A Bioeconomic Model of Community Incentives for Wildlife Management before and after CAMPFIRE

Carolyn Fischer, Edwin Muchapondwa, and Thomas Sterner*

Abstract

This paper formulates a bioeconomic 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 chooses to either aid or discourage 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 sets hunting licenses. A biologically-based quota-setting rule is likely to improve the effectiveness of benefit sharing.

Key Words: bioeconomic, CAMPFIRE, community, poaching, wildlife, benefit sharing

JEL Classification Numbers: H41, Q20

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1. Introduction

Many prominent African species of wildlife—including lions, rhinos, and elephants—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 nongovernmental 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 among 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 affect their success, in terms of both wildlife and community welfare.

Zimbabwe offers an interesting case study 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 land was 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 depends on the size of the parkland (Cumming 1989; Swallow 1990; Emerton 2001).

The Department of National Parks and Wildlife Management owned the wildlife in trust for Zimbabwe and collected the economic benefits produced by wildlife from sale of licenses for consumptive wildlife use (hunting) and from nonconsumptive wildlife services (benign tourism). Hunting is ordinarily disallowed in national parks, but the country has 17 safari areas in which limited hunting occurs, as well as benign tourism.[2] The local people

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[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] Safari areas cover a total of 1,892,724 hectares; examples include Chewore, Chirisa, Matetsi, and Sapi.

near wildlife areas subsisted primarily by engaging in livestock production and marginal crop agriculture, suffering wildlife intrusions. Despite wildlife protections, illegal poaching grew problematic, and locals would often turn a blind eye or even collaborate, since wildlife posed a nuisance.

In 1989, Zimbabwe instituted a benefit-sharing program for wildlife, the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE). It focuses especially on communal areas adjacent to national parks, where wildlife intrusion is most problematic and agricultural productivity is marginal.[3] CAMPFIRE gives communities co-ownership of local natural resources, which generate income through leasing trophy hunting concessions, harvesting 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 profit sharing in a typical wildlife-abundant rural area in Zimbabwe in which wildlife conservation can conflict with agricultural production. Although heavily inspired by CAMPFIRE, our analysis is distilled to represent revenue-sharing programs for wildlife somewhat more broadly.[4] We employ a bioeconomic model similar to those formulated for the East African case in Shulz and Skonhoft (1996), Skonhoft and Solstad (1996, 1998), and Skonhoft (1998), among others.[5] The model in this paper extends that work in several ways, motivated by our application. First, whereas previous papers study poaching by the local communities, this paper studies poaching conducted by outsiders (from the local community's perspective). Second, unlike previous papers that give prominence to antipoaching enforcement by the parks agency, ours emphasizes the antipoaching effort that local communities are exerting. 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.

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[3] In Zimbabwe, more than 90% of the communal lands are located within agriculturally marginal regions plagued by persistent drought, low, erratic rainfall, and poor soil. Crop production in these regions is very low since they are ecologically best suited to extensive cattle and wildlife ranching. The hunger for land is one of the factors that have led to occupation of commercial farms in the more productive regions. It also creates considerable tension over wildlife in the national parks that border on the less productive farm areas.

[4] 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 with many similarities to that analysed here see Sellenthin and Skogh (2004). See Gadgil and Rao (1994, 1995) for work on biodiversity conservation in India.

[5] See also Baland and Platteau (1996) for a general overview of the role of communities in wildlife management.

This approach is inspired by the situation in Zimbabwe in the late 1990s, when the CAMPFIRE project was beginning to settle into a more mature phase. At that time in Zimbabwe, the primary concern was not subsistence poaching by communities, but rather large-game poaching, which was mainly believed to be conducted by foreign nationals. Consequently, we assume that local communities do not benefit directly from poaching proceeds; rather, they choose to what degree to collaborate with or oppose poachers, based on their perception of the value (or cost) of wildlife. Under CAMPFIRE, a significant amount of the antipoaching enforcement was applied by local communities through the hiring of community-based antipoaching units.

This characterization of community effort contrasts with previous studies in which local populations engage in open-access hunting and reap those use benefits directly. For example, Bulte and Horan (2003) develop a model of open-access wildlife exploitation, habitat conservation, and agriculture, in which farmers may either hunt for wildlife or grow crops. They show that increasing wildlife conservation may well be Pareto-superior to equilibria in which agriculture dominates. Kinyua et al. (2000) also deal with wildlife management but focus more on the competition between wildlife and grazing and, in particular, on the incentives 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. Furthermore, cash transfers can have the unintended effect of raising demand for game meat by raising incomes. Substitution and income effects may also depend on access to markets. Muller and Albers (2004) recognize that villagers may extract for subsistence, sell resources to markets, or provide labor to markets. They focus on the role played by incomplete labor and resource markets in shaping optimal management plans, comprising enforcement, agricultural development projects, and conservation payments. Johannesen (2004) considers how integrated conservation and development schemes can leverage opportunities for formal sector work to discourage illegal hunting.

Also relevant to our model is the growing literature on the optimal management of multiuse species, in which wildlife are both resources and pests, such as Zivin et al. (2000) and Rondeau (2001). More recently, Horan and Bulte (2004) consider optimal management in the presence of second-best trade restrictions. As in these models, the shadow value of wildlife to the community may be negative or positive, which can create certain complications. However, we depart from the social planner problem to study the interaction among several agents, taking a particular look at the dynamic game 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 antipoaching effort. 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 1992b; Bulte and van Kooten 1999; Swanson 1993; Khanna and Harford 1996).

Skonhoft (1998) considers the impact of different property-sharing regimes on the incentives of the park manager and 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. The park manager is a profit maximizer, and his incentives derive from the relative marginal values of tourism and hunting licenses; by changing the profit shares, it may be possible to align the park manager's choices with those of the social planner.

Johannesen and Skonhoft (2005) incorporate a response function for communities, who 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 of the CAMPFIRE case, 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 interactions. Engel et al (2006) also study the importance of strategic interactions and intervention design in the case of illegal logging in Indonesia, where communities negotiate with profit-maximizing logging companies in a context of weak property rights.

