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A Bio-Economic Model of Community Incentives for Wildlife Management Under CAMPFIRE

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Abstract This paper formulates a bio-economic model to analyze community incentives for wildlife management under benefit-sharing programs like the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe. Three agents influence the wildlife stock: a parks agency determines hunting quotas, outside poachers hunt illegally, and a local community may choose to protect wildlife by discouraging poaching. Wildlife generates revenues from hunting licenses and tourism; it also intrudes on local agriculture. We consider two benefit-sharing regimes: shares of wildlife tourism rents and shares of hunting licenses. Resource sharing does not necessarily improve community welfare or incentives for wildlife conservation. Results depend on the exact design of the benefit shares, the size of the benefits compared with agricultural losses, and the way in which the parks agency manages hunting quotas.

Keywords Bio-economics · Benefit sharing · CAMPFIRE · Conservation · Elephants · Hunting quotas · Poaching · Renewable resources · Wildlife

JEL Classification H1 · Q20

This paper has not been submitted elsewhere in identical or similar form, nor will it be during the first three months after its submission to the Publisher.

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1 Introduction

Many prominent species of wildlife are threatened with extinction because of habitat loss, poaching, and competition with other land uses. Recent conservation efforts have begun to focus not only on regulation and enforcement of restrictions on trade and use of wildlife, such as the bans on ivory, but also on mitigating some of the economic consequences of human–wildlife conflicts. In particular, major international and non-governmental conservation organizations are supporting initiatives to promote and share the economic benefits of wildlife conservation with local communities.[1] At first glance, benefit sharing seems unquestionably likely to encourage wildlife conservation and to improve incomes in poor rural communities by making wildlife a valuable resource. However, before we can draw this conclusion, more attention must be paid to the institutional and dynamic complexities of wildlife management problems (Brandon and Wells 1992). Important policy questions lie in how the actual design of benefit-sharing initiatives affects their success, in terms of both wildlife and community welfare.

Zimbabwe offers an interesting example of these issues. The establishment of national parks, game reserves, and safari areas in the late 1920s may have helped avert biodiversity and wildlife loss, but it also displaced rural communities from land that was traditionally theirs. Cultivation and grazing lands were expropriated, and the old practice of subsistence hunting became illegal. Although wildlife gained a permanent residence in the parkland, it could also roam freely in surrounding areas, destroying crops and threatening livestock and people. Thus the creation of parklands created a conflict between wildlife conservation and agricultural development, since the growth of the wildlife populations depends on the size of the parkland (Cumming 1989; Swallow 1990; Emerton 2001). While hunting is ordinarily disallowed in national parks, many countries like Zimbabwe have safari areas in which limited hunting occurs alongside benign tourism.

In 1989, Zimbabwe instituted a benefit-sharing program for wildlife, the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE). It focused especially on communal areas adjacent to national parks, where wildlife intrusion was most problematic and agricultural productivity was marginal.[2] CAMPFIRE gave communities co-ownership of local natural resources, which generate income through leasing of trophy hunting concessions, harvesting of natural resources, tourism, live animal sales, and meat cropping. The program went through a period of intense development during the 1990s and has inevitably suffered from the recent crisis in the country; however, in that first decade, there were some important signs of success—but also some considerable difficulties.

In this paper, we analyze welfare implications of resource benefit sharing in a typical wildlife-abundant rural area in Zimbabwe in which wildlife conservation can conflict with agricultural production. Although inspired by CAMPFIRE, our analysis is distilled to represent revenue-sharing programs for wildlife more broadly. Several authors have analyzed bio-economic models to study the role of communities in wildlife management generally (e.g. Baland and Platteau 1996) or in East Africa (e.g. Skonhoft 1998).[3] Skonhoft (1998) considers the impact of different property-sharing regimes on the incentives of the park manager and

[1] See, for instance, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) program on bushmeat (http://www.cites.org/eng/prog/bushmeat.shtml).

[2] In Zimbabwe, more than 90% of the communal lands are located within agriculturally marginal regions plagued by persistent drought, low and erratic rainfall, and poor soil. The hunger for land is one of the factors that led to occupation of commercial farms in more productive regions. It also creates considerable tension over wildlife in the national parks that border the less productive farm areas.

[3] See also Shulz and Skonhoft (1996), Skonhoft and Solstad (1996), and Skonhoft and Solstad (1998),

on the welfare of the communities. Communities are passive in this model, receiving the revenues of the property shares and the burdens of the wildlife interactions, which take the form of intrusions. Of course, the issue of providing incentives for locals to accept wildlife that poses a threat to their crops or cattle is not limited to Africa. For a Swedish case, see Sellenthin and Skogh (2004) or Zabel et al (this issue).

The model in this paper extends earlier work in several ways. First, whereas previous papers study poaching by the local communities, this paper studies poaching conducted by outsiders (from the local community's perspective). This approach is inspired by the situation in Zimbabwe in the mid to late 1990s, when the primary concern was not subsistence poaching of small animals by local communities, but rather large-game poaching, most of which was believed to be conducted not by villagers but by well-organized and armed groups of outsiders, indeed often by foreign nationals.[4] Second, unlike previous papers that give prominence to anti-poaching enforcement by the parks agency, ours emphasizes the anti-poaching effort that local communities may exert, as co-owners of wildlife in their area. In the communal areas, traditional policing is difficult and the only people with a chance to monitor effectively are the local inhabitants. Under CAMPFIRE, a significant amount of the anti-poaching enforcement was applied by local communities through the hiring of community-based anti-poaching units.[5] This characterization of community effort contrasts with previous studies in which local populations engage in open-access hunting and reap those use benefits directly.[6]Third, since the mandate for the parks agency is an important policy itself, we consider the role of the decision rule for determining hunting quotas in influencing community behavior.

