservancy program may not prosper due to excessive human population in an area (Baral and Stern, 2010; Barnes et al., 2002; Barrett and Arcese, 1995; 1998b; Winkler, 2011). Consistently with this opinion, comparative statics show that high human population density negatively affects the prosperity of a conservancy program by reducing participation rates in the long-run and can even hinder the wildlife conservation objectives of the program.

When communities have the same population size the number of communities over the ecosystem size, N/L , can be taken as an indicator to compare the human population density of different areas. Now, if we consider proposition 1, it is straightforward to show that for ESD

⁹ When $P_H q < \varepsilon \gamma$ (a necessary condition for C and F to be simultaneously stable), no equilibrium with hunting communities can be stable. Therefore, omitting the hunting communities has only implications for the set of available trajectories but not for the set of possible long-run outcomes.

programs the stock of wildlife that can be achieved in equilibrium through a conservancy program (i.e. K_C^*) is reduced by higher human population density $\partial K_C^* / \partial (N/L) < 0$ as it entails a higher pressure on wildlife. Moreover, given the values of the rest of parameters, there is a threshold level of N/L , defined as N/L , for which $K_C^* = K_{FC}^\#$. An ESD conservancy program can only prosper in ecosystems with a human population density lower than N/L so that $K_C^* > K_{FC}^\#$. When the human population is N/L , all communities would need to be conservancies to sustain the stock of wildlife that makes conservancy communities at least as profitable as the farming communities

These comments are applicable when the conservancy program is EWD except for situations where FC is the stable equilibrium, i.e. when $K_{FC}^\# \in (K_C^*, K_F^*)$. For that case, since the value of $K_{FC}^\#$ is not affected by the human population density, two ecosystems under the same conservancy program and parameter values but with different human population densities would end up having the same stock of wildlife. Nevertheless, higher population density will result in a lower share of conservancy communities in equilibrium. This is so because a higher population density implies, according to expression 7, higher human pressure on the environment, which can only be (endogenously) compensated with a lower share of communities performing the more damaging strategy, that is, for a EWD program.

4.3. Agricultural productivity.

Some of the previous literature on the success of integrated conservation and development projects defends that policies oriented to increase agricultural productivity show a great promise in facilitating conservation (Barrett and Arcese, 1998b). This result lays on the assumption that agricultural production is static and more complex results emerge when considering, as we do, that agricultural land can be expanded (this effect has been previously studied in Bulte and Rondeau, 2007; Bulte and Horan, 2003; Rondeau and Bulte, 2007; Winkler, 2011), as can land devoted to tourism. In both cases habitat conversion generates a reduction in the carrying capacity of the ecosystem.

Our results support that agricultural productivity not considering wildlife damage, A , has a positive effect on the stock of wildlife in equilibrium FC (stable or not), K_{FC}^* (see proposition 2). The implications of A on the configuration of equilibria and the resulting conservation achievements of a conservancy program differ for three alternative scenarios.

The first scenario is when the conservancy program is EWD and a heterogeneous equilibrium FC exists and is stable. In this case, an increase A results in a larger stock of wildlife. This is because the increase of farmer communities' revenues due to increased agricultural productivity tends to reduce participation in the conservancy program in equilibrium. This in turn increases the equilibrium stock of wildlife, as farmers cause less harm to wildlife than conservancies when they belong to an EWD program.

A second scenario is when the conservancy program is ESD and equilibria F and C are simultaneously stable (that is, as previously discussed, when $K_F^* < K_{FC}^* < K_C^*$). In this case, the agricultural productivity affects the relative size of the basins of attraction of stable equilibria but neither the shares of strategies nor the stock of wildlife in them. Specifically, a higher A implies a smaller basin of attraction to equilibrium C as K_{FC}^* , which defines the location of the stable arm, becomes higher (see figure 3).

INSERT FIGURE 3 ABOUT HERE

Therefore, in this second scenario, higher A makes the conservancy program less resilient to exogenous shocks that may endanger the survival of the conservancy program, such as exogenous falls in K (e.g. due to droughts) or exogenous falls in the share of communities engaged in the conservancy program.

Finally, in a third scenario, a large enough A could make an ESD conservancy program unviable if, given the rest of parameters, equilibrium C becomes unstable. This can occur when raises in A increase the value of $K_{FC}^\#$ to a $K_{FC}^\# > K_C^*$ (see propositions 1 and 2).

In sum, as it is described in the literature (Barnes et al., 2002), a conservancy program is easier to be implemented in areas of low agricultural productivity. Nevertheless, the establishment of conservancies in the Kavango and Caprivi regions of Namibia, where there is room for productive agricultural activities, shows that relatively high agricultural productivity is

not an insurmountable obstacle for a conservancy program (Naidoo et al., 2011b). This is in accord with our model, where a large A compensated with large $\gamma\varepsilon$ (that is, with commercial success of tourism activities) could guarantee a $K_{FC}^\#$ low enough to make the conservancy equilibrium C stable (see propositions 1 and 2).

