Economic Incentives and Wildlife Conservation

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Wildlife exploitation and conservation involves various costs and benefits, which should all be taken into account to achieve an optimal outcome. For this to occur, it will be necessary to develop appropriate economic instruments and incentives. Examining the scope for his is the topic of the current study. The time and funds available to complete this paper were extremely limited, which effectively made it impossible to complete a thorough and detailed analysis. As a result, in the paper we focus on what can be learned from standard economics. The paper lacks the level of detail and data to provide guidance in many operational issues.

Wildlife management poses a particular challenge to the global community because wildlife has an impact not only on people living in areas where wildlife is found, but also on people located considerable distances away. The problem is that the costs and benefits of wildlife exploitation facing "source" states differ substantially from those faced by other countries. From an economist's perspective, the main wildlife problem is that all too often many of the costs of harvesting wildlife are not appropriately taken into account. In particular, the values that wildlife such as elephants, tigers and rhinoceros have for people who may someday view them in the wild and the values that such fauna have for people who are simply delighted to know that such wildlife exist (having no intention of ever viewing them) are ignored in most harvesting decisions. Further, when property rights are insecure, those who harvest wildlife do not take into account the cost of their actions on the future availability of the resource because they do not have a stake in wildlife beyond those accessible to them today. This cost is referred to as the "user cost" and it is typically ignored in harvest decisions unless property rights are clearly stated, and protected. As a result, *in situ* wildlife is undervalued leading to their possible overexploitation (see below).

In essence, there is a divergence between what is optimal from a regional, community or individual perspective, and what is optimal from the perspective of a country or even global society. To address this divergence, a variety of economic instruments can be employed. The term "economic instrument" is used to describe any device/method used by government to achieve an outcome contrary to (other than) the one that occurs in the absence of any government intervention. The government essentially has three categories of economic instruments available to it: (1) common values and norms (threats or moral suasion in economic terms), (2) command and control, and (3) market incentives, which are also referred to as economic incentives (EIs). Moral suasion refers to the ability of the state to convince economic agents (individuals or firms) to act in a fashion that is socially desirable. Voluntary instruments (e.g., product certification/labeling by an industry association), perhaps accompanied by threats, are one aspect, but there also exist opportunities to "convince" citizens to report poachers, protect wildlife habitat and so on. Economic or market incentives and command and control (i.e., regulation) are generally used in combination, often out of necessity.

The objective of this study is to examine the scope of economic incentives in the conservation of wildlife. The focus of the study is on developing countries as these host most of the biodiversity and wildlife. The main results are as follows: While economists often believe that, in general, the best way to conserve wildlife and their habitat is to encourage efficient and sustainable use of these resources, the scope of EIs in such conservation efforts as an 'extra measure' to regulate harvesting pressure may in some cases be limited. Specifically, we argue that there are cases where the usual gains of EIs may be of secondary importance. Whether or not such gains materialize depends on the specific characteristics of a species and the parties involved in its harvesting. This should be assessed on a case-by-case basis. If both the habitat and the harvesters are "homogenous" (in the sense that there is little variation in the area in which the species is harvested and the skills/technologies of those harvesting the species), then the gains from EIs are small. These conditions may hold for (low-tech) open-access harvesting of certain species in Africa, but not for fisheries where "firms" of various sizes from individuals to large corporately-sponsored vessels are engaged in harvesting.

Two important qualifications are in order. First, while the role of EIs in regulation of *harvesting* may (but need not) be modest, we argue that international EIs may be of great importance when it comes to *habitat* conservation (indirectly contributing to wildlife conservation). In this respect we mainly think of means to capture and channel non-use values associated with conservation to affected parties living with (or owning) wildlife in developing countries – an example of international transfers or subsidies. Second, establishing property rights (or secure use rights for extended periods – that is establishing property rights in legal or physical space) is consistently encouraged by economists as a first step towards efficient management of resources – both of land and the wildlife it supports. Whether this first step must be complemented by additional EIs (tax, tradable quota) to arrive at a truly global optimum, however, is not certain. Sometimes additional command and control measures are to be preferred, and sometimes no additional measures are necessary (for example when external effects are small – see the next section).

We begin by examining why society might wish to intervene in the protection and provision of wildlife – the economic theory underpinning public wildlife management. In section 2, we provide a general discussion of the types of EIs that are available for addressing environmental spillovers, focusing on those instruments that may be useful in wildlife management. In section 3 we compare the performance of EIs with that of command and control instruments when addressing the two most important threats to wildlife; overexploitation and habitat conversion. In section 4 we emphasize the importance of the institutional context, and discuss implications for developing countries. Section 5 summarizes and concludes, and we propose a few priorities for targeted follow-up research that can be useful for making progress towards implementation of key issues raised in the paper.

1. ECONOMIC EFFICIENCY AND VARIOUS FAILURES

Environmental economics has become an important subject within economics as people have become increasingly concerned with pollution and other forms of environmental damage. The fact that some wild fauna and flora is threatened and endangered can be considered a special form of environmental damage. Therefore, ideas from environmental economics are relevant to wildlife management. What then does economic efficiency mean in a wildlife context? We consider 'wildlife' as broadly as possible, encompassing all biotic resources, including timber and fish.

First off, economic efficiency refers to the maximization of the well-being or welfare of citizens within a society. Economists measure welfare using a monetary metric and define it in terms of the economic surpluses (or rents) that accrue to economic agents in their capacities as consumers and producers. The surplus accruing to consumers is given by the difference between the benefit that they get from consuming a bundle of goods and services and what they have to pay for those goods and services. In technical terms, the "consumer surplus" is the difference between what people are *willing to pay* for goods and services and what they actually do pay. Likewise, the "producer surplus" is defined as the difference between the revenues from the sale of goods and services and the cost of providing them. Since fixed costs are sunk (i.e., made in the past and unaffected by current decisions), the net benefit accruing to economic agents as owners of factors of production is given as the difference between total revenue and total variable costs. In essence, therefore, the total surplus or economic welfare at any time is given by the difference between the benefit that citizens receive as "consumers" and the costs of providing the goods and services consumed – the area between the demand and supply curves (Figure 1).

To maximize society's overall well being when producing a good, an additional unit of the good should be provided as long as the benefit from this additional unit (the marginal benefit), to whomsoever they accrue, exceed the cost of this additional unit (the marginal cost), no matter who incurs them. Provision should stop when the benefit received from the last unit equals the cost of supplying that unit – marginal costs equal marginal benefits. Marginal benefits (or marginal willingness to pay) define the demand function, and marginal costs determine the supply function (Figure 1). Then society's welfare is maximized at the price and quantity where demand intersects supply – where marginal benefit (demand price) equals marginal cost (supply price). The area between a falling demand function and a rising supply function, up to the point where they intersect, represents the maximum sum of the consumer and producer surpluses, or maximum societal welfare, as shown in Figure 1. However, environmental problems arise because of three sorts of failure: institutional failure, market failure and policy failure. These forms of failure are clearly interdependent. For example, property rights to resources may be insecure (institutional failure) because governments fail to provide the legal environment that supports them (policy failure) or because of public good characteristics associated with the resource (market failure). One result of imperfect property rights can be external effects (another form of market failure). We will explain these concepts below.

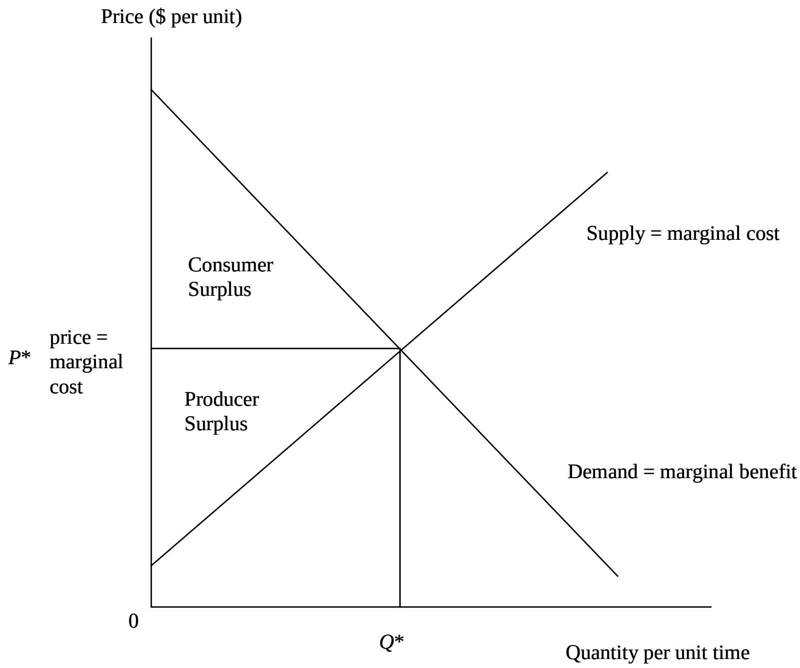

Figure 1: Maximizing Social Welfare when there is No Market Failure

1.1 Institutional failure: Ill-defined and enforced property rights

First, consider the pervasive problem of institutional failure. The most relevant manifestation of institutional failure for the case of wildlife and trade in species is probably insecure property rights, or 'open access' to resources – both the species themselves and the land upon which they live. Property rights can be understood as characteristics that define the rights and duties associated with the use of a particular asset or resource. Four property regimes are usually identified.