Fourth, we depart from the social planner problem to study the interaction among several agents, taking a particular look at the economic game played between the park managers, the communities, and the poachers. An important distinction is that, whereas traditional management problems choose harvesting directly, communities can influence the net offtake (hunting plus poaching) only indirectly, through their anti-poaching effort.[7] Johannesen and Skonhoft (2005) incorporate a response function for communities, whose members are assumed to be the primary poachers, such as for meat cropping. They present a Nash game in which both the communities and the park manager choose harvesting effort in a form of duopoly competition. Although they pose similar questions about resource sharing, their setup contrasts with our characterization, in which the parks agency sets the number of hunting licenses (as opposed to hunting effort), while the community influences poaching by outsiders, who have a distinct response function. These subtle differences change the nature of the strategic

[4] The public perception in Zimbabwe, reinforced by the media and the parks agency, is that commercial poachers are Zambians and, occasionally, Mozambicans (Duffy 2000).

[5] Child (1995):7 states, "Those functions best decentralized, such as game monitors and problem animals reporters, should be the responsibility of the Wards and Vidcos [sub-district entities]. With 80% of wildlife income going to the communities they should bear the burden of paying game-guards, etc. … communities should cover the cost of management at their level while Council should only pay for centralized costs." Thus, the wildlife policy in Zimbabwe clearly articulates and assigns proconservation roles outside the Parks and Wildlife Estate to councils and communities.

[6] See Bulte and Horan (2003) or Kinyua et al. (2000), who focus more on the competition between wildlife and grazing for large commercial ranchers. Barrett and Arcese (1998) show that game meat transfers can discourage poaching by crowding out illegal meat with legal meat, but the sum of the illegal and legal offtake can increase with the program. Substitution and income effects may also depend on access to labour markets; see Muller and Albers (2004) or Johannesen (2004) on how opportunities for formal sector work discourage illegal hunting.

[7] This approach contrasts with other studies of ivory trade, which look at the incentives for enforcement on the part of the government (Milner-Gulland and Leader-Williams 1992; Bulte and van Kooten 1999; Swanson 1993; Khanna and Harford 1996).

interactions. We demonstrate that conservation incentives depend critically on several factors, including the design of the benefit-sharing program, the hunting quota management regime, and the resource itself. We echo the concerns of Ferraro (2001) and Ferraro and Simpson (2002), that incentives created by benefit-sharing programs are indirect, ambiguous, and less cost-effective than direct payments for conservation. However, these programs do have the advantage of being self-financing rather than requiring external funds. Therefore, we point out design features and conditions that are more likely to lead to unambiguous and more effective incentives.

We set up the model in Sect. 2. Thereafter, we analyze how the allocation of the property rights from the two sources of wildlife profits affects conservation incentives and welfare (Sect. 3). The paper concludes in Sect. 4.

2 Model

The bio-economic model comprises three agents (the parks agency, poachers, and a local community), two control variables (hunting quotas and anti-poaching effort), and a stock variable representing wildlife. Economic rents are generated from wildlife (viewing and hunting), which may be distributed between the parks agency and the local community, and from agricultural production. Since the parks agency typically licenses both hunting and many tourism concessions, we consider this agency to represent safari lodges and other such agents as well. From a social planner's perspective, other use and existence values may call for a greater focus on conservation; however, these values do not appear in the objective functions of the agents at hand.

Land use within the park is restricted to wildlife conservation; the main agricultural alternatives outside the park are livestock and crop production. Wildlife tends to roam the lands adjacent to the park. Intruding wildlife damages crops, competes with livestock for the scarce grazing land, transmits diseases,[8] and otherwise reduces agricultural productivity. It also presents a threat to property and to the population itself, although such threats are not the main focus of this paper.

The ecological interaction between wildlife and agricultural productivity is assumed in this paper to be unidirectional—a negative effect from wildlife to agriculture, but not vice versa. For example, wildlife roams into the rangeland, but as in Zimbabwe, the local community is not allowed to take its livestock into the parks. Thus the extent of wildlife conflict can be depicted simply as a function of agricultural rents that decline with the stock of wildlife.

Because of the prominence of large game (particularly elephants and water buffalo) in the CAMPFIRE program, we ignore consumptive benefits to communities, whose hunters target smaller, more abundant species. Implicitly, we assume that the effects of poaching for the pot are separable from large-game management.

The rents from wildlife conservation thus arise in the form of revenue from hunting licenses and from benign tourism. As in Skonhoft (1998), we model two kinds of benefit sharing. Before CAMPFIRE, all of the hunting license rents—and most of the tourism profits—rested with the parks agency (including safari lodges, etc). Under CAMPFIRE, the issuance of hunting quotas to the local community implies giving the local community a share of the hunting profits (α), either by sharing the revenues or by allowing it to sell its quotas directly. The local community may also get a share of the profits from benign tourism (β). We assume that the local community's profit shares α and β are fixed through time and satisfy $0 \leq \alpha \leq 1$

[8] For example, buffalo can infect livestock with foot-and-mouth disease or brucellosis.

and $0 \leq \beta \leq 1$.[9] The remaining profit is assumed to go to the parks agency. An important question addressed in this paper is how the relative allocation of the benefits from wildlife activities affects the conservation and anti-poaching incentives.

Ultimately, the answer depends on the interaction between the community and the parks agency management regime in equilibrium. Furthermore, for benefit sharing to have any effect, requires that the community have a fair amount of sophistication—that they perceive their efforts will influence the biological dynamics, as well as the response of the parks agency in setting hunting quotas. To focus on this interaction, and since we are primarily concerned with equilibrium effects rather than the dynamics, we consider a static problem representing the optimization of a long-run equilibrium. That is, the optimizing agents maximize their objective functions, subject to the constraint that the wildlife stock be in a steady-state equilibrium. The problem is more akin to maximum sustainable yield analysis, in that it ignores single-period deviations that could improve the present discounted value of the objective function, but it has the clear benefits of rendering the agents' response functions transparent, allowing for a straightforward solution to their interaction.[10]

2.1 Wildlife

Assuming that wildlife can be represented as a single composite species, its biomass at a specified point in time (where the time index is omitted) is given by W. In our case, this can be considered the stock of large game in the communal area. The growth in the stock is given by $\dot{W} = F(W) - h - q$, where $F(W)$is the natural growth function of the stock of wildlife, h is the offtake from hunting quotas, and q represents the loss due to poaching. As a result, in the steady-state equilibrium,

$$F(W) - h - q = 0 \tag{1}$$

Additional implicit constraints are $W(t) \geq 0, \quad W(0) = W_0, \quad W(\infty) < \infty$. We assume that $F(W)$ is a non-monotonic, strictly concave function where $F(0) = 0 = F(K)$, so there is no growth if the stock is zero or at carrying capacity K, and $F'(0) > 0 > F'(K)$.