4.4. Damage from wildlife.

It is also interesting to discuss what are the effects on the adoption of the conservancy program of changes in parameter α , which reflects how exposed to wildlife damage are the agricultural activities. This can be modified for example by fencing but also through compensation policies to wildlife damages. This analysis parallels that of parameter A except for the fact that now $K_{FC}^\#$ negatively depends on α . Therefore, according to our model, higher vulnerability of agricultural productivity to wildlife makes the implementation of a conservancy program less difficult. This result is in accord to empirical findings at the household level in Namibia by the World Bank whereby households suffering damage from wildlife to their crops or livestock are more likely to participate in conservancies (Bandyopadhyay et al., 2004).

Our result derives from a higher α reducing agricultural productivity for a given stock of wildlife, favoring the attractiveness of the conservancy strategy compared to the farmer one. This effect that is present in the model should be balanced with the consideration that a higher exposition to wildlife damage exacerbates other costs for the communities not considered in the model, such as human injury and death, restrictions on people's movement and access to water and forest products (WWF, 2006), that could generate opposition to any conservation program, making the implementation of the conservancy program more difficult.

4.5. Restocking policies

Our model predicts situations where, despite favorable conditions in terms of human population density, agricultural productivity and damage from wildlife, a too low current stock of wildlife prevents a conservancy program to prosper. This is in accord with the empirical literature that

stresses the dependence of the financial viability of conservancies on the existence of a sufficiently large wildlife population (Barnes et al., 2002; Weaver and Skyer, 2005; WWF, 2006).

When different equilibria coexist stably and at least one of those includes communities belonging to a conservancy (i.e. for ESD conservancy programs - where stable C and F can coexist), a restocking policy can switch the system to the equilibrium with conservancies. Therefore, restocking policies such as that applied to conservancies in Namibia (Salamba and Nyae Nyae) and in Zambia (Savé Valley) (Barnes et al., 2002; Lindsey et al., 2009) can promote the prosperity of the conservancy program.

When equilibrium F and C are simultaneously stable, the stock of wildlife in the ecosystem when the conservancy program is launched becomes crucial in the long-run adoption of conservancies. Specifically if the initial stock is too low, the system will begin in the basin of attraction of equilibrium F, which implies that despite it is possible to register as a conservancy, there are no incentives to do so. For such low levels of K becoming conservancy members would reduce revenues for communities compared to the farmer strategy. Therefore, the system rests in equilibrium F. In this case, a once and for all restocking policy that sets the stock of wildlife in the basin of attraction of equilibrium C could be effective to foster permanent participation in the conservancy program. After the one-time restocking, further endogenous improvement in wildlife driven by increasing shares of communities registering as conservancies further raises revenues for conservancy members, driving the system to C. The downward slope of the stable arm in Figure 4 also reveals that for a given level of K , the restocking effort needed to guarantee the spread of the conservancy program is lower the higher the share of communities already registered in the program.¹⁰

In other circumstances, restocking is not an effective policy to foster stable participation in the conservancy program. This is the case, again for ESD programs, when C is unstable ($K_C^* < K_{FC}^\#$). This could stem from either large human population density or from a lack of

¹⁰ The existence of these initial conservancies cannot be explained by the mechanisms of the model. They may consist of pilot experiences or be stimulated to join the program by kick-off financial assistance.

capacity to obtain revenues from wildlife through ecotourism and safari hunting (too little $\epsilon\gamma$). Additionally, EWD programs preclude the existence of multiple stable equilibria and, therefore, a once and for all restocking measure can never foster permanent participation to the program.

5. CONSERVANCY PROGRAMS: WIN-WINS, LOSE-LOSES OR TRADE OFFS?

Case studies present conservancies in Namibia as win-win solutions where conservation objectives become compatible with improvements in living conditions of local communities (Barnes et al., 2002; Naidoo et al., 2011b; WWF, 2006). Bandyopadhyay et al. (2004) find that conservancy benefits accrue in form of both communal public goods and household income. In the same direction, Barnes et al. (2002) find that communities can derive very favorable returns of their investments in conservancies, with financial rates of returns ranging between 23-220%. When considering cash, employment and in-kind financial benefits, Naidoo et al (2011a) estimate that tourism and safari hunting operations represent about 80% of communities' benefits from conservancies. Costs associated with wildlife in the form of damage to crops and water points have been estimated to amount less than 5% of wildlife use values (Barnes et al., 2002). More specifically, according Naidoo et al. (2011b) the conservancy program in Namibia has generated economic benefits for 230000 residents in conservancies and at the same time has supported the recovery of populations of wildlife in communal areas in the program.

The possibility of a win-win outcome for community-based natural resource management generally is, however, contentious and has received much criticism. Infield (2001) argues that revenues from tourism and safari hunting are often not enough to compensate communities for the costs associated to investments in wildlife. Gibbons and Marks (1995) and Barret and Arcese (1995) argue that win-win situations are not possible and consequently criticize the use of these programs as a development and/or conservation strategy for Africa. Kiss (2004) takes a similar position with respect to community-based ecotourism. Moreover, Dressler et al. (2010) present cases in South Africa and Philipines where these sort of programs have prioritized conservation to the detriment of human development. For the Namibian case Suich (2010) stresses the differences between different conservancies in their ability to generate

benefits. According to this author, 32 conservancies earned cash income, 13 conservancies covered their operating costs, and 18 did not generate any cash income in 2006.