1. ***Private property*** In this case, the private owner has the right to utilize and benefit from the exploitation, conservation or sale of wildlife, as long as no (socially unacceptable) externalities are imposed on others (e.g., when shooting wildlife endangers the lives of others). Private ownership does not imply absence of state regulation (control), as private property cannot exist without state sanction and protection.

2. ***State property*** The state owns the wildlife and individuals may be allowed to harvest them, but only according to rules imposed by the state or the CITES Management Authority.

3. ***Common property*** In this case, a group owns and manages the wildlife resource, and the group excludes those who are not members. Members of the group have specified rights and duties, while non-members must accept exclusion. Coordination (regulation) of management may or may not be forthcoming, depending on local circumstances.

4. ***No property rights*** (*res nullius*) When a property right is not assigned, open or free access is the result. Under open-access, each potential user of the resource has complete autonomy to utilize wildlife since none has the legal right to keep another potential user out.

A summary is provided in Table 1. In practice, resources are often held in overlapping combinations of these regimes, and it is possible to shift from one (dominant) regime to another when conditions change. Failure to enforce or manage properly a state or common property resource (which is frequent) leads to open-access, which is the case for some endangered large-game species. The switch from common and state regimes to open-access as a result of population growth is well documented (Murty 1994; Bromley 1999).

Table 1: Classification and Characteristics of Property Rights

Type	Characteristics	Implications for economic incentives
Private property	Exclusive rights assigned to individuals	Strong incentives for conservation of resources and for investment as well
State property	Rights held in collectivity with control exercised by CITES authority or designated agency	Creating opportunities for attenuation of rights; managers have incentives for personal gains
Common property	Exclusive rights assigned to all members of a community; approaching private property when coordination arises.	Creating free-riders problem and low incentives for conservation
Open access	Rights unassigned; lack of exclusivity	Lack of incentives to conserve; often resulting in resource degradation

Property rights do not really exist under open access, and if there is no cooperation under communal ownership (or no enforcement under state and private ownership), then property rights are insecure. The absence of secure property rights (or even open-access) has resulted in excessive depletion of resources and biological assets for the following reason. The true cost of exploiting a resource consists of two distinct components: the private extraction costs and the unobserved opportunity cost, or the value of the resource *in situ* – the user cost. The intuition behind user cost in the context of a renewable resource is as follows: harvesting a unit of the resource today means that this unit and the growth (including any offspring) it causes are not available for future consumption. The (future) value of uncaught game depends on many different factors, including the discount rate, future markets for the resource, technological developments, reproductive features and so on. A sole private owner aiming to maximize profits will maximize the discounted value of this rent, and treat the resource as an asset. Hence, the value of unharvested animals and plants prevents a rational wildlife manager from over harvesting the resource, but only as long as she expects to be the one to benefit from this "investment". Private property may result in a conservative harvesting policy. In the absence of externalities and given similar discount rates, the same applies for state ownership.

An open-access resource exists if there is no possibility to exclude firms attracted by excess profits, with their entry competing away those profits. If there is unrestricted access to the resource, no person can be sure of who will benefit from the value of uncaught game. In an open-access situation, no individual harvester has an economic incentive to conserve the wildlife, and none can efficiently conserve the wildlife by delaying harvest. Doing so will only enhance the harvest opportunities of competitors, which is the tragedy of open-access. One might say that the individual does not care about escaped game, and discounts future harvests at an infinite rate (Neher 1990). New harvesters will be attracted to the activity, or existing ones will expand their efforts so long as they earn more than the (opportunity) cost of their effort. In *bionomic equilibrium*, all rent is dissipated, and total cost equals total revenue, rather than marginal cost being equal to marginal benefit. The situation where marginal cost exceeds marginal benefit is usually referred to as *economic overexploitation*.

In terms of Figure 1, failure to account for the user cost implies that agents will not base their harvest decisions on the supply (social marginal cost) curve as drawn, but instead on another marginal cost curve that is below it. This is illustrated in Figure 2 (discussed further below). As a consequence, harvested quantities in the short run will increase and prices will fall.

1.2 Market failure: Spillover effects and public goods

Two general types of market failure may occur and undermine economic efficiency of resource management, even when property rights are secure. First, the supply function may not embody all of the costs of producing goods and services, in which case market prices are no longer reliable as a measure of value. In the context of wildlife conservation we may think of nonuse values associated with *in situ* conservation

– the utility (or well being) that people derive from knowing that certain species exist or thrive, even though they will never "use" or view such species themselves.

One can think of wildlife as providing two sorts of products – products that result in the "consumption" of the specimen (e.g. fiber, wool, caviar, timber, ivory, bones, gall bladders, hides and bush meat, live export to zoos or as pets, ornamental and medicinal plants), and non-consumptive uses like eco-tourism, bird-watching and photography associated with the protection of *in situ* amenities and wildlife. A negative external effect can occur when consumptive use of wildlife reduces their numbers, and, as a wildlife population declines, the total economic (*in situ*) value to "preservers" falls. However, the consumers of wildlife products fail to take this into account in their decisions because there do not exist appropriate economic institutions and incentives to get "consumers" of wildlife to regard the costs they impose on those deriving utility from conservation. This is referred to as an externality, although the term spillover may be more descriptive of what happens and will be used here interchangeably with externality. Externalities can be good or bad, but their effect is that the supply price no longer reflects the true cost to (global) society of the activity. There is a divergence between private and social costs of provision, because one of the inputs in production, namely the environment, is not appropriately priced; the environmental cost or damage is not taken into account. In terms of Figure 2, agents do not base harvesting decisions on the marginal social cost (the 'true' supply curve), but on private marginal costs. When uncorrected, they will supply too much and social marginal cost lies above marginal benefit.

Second, there are many situations where private provision of a good or service does not occur because, once it is provided, no one can be excluded, and "use" or "consumption" by one person does not diminish the amount available to others. This is the definition of a *public good*. Public goods such as national defense, clean air and water, wilderness, biodiversity, and other environmental amenities will not be supplied privately because the provider cannot capture the benefits of so doing – once provided, no one can be excluded, so free riding is possible. Clearly some aspects of wildlife bear the characteristics of a public good. Wildlife contributes to global biodiversity (the "web of life") and enhances the well being of the majority of people (through the provision of "non-use values"). However, no one has the appropriate incentive to provide wildlife habitat or otherwise protect wildlife as they cannot capture the full benefits from the needed investments. Market failure occurs because the amount of a public good is under-provided, and thus marginal social benefits exceed marginal social costs. In this case, more of the (public) good should be provided, but it is forthcoming only if society subsidizes a private supplier, or provides it publicly.

1.3 Policy failure: Perverse government incentives

A final reason why wildlife may be overharvested (or why their habitat is degraded in many regions) has to do with perverse government policies. One well-known form of policy failure is subsidization of harvesting or habitat conversion. As will become clear below, one way to address market failure is through implementation of a tax or user charge. However, rather than charging users to exploit natural resources, there are many real-world examples where exploitation of natural resources is encouraged rather that restricted – policies aggravate rather than mitigate pre-existing distortions. When use of resources is subsidized, the marginal cost curve is pushed downwards and short-term supply expanded. This could be illustrated in Figure 2 (but is not) by adding another marginal cost function that would lie below the 'marginal private harvest cost function with insecure property rights'.

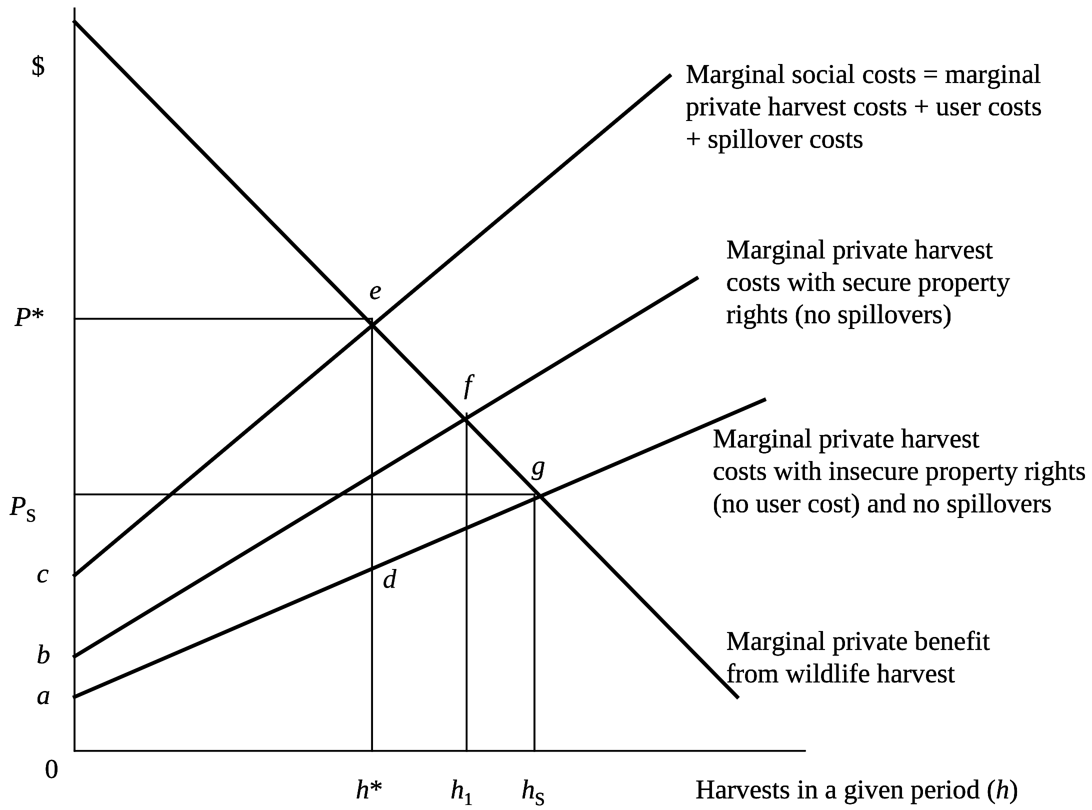

Figure 2: Divergence of Social and Marginal Costs of Harvesting

1.4 Policy and land-use decisions

Figures 1 and 2 suggest that *some* harvesting of wildlife is typically optimal in the sense that it maximizes welfare for society at large – this is the case when the marginal social cost curve and the demand curve intersect for some positive harvest level h^*. But there is more. Economists have long known that allowing profitable use of resources provides incentives for their conservation, and could lower enforcement costs associated with conservation.