Since strategic responses are so important for the efficacy of benefit-sharing programs, we take an agnostic view of park manager objectives and explore several plausible methods for determining hunting quotas. We find that resource sharing with local communities can have ambiguous effects on both conservation incentives and welfare. We demonstrate that conservation incentives depend critically on three factors: the type of resource activity that generates the shared profits, the extent to which these shared profits outweigh agricultural losses from additional wildlife, and the way that the parks agency responds to profit sharing and whether the community internalizes this response.

Because of the prominence this paper assigns to poaching, we briefly discuss poaching, antipoaching activities, and community-based wildlife conservation in Zimbabwe in Section 2 before we set up the model in Section 3. Thereafter, we analyze how the allocation of the property rights from the two sources of wildlife profits affects conservation incentives and welfare (Section 4). The paper concludes in Section 5.

2. Poaching in Zimbabwe and Project CAMPFIRE

It has been argued that poaching may be the economically most important form of wildlife utilization throughout much of Africa (Milner-Gulland and Leader-Williams 1992a, 1992b). Poaching can be subsistence or commercial. In Zimbabwe, the local communities themselves engage in small-scale subsistence poaching mainly for smaller game, such as spring-hare, bushbuck, and guinea fowl, which generally have large stocks and high growth rates. This type of illegal harvest has tended to be overlooked by the parks agency to the extent that it remains poaching for the pot. It is difficult for the parks agency to enforce antipoaching laws against local communities who live close to the wildlife resource and stubbornly claim traditional ownership of it, at least for consumption purposes. Tradition and culture have tended to be the main regulators of this kind of poaching.

Commercial poaching mainly targets larger game for trophy sales and is usually carried out by professional poachers with automatic weapons, typically outsiders employed by dealers. The ultimate customers are international and have considerable financial resources. In Zimbabwe, the public perception, which is reinforced by the media and the parks agency, is that commercial poachers are Zambians and occasionally Mozambicans (Duffy 2000). To succeed, poachers usually make use of a few local informers and accomplices whom they remunerate for their assistance; however, little income from commercial poaching actually reaches the local communities.

Poaching is thought to occur mainly outside the protected areas, where the vastness of habitats and financial constraints make the parks agency an absentee owner. Some antipoaching activities have been carried out on the parkland by the parks agency, police, and defense forces. Even within the parks, these officials have great difficulty enforcing antipoaching measures, given that they do not permanently live close to the safari area. Outside the protected areas, the parks agency has largely withdrawn its limited services in antipoaching enforcement.

The CAMPFIRE program was created to institute sustainable management practices for wildlife, land, and other natural resources by rural communities. The 1982 Parks and Wildlife Act provided the legal structure for the devolution of authority over wildlife resources to the democratically elected rural district councils. Structured into an independent, quasigovernmental framework, CAMPFIRE covers all natural resources, though its focus has been on wildlife management in areas adjacent to the national parks.

At the national level, CAMPFIRE is run by a collaborative group that coordinates policy, training, institution building, research, monitoring, and international advocacy. Membership of the collaborative group is drawn from the CAMPFIRE Association, the

Department of National Parks and Wildlife Management, the Ministry of Local Government, and other organizations, such as WWF and the Africa Resources Trust.[6] CAMPFIRE has relied heavily on donor funds to pay for administrative expenses, with only a small fraction being covered by its activities. From 1989 to 1999, the program received at least US$33 million from international donors (Patel 1998). The day-to-day activities are carried out by the communities themselves through village, ward, and rural district councils (RDCs).

The parks agency remains the guardian of all wildlife in Zimbabwe under CAMPFIRE and sets the quotas for hunting licenses that will be sold by the rural communities and itself, each in a designated area of control. In the mid-1990s, with technical support from WWF, the rural communities became more involved in the quota-setting process, providing ground counts and participating in stakeholder workshops; of course, while the RDCs can propose a hunting quota for the season, the final authority remains with the parks agency. The institutionalization of this participatory approach may be behind a noticeable shift in DNPWLM quota setting policy around 1996, from maximizing returns toward emphasizing more sustainable trophy quality (Taylor 2006).

More than 90% of the revenue earned in CAMPFIRE comes from sport hunting,[7] of which elephants attract 60% of the revenues (Bond 1994). The proceeds from hunting (licenses) and tourism fees go to the RDCs and the rural communities, although somewhat less than the target of 50% of the revenues reaches the local communities.[8] They engage in wildlife conservation and utilize the revenues to benefit themselves as communities, more so than individuals, such as building schools, clinics, or electric fences to deter wildlife intrusions (Patel 1998, Maveneke et al. 2000).

The motivation for CAMPFIRE was to integrate the local communities into decisions about wildlife conservation and give them shares of the benefits. Absent such benefits, the local communities have an interest in getting rid of the nuisance from wildlife, particularly large game, and they are likely to tolerate poaching. With integration, the local communities begin to perceive game as a resource and the activities that harm it as poaching. This change in local norms and peer enforcement alienates accomplices and makes poaching more difficult and expensive because the poachers cannot count on local support. It eventually becomes natural for neighbors to monitor, report, or discourage poaching and related behavior—activities that we refer to as antipoaching effort. Some communities have

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[6] WWF support to CAMPFIRE is described and evaluated in Goredema et al. (2005).

[7] Several sources cite Bond (2001) and CAMPFIRE Monitoring & Evaluation Database, WWF SARPO Harare.

[8] ibid.

employed and trained game guards to monitor the state of the resource, carry out problem animal control, implement antipoaching campaigns, and monitor the interaction between local communities, safari operators, safari clients, and the resource (Metcalfe 1994).[9]

Evidence from some areas in Zimbabwe shows that poaching was rampant prior to CAMPFIRE. Since then, poaching has been drastically reduced in some areas as the neighboring communities started reaping economic benefits from legal wildlife utilization and consequently began to make public arrests of commercial poachers (Child et al. 1997). However, in other areas, poaching subsided only temporarily with CAMPFIRE and then bounced back after a few years.

3. Model

The bioeconomic model comprises two agents (the parks agency and a local community), two control variables (hunting quotas and antipoaching 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, which solely benefits the community.

Each agent has a fixed amount of land. Parkland is the permanent residence of wildlife, and the local community has user rights over the remaining land. 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 around the lands adjacent to the park. Intruding wildlife damages crops, competes for the scarce grazing land with livestock, transmits diseases,[10] 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 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, which tends to increase the likelihood of intrusions.