Using our model we cannot give a full account of the consequences for communities' welfare of a conservancy program, but we can take a narrower focus on the effects on communities' revenues. In our model, the conservancy program can only affect long-run revenues and the stock of wildlife when there is a stable equilibrium with at least some communities being members of the conservancy. Consequently in the following proposition we focus on the revenue and conservation implications of equilibrium situations where a positive proportion of communities join the conservancy program in the long-run:

Proposition 4. *For ESD conservancy programs win-win situations are possible. Yet a combination of a positive effect on conservation and a negative effect on revenues is also possible. For EWD programs only a combination of a negative effect on conservation and a positive effect on revenues is possible. Therefore, for no type of conservancy program lose-lose outcomes are possible.*

When conservancy programs are ESD, the program either has no effect (if it does not succeed in stimulating stable participation) or improves the wildlife stock in equilibrium with respect to the situation previous to the conservancy creation. This is so because, as presented in section 4.1, the only possible stable equilibrium with conservancies is that where all communities living in the ecosystem join the conservancy program; and, second this equilibrium maximizes the stock of wildlife.

Regarding communities' revenues, when the conservancy program is ESD, two possible situations have to be distinguished. Let us first consider that despite being conservancies, communities are still very dependent on agriculture to raise revenues. In this case, it is possible that overall revenues of a conservancy negatively depend on the stock of wildlife. Then, since ESD conservancy programs improve wildlife, the prosperity of the program could reduce communities' revenues. Nevertheless, evidence for conservancies in Namibia reflects that wildlife damage on agricultural revenues is small compared to the ecotourism and safari hunting earnings (Barnes et al., 2002) . Therefore, it seems more sensible to assume that conservancy

revenues are increasing in the wildlife stock. Then, if equilibrium C is the only possible stable equilibrium, revenues of communities will unambiguously improve with the conservancy program. Therefore in this case the result will be a win-win outcome. The reason is that, whatever is the initial situation when the conservancy program is created, the conservancy strategy yields higher revenues than the other two remaining possible strategies from the very onset (that is why equilibrium F is not stable).¹¹ Moreover, as additional communities join the program, revenues for conservancies keep improving as the stock of wildlife increases. This dynamics continue until all communities join the conservancy program, reaching the maximum revenues and wildlife stock in steady-state. This result is consistent with previous theoretical literature supporting that integrated conservation and development programs can encourage conservation and increase economic well being (Bulte and Horan, 2003; Winkler, 2011).

However, the fact that the conservancy program is ESD and conservancies' revenues are positively affected by the stock of wildlife does not warrant a win-win effect. To show this, let us consider a situation where both equilibrium F and C are stable. In this case, conservation objectives might come at the expense of communities' well-being. This is because communities' revenues in C can be lower than in F despite conservancy revenues are increasing in the wildlife stock. Starting prior to the creation of the conservancy from an equilibrium F, when the program is launched the stock of wildlife is too small for conservancies' revenues to be higher than those of farmer communities. As explained in section 4.5, a restocking policy could increase conservancies' revenues above farmers' revenues if it sets the system in the basin of attraction of equilibrium C. Then, although for these new conditions of the system certainly in equilibrium C conservancies' revenues are above those from the farmer and hunter strategies (and that is why C is stable), nothing warrants that these revenues are above what communities were earning at equilibrium F, when all were farmers. Thus, the revenue implication of conservancies is ambiguous. In these situations the conservancy program might improve wildlife at the expense of local communities' revenues. As such, the model suggests

¹¹ See appendix 2 for a formal proof.

that the need of restocking for the long-run prosperity of a conservancy program might be a warning signal that the program could be detrimental for communities' revenues.

When the conservancy program is EWD, we discussed in section 4.1 that whenever $P_g q < \gamma \varepsilon$, only one stable equilibrium can exist that may be either equilibrium F, C or FC. It is clear that when F is the only stable equilibrium, the conservancy program cannot have any permanent effect on wildlife. On the contrary, if C is the only stable equilibrium, the conservancy program will have a negative effect on wildlife, as it implies the substitution of farming and hunting for tourism, which is more harmful for wildlife in the EWD scenario. Although not so evident, appendix 2 shows that the conservancy program will be also detrimental for conservation when FC is the stable equilibrium. Given the negative impact on wildlife of an EWD conservancy program, this scenario precludes a win-win result. Yet, our model also shows that lose-lose results are also not possible. This is proved in appendix 2, where it is shown that for EWD conservancy programs with positive participation in equilibrium the program improves community's revenues at the expenses of wildlife.

6. CONCLUSION

This paper presents a model of conservancy programs where communities can choose between three strategies; two of them are specializing in agriculture or hunting and the third one implies registering as a conservancy. The conservancy program promotes new economic activities dependent on wildlife, which are based on tourism that include both ecotourism (i.e., non-consumptive tourism) and safari hunting.