2.2 Poachers

Implicitly, poachers base their behavior on the effort needed to poach wildlife and evade enforcement in relation to the return to poaching. Commercial poaching is presumed to be an open-access business; as such, poachers do not take into account their impact on the future wildlife stock. Consequently, it suffices to represent the result of their maximized objective function as $q = Q(W, A)$, a function of the current wildlife stock and anti-poaching effort A. It is natural to assume that $Q(0, A) = 0$ (there is no poaching without wildlife), $Q_W(W, A) \geq 0$ (poaching is at least weakly increasing in the wildlife stock, though there may be some levels that are too low to make it profitable), and $Q_A(W, A) < 0$ (anti-poaching effort decreases poaching). The second-order derivatives are such that $Q_{AA}(W, A) > 0$, and $Q_{AW}(W, A) = Q_{WA}(W, A) < 0$, and we assume a concave net offtake function, which

[9] Communities tend to be allowed a larger offtake compared to the Parks and Wildlife Estate ("parks agency"). Thus one might conjecture α is probably greater than 0.5. On the other hand, communities tend to lack the infrastructure to capture tourism revenues, and it seems they do not receive major revenues from the national park system; thus beta may be quite small. Early data on CAMPIRE show that tourism generated less than 1%; trophy hunting is the serious source of money.

[10] Given the complexity and generality of the model, a closed-loop solution is not possible in a fully dynamic optimization framework.

implies that $Q_{WW}(W, A) > F_{WW}(W)$ (i.e. poaching is less concave than regeneration). Poaching will not necessarily be zero, however large A becomes.

2.3 Parks Agency

The commercial wildlife sector is represented by the parks agency, which determines the hunting quota, h. The parks agency generates revenue from wildlife by selling licenses to hunters and collecting fees (and other income) from tourists.

We assume that the market price of hunting licenses is a function of the wildlife stock $p(W)$ but constant per unit hunted. The fact that Zimbabwe is only one of the many countries offering sport-hunting opportunities motivates the price-taking assumption. However, the “quality” of a license does depend on the stock of wildlife, where $p'(W) > 0$ and $p''(W) < 0$. In particular, elephant trophy prices are typically differentiated by tusk weight, and larger herd sizes allow more elephants to mature longer and achieve larger tusk sizes (to a natural limit).[11]

Revenue from non-consumptive tourism, $T(W)$, will increase with the stock of wildlife; that is, $T(0) = 0$, $T'(W) > 0$, and $T''(W) < 0$. We abstract from the fact that non-consumptive tourism depends on biodiversity, since we represent all wildlife by one stock.

An important question is how to characterize the objective function of the parks agency, which could well range from maximizing its own revenues, to achieving preservation goals, to managing the public good aspects of wildlife that range across the jurisdictions of several communities. Even preservation and public good goals could be formulated in many different ways. The agency might want to maximize population numbers, or it could aim for some “healthy” stock level. It could also be more or less receptive to local opinion and variations in local circumstances. Another plausible scenario is one of inertia, in which the agency issues a constant number of hunting licenses despite a (rapidly) changing stock. As noted by Taylor (2006), there was a shift in quota-setting strategy over the course of the CAMPFIRE program. Thus, we will entertain different decision rules to understand the agency’s influence on community decisionmaking. These rules determine the parks agency hunting response function, which in combination with the biological equilibrium makes the equilibrium wildlife stock an implicit function of anti-poaching effort.

2.3.1 Fixed Quotas Rule

First, consider a fixed quotas rule, where $h = \bar{h}$. This rule is in many ways the simplest, and it might reflect the behavior of an agency that is understaffed and lethargic or does not dare take new initiatives. In the steady state, from the wildlife stock equilibrium, $F(W) - Q(W, A) - \bar{h} = 0$. Totally differentiating (and using superscript F for fixed quotas rule) we derive the steady-state wildlife response to anti-poaching effort,

$$\frac{dW^F}{dA} = \frac{-Q_A}{-(F_W - Q_W)} > 0 \tag{2}$$

Thus, with this rule of thumb, the response of the equilibrium wildlife stock is unambiguously increasing. Furthermore, since hunting quotas do not increase with reductions in poaching, the wildlife stock tends to be most responsive to anti-poaching effort under this regime.

[11] For example, one safari operator lists the following prices for 2007: \$10,500 for an elephant up to 49 lbs/side, \$12,500 for 50–59 lbs/side, \$14,500 for 60–69 lbs/side, and \$15,500 for more than 70 lbs/side. (http://www.zindelesafaris.com/zimbabwe_prices.htm. Accessed June 2007.)

2.3.2 Biologically Based Rule of Thumb

A principal idea behind CAMPFIRE was to foster more involvement with the communities, in addition to better wildlife management practices. After 1996, greater stakeholder participation and external technical support placed more focus on improving the wildlife census and calculating a sustainable offtake.