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[9] Examples include Muzarabani, Guruve, Chipinge, Gokwe North, UMP Zvataida, Binga, Hwange, and Nyaminyami.

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

Due to the prominence of large game (particularly elephants and buffalo) in the CAMPFIRE program, we ignore consumptive benefits to communities, whose harvesting targets smaller, 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 profit sharing. Before CAMPFIRE, all of the hunting license rents—and most of the tourism profits—rested with the parks agency, although this agent may effectively also represent safari lodges and other actors. Under CAMPFIRE, the issuing of a hunting quota to the local community implies giving the local community a share of the hunting profits (α), either by sharing the revenues or by allowing them to sell their 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 $0 \le \alpha \le 1$ and $0 \le \beta \le 1$. 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 antipoaching incentives.

Wildlife

Assuming that wildlife can be represented as a single species, its biomass at a specified point in time (where the time index is omitted) is given by W. The growth in the stock is given by Equation (1), where $F(W)$ is the natural growth function of the stock of wildlife, h is the offtake from hunting, and q represents the loss due to poaching:

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

Additional implicit constraints are $W(t) \ge 0,\ W(0) = W_0,\ W(\infty) < \infty$. One of the potential specifications of the natural growth process is the logistic function

$$F(W) = gW\left(1 - \frac{W}{K}\right) \tag{2}$$

where g is the intrinsic growth rate and K is the carrying capacity.

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, as any benefits of foregoing current harvest would just accrue to future

competitors. Consequently, with poachers as third parties, it suffices to represent the result of their maximized objective function as $q = Q(W, A)$, distilling poaching as a function of the current wildlife stock and antipoaching effort A. The main properties of this function are as follows. No poaching takes place with a zero stock of wildlife, $Q(0, A) = 0$; more wildlife means a higher illegal offtake, $Q_W(W, A) > 0$; more antipoaching effort increases the possibility of detection hence results in lower illegal offtake, whereas more collaboration increases the offtake, implying in both cases $Q_A(W, A) < 0$. The second-order derivatives are such that $Q_{WW}(W, A) < 0$, $Q_{AA}(W, A) > 0$, $Q_{WA}(W, A) < 0$, and $Q_{AW}(W, A) < 0$. It is important to note that poaching will not necessarily be zero however large A becomes, since a certain amount of poaching cannot be detected even with the cooperation of the local community. Also, poaching will have a maximum bound because of the standard decreasing returns to effort and opportunity costs in the implicit poacher objective function.

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 from tourists who engage in nonconsumptive tourism. Although the beneficiaries of these revenues are generally more diverse, including safari lodges and operators, we will allow these gains to be considered by the parks agency in setting the optimal quota.

We assume that the market price of hunting licenses per unit of harvest has been fixed at *p*. The fact that Zimbabwe is only one of the many countries offering sport-hunting opportunities motivates the price-taking assumption. We further assume for simplicity that the "quality" of a license does not depend on the stock of wildlife (i.e., licenses can always be sold at the market price, whether wildlife is so plentiful as to make tracking unnecessary or so scarce that a trophy may not be found).

Costs and demand conditions also are assumed to be constant through time. Revenue from nonconsumptive 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 nonconsumptive tourism depends on biodiversity, since we represent all wildlife by one stock.

An important question will be how to characterize the objective function of the parks agency. For now, since we want to focus on the community incentives, we will begin by representing the harvesting quota decision as a general function $h = H(W, A)$. We assume that harvesting is positive and weakly increasing in these variables $[\, H(W, A) \geq 0, \quad H_W(W, A) \geq 0, \quad H_A(W, A) \geq 0 \,]$ and weakly concave in both. An additional

logical restriction is that any decrease in poaching will not be more than offset by an increase in quotas; that is, $H_A(W, A) \leq -Q_A(W, A)$.

By studying various strategies for the parks agency (like profit maximization or simple harvesting rules), we will later be able to address different strategic responses on the part of the community. As noted by Taylor (2006), there was a shift in quota-setting strategy over the course of the campfire program. Thus, our ambivalence is motivated by real ambiguity regarding the motivations of the park managers, which can well range from concern over their own revenues, to preservation goals, and to managing the public good aspects of wildlife that range across the jurisdictions of several communities. Furthermore, as noted, hunting and tourism revenues are captured by more agents than the local park, including national agencies and safari lodges. Given this ambiguity, we entertain different decision rules is to understand the important influence they have on the community decisionmaking.

Local Community

The community is assumed to maximize the present value of its collective rents.[11] It engages in agriculture, which provides rents $R(W)$ that tend to be affected negatively by larger wildlife stocks. This function could represent the profit-maximizing production frontier of livestock rearing or of crops on the community's fixed factor, land. The key property is that it is declining and convex in the stock of wildlife. This function can represent the productivity of grazing land, which declines with wildlife ranging, or of agricultural land, from which the crops can be eaten or trampled by intruding wildlife. Since in Zimbabwe more than 90% of the communal lands is in semiarid regions, which are ecologically better suited to wildlife (or cattle), crop production is likely to be at subsistence levels.

Our model emphasizes the antipoaching effort of the local communities rather than the parks agency. We assume the parks agency does not carry out any antipoaching activity (or at least does not change its enforcement behavior), which seems appropriate for areas adjacent to parklands. The community has a range of antipoaching activities at its disposal: encouraging members to withdraw their services to the commercial poachers as informers and accomplices, active antipoaching enforcement by monitoring and protecting wildlife, reporting inappropriate behavior with respect to wildlife conservation, and employing and

[11] Of course, an implicit assumption is that communities can overcome the problems of collective action and common property management. Ostrom (1990) has documented the ability of communities to manage common pool resources in such a way as to give sustainable and satisfactory levels of benefits.

equipping antipoaching units through district administrative structures. They also have options for assisting poachers. Although they do not benefit directly from commercial poaching—an assumption that differs from other models of subsistence poaching—they may benefit indirectly by reduced intrusions.