Prins et al. (2002) argue that arbitrary restrictions on use of wildlife in regions in Africa have contributed to the demise of these species – a rather paradoxical statement perhaps. The reason is that restrictions on use erode the incentive that people have to invest in the protection of the species for potential future harvesting. People have no incentive to invest in the conservation of the species. The result will be that base resources on which wildlife depend for their survival (such as land) will be allocated to other, more profitable uses. The key insight is that taking away the short-term incentive to harvest may not be in the species' best interests. In effect, this is another manifestation of policy failure.

Local people make the decisions concerning land use and resource exploitation. It is costly to enforce prohibitions against their chosen activities, and so prohibitions often increase monitoring cost without conservation benefits. This is because the conversion of reserve lands and incursions for wildlife poaching halt when local people want it to do so, not when they are told to do so. The least costly policies provide incentives for the local people to support the reserve's designated uses, not bans against non-designated uses. This implies that the most successful policies for the conservation of wildlife and wildlands have been those that encourage their limited and managed development. For a review of experiences in the context of crocodile management, roughly consistent with this insight, refer to Hutton et al (2001).

The allowance of restricted use of wildlife encourages the local community to view the wildlife as an asset for development. The allowance of restricted uses of reserve lands allows the local people to receive some use of the lands while affording other uses to the wildlife. In any event it is necessary for the benefits of wildlife conservation to be distributed across the local community, by means of benefit sharing policies. This might also take the form of sharing tourism revenues from the reserve with locals, or the maintenance of a certain share of the jobs in the reserve for locals. The locals must be encouraged by such policies to view the designation of the reserve for wildlife uses as a specific form of local development policy for their benefit, not as a constraint on their development of the reserved lands.

In the context of lands that are heavily used by local peoples, the designation of protected areas and reserves simply acted as a mechanism for generating hostility amongst the local populations. This hostility then became translated into management costliness, as park managers attempted to enforce the restrictions on the use of the lands. When local peoples viewed these restrictions with hostility, they simply made even greater efforts to make use of lands that they believed to be theirs. Park managers had a near impossible job of enforcement, and always insufficient funding to accomplish it.

An important caveat is in order here. There are exceptions to the statement that use restrictions should not be too stringent. In certain cases zero use is economically optimal. This is the case, for example, when full internalization of costs (including spillover cost) would lead the market to break down – that is; the relevant supply curve would lay above the demand curve such that there is no intersection where $h^*>0$. In that case, assuming there is no pressure for habitat conversion, strict preservation (possibly supported through a trade ban) is globally optimal. Alternatively, strict preservation may also be optimal when values derived from using the resource conflict with non-use values. Economists typically assume that non-use values are associated with the size of the wild stock, but recent research suggests that there may also be *direct disutility* following from uses that are harmful to individual animals. For example, people may care about the fact that individual whales are shot, rather than care about the fact that the whale population becomes a bit smaller as a result. When direct disutility is sufficiently large, global welfare is maximized by refraining from use altogether. Swanson and Kontoleon (2003) have established that this condition holds for the black rhino, where intrusive uses include trophy hunting and seducing rhinos to remove their horn. However, the earlier statement about benefits sharing still applies in this context: If non-use values are large, they should be captured (through transfer payments from the North, say) and channeled to those who bear the burden of living with the wildlife. In the absence of such transfers, advocating zero use may simply be non-sustainable.

1.5 Summary

Economics prescribes that wildlife should be harvested as long as the marginal social benefits of so doing exceed the marginal social costs. Included in (marginal) benefits are values of wildlife products (e.g., caviar, medicinal plants, ivory, bush meat, hides) or the live specimen (if sold to a zoo or herbarium). In addition, (marginal) social costs include (i) the loss *in situ* (existence, viewing) value that wildlife provide citizens who may be located in countries other than the source country, (ii) the opportunity cost from harvesting the wildlife today rather than waiting for a more opportune time in the future when the specimen(s) may fetch a higher price, (iii) the lost future value of offspring that might result from leaving the specimen(s) in place, and (iv) the opportunity cost of the resources employed in the harvest activity. This is illustrated in Figure 2, where failure to include all costs and benefits leads to suboptimal levels of harvest, h_S, that are likely well above those desired by global society, h^*. The loss to global society from harvesting h_s rather than h^* is area *deg* in Figure 2.

We can compare outcomes relative to the suboptimal level of harvest h_s (and even higher harvests if harvest subsidies are in place). If property rights to wildlife (and habitat) are clearly spelled out and protected by the courts, then the harvest level would fall to h_1. In that case, global benefits would increase by area *abfg* in Figure 2. If in addition it is possible to pay wildlife owners for the nonuse benefits of *in situ* wildlife, harvest levels would decline further to h^*, the globally optimal level of harvest. In that case, global well-being would increase by an additional amount given by area *bcef* in Figure 2. Our contention based on previous research concerning marginal willingness to pay for increased numbers of wildlife and minimum viable populations required for preservation of the species (e.g., see van Kooten and Bulte 2000) is that area *abfg* is larger than *bcef* – that the benefits for wildlife protection of specifying and protecting property rights is greater than those from attempting to subsidize “owners” of wildlife for protecting *in situ* numbers. Indeed, without appropriate protection of property rights, transfer payments to protect *in situ* wildlife cannot even be attempted. This is discussed further below.

2. WILDLIFE AND ECONOMIC INSTRUMENTS

In this section, we examine the various economic instruments that are available to countries, and discuss their advantages and disadvantages. As noted in the introductory section, economists generally identify three economic instruments for addressing market failure due to environmental externalities (or spillovers): (1) command and control (hereafter C&C), (2) common values and norms (or more cryptically moral suasion), and (3) market incentives. Conceptually, common values and norms are intermediary between the "extremes" of the market and C&C (Loasby 1990; Stavins 2002). Common values and norms develop more easily in a homogeneous society, while markets are more appropriate in a heterogeneous society (CPB 1997, pp.42-44). However, there is much confusion about the different instruments that are available and which are preferred (see, e.g., Richards 2000).

One way to classify economic instruments for resolving environmental spillovers and user cost is illustrated in Table 2, where instruments are classified according to two dimensions – whether control of the means used to address the externality resides with the private party or with the state, and who bears the costs. Market incentives include subsidies, contracts, taxes and rights trading. (Rights are defined as an entitlement, whether to a harvest quota of a wildlife species or fish stock, or the ability to develop or conduct other activities on land, such as plowing or harvesting before a certain date.) These give private parties complete discretion over the actions taken.

In contrast, C&C regulations generally provide much less discretion. As will become clear below, this will lead to inefficiencies in the context of asymmetric information between agent and regulator. At one extreme, regulations may specify technology-based standards that regulated firms must use or, in the case of wildlife perhaps, prescribe management standards – the "party-on-the-ground" (individual, firm, wildlife management agency) has no degrees of freedom in decision-making. Alternatively, regulations could provide the party-on-the-ground some degree of freedom on how to proceed, as would be the case if the regulation only specified the number of specimens that can be harvested each period (a quota of h^* in Figure 2, say). The regulator or CITES authority could then employ a market instrument (e.g., tradable quota) to allocate the harvest in an efficient manner. In either case, the cost is borne by the private party.

Table 2: Classification of Instruments for Addressing Wildlife Conservation

Who Bears the Costs?	*Private Party Control*		*Government Control*
	Price Based	Quantity Based	
Government/ Society	Subsidies, transfers	Grandfathered (tradable) quota Contracts	Public provision
Private Party	Taxes, fees, charges, tariffs	Auctioned (tradable) quota	C&C regulation Harvest quota

The instruments included in Table 2 do not exhaust the full range of instruments for environmental protection. For example, the literature contains discussions of liability systems, and bond-and-deposit systems. However, neither of these types is likely to be important for the case of wildlife conservation, and they will therefore be ignored in what follows. We will focus on the most important economic instruments in the context of the protection of wildlife – taxes/charges and tradable quota or rights. In addition, we consider physical property rights, since tradable quota constitutes a legal right. Before we turn to a discussion of these EIs, however, we will briefly evaluate the subsidy instrument.