With a biologically (rather than economically) based decision rule, the parks agency determines a sustainable harvest depending only on the current wildlife stock, ignoring the revenue implications. This characterization reflects the recommendations of the WWF Quota Setting Manual, which was developed for CAMPFIRE to guide stakeholders in choosing an offtake rate. That rate depends on the growth rate of the wildlife population, as well as other factors, including the relative importance of sport hunting and maintaining trophy quality (WWF 1997). Quotas are then determined by a fixed offtake rate, designed to maintain a stable population (in equilibrium), typically about 1% for elephants.[12]

Let us suppose the parks agency allows a share ϕ of the stock (perhaps above some threshold $\underline{W}$) to be hunted, so $h^B = \phi(W - \underline{W})$. Then the biological equilibrium requires

$$F(W) - Q(W, A) - \phi(W - \underline{W}) = 0 \tag{3}$$

Totally differentiating, we derive the steady-state wildlife response to anti-poaching effort (with superscript B for biological rule),

$$\frac{dW^B}{dA} = \frac{-Q_A}{\phi - (F_W - Q_W)} > 0 \tag{4}$$

Thus, with this rule of thumb, the response of the equilibrium wildlife stock is also unambiguously positive, but it is smaller than if hunting allocations were fixed, since part of the increase in wildlife population is taken through additional hunting quotas:

$$\frac{dW^B}{dA} = \frac{-Q_A}{\phi - (F_W - Q_W)} < \frac{-Q_A}{-(F_W - Q_W)} = \frac{dW^F}{dA}.$$

2.3.3 Profit Maximization

In CAMPFIRE, prior to 1996, revenues seemed to be the primary concern in quota setting (Taylor 2006). Suppose, first, that the parks agency adjusts to market incentives, intending to maximize its own profits, as in Skonhoft (1998) and Johannesen and Skonhoft (2005). This could well characterize decisionmaking in a weak state, where ministries and public agencies are not necessarily just implementing welfare-oriented government policies, but rather fending for themselves as best as they can, since they get little resources or guidance from government. In this case, it maximizes with respect to h the total revenues from wildlife activities—after benefit sharing:

$$\pi = p(W)(1 - \alpha)h + (1 - \beta)T(W) \tag{5}$$

subject to the steady-state wildlife stock constraint of Eq. (1). Furthermore, we assume that the parks agency takes the community's contribution of anti-poaching effort as given.

[12] The manual does recognize the option to adjust the rate to account for drought or management targets, but the essence of the guidelines is to choose a sustainable offtake rate and then monitor the population to determine the quotas.

This problem is tantamount to choosing the wildlife stock to maximize annual steady-state profits:

$$\pi = p(W)(1-\alpha)(F(W) - Q(W, A)) + (1-\beta)T(W) \tag{6}$$

We note that in the case where $\alpha = \beta$ and the community gets the same share of all revenue sources, the problem is equivalent to a management regime of maximizing total revenues, regardless of how much or little the parks agency retains.

The first-order condition for the parks agency is

$$\frac{\partial \pi}{\partial W} = (1-\alpha)\left(p_W(F(W) - Q(W, A)) + p(F_W - Q_W)\right) + (1-\beta)T_W = 0 \tag{7}$$

Since in equilibrium $h^\pi = F(W) - Q(W, A)$, by substituting and solving (and assuming an interior solution with positive quotas), we see that the parks agency's optimal hunting allocation is

$$h^\pi = -\left(p(F_W - Q_W) + \frac{1-\beta}{1-\alpha}T_W\right)/p_W \tag{8}$$

From this we infer that $(F_W - Q_W) < 0$ if a positive number of quotas are to be allocated. Furthermore, it must be that $\frac{1-\beta}{1-\alpha} < \frac{-p(F_W - Q_W)}{T_W}$; else $h^\pi = 0$.

Assuming an interior solution (positive quotas), the profit-maximizing parks agency will be more concerned with conservation if the marginal return to tourism is higher than the marginal return to hunting, after benefit sharing. In this manner, as Skonhoft (1998) first showed, changing the relative hunting and tourism revenue shares can change the manager's incentive for setting quotas.

Totally differentiating the first-order condition,

$$\begin{aligned} &s p_{WW} h^* dW + 2p_W(F_W - Q_W)dW - p_W Q_A dA + p(F_{WW} - Q_{WW})dW \\ &+ \frac{1-\beta}{1-\alpha}T_{WW}dW - pQ_{WA}dA = 0 \end{aligned}$$

we get

$$\frac{dW^\pi}{dA} = \frac{\overbrace{p_W Q_A + pQ_{WA}}^{-}}{\underbrace{p_{WW}h}_{-} + 2p_W \underbrace{(F_W - Q_W)}_{-} + p\underbrace{(F_{WW} - Q_{WW})}_{-} + \underbrace{\frac{1-\beta}{1-\alpha}T_{WW}}_{-}} > 0 \tag{9}$$

Whether $dW^\pi/dA < dW^F/dA$ (or dW^B/dA) is not clear, since both the numerator and denominator are larger.

Also from this equation, we see that benefit sharing does not have a big effect on the profit-sharing equilibrium response to changes in anti-poaching effort other than changing the relative weight of the slope of marginal tourism revenues. (Recall that with simple revenue maximization, $(1-\beta)/(1-\alpha) = 1$). If the parks agency gets a larger relative share of tourism revenues, the equilibrium wildlife stock becomes less sensitive to changes in A.

However, benefit sharing does have a more direct effect on the equilibrium wildlife stock. Totally differentiating the equilibrium first-order condition again, conditional on the level of anti-poaching effort,

$$p_{WW}h^*dW + 2p_W(F_W - Q_W)dW + p(F_{WW} - Q_{WW})dW + \frac{1-\beta}{1-\alpha}T_{WW}dW$$
$$+\frac{1-\beta}{(1-\alpha)^2}T_W d\alpha - \frac{T_W}{1-\alpha}d\beta = 0$$

we see that

$$\left.\frac{dW}{d\alpha}\right|_{dA=0} = \frac{-\frac{1-\beta}{(1-\alpha)^2}T_W}{(p_{WW}h + 2p_W(F_W - Q_W) + p(F_{WW} - Q_{WW})) + \frac{(1-\beta)}{(1-\alpha)}T_{WW}} > 0 \quad (10)$$
$$\left.\frac{dW}{d\beta}\right|_{dA=0} = \frac{T_W/(1-\alpha)}{(p_{WW}h + 2p_W(F_W - Q_W) + p(F_{WW} - Q_{WW})) + \frac{(1-\beta)}{(1-\alpha)}T_{WW}} < 0 \quad (11)$$

Thus, absent a change in community effort, the profit-maximizing parks agency responds to less tourism revenue by targeting a lower wildlife stock, while less hunting revenue raises the equilibrium stock, as tourism becomes relatively more important.