Engaging in antipoaching 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 U-shaped: positive, increasing, and convex with positive effort and positive, decreasing, and convex for negative effort. Allowing $A < 0$ implies that communities may be willing to assist poachers to reduce wildlife intrusions; $A > 0$ means efforts against poaching. Both kinds of efforts are assumed to be costly, as either assisting or defending against poachers entails time and other resources that could be expended elsewhere, such as in agriculture. However, while the marginal cost of increasing antipoaching effort is positive when A>0, it is negative when collaboration is occurring, since increasing enforcement implies reducing costly assistance effort. Mathematically, these assumptions imply $c(0) = 0, \quad c'(0) = 0; \; c(A) > 0, \quad A \neq 0; \; c'(A) > 0, \quad c''(A) > 0, \quad A > 0;$ and $c'(A) < 0, \quad c''(A) > 0, \quad A < 0$.[12]

The community perceives that the allocation of hunting licenses may be influenced by the wildlife stock and by poaching, which it, in turn, influences through its antipoaching efforts. With profit sharing, the local community's utility $u(W, A)$ combines the rents from agriculture with profit shares from hunting and benign tourism, less the costs of antipoaching activities:[13]

$$u(W, A) = R(W) + \alpha p H(W, A) + \beta T(W) - c(A) \tag{3}$$

The local community maximizes the present value of its income by choosing A subject to the dynamics of the wildlife stock. The current value Hamiltonian is

$$\mathcal{H} = u(W, A) + \gamma \left(F(W) - H(W, A) - Q(W, A) \right) \tag{4}$$

The first-order condition with respect to antipoaching effort is

$$c'(A) = \alpha p H_A(W, A) - \gamma \left(Q_A(W, A) + H_A(W, A) \right) \tag{5}$$

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[12] By allowing for collaboration, we also ensure an interior solution. Furthermore, the absence of fixed costs or economies of scale should avoid a pulsing equilibrium, as in Rondeau and Conrad (2003).

[13] Although some individuals in the local community may also get economic benefits from the poaching activity, such income does not enter the local community's welfare function, which the community council uses for decisionmaking.

Positive effort will be exerted if the net gains are positive. Net gains include both the shadow value of the net increase in the wildlife stock and the value to the community of any additional licenses. Negative effort (collaboration) is implied by a sufficiently negative shadow cost.

Henceforth, for brevity of notation, we shall write $T'(W)$ as T_W, $c'(A)$ as c_A, $Q_A(W, A)$ as simply Q_A, etc.

The dynamics of the shadow value of the wildlife stock to the community are defined by

$$\dot{\gamma} = -R_W - \alpha p H_W - \beta T_W + \gamma\left(\delta - F_W + H_W + Q_W\right) \tag{6}$$

Community Requirements for a Steady State

In a steady state, $\dot{W} = 0$, and offtake from hunting and poaching must equal growth:

$$H(W, A) + Q(W, A) = F(W) \tag{7}$$

Additionally, $\dot{\gamma} = 0$ and the shadow value of wildlife equals

$$\gamma^* = \frac{R_W + \beta T_W + \alpha p H_W}{\delta - F_W + H_W + Q_W} \tag{8}$$

Note that this shadow value can be positive or negative, depending on the sign of the marginal net benefits to the community (let $MNB = R_W + \alpha p H_W + \beta T_W$) and on the excess of the discount rate over the marginal net growth (natural replenishment rate net of marginal harvesting and poaching, or $MNG = F_W - H_W - Q_W$). As Horan and Bulte (2004) discuss, the ambiguity in the sign of the shadow value leads to the possibility of multiple equilibria. They call wildlife an "asset" when the marginal net benefits to the community are positive and a "liability" when they are negative. Meanwhile, if the shadow value (which incorporates the stock dynamics as well as the marginal benefits) is positive, wildlife is a "commodity," whereas a negative shadow value denotes wildlife as a "nuisance." Using their terminology, these combinations form four potential classes of steady states:

I. Commodity/asset ($MNB > 0$, $\delta > MNG$ and thus $\gamma > 0$)

II. Commodity/liability ($MNB < 0$, $\delta < MNG$ and thus $\gamma > 0$)

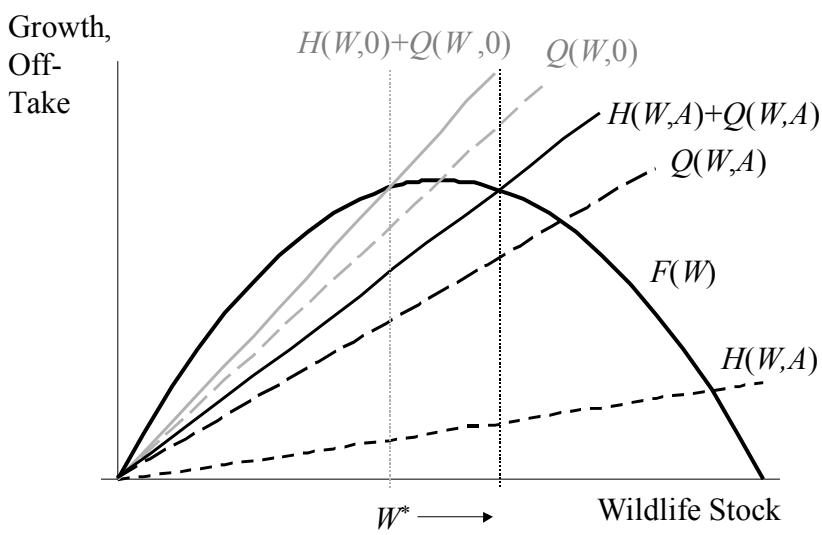
III. Nuisance/asset ($MNB > 0$, $\delta < MNG$ and thus $\gamma < 0$)

IV. Nuisance/liability ($MNB < 0$, $\delta > MNG$ and thus $\gamma < 0$)

Our model is somewhat more complicated, since the community does not directly choose harvesting as a control variable but rather influences the offtake (hunting plus