Economists are typically critical about the use of subsidies to achieve conservation. Consider the case where harvesters are subsidized to *lower* their harvest rate (this is the logical counterpart of the literature on subsidies and pollution, where firms are paid to lower their emissions). Assuming such agreements can be enforced, subsidies would "work" in the sense that they tend to lower the optimal harvesting level of individual harvesters. But there is a large potential problem with such subsidies when property rights to the resource are imperfect: they can encourage entry into the harvesting sector that the government aims to control. That is, even though harvesting per harvester goes down, the number of harvesters will likely go up, compromising conservation objectives. Unless the number of harvesters is somehow fixed such that new entry does not occur (e.g., when property rights are secure), subsidies are a poor instrument to regulate harvesting. But, importantly, there is another issue to consider in this context. In addition to suffering from excess harvesting, many wildlife species are threatened by habitat conversion. Subsidies can be an efficient, effective and equitable instrument to deal with habitat conversion. By basing transfer flows on habitat made available by landowners, habitat conversion (and, thus, indirectly also wildlife conservation) will be promoted. The fact that "entry" in the habitat sector is promoted by subsidies is, of course, no problem – quite to the contrary; this is the main intention. We return to this in section 3.3.

Following Panayotou (1994), we distinguish between property rights in *physical* space (land ownership, ownership of wild fauna and flora on one's land) and property rights in *legal* space (e.g., a right to hunt or collect one or more specimens, trade live specimens or parts or derivatives of them). The latter right specifies a narrower "bundle of rights" to the resource than the former. Many species are migratory, so it is not possible to establish full property rights (i.e., rights in physical space) as access will be shared with others. But rights can still be established in legal space by defining an allowable harvest level for individuals when the migratory species is on their land. Note that, when property rights are established in legal space, the regulator can set the quota by taking external effects into account. In contrast, if a private agent has the property rights to the resource in physical space, the agent will fail to take these effects into account. When such spillovers are important, assignment of property rights in physical space should be complemented with other instruments in Table 2.

Establishing and enforcing property rights to resources in physical space can provide an important impetus for sustainable use and conservation. In Figure 2, a private agent given the right to a resource will change harvests from h_S to $h1$, still below the optimal harvest level h^*, but movement is in the "right" direction. Other instruments are needed to move from h_1 to h^*. Whether or not the change from h_1 to h^* can be accomplished via EIs is debatable, but it may also be the case that the "effort" required to go from from h_1 to h^* is not worth making: By appropriate specification of property rights in physical space, most of the spillover problem may be overcome and the species spared from potential extinction. Property rights depend on cultural conditions, so it may be better in some sense to allocate them to a well-defined group or community rather than private individuals/firms (see Table 1 and the excellent book by Baland and Platteau 1996).

The existence of property rights and the associated 'right' to exclude others from using the resource implies that the user cost will no longer be ignored by those with access to the resource. When property rights are secure, owners know that the fruits of their 'investments' (such as refraining from current harvesting, or postponing the decision to convert habitat) will accrue to them. This means that they are more inclined to make such investments. Addressing this institutional failure therefore enhances efficiency, and comprises an important first step in enhancing efficiency and sustainability of resource management. This is illustrated in the following case study, which illustrates the benefits of defining property rights in legal space, and of benefits sharing.

2.1 Establishing secure property or use rights – The CAMPFIRE case study

In Southern Africa there was a widespread problem of poaching in designated parks and reserves until government officials began to institute benefit-sharing programs. These programs have taken many different forms. Sometimes they simply allow the local community to set up tourist related facilities within the park (Natal's Good Neighbour Policy), other times they give the local community a share in the value of wildlife that wanders onto their neighbouring lands (Zimbabwe's CAMPFIRE program), and sometimes the community is allotted a share of the receipts from wildlife management on reserve lands (Wildlife Management Trusts). It is important to note how these community funds were channeled back to the community in a manner that is widely visible throughout the community. Sometimes this can be accomplished by means of purchasing community goods such as schools etc. Other times it is best to send the benefits back to the individuals in the form of jobs or money.

Zimbabwe's approach of sustainable wildlife utilization has now been extended to all of the communal areas by the CAMPFIRE program. Communities have been granted the rights to manage as well as the means to capture the benefits from wildlife use. Since its introduction, CAMPFIRE has managed to promote cooperation among village members and has enhanced the institutional capacity of other community programs.

During the colonial times and up to 1978, legislation in Zimbabwe prohibited all utilisation of wildlife for commercial as well as traditional hunting. Locals were even relocated to make way for National Parks. As a result, many communities have been disenfranchised from their natural resources and wildlife became not only valueless, but a symbol of oppression and its destruction was encouraged. This alienation of people from wildlife was clearly unsustainable. In 1955, the Department of National Parks and Wildlife Management allowed commercial, (mainly white) farmers to utilize their wildlife commercially. Consequently, farmers began to benefit from wildlife and started to look after it. The value of wildlife products combined with the marginal economic viability of conventional agriculture induced a shift from livestock to natural ecosystems accommodating a wide range of species. While cattle could only be sold for meat, wildlife could be photographed, sold as hunting trophies, as well as being sold as meat. At present, some 75 percent of Zimbabwe's commercial ranches now participate in the wildlife industry.

The first attempt to extend this system to communal areas was a program called WINDFALL. The program involved allocating revenues from wildlife culling in National Park and from safari hunting to district councils, but overall wildlife management remained with the State. The results of this program were disappointing since the councils kept all the money and local people saw few benefits. In 1975, a further step was taken which granted councils the same rights as private landholders to appropriate the value from wildlife. In order to increase the accountability of the councils the CAMPFIRE program was established. The program ensured that producer communities rather than councils, managed and benefited from wildlife.

Consider the impact of CAMPFIRE at the local level. Chikwarakwara is a small village. Its population is exceedingly poor, largely uneducated and aging, since many of the young people have migrated out of the area in search of work. As with many other villages, disenfranchisement from its resource resulted in open access and over exploitation of their wild resources. Chikwarakwara was characterised by an erosion of traditional controls on resource use, growing population pressure, open access resources and unsustainable resource use.

In 1989, there was a major step towards the implementation of CAMPFIRE principles including the appropriation of wildlife revenues by the villagers. In the process, special care was taken to ensure that villagers related the revenues they received to the actual value of wildlife in their area. Moreover, the revenues were allocated to individuals rather than to the community as a whole. This not only helped to increase the perception of important individual revenues to be gained from wildlife management, but also boosted accountability of the project.

As a result of this approach, more positive attitudes were fostered towards wildlife and towards the management of the wildlife revenues. Villagers were able to carry out better resource trade-offs and gained self-esteem. New institutions were created including wildlife committees to ensure accountability and transparency. With stronger community unity, a number of new opportunities began to open. Snaring was reduced as informal social controls were established and strengthened. Entrepreneurial skills learned in wildlife management were transferable to other projects such as the expansion of the irrigation system and the management of the grinding mill.

To control levels of wildlife use, each council develops a sustainable hunting quota in collaboration with the state departments. Middle agents who have the capital and skills are employed to attract international clients. In order to avoid excessive monopoly power and appropriation of the wildlife rents by the middle agents, a system of tenders was established. Through time, the communities have improved their marketing skills, managing to double their incomes between 1989-93. In fact, they have managed to capture better prices than the government in key safari areas in Zimbabwe. The program has shown that communities have rapidly learnt the necessary skills for natural resource management despite the limited capacity of the state to provide technical assistance. In fact, districts with donor support tended to be slower to develop and have suffered from excessive overhead costs.

The philosophy of CAMPFIRE has been to set initially the conditions right for sustainable wildlife management by local communities. The communities have started to cooperate and build institutions for management of resources. A key insight is that allowing use by well-defined groups (akin to establishing property rights) may go a long way towards achieving efficiency. However, to arrive at the optimal outcome property or use rights will generally have to be complemented with additional policies. As will become clear below, economists usually prefer economic incentives in this case.

2.2 Preference for economic incentives

Economists generally express a preference for private party control, or market incentives. A common feature of such incentives is that the market allocates resources, with the role of the government or regulator restricted to providing the legal and institutional framework, rather than interfering with the conduct of business itself, as is often the case in C&C approaches. Economic or market incentives consist primarily of taxes (or charges) and tradable rights.[1] Before taxes can be levied or (tradable) rights issued, the authority must have in mind some target. In the context of Figure 2, the target is h^*, which is much lower than the unregulated harvest h_S. (Clearly, government intervention is only required if $h^* \neq h_S$.)

In this subsection, we present the main arguments in favor of economic instruments found in the literature. These four arguments are often advanced to manage polluting industries, rather than wildlife, and we will explore to what extent these insights spill over to the realm of CITES in section 2.3.

A. Least cost approach: It is realistic to assume that agents in the economy are heterogeneous in their ability to produce commodities – some people will be more efficient than others in producing output, harvesting wildlife or abating emissions. It is also realistic to assume that knowledge about the capacities of different agents is not public; economic agents themselves have more information about their 'true type' than the government, and it may not be in the interest of private parties to reveal their true type to the regulator. If these two assumptions are met, then market incentives are more

[1] The choice between a harvest tax and quota depends on uncertainty. If there is no uncertainty about the marginal cost and marginal benefit functions (about species growth, their future value, etc.), it does not matter whether the authority chooses a tax or quota to achieve its goal (h^* in Figure 2, say). If there is uncertainty, picking a harvest tax can lead to the 'wrong' level of harvests, while choosing a quantity can result in a mistake about the forecasted 'price' that agents will have to pay for harvesting rights (Weitzman 1974). Such errors have social costs. However, unless high values for rights encourage 'excessive' poaching, it would seem that a quota is preferred from an uncertainty standpoint because, subject to the effectiveness of policing efforts (which is also required with a tax), it guarantees that harvests do not exceed socially desired levels.

efficient than across the board regulatory C&C measures. If these conditions are not met, the regulator can 'mimic' the market outcome by tailoring regulations for each and every firm/agent. However, even in such an unlikely full-information context, the administrative requirements and costs of such regulations exceed those of market incentives. To circumvent such costs, the regulator typically ignores heterogeneity at the agent level and treats each agent the same (e.g., requiring each one to harvest the same quantity). As will become clear, this comes at a cost.