These different kinds of responses matter for the community, which must take into account the full effects of its actions on the new wildlife equilibrium.

2.4 Local Community

The community is assumed to maximize the present value of its collective rents.[13] It engages in agriculture, which provides rents $R(W)$ that decline with the stock of wildlife ($R'(W) < 0$). This function can represent the productivity of grazing land, which declines with roaming wildlife, or of agricultural land, where crops risk being eaten or trampled by intruding wildlife.

The community has a range of anti-poaching activities at its disposal: monitoring wildlife, reporting inappropriate behavior, and employing and equipping anti-poaching units through district administrative structures. Our model is thus somewhat more complicated than conventional harvesting models, since the community does not directly choose hunting or poaching effort as a control variable but rather influences the offtake (hunting plus poaching) indirectly through its choice of anti-poaching effort.

Engaging in anti-poaching effort, A, entails costs—the value of time lost, wages for private enforcement agents, and so forth—represented by the function $c(A)$. This function is assumed to be positive, increasing, and convex: $c(0) = 0$, $c'(0) \geq 0$, $c(A) > 0$, and $c'(A) > 0$ for $A > 0$; $c''(A) > 0$.[14]

With benefit sharing, the local community's utility $u(W, A)$ combines the rents from agriculture with revenue shares from hunting and benign tourism (α and β, respectively), less the costs of anti-poaching activities:[15]

$$u(W, A) = R(W) + \alpha p(W)h + \beta T(W) - c(A) \quad (12)$$

[13] Of course, an implicit assumption is that communities can overcome the problems of collective action and common property management. Ostrom (2000) has documented the ability of communities to manage common pool resources in such a way as to give sustainable and satisfactory levels of benefits. Many of the communities in rural Zimbabwe, for example, have a fairly strict internal hierarchy and obey community council resolutions facilitated by the elders or chiefs.

[14] The absence of fixed costs or economies of scale should avoid a pulsing equilibrium, as in Rondeau and Conrad (2003).

[15] Depending on the sign of $R''(W)$, the objective function may or may not be concave in W, although it is concave with respect to the control variable, A.

The local community maximizes the present value of its income by choosing A subject to the dynamics of the wildlife stock equilibrium, including the response of the parks agency. In general form, the complementary slackness conditions are $A \geq 0$,

$$\frac{\partial u}{\partial A} = (R_W + \alpha p_W h + \beta T_W)\frac{dW}{dA} + \alpha p \frac{dh}{dA} - c_A \leq 0 \tag{13}$$

The net effect of benefit sharing thus depends on the parks agency's response.

2.4.1 Fixed Quotas Rule

In the first case, if hunting allocations are fixed, then *dh*/*dA* = 0, and the complementary slackness conditions are $A \geq 0$,

$$c_A \geq \left(R_W + \alpha p_W \bar{h} + \beta T_W\right) \frac{-Q_A}{-(F_W - Q_W)} \tag{14}$$

Here, we see that the level of antipoaching effort depends on whether $R_W + \alpha p_W \bar{h} + \beta T_W > 0$. In the terms of Horan and Bulte (2004), wildlife is an "asset" when the marginal net benefits to the community are positive and a "liability" when they are negative. Without revenue sharing, wildlife is a nuisance, and there is no effort.

When the hunting quotas do not change, sharing hunting revenues only improves the asset value of wildlife through the responsiveness of prices to trophy quality, while the sharing tourism revenues depends on the sensitivity to changes in the wildlife stock.

2.4.2 Biologically Based Rule

In this case, $dh/dA = \phi dW/dA$, and the complementary slackness conditions are $A \geq 0$,

$$c_A \geq \left(R_W + \alpha p_W h^B + \alpha p \phi + \beta T_W\right) \frac{-Q_A}{\phi - (F_W - Q_W)} \tag{15}$$

Compared with the fixed quotas rule, we observe several points. First, the value of the additional quota allocations ($\alpha p \phi$) raises the marginal net benefits of wildlife, making it more likely that anti-poaching effort will be positive if hunting revenues are shared, given comparable quotas. Second, that adjustment also appears in the denominator of *dW*/*dA*, which tends to diminish the incentive effect from those benefits.

Whether anti-poaching efforts are higher with the biologically based rule than with the fixed allocation then depends on whether $\alpha(p_W(h^B - \bar{h}) + p\phi)(Q_W - F_W) > \phi(R_W + \beta T_W)$. Of course, this is more likely to hold if allocations are larger with the biologically based rule. But if we consider allocations that are otherwise identical (i.e. $\bar{h} = h^B$), this reduces to the question of whether $R_W + \beta T_W + \alpha p(F_W - Q_W) < 0$, which holds unless tourism revenues are sufficiently important. Thus, when the community gets a large share of the tourism revenues, it may contribute more anti-poaching effort when quotas do not adjust than when they do. However, when the community receives a relatively large share of hunting revenues, effort is more likely to be larger when those quotas do adjust.

2.4.3 Profit Maximization

From Eq. (13), we can see that the profit-maximizing regime is most easily compared with the preceding biological rule. The relative incentive effects depend on the relative magnitudes of h, *dW*/*dA*, and *dh*/*dA*.

In general, allocations in equilibrium must follow the stock constraint, implying $\frac{dh}{dA} = -Q_A + (F_W - Q_W)\frac{dW}{dA}$. Thus, the first-order condition for the community becomes $c_A = \alpha p(-Q_A) + (R_W + \alpha(p_W h^\pi + p(F_W - Q_W)) + \beta T_W)\frac{dW^\pi}{dA}$, and substituting the equilibrium value of h^π and simplifying, we get

$$c_A = \alpha p(-Q_A) + \left(R_W + \frac{\beta - \alpha}{(1-\alpha)} T_W\right)\frac{dW^\pi}{dA} \tag{16}$$

Compared with the biological rule, there is a stronger direct incentive to reduce poaching from the equilibrium quota adjustment, in the form of $\alpha p(-Q_A) > \alpha p(-Q_A)\phi/(\phi - (F_W - Q_W))$. However, the incentive effect from adjustments in the wildlife population is tempered by $\alpha p(F_W - Q_W) < 0$, and by the fact that *dW/dA* is smaller when the parks agency responds by appropriating additional wildlife that goes unpoached for additional hunting quotas.