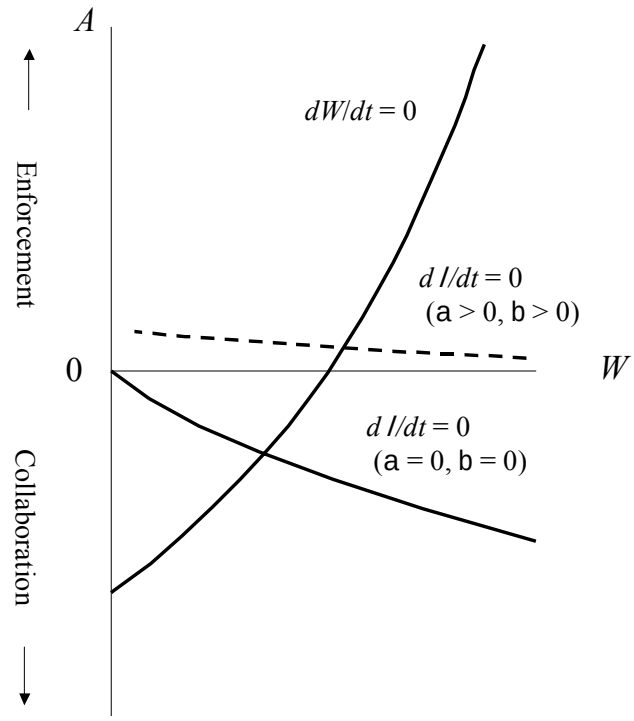
poaching) through its choice of antipoaching effort. Correspondingly, harvesting is influenced not directly by the community's shadow value of wildlife but indirectly by its choice of effort. The phase diagram for this problem can be drawn in two parts. First, we map the offtake and growth functions against the wildlife stock to see the equilibrium—conditional on a given level of antipoaching effort. That effort, A, shifts those functions, thereby affecting the equilibrium stock, W^*, that satisfies $\dot{W} = 0$, as shown in Figure 1.

Figure 1. Effort Shifts Offtake Functions in a Potential Harvesting Equilibrium



Next, we map the equilibrium wildlife stock against the level of antipoaching effort. In Figure 2, the revenue shares affect the $\dot{\gamma} = 0$ isocline, as seen in Equation (6). It depicts a possible form for these functions. This hypothetical scenario portrays benefit sharing as encouraging more effort, but this result is not guaranteed, as we discuss next.

Figure 2. Wildlife Stock and Effort



In our model, the community will engage in active antipoaching effort if wildlife has commodity status, and they will collaborate if it is a nuisance. It is normal to think of Classes I and IV, in which the marginal benefits and shadow values of wildlife express the same sign. This situation requires $\delta > MNG$, which tends to imply a larger stock. Although these scenarios are more intuitive, Classes II and III are theoretically possible. Since the *MNG* is likely to be diminishing in the wildlife stock, at low initial stock levels, the *MNG* may exceed the discount rate. In this kind of equilibrium, the marginal effects of revenue sharing would be reversed. In Class III, the community benefits on the margin from wildlife stock but prefers to limit the stock because people know growth is fast and fear that this can become a liability at higher stock levels. Thus they may encourage the poachers and will not engage in enforcement. In Class II, the community suffers net damages on the margin but wants to increase the stock since people know it grows fast (which implies a potential source of revenue through the revenue sharing), and thus they will engage in enforcement.

The possibility of multiple equilibria itself means that resource-sharing schemes can have ambiguous effects, depending in part on the initial stock of wildlife, as well as the parameter values and forms of the key functions. Stability of the equilibria is another important issue, which Bulte and Horan (2003) discuss and show to be dependent in part on functional forms. For our purposes, we will acknowledge and abstract from questions of the dynamics. We will show that $\delta > MNG$ is necessary for the existence of an equilibrium with

a profit-maximizing parks agency in Equation (14), so we will primarily deal with Classes I and IV rather than II and III. These cases are more favorable to resource-sharing programs, and we will focus on how a steady state, assuming it is locally stable, might respond to a change in the resource-sharing regime.

4. Property Rights and Conservation Incentives

Next we consider how the allocation of the property rights from the two sources of wildlife profits—hunting and tourism—affects conservation incentives. An important factor involves the strategic interaction between the communities and the parks agency: does the community take into account the reoptimization of hunting allowances, and how does the parks agency react?

Revenue Sharing: Hunting Licenses versus Tourism Profits

No Sharing

In a resource use regime in which the local community does not reap any benefits from wildlife (i.e., $\alpha = \beta = 0$), the shadow value of wildlife to the local community is negative, since the wildlife stock merely decreases agricultural rents. Consequently, residents will not engage in any antipoaching effort; if anything, they will tolerate poachers and possibly assist them to some degree, even without remuneration. This complicity results from the fact that their preferred stock of wildlife is effectively zero, since they bear only costs and no benefits.[14]

Profit Sharing

According to Equations (5) and (8), the community will want to assist conservation by engaging in active antipoaching efforts only to the extent that people gain additional license revenues and benefit from additional wildlife stock. This requires that the additional revenues from wildlife activities outweigh the costs of additional intrusions: $\alpha p H_W + \beta T_W > -R_W$.

Raising tourism shares always improves incentives for conservation and antipoaching effort—or at least it reduces incentives to collaborate with poachers. From Equation (8), we

[14] We have assumed that the communities have no existence value for wildlife. In fact, Muchapondwa (2003) does find such values, but their inclusion here would not add anything very substantial to the model. We would still assume these existence values are small compared with agricultural damage, and thus the communities would presumably have a small rather than a zero preferred stock.

see that the equilibrium shadow value of wildlife to the community is unambiguously increasing in β, and from Equation (5) we see that effort is strictly increasing in the shadow value of wildlife. Consequently, $dA/d\beta > 0$, and sharing tourism revenues necessarily increases antipoaching effort and the equilibrium wildlife stock.

However, the model reveals that raising hunting quota shares α increases incentives only to the extent the community thinks that its efforts will result in an increase in quotas—that is, that $H_W > 0$. It may seem a reasonable hypothesis that less poaching will give more hunting quotas, but this number depends both on the growth rates and on the way the agency reacts, which we explore next. If the communities have little faith in the willingness of the parks agency to raise quotas, their incentives will be considerably weakened. In the extreme case in which the community believes the quota allocation to be fixed, increased hunting shares act merely as a lump-sum increase in income and welfare of the communities, with no impact on conservation incentives.

We can also get a sense of the dynamics of instituting profit sharing. From Equation (8), we see that going from a steady state with no profit sharing to positive shares of wildlife revenues causes the shadow value to jump up, since the stock is then lower than is preferred. At that point, according to Equation (6), it will decline toward the new steady state.[15]

Parks Agency's Response

The parks agency's decision process is important for two reasons. First, the allocation of hunting licenses has direct effects on the communities and determines how they respond. Second, it influences the wildlife stock in ways outside the control of the community, which can affect people's welfare.