The main market incentives are harvest taxes and quota trading. Harvest taxes are a way to internalize the spillovers associated with harvesting. The optimal level of the tax is determined by the institutional setting. When property rights are not secure, the tax should be set at a relatively high level to account for both external effects and user costs. When property rights are secure, resource owners already account for the user cost, such that the optimal tax only reflects the external cost (nonuse values foregone or possibly ecological costs) associated with harvesting at the optimal level.

Quota trading first requires the establishment of an aggregate cap (or quota) on harvesting, followed by the 'issuance' of 'rights to harvest' (quota or permits) that can then be traded. Tradable emission permits can be allocated to existing harvesters at no cost to them (referred to as "grandfathering") or sold via an auction to the highest bidders, thereby generating revenue for the government.

The common feature of taxes and tradable rights is that, in equilibrium, *harvesting occurs by the most efficient firms/agents*. With taxes, only agents with low harvesting costs will be able to pay the tax and stay in business. The important thing to note here is that agents themselves reveal their costs by their actions (exiting or entering the 'industry' by harvesting wildlife) – the regulator does not need to know anything about individual agents. With tradable quota, harvesting rights will gravitate to firms with the lowest marginal harvest costs. They are the ones that stand to benefit most from acquiring these permits (as their profits per unit of harvesting are greatest), and therefore they are able to either outbid other agents at an auction, or simply purchase quota from less efficient firms/agents directly. Again trade among firms themselves, with superior knowledge about their costs than the regulator, ensures that an efficient outcome emerges. This is an important result: with taxes and quota the total harvesting costs are minimized for a given (pre-determined) total harvest level.

B. Easy to enforce: Market incentives are no panacea. Regardless of whether the authority chooses regulations or market incentives, or a mix of both, there is no escaping the need to monitor agents' behavior and enforce the regulations. Regardless of the type of regulation chosen, agents have an incentive to ignore it and free ride on the conservation efforts of others – over-harvest their quota, understate their catch to save on taxes, and so on. However, it can be argued that enforcement requirements can be lowered when property rights are created (regardless of whether they exist at the level of the individual or the community). This will have two important effects. First, as stated above, secure property rights will induce owners to take the user cost of extraction into account, and therefore it diminishes the incentive to over-harvest in the short run. Second, while access to the resource must still be enforced to restrict usage of non-owners and control extraction by potential 'cheaters' in the case of a community-owned resource, the costs associated with such enforcement will now to a large extent be borne by the owner rather than the regulator. Since the resource owner likely has better knowledge about local enforcement issues than a regulator, costs may also be lower. To some extent similar devolution of enforcement is expected to occur when a multi-year tradable quota system is in place, with a multi-year quota amounting to just another form of property right. For example, when it is agreed that the aggregate cap or total harvest in any particular season is a function of the wild stock, then it is in the interest of all quota holders to monitor each other's behavior and make sure that others do not over-harvest. Allowing others to over-harvest would be to one's own detriment as this lowers next season's stock and quota.

C. Dynamic incentives: Economists generally like economic incentives because they provide agents with an incentive to adopt technical changes that lower costs. Consider the textbook case of abating emissions. A tax or tradable emission permit system gives an incentive to develop and adopt new and clean technologies because such technologies will enable firms to sell permits (or avoid buying them), or avoid paying the tax. Further, market instruments provide incentives to change products, processes and so on, as marginal costs and benefits change over time. Because firms are always trying to avoid the tax or paying for emission rights, they tend to respond quickly to technological change.

D. Economic instruments may raise revenues: As mentioned under B; regulating firms requires monitoring and enforcement – costly activities for the regulator. When market instruments are used, some of these costs might be returned to the authority in the form of revenue. Indeed, regulation can turn into a net revenue-raising activity, and there is ample evidence that many environmental taxes are used exactly for this purpose in OECD countries. It is evident that a tax system raises revenues, but creation of property rights or tradable quota may have a similar effect. This happens when rights are auctioned off rather than grandfathered (provided free of charge to agents already involved in the activity). However, grandfathering of rights has a certain political appeal since the allocation mechanism can be used to make the trading system acceptable. For example, if it were possible to identify all harvesters of a certain wildlife species, allocating them a certain number of harvest rights (and guaranteeing those) might cause them to mend their ways, harvest only the allowable quota granted them, and, at the same time, encourage them to aid in the protection of the species.

2.3 Relevance for the case of wildlife conservation

How important are these four arguments in favor of economic (market) incentives for the particular context of wildlife conservation? First, consider the issue of least cost harvesting. It has been demonstrated that extensive cost savings may occur in the context of commercial fisheries management and pollution control (Weninger and Waters 2003, Cropper and Oates 1992). But it is important to realize that commercial fishers and polluting firms (possibly producing different goods) may constitute a much more diverse or heterogeneous set of actors than, say, those harvesting certain wildlife species in a developing country.

When (i) harvesting techniques are low-tech, labor-intensive and capital-extensive, and fairly uniform across all contributing agents, and (ii) the ecological conditions under which the species lives, and their local densities, are rather similar across space, then the efficiency gains in terms of equating marginal harvesting costs across agents must be small. In other words, if marginal costs across harvesters are rather similar, the efficiency gains from trade or EIs will be small. How "similar" are marginal harvesting cost functions for wildlife harvesters? This should be assessed on a case-by case basis. There are likely cases where efficiency gains from tradable quota or taxes are considerable. When harvesting technologies vary greatly (WWI rifles versus helicopters and high precision rifles); when habitat is diverse (common lands versus private reserves, game ranches); or when demand for wildlife harvesting originates from different markets (e.g. commodity versus trophy markets, nuisance harvesting), such gains can be considerable. In contrast, when a homogenous group of people is harvesting a single species for a common market, using similar techniques, then the gains are small. The conclusion is that efficiency gains are context-dependent, depending on conditions of heterogeneity. They may be smaller for the management of many wildlife species than have been observed in many other sectors, but this is not certain.

Next, the case for *dynamic* efficiency (i.e., the incentive to spur technical change) is not very compelling in the context of species harvesting. It can be debated whether encouraging harvesting efficiency is to be applauded in this context. More important, however, is the following observation: perhaps economic incentives will have no impact on dynamic efficiency whatsoever. The important textbook insight discussed above depends on the assumption that firms produce output and create pollution as a byproduct (that can be controlled or abated at some cost). It does not spill over to the case of wildlife conservation where the separation between product and byproduct does not exist. By inventing more efficient harvest techniques, firms cannot sell quota (in fact, they will presumably buy more of them), nor can they save on their tax obligations.

This implies that the main advantages of market incentives will likely be the impact on enforcement costs and the potential to raise revenues, and possibly (in certain cases) standard efficiency gains from adopting EIs. The former is clearly important for conservation purposes, as emphasized above. The relevance of revenue raising is slightly more dubious. Economists usually consider distributional issues (like transferring money from private parties to the government) of secondary importance, but in the context of a developing country facing difficulties in raising revenues to provide basic public goods, this issue obviously becomes relevant as well (as already noted above). In addition, as will become clear in section 3.2, distributional issues may be of importance when the amount of habitat is endogenous, or chosen by local agents.

2.4 Summary

To achieve optimal use of resources typically involves regulation of users. Regulation is necessary to internalize spillover effects and, when use or property rights are not secure, to account for the inter-temporal user cost associated with harvesting. We argue that defining property rights in physical or legal space is an important first step towards optimal resource management. In this sense, EIs can often be fruitfully applied. To capture spillover benefits (if any) the regulator can choose either *additional* economic instruments, or command and control. It has been documented in other sectors such as commercial fisheries that adopting EIs may result in substantial efficiency gains. In this section we argue that the scope for such additional efficiency gains may be modest in the case of wildlife harvesting. Whether no additional regulation is preferable, or intervention through C&C or EIs instead, should be determined at the case study level. The costs and benefits of the various options will vary greatly, depending on characteristics of the species, the habitat and the parties involved in harvesting.

3. ECONOMIC INSTRUMENTS: WILDLIFE HARVESTING AND HABITAT

The main threats to wildlife are introduction of exotic species (invasives), overexploitation and habitat conversion. Trade arguably affects all three threats, for example, by shipping species from one location to another or by changing relative prices of factors and commodities. For trade in threatened species and/or wildlife, that is CITES, the most important threats are overexploitation and habitat conversion. Economic incentives may affect both the incentive to harvest species, and the incentive to convert natural habitat for some competing purpose. We will return to these two threats in this section.