Of course, that assumes that the marginal net benefit of the stock is still positive, which requires sufficient benefit sharing—and greater benefit sharing of tourism revenues ($\beta > \alpha$). If wildlife remains on balance a nuisance, then a larger response by the parks agency tends to decrease community incentives to engage in anti-poaching efforts, even if they do get additional benefits from hunting license revenues. In general, then, the net effect of different management regimes depends both upon the equilibrium level of quotas and upon *dW/dA*.

3 Property Rights, Wildlife, and Welfare

3.1 Conservation Incentives

Next we consider how the allocation of the property rights from the two sources of wildlife revenues—hunting and tourism—affects conservation incentives. We have already seen that some benefit sharing is required to induce community cooperation. But how effective is increasing the community benefit shares for people's incentives to engage in anti-poaching efforts in the long run? It is complex to derive $dA/d\alpha$ and $dA/d\beta$ fully, but we note that they follow the direction of $\partial\{c_A\}/\partial\alpha$ and $\partial\{c_A\}/\partial\beta$.

From the community's perspective, the incentive effect of additional benefit shares hinges on the steady-state wildlife stock and quota allocation responses.

3.1.1 Rules of Thumb

First, consider all but the profit-maximizing management regime. With the simple rules of thumb, the parks agency's target wildlife stock is not directly affected by revenue sharing, and the change in anti-poaching incentives is straightforward. For the fixed quotas rule, an increase in the shares of hunting revenues for the community enhances the incentive to improve trophy quality:

$$\frac{\partial\{c_A\}}{\partial\alpha} = p_W \bar{h}\frac{-Q_A}{-(F_W - Q_W)} > 0$$

Meanwhile, a larger share of tourism revenues enhances incentives to protect wildlife and expand tourism rents:

$$\frac{\partial\{c_A\}}{\partial\beta} = T_W \frac{-Q_A}{-(F_W - Q_W)} > 0.$$

With the biologically based rule, additional hunting revenue shares encourage the community to further protect wildlife, not only to promote trophy quality but also to garner additional licenses:

$$\frac{\partial\{c_A\}}{\partial\alpha} = \left(p_W h^B + p\phi\right)\frac{-Q_A}{\phi - (F_W - Q_W)} > 0.$$

Additional tourism revenues also create positive incentives for anti-poaching:

$$\frac{\partial\{c_A\}}{\partial\beta} = T_W \frac{-Q_A}{\phi - (F_W - Q_W)} > 0.$$

However, both of these are tempered by the fact that the wildlife does not adjust upward as much as in the fixed quotas rule.

Thus, we find that sharing in tourism revenues enhances community conservation incentives if people know that the parks agency's response to their effort is to increase the steady-state wildlife stock. Furthermore, the effect of tourism revenue sharing is stronger when quotas are fixed (all else equal). On the other hand, the effect of hunting revenue sharing is stronger with the biologically based rule if the value of additional quotas outweighs the value of higher increases in trophy quality without quota adjustments.

The choice of offtake rate in the biologically based rule, ϕ, also represents a potential instrument but has a more ambiguous effect. On the one hand, the community receives additional revenue shares; on the other hand, quotas reduce the stock of wildlife, which now has a net positive value to the community. From (15) and (4), we have

$$c_A = (R_W + \alpha p_W \phi (W - W) + \alpha p\phi + \beta T_W)\frac{-Q_A}{\phi - (F_W - Q_W)}$$

so

$$\begin{aligned}\frac{\partial\{c_A/(-Q_A)\}}{\partial\phi} &= \frac{\alpha(p_W W + p) - c_A/(-Q_A)}{\phi - (F_W - Q_W)} \\ &= \frac{\alpha(p_W W + p)(-(F_W - Q_W)) - (R_W - \alpha p_W \phi W + \beta T_W)}{(\phi - (F_W - Q_W))^2} \qquad (17)\end{aligned}$$

In other words, ignoring the community revenue from licenses (or if $\alpha = 0$), increasing the harvest rate serves to dampen community efforts. When the community recognizes that the parks agency will increase quotas as the wildlife stock increases, the community responds by saving its costs of antipoaching activities, since part of the stock increase would be lost to more hunting.

3.1.2 Profit Maximization

In the case of a profit-maximizing parks agency, however, the question is not merely how anti-poaching effort responds to revenue shares but also how the parks agency responds. Differentiating (16) with respect to the benefit-sharing parameters (given a level of A and W) yields the direct effects on community incentives. Additional hunting revenue shares now have an ambiguous effect:

$$\frac{\partial\{c_A\}}{\partial\alpha} = -Q_A p - \left(\frac{1-\beta}{(1-\alpha)^2}T_W\right)\frac{dW^\pi}{dA}.$$

They enhance the value to the community of reducing poaching, for which the direct effect is to allow additional licenses, but there is a strictly negative effect from the adjustment of the wildlife stock. Sharing tourism revenues, on the other hand, has an additional incentive effect for conservation when $\alpha > 0$:

$$\frac{\partial\{c_A\}}{\partial\beta} = \frac{T_W}{1-\alpha}\frac{dW^\pi}{dA}.$$

However, there may still be differences between dW^π/dA and dW^B/dA or dW^F/dA. Furthermore, with the profit-maximizing parks agency, revenue shares also influence the targeted wildlife stock, and the ultimate effect on community incentives depends also on these indirect effects through $\partial\{c_A\}/\partial W$.

Ultimately, in the new equilibrium from a change in hunting quota shares, $\frac{dW}{d\alpha} \approx \left(\frac{dW^\pi}{d\alpha}\Big|_{dA=0} + \frac{dW^\pi}{dA}\Big|_{dA=0}\frac{dA}{d\alpha}\right)$. The net effect is unambiguously positive if $dA/d\alpha > 0$: then the community supports a larger population for the additional revenues *and* the parks agency prefers a larger population as tourism revenues become relatively more important to it. This latter effect can offset the fact that community incentives may be somewhat weakened by the anticipated response of the parks agency.