We first explore the results of a restricted model where only farming and hunting activities are available to communities. When communities choose between these two activities we obtain consistent results to those presented by Bulte and Rondeau (2007) whereby existence and stability of homogeneous and heterogeneous equilibria depend on whether hunting or agriculture dominate in the impact of human activities over wildlife. Similarly, our results on the implications of compensation policies for wildlife damages on farming activities are also in line with those of Bulte and Rondeau (2007).

We then introduce the possibility for communities to become members of a conservancy program. The model allows for conservancies to enhance or to be detrimental to the population of wildlife, and consequently we envisage two different possible scenarios: that of *ecologically strongly-designed* conservancy programs, where registering as a conservancy reduces human pressures over wildlife, and that of *ecologically weakly-designed* conservancies, where tourism brings larger environmental impacts than previous economic activities. Whether the former or the latter case applies determines the resulting configurations of equilibria as well the effects of key parameters on the long-run behavior of the system. For ecologically-strongly designed conservancy programs our model shows that even if conservancies achieve favorable conditions in terms of number of visitors and willingness to pay for ecotourism services and safari hunting, high human population density hinders participation to the conservancy program and can be an obstacle for a conservancy program to produce positive effects in terms of wildlife conservation. Similarly, a higher agricultural productivity makes the conservancy program less resilient to exogenous shocks that may endanger the prosperity of the program, such as an exogenous fall in the population of wildlife due to droughts. Similarly, lower vulnerability of agricultural productivity to wildlife damages (e.g. by fencing) makes the implementation of a conservancy program more difficult. We also explore the mechanisms upon which restocking policies may help to foster participation to the conservancy program. The model shows situations of multiple stable equilibria where despite favorable conditions in terms of human population density, agricultural productivity and damage from wildlife, a too low current stock of wildlife prevents a conservancy program to prosper. In these cases, a one and for all restocking measure can switch the system to the equilibrium with conservancies. Therefore, restocking policies may be in some circumstances critical for prosper of the conservancy program.

Finally, we explore to what extend the two targets that conservancies aim to achieve, namely wildlife conservation and communities' welfare (that we rather address by the narrower measure of revenues), are compatible. Our results show that when conservancy programs are ecologically strongly designed, the program either has no effect or improves the wildlife stock

whereas it may improve or reduce communities' revenues. Therefore, despite win-win situations can occur, these are not guaranteed. It is possible to identify situations where an ecologically-strongly designed program improves wildlife at the expense of local communities' revenues. Moreover, the model suggests that a need for restocking to support the launch of a conservancy program might be a warning signal that the conservancy program could be detrimental for communities' revenues. As for ecologically-weakly designed conservancy programs, they either have a negative or a nil permanent effect on wildlife. However, a negative effect on revenues is precluded. Then, for the case of weakly-designed conservancy programs, neither win-win or lose-lose situations between conservation and welfare can occur.

All in all, our model integrates key variables that according to previous empirical evidence influence the prosperity of conservancy programs. In doing so we can provide a theoretical background for part of the variability in success of community-based programs for wildlife management. In this regard not only design attributes of the program are relevant but also the context of implementation and the initial conditions in which a system is when a program is launched.

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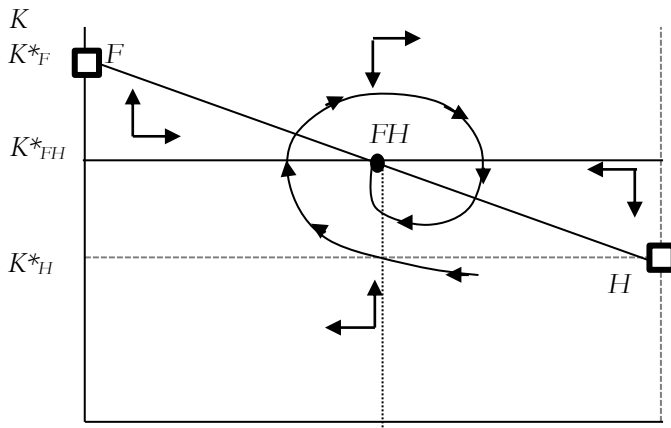
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FIGURES

Figure 1. Open access dynamics for the heterogeneous equilibrium with hunting and farming communities.

i- Hunting effect dominates habitat effect



ii- Habitat effect dominates hunting effect

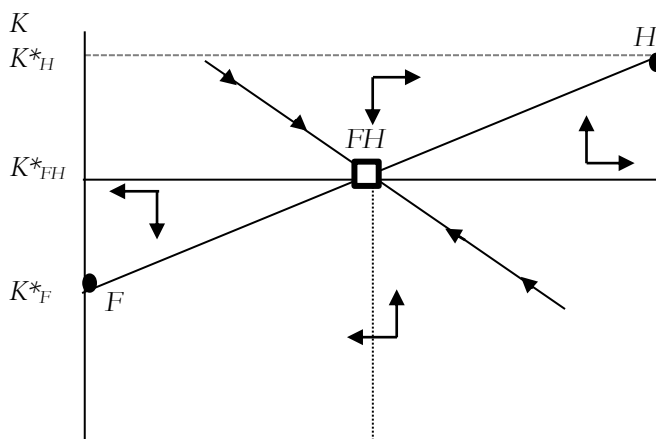


Figure 2. Equilibria configuration for an ecologically strongly-designed conservancy program.