3.1 Regulating harvesting

Under open-access no individual harvester has an economic incentive to conserve the wildlife, and none can efficiently conserve the wildlife by delaying harvest. Doing so will only enhance the harvest opportunities of competitors. New harvesters will be attracted to the activity, or existing ones will expand their efforts so long as they earn more than the (opportunity) cost of their effort. The consequence of ignoring user costs by individuals is that all rents are dissipated, and eventually total cost equals total revenue. Excessive hunting effort and too small resource stocks represent the fundamental problem of open access. Various management instruments can be used to combat rent dissipation and protect wild stocks. It will become clear that while most instruments are theoretically able to protect stocks, only some will actually be able to maximize resource rents.

Most textbooks on resource economics (e.g. Conrad and Clark, 1987) demonstrate that management agencies or the CITES management authority can force harvesters to recognize user costs by either imposing the appropriate tax on harvests (reducing revenues) or harvesting effort (raising costs by either a license fee or effort tax). While the resulting outcome is theoretically efficient and does not involve tedious monitoring of effort, a few major problems remain. First and foremost is that taxes may be politically infeasible as it transfers all of the economic rent to the government, and harvesters will use their (political) power to prevent such a policy from being implemented. This is the most important reason why tax policies are hardly implemented anywhere in the world to regulate commercial fisheries (Brown 2000). Second, the authority may have difficulty in computing the optimal tax, which depends on factors such as demand for wildlife products and biological processes. Taxing harvesting effort can be difficult because fishers have an incentive to substitute types of effort that are not taxed for types that are taxed. Finally, enforcement of a harvest tax and its collection may be difficult.

Much more common than tax schemes in actual renewable resource management policies are quota schemes. In the case of wildlife, an annual harvest quota can be determined from information about the species' population dynamics and minimum viable population, and other (economic) factors, and then allocated in some fashion. At the national level, quota can be distributed amongst individual hunters or communities, or hunting can remain open until the national quota is reached. While a quota system may result in conservation of the stock and optimal harvesting levels (provided that the authority has access to all the relevant data, and that monitoring and enforcement occur), a quota system will not always result in efficient allocation of effort. For example, if the hunting is opened up until the country's quota is reached, it is possible to end up in a situation where the wrong animals (e.g., females of child bearing age) are taken with more effort than needed as hunters/communities rush to capture quota before others get there first. Such rushing is likely to dissipate rents as the situation is not unlike (controlled) open-access. The only difference is that wildlife stocks are protected from over-exploitation by the quota.

Open-access problems can be overcome if property rights are allocated to individuals or communities/groups. If a hunter has the right to harvest a certain quantity in a specified time interval (say, per year), she will decide to use her effort so that harvest costs are minimized if discounted prices are constant, for example, or that her supply is concentrated in periods of high demand and high prices. Economic efficiency occurs at the firm level, but from society's point of view it is still possible to improve the allocation of effort by allocating harvest to least-cost agents. This may be accomplished by auctioning them off or by allowing trade in harvest rights.[2] If either of these options is implemented, the quota scheme is both efficient (maximizes resource rents) and conserves the wild stock.

If a particular wildlife species is found in more than one country and global management is desired (as may be the case for elephant), an overall harvest quota can be determined and allocated among the individual countries which would then allocate quota internally. Again each nation's harvest rights can be traded domestically or, perhaps, even internationally. The latter option enables society to earn further gains from trade by exploiting international differences in harvesting cost. A condition for such a scheme to maximize welfare, however, is that the conservation value of elephants is the same for elephants in different countries, If this condition is violated, trading with an "exchange rate" reflecting differences in spillover values may be introduced.

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[2] Quota constitutes a property right that has value. The price of quota is the value of the *in situ* resource, which is simply the market price minus the marginal harvesting cost, or the scarcity rent. Of course, enforcement of quota rights is a necessary condition for quota prices to reflect scarcity rent. Agents with low costs will bid more for quota; likewise, if quota is tradable, low cost 'firms' will buy quota from high cost ones, thereby making everyone better off. In equilibrium, the price of transferable quota is equal to the resource rent (Anderson 1995).

The allocation of quota can be used as a policy tool. Quota can be auctioned each year to the highest bidders, thereby earning rents that the government can use to monitor and enforce the scheme and fund wildlife management and habitat protection programs. Revenue can also be used to reduce tax distortions elsewhere, or finance the provision of other public goods. Quota can also be allocated to local communities that can then sell the quota, harvest specimens themselves or protect them from harvest (perhaps to enhance tourism). In this case, the local communities have a greater interest in wildlife management than if they are left out of the system entirely.

To sum up, when backed by sufficient enforcement and monitoring effort, economic instruments (and C&C measures alike) are capable to contribute to the conservation of wildlife by restricting harvest effort. In theory, an ‘optimal’ or efficient level of harvesting effort can be implemented by appropriate choice of regulatory stringency. However, only some economic instruments (notably taxes and tradable quota) are able to maximize the resource rent associated with harvesting.

3.2 Case study: Commercial use of The Vicuña

The vicuña (*Vicugna vicugna*) is one of the South American camelids along with the guanaco (*Lama guanicoe*), the llama (*Lama glama*) and the alpaca (*Lama pacos*). While vicuña and guanacos are wild, the llamas and alpacas are their domesticated counterpart, a process of selection that appears to have started between 7,000 and 6,000 years ago. The vicuña inhabits the Andean highlands, between 3,000 and 4,600 m. Its range currently extends over large areas of Perú (80,000), north of Argentina (23,000) and Chile (25,000), and west of Bolivia (12,000).

Hunted for their precious wool, which is the one of the finest in the world, the vicuña was near to extinction by the late 1960s. With the European invasion, a trade in fibre was developed, involving the killing of the animal. The few attempts to regulate the use of vicuñas up to this century failed and uncontrolled hunting continued until the species reached near extinction, with just an estimated 10,000 individuals left in the 1950's.

Vicuña wool has been long praised for its softness and fineness. Its current scarcity also adds to the high prices commanded by the few items traded internationally. Vicuña wool (or rather fleece) is regarded as a luxury fibre along with Alpaca, Angora, Cashmere, Camel hair, Mohair, Musk Ox and, Yak, which are noted for their fineness, scarcity, unique appearance and status. It is a very exclusive market, with production of all luxury fibres representing less than 3% of annual sheepwool production by weight. Vicuña is considered the finest and rarest of all, and its softness and colour are highly valued, commanding the highest prices. Archaeological findings and ethnic history archives indicate three distinct phases of interaction between the vicuña and human populations. In a first stage, in the Arcaic era of the Central Andes (7,000–2,000 B.C.), human population in the highlands was significant and the vicuña was popular prey of the highland hunters (Hurtado 1987).

In the second stage, from the late Arcaic area to the advent of agriculture, in parallel, hunting and livestock rearing took place, with hunting being reduced in importance. Llamas and alpacas, both domesticated species, provided food, wool and fuel, and the llama could be used to carry loads. A more complex sociopolitical system emerged and the hunting of vicuña was banned for religious reasons. Wool was still obtained, although this was done through a management system imposed from political authorities. A live capture technique called chaku was used because it allowed the shearing and release of the animal with little impact on the population. These practices were clearly directed to the conservation and sustainable use of resources, where the vicuña wool was only used for special robes for the nobles and royals (Hurtado 1987).

The system, however, was affected by the European invasion, giving way to a third phase where the planned chaku was gradually dismantled and hunting of vicuñas increased, coupled with a regional land struggle between native communities and the Europeans. The few attempts to regulate the use of vicuñas failed (Hurtado 1987) and uncontrolled hunting continued until significant control measures were set in place in the 1950s by which time the population was nearing extinction, with an estimated 10,000 individuals left (Torres 1992).

Conservation efforts to protect the vicuña started in Perú in 1969, with the creation of the Pampa Galeras National Reserve. Subsequently, range states have coordinated conservation efforts through several agreements. In 1969 the first agreement for the protection of the *vicuña* was signed. Peru and Bolivia signed in 1969, with Argentina joining in 1971 and Chile in 1972. The agreement banned all international and internal trade in vicuña products and prohibited the export of fertile individuals to third parties. The vicuña was also listed in Appendix I of CITES in 1975, ratified by all range states and banning all international trade in the species.

These coordination efforts for conservation at the international level created a strong base for cooperation among range states. As a result, the *vicuña* experienced an impressive recovery during the last 30 years, particularly in Perú. From an estimated 6,000 over the four range countries in 1965, the vicuña reached 10,000 by 1970, 101,215 in 1983 and around 154,000 by 1992. Management areas have also increased from 248,000ha in 1965 to more than 7,289,896 ha in 1982 to some 20,800,000 ha currently under protection status. Conservation efforts have been particularly successful in Peru and Chile, where population levels increased significantly during the early years. However, financial and physical requirements to effectively protect those areas have not grown at the same rate.

Although vicuñas have natural predators such as pumas and foxes, the most important limiting factors appear to be poaching by humans and the availability of food, for which they compete with other livestock, like llamas and alpacas. It is the first factor which motivated governments to protect the species and ban all use; it is the second, however, which has caused most social conflict as communities resent the competition of the vicuña for the scarce bofedales in the highlands. Studies in Chile suggest that the vicuña population has reached the carrying capacity of the habitat (given the existing livestock densities), which would account for the oscillating pattern in the population levels registered since 1990 (Torres and Nuñez 1994). Every hundred vicuñas in the highlands eat the same quantity of food as 75 alpacas, or 61 llamas or 72 sheep. The total stock of domesticated livestock in the management zones, is estimated to be the equivalent of 51,864 heads of vicuña. The 21,620.2ha of bofedal available in the management areas of the Parinacota Province, is therefore estimated to be capable of supporting 25,969 vicuñas in the management areas of the Parinacota Province. However, in 1992 the vicuña population in the area was estimated in 26,144, indicating that it is at carrying capacity and is in competition with domestic livestock for food.