Meanwhile, $\frac{dW}{d\beta} \approx \left(\frac{dW^\pi}{d\beta}\Big|_{dA=0} + \frac{dW^\pi}{dA}\Big|_{dA=0}\frac{dA}{d\beta}\right)$; again, the net effect is unclear. Since $dW^\pi/d\beta < 0$, higher tourism revenues to the community reduce the parks agency's wildlife target, but the community anti-poaching incentives are strengthened.

For the other cases—including a revenue-maximizing parks agency—the equilibrium wildlife stock response to tourism revenue sharing depends on the relative magnitude of *dW/dA*, in which case the fixed quotas regime has the strongest response. On the other hand, greater sharing of hunting license revenues may have a stronger conservation impact when the parks agency is a profit maximizer. Finally, we must also recall that at some point (if α becomes too large relative to β), the parks agency might no longer want to issue hunting quotas, and the community problem reverts to that of fixed quotas, where $h = 0$, and only tourism shares affect community conservation incentives.

Table 1 summarizes these results, that the parks agency's management strategy and its interaction with the community is important for anti-poaching effort and wildlife outcomes. Indeed, those factors determine whether greater resource sharing engenders greater conservation. Theoretically, it may be possible to encourage more conservation activity with a delicate balance of the revenue shares in the profit-maximizing game than with the biologically based rule; however, finding that balance in practice is likely to be challenging. By constraining the parks authority's decision rule, the effects of benefit sharing become transparent.

Given these conditions involved in this analysis, it becomes evident that other kinds of equilibria and corner solutions are possible, and not all will lead to an outcome in which benefit sharing leads to a larger wildlife stock. In a sense, we consider the most promising classes of equilibria and still find that some scenarios lead to ambiguous outcomes.

3.2 Welfare

The formulation of the quota-setting decision also has important implications for welfare. When the decision rule for setting hunting quotas does not change with benefit sharing, as with the biological rule or fixed quotas, allocating more of either revenue source can serve only to enhance community welfare. In essence, all tools for community influence remain in place; the only difference is that the community controls more assets under CAMPFIRE. However, this result does not necessarily hold if the parks agency changes its quota rules according to the revenue shares it receives.

Table 1 Summary of effects under different management regimes

Management regime	Parks agency response	Community response	Net effect on wildlife stock
Fixed quota	Strong wildlife target stock response to anti-poaching effort	Effort clearly increases with tourism revenue shares	Strong positive response to tourism shares
	Unaffected by benefit sharing	Effort increases with hunting shares to improve trophy quality	Positive response to hunting shares
Biologically based rule	Less strong wildlife stock response to anti-poaching effort	Effort increases with tourism revenue shares	Positive but weaker response to tourism shares
	Additional hunting licenses given	Effort increases with hunting shares to improve trophy quality and garner more licenses	Positive, likely stronger response to hunting shares
	Unaffected by benefit sharing		
Profit-maximizing parks agency	Positive wildlife stock response to anti-poaching effort	Stronger effect of tourism revenue shares on effort	Ambiguous response to tourism shares
	Additional hunting licenses given	Ambiguous effect of hunting revenue shares on effort	Ambiguous response to hunting shares
	Licenses decrease and wildlife stock increases with larger α relative to β		

When the parks agency is a profit maximizer, the question is whether the shared revenues from an increase in the wildlife stock outweigh the corresponding agricultural losses. To take an extreme case, if the community were given all the hunting licenses but none of the tourism revenues, the profit-maximizing parks agency would authorize no hunting and reap all the benefits from tourism. Meanwhile, the community would receive no revenues, yet it would lose agricultural rents. In that case, the community would be strictly worse off.

At the other extreme, if the community gets all the tourism profits while the parks agency gets the hunting revenues, the community necessarily benefits. We can understand this result by first ignoring the community's incentives to promote tourism. By removing tourism from the parks agency's objective function, the park manager expands hunting, which improves agricultural rents and saves anti-poaching costs. Then, on top of those gains, the community gets tourism revenue. To the extent that it adjusts its anti-poaching behavior to reoptimize and encourage more wildlife in this scenario, the community benefits further.

The take-home message from this analysis is that sharing resource profits does not automatically confer both benefits and conservation incentives on local communities. That depends importantly on which resource profits are shared, how much is shared, and how the resource management practices outside the communities respond. The change in CAMPFIRE practice toward a biological rule for quota setting seems to make the program more likely to confer welfare benefits than a situation in which the parks agency has its own, self-interested objective function.

4 Conclusion

In many respects, CAMPFIRE seemed successful, at least initially. Poaching, seen as rampant prior to CAMPFIRE, fell drastically afterward, with evidence of community help. As participatory approaches to management and quota setting gained hold, trophy quality was better maintained (Taylor 2006). However, in some areas poaching subsided only temporarily, and the situation deteriorated again when communities did not receive the promised benefits and rural district councils did not generate enough money to operationalize the anti-poaching units.

This analysis demonstrates that mere resource sharing does not automatically confer benefits and conservation incentives on local communities. Those incentives depend critically on the type of resource activity that generates the shared profits, the extent to which these shared profits outweigh agricultural losses, and also how benefit sharing and community responses affect the resource management practices of the parks agency—and the delivery of benefits to the communities.

It is worthwhile understanding the conservation incentives of benefit sharing schemes, since society may place values on wildlife protection that are not reflected in the relevant agents' objective functions. (Indeed, without such values, the optimal strategy in our model would allocate all property rights to the communities.) The model reveals that the effects of allocating shares of hunting quotas interact with the response of the parks agency; while there may be an incentive to improve trophy quality and prices, the incentive to make more wildlife available for hunting depends on the responsiveness of the quota-setting regime.