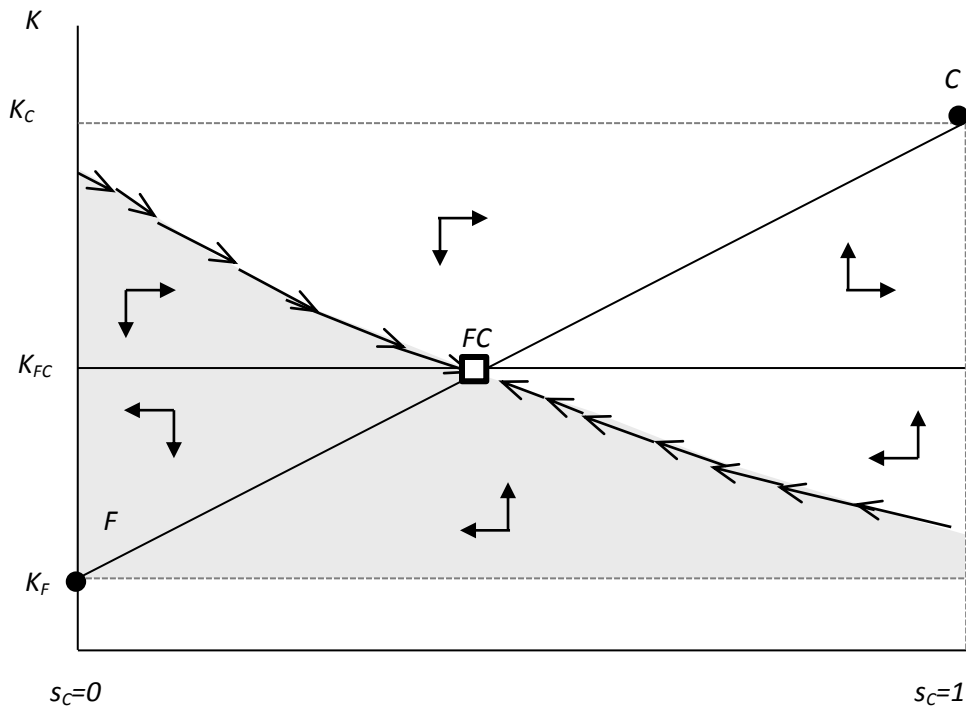
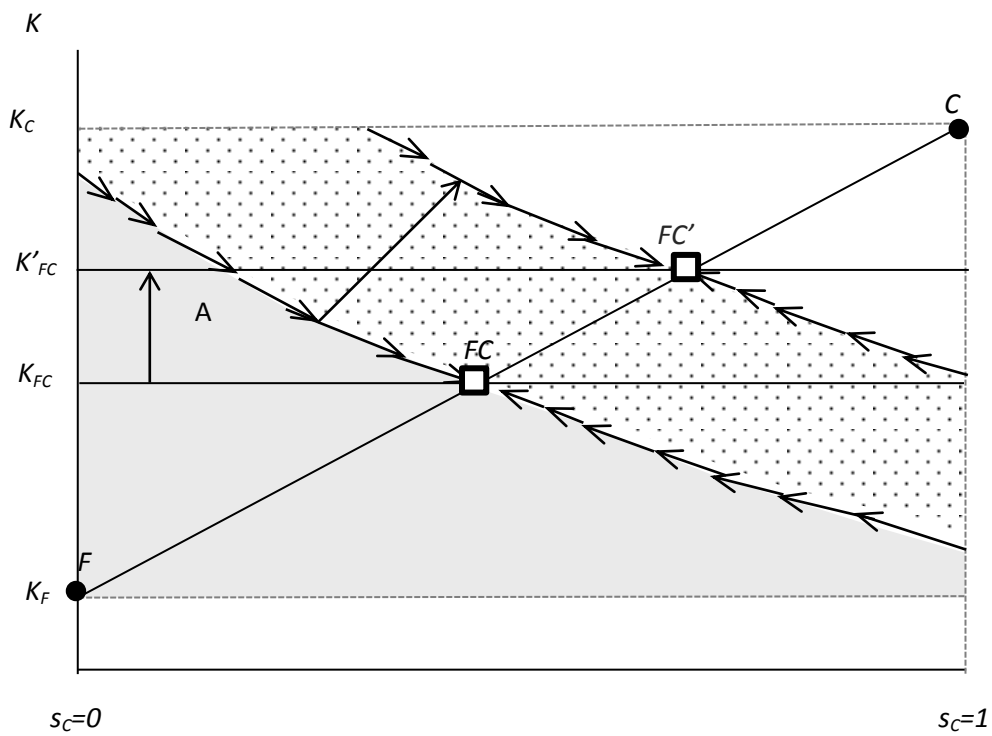


Figure 3. Implications of an increase in agricultural productivity in an ecologically strongly-designed conservancy program.