At the regional level, some areas show significant overstocking, as in the Lauca National Park. As populations recovered, the competition over habitat with domestic livestock (llamas and alpacas) increased, this being one of the factors behind the increase in poaching. These factors made the involvement of the local communities essential for the long term protection of the species. One way to create incentives for conservation and protection of the vicuña at the local level was to reopen trade in *vicuña* wool, which can be extracted by shearing live *vicuña*s with little impact on wild populations, and generating revenue for local communities. This was the philosophy behind the second *vicuña* agreement in 1979. It provided a use-based rationale for the local communities to become interested in the conservation of the vicuna.

In 1987, Vicuña populations in the Laguna Blanca Reserve (Catamarca province) were examined to assess their potential contribution to the indigenous peasant economy. This is primarily a subsistence economy, with a small but increasing involvement in the market economy. The two main sources of income are from sheep and llama spun wool. The potential harvest of the Vicuña population was estimated using simulation techniques, calculating the maximum sustainable yield and the carrying capacity of the area (Rabinovich *et al.*, 1991). If the Vicuña population were allowed to grow from its current size of 5,000 animals to around 8,000, 15.2% of that population could be harvested each year. The monetary value of each Vicuña is estimated at US$64: $19 for the wool, $10 for the meat (assuming a 20kg animal fetches $0.50 per kg) and $35 for the hide. The estimated total income that could be derived from sustainable management of the Vicuña is US$94,464 per year. This would provide an annual household income to the peasant community of the Laguna Blanca Reserve of almost US$1,000 if equally distributed among the 95 families. This would clearly provide an incentive for these peasant farmers to share their lands with the wild vicuna.

To sum up, strict conservation through use restrictions, when properly enforced, will conserve wildlife. However, in the long run strict conservation can undermine the stated objectives. Vicuñas and livestock compete for forage, and use restrictions on the former remove the incentives of peasant farmers to share their land with Vicuñas. Establishing property rights in land and wildlife provides such an incentive and therefore represents a major step towards sustainable development. Whether tradable quota (an EI) or non-tradable quota (a form of command and control) are implemented to regulate management is of secondary importance. A main point of this study is that establishment of property rights is important, and EIs or government regulation will contribute little more to the protection of species.

3.3 Instruments and habitat conversion

The problem of wildlife conservation is intimately related to the protection of wildlife habitat, which implies that it is bound up in land use and land ownership. In the previous section, we examined economic instruments and incentives related to the harvest of wildlife. In this section, we consider wildlife habitat and land use. Of course, property rights to wildlife, regulations concerning take and incentives to ensure that wildlife are not over harvested affect the value of land. That is, any harvest and wildlife protection policies that increase the value of wildlife might increase the value of land in habitat.

Economists usually consider distributional issues of secondary importance. The focus is generally on maximizing social surplus, and whether that surplus accrues to the regulator or private agents typically matters less. In this section, however, we argue that distribution may be of the utmost importance for the case of wildlife conservation. The reason is as follows. In any economy, there are agents (private or public) that decide about land use. Supposedly such agents compare the present values of net returns from alternative land uses – they compare the returns of habitat conservation and sustainable resource management to those of agricultural conversion. When intervention lowers the decision maker’s returns to habitat conservation and resource harvesting, it becomes more likely that habitat will make place for other uses of the land.

Above we established that taxing, auctioned quota, subsidies and grandfathered quota are equally efficient in restricting harvest effort. However, as mentioned, there is a distributional difference. Taxing and auctions imply resource rents for the regulator, whereas subsidies and grandfathered quota imply rents for the harvester. This translates into different incentives to conserve habitat.

Often landowners have little incentive to protect wildlife habitat because the value of land in habitat for agricultural producers and foresters may be very small or non-existent. As noted earlier, wildlife and wildlife habitat are a public good and private landowners have little if any incentive to protect wildlife habitat on their land. Indeed, as the enactment of the Endangered Species Act in the United States has demonstrated and as we argue further below, landowners may have every incentive to do the opposite – convert habitat to crops. Therefore, economic instruments are required to ‘encourage’ landowners to protect wildlife habitat.

In many political jurisdictions, rural land continues to be largely publicly owned, or, if not owned outright, agricultural and other users of rural land have ill-defined or weak property rights. Peasants lack property rights to wildlife and often gain the right to land only by actively farming it. Even productive forestland might be sacrificed and wildlife habitat lost because peasants cannot demonstrate ownership of land unless they ‘develop’ it – that is, conduct cropping or grazing activities – and this inevitably results in conflicts with wildlife. The appropriate assignment (and protection) of property rights to undeveloped land that also serves as wildlife habitat might encourage peasants not to develop it; peasant landowners might earn a living through sustainable, small-scale forestry and/or harvest of non-timber forest products, including wildlife if they are given a right to animals on their property.

However, this would require a change in the way most developing countries allocate land and other property rights. Moreover, and importantly, the earnings from habitat and wildlife exploitation must exceed that of agriculture. While government policies related to wildlife (and forestry) can affect returns, it can be the case that such activities cannot compete with cropping, even supposing that the ‘correct’ institutions were in place to enable landowners the rights to all the products produced on their land. When the social benefits of habitat conversion exceed the social benefits of conservation (including international positive external effects), economists recommend conversion of natural lands into alternative uses.

The most interesting case exists where habitat conservation “does not pay” from a private perspective, but would be optimal from a social (global) perspective. In other words, when the positive external effects associated with conservation of habitat and wildlife are sufficiently great to topple the balance from conversion to conservation. In this case economic instruments can be used to encourage private landowners or land users to take into account the negative external effects of their land-use decisions on wildlife. What instruments might be employed that directly affect land management?

Regulation

Regulations specify what landowners can and cannot do on their land. The Endangered Species Act is an example of regulation in that it prohibits destruction of the habitat of wildlife on private land. Regulatory approaches often entail expensive monitoring and enforcement, and can still be ineffective if social norms and formal rules do not coincide (Ostrom 1990; Nielson 2003). In fact, it is possible that regulations may lead to perverse incentives that discourage conservation (‘shoot, shovel, and shut up’) if restrictions on established property right owners are onerous (Polasky 2001).

Taxes and subsidies

Tax incentives can be designed to give farmers an incentive to protect wildlife habitat on farmland. However, evidence from developing countries indicates that tax policies are not, by themselves, capable of compensating rural landowners for providing a public good (wildlife habitat) at private expense. As evidence has accumulated that preferential tax assessments do more to subsidize farmland owners than to conserve farmland, governments have increasingly initiated programs to purchase development rights and conservation easements (Wiebe et al. 1996). These programs involve separating and purchasing some but not all of an owner’s rights to a property: separated rights might include, for example, the right to build residential or commercial buildings, to drain sloughs, to burn associated uplands, or to remove endangered species of trees. In the United States, most purchases have been in the form of agricultural conservation easements that restrict residential, commercial or industrial uses, but that allow active farming (Hardie et al. 2004).

Subsidies are perhaps better than tax incentives for protecting nature on agricultural lands. In developed countries, subsidies are used to take land out of production, keep extant wetlands or other critical wildlife habitat from being converted to agriculture, or establish wildlife habitat through tree planting, plant of dense nesting cover for migratory waterfowl, et cetera. Similar programs can be used in developing countries, although financing such programs will pose a greater challenge and likely prevent them from being implemented.

The subsidy approach most-often mentioned in the literature is that of compensating farmers for losses from wildlife depredation. While not providing incentives to prevent legal and illegal taking of wildlife, compensation may at least reduce the incentives of local peasants to go out and destroy wildlife to prevent the agricultural damage that they may cause. On the other hand, wildlife damage programs may encourage additional conversion of habitat into cropland as they essentially amount to a subsidy to agriculture (Rondeau and Bulte 2003).

Finally, one way to arrive at a globally optimal level of habitat conversion is through subsidies at the international level. Fair compensation for positive external effects of conservation implies a transfer flow from North to South. While some of this could presumably be arranged through NGO involvement (see below) and current opportunities provided by the Global Environmental Facility (GEF), it is an open question whether this is enough to safeguard sufficiently large areas of nature in the long run. The public good characteristics of nature conservation, and the implied incentives to free ride on other's efforts, could mean additional, cooperative efforts, should be undertaken. One can think of large-scale programs to finance the provision of ecological services (such as now pioneered in Costa Rica), funded through taxation in the North.

Transfer of development rights

Transferable development rights and wildlife habitat banking constitute cases where separation of development rights can be integrated with land use planning. Wildlife habitat banking (WHB) allows landowners to develop wildlife habitat on their property if they have sufficient credits from investment in the completed rehabilitation of a WHB site. Land use planning enters this program through the designation of the WHB sites (see Fernandez and Karp 1998). Sites can be chosen that provide large high-quality habitats with superior potential to sustain desired ecosystems. Given good choices, the investments in the WHB can provide greater community-wide environmental benefits than equivalent investments in the maintenance of habitat on sites that are being developed. Good planning is crucial to obtain higher benefits, because WHB is a 'no net loss' program that links area restored to wildlife habitat area removed by conversion of habitat to agriculture (Hardie et al. 2004).