In reality, the parks agency is a government department, which is to some extent commercialized: it retains revenues from its operations and gets additional grants from the central government. It has offices that run various national parks and safari areas. As the manager of Zimbabwe's wildlife trust, the agency determines the hunting quota for both the safari areas under its direct control and also for the rangelands. The manager of a safari area may want to maximize the present value of profits. However, as the same agency also manages the national parks and safeguards all wildlife in Zimbabwe, it will also take general tourism and biodiversity effects into account, and the offtake from individual safari areas is unlikely to be determined independently of dynamics in the national parks.

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[15] Starting from the steady state without profit sharing, $\delta - F_W + H_w + Q_w = 0$, so $\dot{\gamma} = -R_W - \alpha p H_w - \beta T_W < 0$.

Indeed, communities typically do consider hunting quotas in CAMPFIRE areas to be conservative. This may be because the parks agency does not take into account the number of animals that the communities want in their areas, and hunting quotas are primarily designed for the trophy-hunting industry and as such are generally lower than quotas for nonselective hunting. However, one of the advantages of the CAMPFIRE program is that creates dialogue among stakeholders, leading to an increasing convergence between the quotas that the Department of National Parks and Wildlife Management sets and what the rural district councils desire.

Given the considerable ambiguity over the influence of economic incentives on park manager decisionmaking, we explore two types of decision rules for determining the hunting quotas, representing the extremes of this range of objectives. First, we explore the effects of a parks agency that adjusts to market incentives, intending to maximize its own profits. Second, we consider a simple decision rule, wherein the parks agency determines quotas based on the biology of the wildlife stock, ignoring revenue effects. We see that the parks agency's response to profit sharing has important implications for community incentives and welfare.

Profit Maximization

The park agency may make its decisions based primarily on economic grounds; in CAMPFIRE, prior to 1996, revenues seemed to be the primary concern in quota setting (Taylor 2006). Suppose now that the parks agency is a profit maximizer, as in Skonhoft (1998) and Johannesen and Skonhoft (2005). In this case, it maximizes with respect to *h* the total revenues from wildlife activities—after profit sharing:

$$\pi = p(1-\alpha)h + (1-\beta)T(W) \tag{9}$$

subject to the stock dynamics of Equation (1), to which we assign the shadow value μ for the parks agency. We assume that the parks agency engages in an open-loop Cournot game in which the it takes the community's path of antipoaching efforts as given (i.e., it perceives that $dA/dh = 0$).[16] Although we will explore different reactions to profit-sharing regimes on the part of the parks agency, we will retain this assumption that the parks agency does not take into account any strategic impact on the part of the community's antipoaching choice.

[16] This assumption is necessarily valid in Skonhoft (1998), since no antipoaching activity takes place. In Johannesen and Skonhoft (2005), each party takes the other's hunting effort as given.

The first-order conditions from the resulting Hamiltonian are the complementary slackness conditions

$$h \geq 0, \quad (1-\alpha)p \leq \mu \tag{10}$$

implying that licenses will be allocated if the price received by the agency is at least equal to the shadow value of the wildlife stock, and

$$\dot{\mu} = \mu\left(\delta - F'(W) + Q_W(W, A)\right) - (1-\beta)T'(W) \tag{11}$$

which states that the rate of change in the shadow value reflects the rate of discount, growth, poaching impacts, and the relative benefit from increased tourism.

In a steady state, $\dot{\mu} = 0$, implying $F_W - Q_W = \delta - (1-\beta)T_W / \mu^*$. Furthermore, for the steady state to include both hunting and tourism, with (10) this implies

$$F_W = \delta - \frac{(1-\beta)T_W}{(1-\alpha)p} + Q_W \tag{12}$$

This interior solution thus requires a certain balance between the rents to the parks agency from tourism and from hunting. Essentially, the parks agency allows the wildlife stock to adjust such that the marginal discounted rents from additional tourism equals the value to the parks agency of selling an additional quota. From Equation (12), we see that the steady-state wildlife stock is decreasing in additional poaching, decreasing in the rate of time preference (which makes current harvesting more valuable), and increasing in the relative return to tourism. In other words, the absolute returns to the alternative wildlife activities do not matter (assuming an interior solution); rather, 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 profit 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.

Note here that the existence of an interior solution in the parks agency problem guarantees that the community problem will fall into Class I or Class IV:

$$\delta - F_W + Q_W + H_W \geq \delta - F_W + Q_W = \frac{(1-\beta)T_W}{(1-\alpha)p} \geq 0 \tag{13}$$

This result must hold since the parks agency derives only positive benefits from wildlife use.

One also must beware of corner solutions, since no licenses will be given if $\alpha > 1 - \frac{(1-\beta)T_W / p}{\delta - F_W + Q_W}$.[17] In other words, if tourism values are relatively high and the parks get little hunting revenue, the agency might actually want to restock wildlife up to its ideal level, although we assume they are constrained from doing so (leaving $h = 0$, not an interior solution). On the other hand, if the parks agency gets no tourism rents, the steady state implies $F'(W) = \delta + Q_W(W, A)$, meaning that the stock is more heavily harvested, down to a level at which its growth rate is sufficiently high to match the joint effect of discounting and poaching.

To compare the different resource-management and profit-sharing regimes and strategic games, let us focus on the steady-state equilibria suggested by an interior solution. In a steady state with hunting, hunting quotas must equal net growth:

$$h = F(W) - Q(W, A) \tag{14}$$

With Equations (5), (8), and (12), we see that in equilibrium, the steady-state shadow value and antipoaching effort by the community must satisfy

$$\begin{aligned} \gamma^* = -\frac{c_A - \alpha p H_A}{Q_A + H_A} &= \frac{R_W + \beta T_W + \alpha p H_W}{\delta - F_W + H_W + Q_W} \\ &= \frac{(1-\alpha)p(R_W + \alpha p H_W + \beta T_W)}{(1-\beta)T_W + (1-\alpha)pH_W} \end{aligned} \tag{15}$$

The next question is then whether and how the community takes the parks agency's decision into account.