Tourism revenue shares may more recognizably offer conservation incentives, which may explain some of the success stories of safari lodges that engage local communities. However, the overall effect of this incentive still depends on the response of the parks agency. The community's additional efforts to promote tourism through more conservation can be tempered after a while—even undone completely—if the parks agency does not have enough incentives of its own to maintain the wildlife stock for tourism. Furthermore, to the extent the community recognizes that some of an increase in the wildlife stock will be taken for hunting instead, the tourism incentive is diminished.

Finally, community welfare is not necessarily enhanced if the parks agency changes its quota-setting decisions when shares of its revenues are diverted to local communities. Given any fixed decision rule for hunting, profit shares transfer valuable assets to communities, and any changes in their behavior further enhance their welfare (even if they do not enhance conservation). However, if benefit sharing induces the parks agency to change the way it determines hunting quotas, and that runs counter to the new wildlife stock objectives of the local community, the community can be made worse off. This welfare reduction can result because the parks agency does not consider the impacts of wildlife on agricultural rents, and although the community might share in the profits from hunting, it cannot set the quotas directly.

Thus, a critical feature of the CAMPFIRE program was not just the sharing of resource revenues, but also the technical support. By providing advice for biologically sound quota setting, as well as the development of institutions for community participation, the program was able to improve transparency and enhance the incentive effects of the hunting revenue shares, which form the bulk of the CAMPFIRE revenues. Zimbabwe may also have been fortunate to set out with a sufficient stock of large game such that the biological dynamics lent themselves toward a successful outcome. On the other hand, the political institutions ultimately proved equally important to the incentives we have modeled here. In particular, the filtering of the revenues through the rural district councils, which were removed from the local communities and did not always live up to their financial management obligations, served to temper those incentive effects, as well as the impact of the program on community welfare.

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References

Baland JM, Platteau JP (1996) Halting degradation of natural resources: is there a role for rural communities. Clarendon Press, Oxford

Barrett CB, Arcese P (1998) Wildlife harvest in integrated conservation and development projects: linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti. Land Econ 74:449–465

Brandon K, Wells M (1992) Planning for people and parks: design dilemmas. World Dev 20:557–570

Bulte EH, Horan RD (2003) Habitat conservation, wildlife extraction and agricultural expansion. J Environ Econ Manage 45(1):109–127

Bulte EH, van Kooten GC (1999) Economics of antipoaching enforcement and the ivory trade ban. Am J Agric Econ 81(2):453–466

Child B (1995) Guidelines for managing communal lands wildlife revenues in accordance with policy for wildlife. Department of National Parks and Wildlife Management, Harare, Zimbabwe

Cumming DHM (1989) Commercial and safari hunting in Zimbabwe. In: Hudson RJ, Drew KR, Bodkin LM (eds) Wildlife production systems: economic utilisation of wildlife. Cambridge University Press, Cambridge

Duffy R (2000) Killing for conservation: wildlife policy in Zimbabwe. The International African Institute and Weaver Press, Harare

Emerton L (2001) The nature of benefits and benefits of nature: why wildlife conservation has not economically benefited communities in Africa. In: Hulme D, Muphree M (eds) African wildlife and livelihoods: the promise and performance of community conservation. Weaver Press, Harare

Ferraro PJ (2001) Global habitat protection: limitations of development interventions and a role for conservation performance payments. Conserv Bio 15(4):990–1000

Ferraro PJ, Simpson D (2002) The cost-effectiveness of conservation payments. Land Econ 78(3):339–353

Horan RD, Bulte EH (2004) Optimal and open-access harvesting of multi-use species in a second-best world. Environ Resour Econ 28(3):251–272

Johannesen AB (2004) Designing integrated conservation and development projects (ICDPs): illegal hunting, wildlife conservation and the welfare of the local people. Working paper series no. 2/2004. Norwegian University of Science and Technology, Department of Economics, Trondheim, Norway

Johannesen AB, Skonhoft A (2005) Tourism, poaching and wildlife conservation: what can integrated conservation and development projects accomplish? Environ Resour Econ 27:208–226

Khanna J, Harford J (1996) The ivory trade ban: is it effective. Ecol Econ 19:147–155

Kinyua PID, van Kooten GC, Bulte EH (2000) African wildlife policy: protecting wildlife herbivores on private game ranches. Eur Rev Agric Econ 27(2):227–244

Milner-Gulland EJ, Leader-Williams N (1992) A model of incentives for the illegal exploitation of black rhinos and elephants: poaching pays in Luangwa Valley. Zambia J App Ecol 29:388–401

Muller J, Albers HJ (2004) Enforcement, payments, and development projects near protected areas: how the market setting determines what works where. Resour Energ Econ 26:185–204

Ostrom E (2000) Collective action and the evolution of social norms. J Econ Perspect 14(3):137–158

Rondeau D, Conrad J (2003) Managing urban deer. Am J Agric Econ 85:266–281

Sellenthin MO, Skogh G (2004) Property rights in endangered species: the wolverine case. Eur J Law Econ 18(2):239–247

Shulz C-E, Skonhoft A (1996) Wildlife management, land use and conflicts. Environ Develop Econ 1(3):265–280

Skonhoft A (1998) Resource utilisation, property rights and welfare—wildlife and the local people. Ecol Econ 26:67–80

Skonhoft A, Solstad JT (1996) Wildlife management, illegal hunting and conflicts. Environ Develop Econ 1(2):165–181

Skonhoft A, Solstad JT (1998) Investing in wildlife: can wildlife pay its way?. J Afric Econ 7(2):237–262

Swallow S (1990) Depletion of the environmental basis for renewable resources: the economics of interdependent renewable and non-renewable resources. J Environ Econ Manage 19:281–296

Swanson T (1993) Regulating endangered species. Econ Policy 8(16):185–205

Taylor RD (2006) Case study: CAMPFIRE (communal areas management programme for indigenous resources), Zimbabwe. Available at http://www.irgltd.com/Resources/Publications/Africa/Zimbabwe%20paper%20final%20draft%20v3.pdf

WWF (1997) Quota setting manual. Wildlife management series. World wide fund for nature (formerly world wildlife fund). Programme Office, Zimbabwe, Zimbabwe Trust and Safari Club International