APPENDIX 1

I. System with three strategies (section 4.1)

i. Existence of equilibrium

A mere inspection of equations 5 and 6 is needed to verify that homogeneous equilibrium C, F and H always exist. Moreover, when we set $K = 0$, equation 7 reveals that wildlife in C, F and H is, respectively:

$$\begin{aligned} K_A^* &= L \left[1 - \frac{N}{L} \frac{1}{r} r \frac{w_A}{\lambda_A} s_A a_A - a_H + a_H + q \right] (1 - a_A) \quad . \\ K_H^* &= L \left[1 - \frac{N}{L} \frac{1}{r} r \frac{w_A}{\lambda_A} a_H + qNK \right] (1 - a_A) + s_H a_A - a_H \\ K_C^* &= L \left[1 - \frac{N}{L} \frac{1}{r} r \frac{w_A}{\lambda_A} a_H + q \right] (1 - a_A) + a_A - a_H \left[r \frac{w_T}{\lambda_T} + \gamma\theta q' \right] \end{aligned}$$

Provided that they are positive. Otherwise, $K_i^* = 0, i=C,F,H$.

As to heterogeneous equilibria, these require profit equalization of stable strategies, which for equilibria FH and FC implies $K = K_{FH}^\# = \frac{A}{\alpha + p_g q}$ and $K = K_{FC}^\# = \frac{A}{\alpha + \gamma \varepsilon}$, respectively, as it can be verified with expressions 1 and 4; whereas for equilibria HC and FCH implies $p_g q = \gamma \varepsilon$ and $K = K_{HC}^\# = K_{HCF}^\# = \frac{A}{\alpha + p_g q}$, as can be verified using the same expressions.

Equilibrium shares for each heterogeneous equilibrium can be worked out using equation 7 for the dynamics of wildlife, considering $K = 0$ and the corresponding equilibrium level of wildlife. Specifically, for equilibrium FC it can be verified that, according to expression 7 the steady state stock of wildlife can be expressed as the weighted average $K_{FC}^* = s_C^* K_C^* + (1 - s_C^*) K_F^*$, which implies $s_C^* = \frac{K_{FC}^* - K_F^*}{K_C^* - K_F^*}$. Using this expression, and given that in the FC equilibrium $K_{FC}^* = K_{FC}^\#$, it can be easily verified that $0 < s_C^* < 1$ if and only if $K_{FC}^\#$ is strictly between K_C^* and K_F^* .

ii. Stability of equilibria.

The characteristic equation of the Jacobian of a three-dimensional system is

$$x^3 - c_1 x^2 + c_2 x - c_3 = 0$$

where $c_1 = \text{Trace}J$, $c_2 =$ sum of all second-order principal minors of J , and $c_3 = J$. There are three possibilities regarding stability. If $\text{trace}J < 0$, $c_2 > 0$, and $J < 0$, the steady state is stable. If $\text{trace}J > 0$, $c_2 > 0$, and $J > 0$, the steady state is unstable. Otherwise, the steady state is conditionally stable, that is, there is a stable manifold or stable arm associated with that equilibrium.

The model is defined by non-linear differential equations 5, 6 and 7. Linearization results in a system which Jacobian is

$$J = \begin{pmatrix} J_{11} & J_{12} & J_{13} \\ J_{21} & J_{22} & J_{23} \\ J_{31} & J_{32} & J_{33} \end{pmatrix},$$

where:

$$\begin{aligned} J_{11} &= \frac{\partial s_F}{\partial s_F} = -a_F - a_H s_C \gamma \varepsilon - p_g q K + 1 - 2s_F a_F - a_H A - \alpha + p_g q K; \\ J_{12} &= \frac{\partial s_F}{\partial s_C} = -s_F a_F - a_H \gamma \varepsilon - p_g q K \\ J_{13} &= \frac{\partial s_F}{\partial K} = s_F(a_F - a_H) - 1 - s_F \alpha + p_g q - s_C \gamma \varepsilon - p_g q; \\ J_{21} &= \frac{\partial s_C}{\partial s_F} = -s_C(a_F - a_H) A - \alpha + p_g q K \\ J_{22} &= \frac{\partial s_C}{\partial s_C} = (a_F - a_H)s_F A - \alpha + p_g q K - 1 - 2s_C a_F - a_H \gamma \varepsilon - p_g q K \end{aligned}$$

$$J_{23} = \frac{\partial s_c}{\partial K} = s_c a_f - a_H (1 - s_c) \gamma \epsilon - p_g q + s_A \alpha + p_g q$$

$$J_{31} = \frac{\partial K}{\partial s_f} = N a_f - a_H K q - r \frac{w_A}{\lambda_A}; J_{32} = \frac{\partial K}{\partial s_c} = N(a_f - a_H)K q - \gamma \theta q' - r \frac{w_T}{\lambda_T}$$

$$J_{33} = \frac{\partial K}{\partial K} = r L - K - rK - rN \frac{w_F}{\lambda_F} s_f(a_f - a_H) + a_H - qN (1 - a_f + s_H(a_f - a_H)) - N(a_f - a_H)s_c r \frac{w_T}{\lambda_T} + \gamma \theta q'$$

where K , s_c , and s_f take different values depending on the specific equilibria we consider.

ii.1. Stability of homogeneous equilibria.