Symmetric Open-Loop Game

In a symmetric open-loop Cournot game, the community also takes the license path being offered by the parks agency as fixed. Correspondingly, the community does not believe it can influence the hunting quota decision on the margin, and $H_W = H_A = 0$. Simplifying Equation (15), we see that in the Nash equilibrium, in addition to (14):

$$-c_A / Q_A = (R_W + \beta T_W)\frac{(1-\alpha)p}{(1-\beta)T_W} \tag{16}$$

If the marginal tourism benefits outweigh the marginal agricultural losses to wildlife, effort is positive. The relative preference of the park manager for hunting over tourism rents

[17] This corner also implies a class I or IV equilibrium, since $\alpha \le 1$.

reinforces the incentive for effort, be it for antipoaching enforcement or for propoaching collaboration.

Here, we see that in equilibrium with the parks agency—quite differently from the community incentive effects alone—the antipoaching effort does not necessarily increase with greater sharing of tourism profits. The marginal incentive for effort only increases with the community's tourism share if the total marginal rents from tourism outweigh the marginal damages from additional wildlife:

$$\frac{\partial\{-c_A/Q_A\}}{\partial\beta} = \frac{(1-\alpha)p}{(1-\beta)T_W}\left(\frac{T_W + R_W}{(1-\beta)}\right) \tag{17}$$

In part, this result derives from the fact that, all else equal, although community effort is increasing with respect to tourism revenue, the parks agency conservation incentive is decreasing in the share of tourism sent to communities. In the extreme case when the community always perceives the wildlife as a net nuisance ($R_W + T_W \leq 0$, implying that agricultural losses from wildlife outweigh benefits to tourism), the loss of the parks agency's stronger incentives for conservation by the transfer of the tourism revenues to the community could have significantly negative impacts on the wildlife stock.

Similarly, the change in equilibrium effort is ambiguously related to the community's share of hunting profits, displaying the opposite sign of the marginal net benefits from agriculture and tourism. The community takes hunting profits as fixed, but the parks agency adjusts to losing hunting shares by emphasizing relatively more conservation and a higher wildlife stock in equilibrium. The effect is to dampen the community's influence on the equilibrium offtake and stock. They engage in less antipoaching effort if wildlife is a commodity and collaborate less if wildlife on the margin is a nuisance.

With changes in effort ambiguous, the impact of revenue sharing on the equilibrium wildlife stock is also ambiguous in the symmetric game. Since increasing β reduces the parks agency's incentive to conserve, for more sharing of tourism rents to improve the wildlife stock, the community has to engage in sufficiently more effort so as to outweigh the reduction in care by the parks. Any more effort requires positive total marginal benefits from agriculture and tourism—excluding hunting profits.

On the other hand, raising α increases the parks agency's desire to conserve. It unambiguously increases the equilibrium wildlife stock when wildlife remains a nuisance to the community. Otherwise, if the community values the marginal tourism rents more than the agricultural damages, allocating them a larger hunting share detracts from their incentives to conserve, limiting (or possibly eliminating) the gains to the wildlife stock.

These results echo those of Johannesen and Skonhoft (2005), who also find ambiguous effects of benefit sharing in a similar game between the parks agency and community poachers.

Stackelberg Open-Loop Game

On the other hand, the community may recognize its influence over the parks agency's quota setting behavior, rather than expecting quotas to be fixed. To consider this case, we use a dynamic, open-loop Stackelberg game with the community as a leader. If the community knows the equilibrium decision from the profit maximization function, $H_W = F_W - Q_W = \delta - \frac{(1-\beta)T_W}{(1-\alpha)p}$, which may be positive or negative, depending on the relative size of tourism rents. Furthermore, $H_A = -Q_A$: a reduction in poaching on the margin is offset by an increase in hunting licenses.

Substituting these incentives, Equation (15) simplifies to

$$c_A = -\alpha p Q_A \tag{18}$$

In other words, although the community cannot on the margin influence the wildlife stock targeted by the park manager, it can influence the portion of the total offtake that is harvested through hunting rather than poaching. Consequently, when the community is strategic in influencing the hunting quotas, it combats poaching to secure a share of those licenses for itself, not to manage the wildlife stock. In this case, to the extent the community receives hunting shares, they have the incentive to stop poaching; tourism, however, has no effect.

This strategic influence could raise or lower effort in equilibrium (relative to the symmetric case), depending on whether $\alpha/(1-\alpha) > (R_W + \beta T_W)/\left((1-\beta)T_W\right)$. That is, if tourism shares are relatively less important (particularly if $R_W + \beta T_W < 0$), strategic interests raise conservation incentives. It would seem likely that this is closer to reality in Zimbabwe where tourism revenues from CAMPFIRE have been small compared with hunting revenues.[18] However, if the incentive from tourism shares is sufficiently strong,[19] the community may engage in less effort in equilibrium as a Stackelberg leader, recognizing that on the margin it affects only hunting licenses rather than the wildlife stock.

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[18] Personal communication with the former director of the CAMPFIRE Association, Stephen Kasere, June 2004. See also Bond (2001).

[19] This could be the case in countries with a large amount of wildlife tourism, such as Kenya.

Corner Solution

Finally, if the parks agency gets no (or relatively few) hunting rents, its solution collapses to the corner $h = 0$, regardless of whether the community is strategic. The parks agency's equilibrium conditions become irrelevant, since the community is left alone to influence the wildlife stock, implying

$$\frac{c_A}{-Q_A} = \frac{R_W + \beta T_W}{\delta - F_W + Q_W} \tag{19}$$

In this case, the tourism share unambiguously improves community incentives for effort.

Biologically Based Decision Rule

With a biologically (rather than economically) based decision rule, the parks agency determines a sustainable harvest depending only on the current wildlife stock. This rule may better represent decision making after 1996, when greater stakeholder participation and external technical support placed more focus on improving the wildlife census and calculating a sustainable offtake. To a large extent, the rule characterizes 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 population, as well as other factors, including the relative importance of sport hunting and maintaining trophy quality (WWF 1997). Quotas then equal the offtake rate, multiplied by the population.[20]

Let us suppose the parks agency allows a share of the stock to be hunted, so $H(W, A) \equiv \varphi W$.[21] In this case, quota allocation is not directly affected by antipoaching effort, since $H_A(W, A) = 0$.

With this kind of simple decision rule, the community's equilibrium antipoaching effort is given by

$$-\frac{c_A}{Q_A} = \frac{R_W + \beta T_W + \alpha p \varphi}{\delta - F_W + \varphi + Q_W} \tag{20}$$

and

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[20] 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.

[21] One could also allow for a minimum threshold, above which the stock may be hunted, so that $H(W, A) \equiv \varphi(W - \underline{W})$.

$$F(W) - \varphi W - Q(W, A) = 0 \tag{21}$$

Totally differentiating (21), we get

$$\frac{dW}{dA} = \frac{Q_A}{F_W - \varphi - Q_W} \tag{22}$$

For a stable equilibrium at a given A, $F_W - \varphi - Q_W < 0$: if the wildlife stock is shocked upward, it will decline back to the steady state, not continue growing away from it. Therefore, $dW/dA > 0$; i.e., an equilibrium with more enforcement has a higher wildlife stock.

This further implies that the denominator of (20) is positive. From (20), we see that given an equilibrium wildlife stock, additional agricultural losses decrease antipoaching effort, while benefits from tourism increase it. In this case, we also find that an increase in the share of hunting revenues, α, unambiguously increases antipoaching effort.

In the new equilibrium, of course, the wildlife stock will adjust to the change in antipoaching effort, and vice-versa. While these adjustments may attenuate the initial impact of a change in revenue shares, they cannot more than undo the first-order effects.

The adjustment in hunting allocations, φ, also represents a potential instrument but has a more ambiguous effect.

When communities would otherwise prefer to collaborate rather than discourage poachers (when $MNB < 0$), the quota adjustment φ serves unambiguously to improve conservation incentives (or reduce collaboration with poachers), since additional hunting licenses both raise revenues for the community and reduce the stock of nuisance wildlife.

However, when some antipoaching effort is already worthwhile, the effect of additional hunting quotas can raise or lower that effort. On the one hand, the community receives additional revenue shares; on the other hand, quotas reduce the stock of wildlife that now has a net positive value to the community. In other words, ignoring the community revenue from licenses (or if $\alpha = 0$), the quota response serves to dampen community efforts. When the community recognizes that the parks agency will increase harvesting as the wildlife stock increases, the community responds by saving its costs of managing the stock either through facilitating or discouraging poaching. It saves costs of antipoaching efforts when they prefer a larger stock, since part of that stock increase is lost to more harvesting, and it saves collaboration costs when it prefers a smaller stock, since some of that effort just results in fewer quotas.

Summary

In Table 1, we summarize the predicted response of community effort to greater revenue shares, according the hunting quota adjustment process on the part of the parks

agency. Note that for the fixed adjustment process and the corner solution, we assume that $\delta > MNG$ still holds; else, those results could be reversed.

Table 1. Community Response to Resource Sharing and Parks Agency Policy

Parks Agency Process for Adjusting Hunting Quotas	$dA/d\alpha$	$dA/d\beta$
Fixed Rate	+ (if $\varphi > 0$)	+
Profit Maximization		
Symmetric Game	? (+ if $R_W + \beta T_W < 0$)	? (+ if $R_W + T_W > 0$)
Stackelberg Game	+	0
Corner Solution	0	+

This summary reveals the importance of the parks agency's management strategy and of its interaction with the community. Indeed, those factors determine whether greater resource sharing engenders greater conservation or, perversely, more collaboration.

Welfare

In terms of community welfare, the formulation of the harvesting decision has important effects. When the decision rule for setting hunting quotas does not change, as with the biological rule, 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.

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 [from Equation (10)] and reap all the benefits from tourism. Meanwhile, the community would receive no revenues yet would lose agricultural rents, as well as incurring costs while assisting poachers toward thinning the unwanted stock. 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 antipoaching (or collaboration) costs. Then, on

top of those gains, the community gets tourism revenue. To the extent that it adjust its antipoaching 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 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.

5. Conclusion

In this paper we formulated a bioeconomic model of wildlife management in a typical rural area in Zimbabwe, where a local community lives adjacent to a safari area. Two agents have influence over the wildlife stock: a parks agency, which determines hunting quotas, and a local community, which chooses to either collaborate with or discourage poachers from outside the area. Wildlife generates economic benefits both from the sale of hunting licenses and from nonconsumptive tourism; however, it also intrudes on the agricultural rents of the local community. Since a larger wildlife stock reduces agricultural returns, the community will engage in antipoaching efforts only if they reap benefits from wildlife activities.

The CAMPFIRE program in Zimbabwe directed shares of the profits from hunting and benign tourism toward the local community, in part to offer direct compensation for the nuisance suffered from wildlife and in part to induce antipoaching effort. 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 antipoaching units. Indeed, Goredema et al. (2005) note that CAMPFIRE served to empower the RDCs more than the community-based organizations.

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 profit sharing and community responses affect the resource management practices of the parks agency—and the delivery of benefits to the communities.

The model reveals that allocating shares of hunting quotas is not enough; communities must believe that the parks agency will afford them more licenses if their conservation efforts lead to a larger wildlife stock. If they perceive the hunting quota to be fixed, their profit share is treated as a transfer and does not affect their conservation incentives.

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. If less poaching merely translates into more licenses—and the communities know this—the incentives to resist poaching then derive primarily from the hunting revenues. If, on the other hand, additional licenses do not completely crowd out reductions in poaching, the community will expend more effort against poaching (or at least collaborate less) to the extent they receive more revenues from tourism. When the communities do not expect the parks agency to change its allocation of hunting quotas, their additional efforts to promote tourism through more conservation can be tempered after a while—even undone completely—if the parks agency does not itself have enough incentive of its own remaining to protect the wildlife stock for tourism.

Finally, community welfare is not necessarily enhanced if the parks agency changes its harvesting decisions when shares of its revenues are diverted to communities. Given any decision rule for harvesting, 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 the parks agency changes the way it determines hunting quotas, and that runs counter to the new wildlife stock objectives of the community, the local community can be made worse off by profit sharing. This welfare reduction can result since 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 the incentive effects of the revenue shares. On the other hand, the political institutions proved equally important. In particular, the filtering of the revenues through the RDCs, 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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