It can be verified that for homogeneous equilibria $DET J = J_{11} \cdot J_{22} \cdot J_{33}$, $TRACE = J_{11} + J_{22} + J_{33}$ and $c_2 = J_{11} \cdot J_{22} + J_{11} \cdot J_{33} + J_{22} \cdot J_{33}$. Then, necessary and sufficient conditions for stability are $J_{11} < 0$, $J_{22} < 0$, $J_{33} < 0$, which imply:

Equilibria	$J_{11} < 0$	$J_{22} < 0$	$J_{33} < 0$
F	$K_F^* < \frac{A}{\alpha + p_g q}$	$K_F^* < \frac{A}{\alpha + \gamma \epsilon}$	$F'(K) < D'(K)$
H	$K_H^* > \frac{A}{\alpha + p_g q}$	$p_g q > \gamma \epsilon$	$F'(K) < D'(K)$
C	$K_C^* > \frac{A}{\alpha + \gamma \epsilon}$	$\gamma \epsilon > p_g q$	$F' K < D'(K)$

where:

$$F' K = r L - K - rK \text{ and } D' K = rN \frac{w_A}{\lambda_A} a_H + qN (1 - a_f + N(a_f - a_H)(r \frac{w_T}{\lambda_T} + \gamma \theta q'))$$

The condition of the third column is redundant for equilibria C and H since conditions $K_C^* > \frac{A}{\alpha + \gamma \epsilon}$ and $K_H^* > \frac{A}{\alpha + p_g q}$ imply that stability requires $K_i^* > 0$, $i=C,H$. But when $K_i^* > 0$ it results that $F' K_i^* = D'(K_i^*)$, which it is easy to show that implies $D' K_i^* = r L - K_i^*$, which, in turn, implies $F' K_i^* < D'(K_i^*)$.

ii.2. Stability of heterogeneous equilibrium $0 < s_c, s_f < 1$; $s_H = 0$; $K = K_{FC}^{\#} = \frac{A}{\alpha + \gamma \epsilon}$

Given that $J_{11} = J_{12} = \frac{s_f}{1-s_f} J_{21} = \frac{s_f}{1-s_f} J_{22}$, $J_{13} = -J_{23}$, it can be shown that $DET J = \frac{J_{11} J_{13}}{s_f} (J_{32} - J_{31})$, $TRACE(J) = J_{11} + J_{22} + J_{33} = \frac{1}{s_f} J_{11} + J_{33}$ and $C_2 = \frac{1}{s_f} J_{11} \cdot J_{33} + J_{13} J_{32} - J_{31}$

If $J_{11} > 0$, and given that $J_{13} < 0$, then $J_{32} > J_{31}$ is a necessary condition for $DET(J) < 0$ and $J_{33} < 0$ a necessary condition for $TRACE < 0$; But if $J_{11} > 0$, $J_{32} > J_{31}$ and $J_{33} < 0$, then $c_2 < 0$. Therefore, $J_{11} < 0$ and $J_{32} < J_{31}$ are necessary conditions for stability, which imply, respectively, $\gamma \epsilon > p_g q$ and $\gamma \theta q' + r \frac{w_T}{\lambda_T} > r \frac{w_A}{\lambda_A}$. Condition $J_{33} < 0$ (that is, $F' K < D'(K)$) is not necessary for stability but the three previous conditions jointly considered are sufficient for stability.

ii.3. Stability of heterogeneous equilibrium $0 < s_f, s_H < 1$, $s_c = 0$, $K = K_{FH}^* = \frac{A}{\alpha + p_g q}$

Given that $J_{11} = J_{21} = J_{23} = 0$, then $DET J = J_{22} \cdot J_{13} \cdot J_{31}$, $TRACE(J) = J_{22} + J_{33}$ and $C_2 = J_{22} \cdot J_{33} + J_{31} \cdot J_{13}$. Given this, it can be shown that necessary conditions for stability are:

$$J_{22} < 0 \rightarrow \gamma \epsilon < p_g q$$

$$J_{31} < 0 \rightarrow r \frac{w_A}{\lambda_A} < q$$

As in the previous case, $F' K < D'(K)$ is not necessary for stability but jointly with the previous two conditions form a set of sufficient and necessary conditions for stability.

II. The model with two strategies (section 2.3)

Most of previous analysis referring to the model with three strategies can be applied to the model with two strategies. Specifically, what is said regarding existence of equilibria F, H and FH is correct when the conservancy strategy is absent. As to stability, the Jacobian of the model with two strategies is composed by derivatives $\frac{\partial s_F}{\partial s_F}, \frac{\partial s_F}{\partial K}, \frac{\partial K}{\partial s_F}, \frac{\partial K}{\partial K}$ calculated above, setting $s_C = 0$. Considering this and the analysis performed for the model with three strategies it is easy to show that necessary and sufficient conditions for stability of homogeneous equilibria F and H are, respectively, $K_F^* < \frac{A}{\alpha + p_g q}$ and $K_H^* > \frac{A}{\alpha + p_g q}$ and $F'(K) < D'(K)$ for both, whereas necessary and sufficient conditions for stability of equilibrium FC are $r \frac{wA}{\lambda_A} < q$ and $F'(K) < D'(K)$.

APPENDIX 2

I. Proof that the prosperity of a EWD conservancy program reduces wildlife conservation.

This is directly evident when the stable equilibrium with positive participation to the program is equilibrium C, since, by construction, K_C^* is the lowest of all possible steady state wildlife stock levels.

If FC is the stable equilibrium with positive participation to the program, then $K_{FC}^\# < K_F^*$ (see proposition 2). This implies that if equilibrium F is the initial situation when the conservancy program is launched, its prosperity reduces the wildlife stock from K_F^* to $K_{FC}^\#$.

However, it is possible that FC is the stable equilibrium with positive participation to the program and $K_{FC}^\# > K_H^*$. If this is the case and equilibrium H represents the situation prior to the launching of the program, then the conservancy program could produce a positive effect on wildlife as K would converge from K_H^* to $K_{FC}^\#$. Several pieces of evidence preclude this possibility. First, equilibrium H can only be stable in the open access (that is, prior to the launching of the conservancy program) if $K_{FH}^\# < K_H^*$. Second, a condition for equilibrium FC to be stable is $p_g q < \varepsilon \gamma$, which implies $K_{FC}^\# < K_{FH}^\#$. Therefore, a situation where H is stable prior to the launching of the program and FC is stable after this event requires $K_{FC}^\# < K_H^*$ and, therefore, implies that the prosperity of the program is to the detriment of wildlife.

II. Proof that lose-lose scenarios are not possible.

As explained in the main text, when the conservancy program is ESD, the program cannot diminish wildlife conservation and, therefore, a lose-lose situation is not possible.

As explained in this appendix, the prosperity of an EWD conservancy program reduces wildlife conservation. Here we show that it cannot reduce communities' revenues in the steady state. To show this, let us first remember that for the program to prosper it is necessary that $p_g q < \varepsilon \gamma$ (see propositions 1 and 2). This implies that unitary revenues from tourism are higher than unitary revenues from hunting for any K . Therefore, any hunter community would increase revenues if joining the conservancy program.

Then, when initially all communities' are hunters (equilibrium H), the prosperity of the conservancy program improves revenues.

When equilibria F or FH represent the initial situation prior to the launching of the program, for the conservancy program to prosper either equilibria C or FC must be stable and, according to corollary 2, this renders equilibria without conservancies' unstable (either F or FH). This in turn implies that initially (that is, in equilibrium F or FH) conservancies' revenues are higher than farmer communities' revenues. Given that the conservancy program is EWD, wildlife will deteriorate as more and more communities join the conservancy program. As revenues of farmer communities negatively depend on wildlife, lower stocks of wildlife induced by the prosperity of the conservancy program will increase farming revenues. In the new stable equilibrium, C or FC, conservancies' revenues will be equal (in equilibrium FC) or higher (in equilibrium C) than farmer communities' revenues. These farmer communities' revenues are in turn higher than farmer communities' revenues in the initial situation. Therefore, conservancies will enjoy higher revenues compared to the initial situation.

III. Proof of the effects of an ESD conservancy program on communities' revenues when equilibrium C is the only stable equilibrium.

If equilibrium C is stable, then $\gamma\varepsilon > p_g q$, which implies $\pi_C > \pi_H$ for any possible values of the composition of communities and wildlife stock (see expressions 1 and 4). Therefore, the conservancy program implies an improvement in revenues if the system was in equilibrium H previous to its creation. Moreover, when $\gamma\varepsilon > p_g q$ and equilibrium F is not stable, then $K_F^* > K_{FC}^\# = \frac{A}{\alpha + \gamma\varepsilon}$ (see stability conditions for homogeneous equilibria in appendix 1), which implies that in equilibrium F $\pi_C > \pi_F$ (again compare expressions 1 and 4). If π_C is monotonously increasing in K and the conservancy program is ESD (which implies $K_F^* < K_C^*$), then revenues for conservancies improve as a result of the increase in K induced by more and more communities joining to the program, until reaching a maximum in equilibrium C. However, if π_C is decreasing in K , the increase in K reduces conservancies' revenues, leaving the possibility of π_C evaluated in equilibrium C being smaller than π_F evaluated in equilibrium F.

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Evolutionary success and failure of wildlife conservancy programs

Abstract

This paper develops an evolutionary bio-economic model for hunting, farming and tourism (non-consumptive and safari hunting) to study the determinants of the prosperity of conservancy programs. The model is inspired in the Conservancy program of Namibia, despite it is of more general applicability to other contexts. We explore the relevance of the design attributes of conservancy programs in their prosperity in the long-run as well as the relevance of variables of the context of application highlighted in empirical literature. In addition, we explore the welfare implications of conservancies for local communities and its compatibility with conservation objectives. We discuss the results of the conservancy model with respect to the benchmark of open access and of compensation policies for agricultural losses of wildlife.

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