An important difference between preferential tax assessments and purchase of development rights is the potential role of planning. Preferential tax assessments are typically extended to all eligible landowners regardless of the location of their property. However, purchases can be targeted to sites where the social or environmental benefits are deemed to be particularly high, such as along a wildlife corridor or within a region under particular agricultural pressure. While the potential for targeting exists, it generally is not realized (see Hardie et al. 2004). Zoning-based transferable development right (TDR) programs are initiated by dividing an area that is being opened for agricultural conversion, or one that has already been converted, into a zone where agricultural development is permitted and one where agriculture is limited or prohibited entirely, thereby protecting crucial habitat. The government partially takes private property rights in ‘down-zoned’ area in order to protect an environmental amenity – wildlife habitat (see Johnston and Madison 1997; Hardie et al. 2004). When the down-zoning occurs, landowners in the affected (source) areas are granted the option to sell the separated development rights to landowners in designated agricultural development (‘up-zoned’) areas or sinks. It is the owners of property in the up-zoned or target areas that must purchase the transferable development rights in order to be able to farm their land. Landowners who lose property rights are compensated in a development rights market, but at rates driven by the opportunity costs created by zoning instead of by willingness to pay for cropland. Of course, governments incur costs of planning and administration of such a TDR program, and the TDR system is only meant to make the separated zoning politically palatable. It is unlikely that this type of instrument will work to protect wildlife habitat in developing countries unless property rights of all kinds are made stronger (see section 4).

One variant that might work in areas where forest concessionaires are active is to require the forest companies to purchase TDRs from landowners who have been down zoned. That is, a forest concessionaire would be required to purchase a certain number of TDRs that protect wildlife habitat in exchange for the right to harvest a certain volume of timber.

Direct purchase of conservation easements to protect wildlife habitat also constitutes a form of property rights purchase. In this case, the state simply purchases the right to develop land for agriculture from the landowner. Since this might be too costly for many developing countries, one alternative is to permit NGOs (or even foreign governments) to purchase these rights, as indicated above. Like the case of TDRs, this option requires that economic institutions exist so that development rights can be separated from ownership of land (and that ownership of land is well defined and protected by the courts – see section 4 below). There may also be opposition to the idea of selling development rights to foreigners, whether foreign governments or NGOs.

The problem is that there is no guarantee that earnings from (perhaps marginal) agricultural land are sufficient to enable compensation to take place. If this is the case, illegal conversion of all land capable of producing crops will still occur. It is not a simple matter to construct a land protection scheme that includes restrictions on land use with transferable development rights to compensate losers (see van Kooten 1993). In developing countries, the obstacles standing in the way of implementing such a scheme may again be too large to surmount.

Transferring income through NGO involvement

The private sector might also be relied upon to a greater extent than currently. Environmental NGOs are perhaps the best means for transferring wildlife conservation funds from rich to poor countries. Nonprofit private land trusts, such as the Nature Conservancy, The Conservation Fund and the Trust for Public Land have become active in the conservation of open space and wildlands in the United States (see Hardie et al. 2004). These organizations purchase properties or easements on lands that provide environmental benefits (such as wildlife habitat) and seek to protect land slated for urban development. Purchased land may be turned over to state and/or local governments, but might be managed by the NGOs in order to guarantee that contributors in developed countries receive the non-market amenity values purchased in developing countries where the record of government management of public lands is perhaps not as good. Of course, for this option to work, it is important that property rights are clearly delineated and protected by the courts in the developing countries. NGOs are unlikely to purchase property or wildlife easements on land if these property rights are non-enforceable.

Kontoleon and Swanson (2003) have shown that, in the context of giant panda preservation (in the Wolong reserve, China), the non-use values associated with panda conservation in the “wild” are sufficiently large to warrant setting aside extensive stretches of land as a reserve – such that not only the flagship ‘panda’, but many other species as well can be supported. However, when such elusive non-use values are not backed up by true transfer flows, it will be in the interest of local people to allocate the land to other uses. Capturing and channeling non-use values through international transfer payments, perhaps actual purchase or lease of land by environmental NGOs, may be one good means to protect species.

3.4 Summary

In this section we, again, demonstrate that defining property rights and benefit-sharing programs are vital in promoting conservation of wildlife. We show that EIs are in theory capable of maximizing resource rents, but argue that their main role could be in promoting habitat conservation. There are various EIs that can be used to make sure that habitat conservation occurs at the lowest cost (tradable development rights, habitat conversion taxes). Equally important, to our opinion, will be the use of international EIs that capture and channel nonuse values from North to South, and to promote habitat conservation through transfers and subsidies.

4. IMPLEMENTING ECONOMIC INSTRUMENTS TO PROTECT WILDLIFE

What is the scope for adopting EIs in developing countries to promote conservation of wildlife and enable a transition towards sustainable development? We argue that the perspective is mixed. EIs are not a panacea, and it is an open question whether they can be effectively employed in all contexts. Institutions and social capital are important if economic incentives are to be used to manage and protect wildlife populations. For example, in their review of emissions trading, Tietenberg et al. (1998) indicate that it is impossible to institute any system of emissions trading unless the requisite legal and other institutions are in place for monitoring, measuring, certifying and enforcing trades, and that lack of appropriate institutions is probably the most important obstacle to the use of market incentives for addressing climate change. For a democratic market economy to function properly, or for market-oriented economic policies to have effect, three criteria or factors other than markets and private property are required (Fukuyama 2002). These criteria relate to economic institutions, the role of the state, and culture.

While a full-fledged analysis of these issues is far beyond the scope of the current study, we would like to note that it is by no means guaranteed that the current state of economic institutions (be it formal or informal) and governments in many resource-rich countries is sufficient to exploit the gains from employing EIs. This can be illustrated for the case of elephant harvesting and ivory trade. In Table 3 we summarize key institutional indicators for (i) OECD countries, (ii) Asian consuming states, and (iii) main ivory producers. An examination of the Table suggests that the prospects of implementing EIs in producer states are not promising. By all measures, range states are the least capable of preventing illegal harvests and sales of ivory. They lack the required economic institutions (courts, rule of law, government effectiveness) and social capital (control of corruption) for enforcing and policing ivory trade. Establishing the infrastructure to guide successful implementation of EIs comes at a cost that is unknown.

Table 3: Measures of the Effectiveness of Economic Institutions and Levels of Social Capital in Industrial Countries, Ivory Importing States and Elephant Range States, 2000-2001

Measure	Eight Industrialized Countries	Five Major Asian Buyer States	30 Range States (Africa & Asia)
Voice & Accountability	1.453	0.106	-0.563
Political Stability	1.275	0.971	-0.801
Government Effectiveness	1.586	1.048	-0.625
Regulatory Quality	1.165	0.899	-0.337
Rule of Law	1.628	1.073	-0.516
Control of Corruption	1.878	0.946	-0.524

Source: World Bank (2002) and calculation

5. CONCLUSIONS AND POLICY RECOMMENDATIONS

Economic instruments have great potential to address spillovers associated with wildlife management. Economic incentives appear particularly useful for the following reasons. First, they are theoretically able to achieve objectives at the lowest cost. They encourage efficient use of resources and therefore have the smallest impact on economic growth and development. The flexibility, efficiency and cost-effectiveness associated with the use of

EIs is presumably nowhere more important than in developing countries. Second, instruments such as auctioned tradable quota and taxes are capable of generating government revenues, enabling the government to provide a variety of public goods. In countries where the scope for raising revenues is small because of limited administrative capacity, this effect could be important. The administrative requirements of EIs are different – for example, raising revenues through auctioning off trophy quotas is much easier than raising funds through taxing households involved in harvesting of a species. Third, compared to command and control measures, the information requirements of EIs are modest. C&C requires the planner to make decisions that allocate resources across activities. In contrast, by simply providing a setting or context, "the market" will take care of an efficient allocation of resources when EIs are used. EIs have lower institutional and human resource requirements than C&C – an important advantage in an information-sparse environment.

In spite of these advantages, EIs are not a panacea that can be straightforwardly implemented across the board. The following critical comments are a useful reminder of the key restrictions. First off, and perhaps obviously, *EIs are not a substitute for monitoring and enforcement*. Conservation of wildlife will critically depend on these activities, regardless of whether C&C or EIs are used to allocate resources. (However, as argued in section 2, public authorities may be able to shift the burden of enforcement and monitoring onto private agents if property rights to resources are defined in either a physical or legal sense.) Without effective monitoring and enforcement, economic incentives cannot work – prices paid for tradable quota will be too low and could approach zero (as harvesting without quota is also feasible) and tax evasion will occur on a large scale. One potential advantage of some EIs is that they can generate the resources that are required to support the enforcement effort that is needed to enable the EIs to work. However, there may be circumstances where adequate enforcement and monitoring is too costly. In that case, it may be optimal to opt for the private optimum rather than the social one; that is, establish property rights to wildlife and ignore spillover benefits associated with conservation. While harvest levels will be "too large" in the short run, such that species stocks will be "too low" eventually, the costs associated with this imperfection may be small compared to the costs of achieving the first best outcome.

Next, a key obstacle to the implementation of EIs is the availability of economic institutions and social capital in many wildlife-rich countries. As of yet it is unclear what "minimum" level of institutional infrastructure is necessary to successfully implement EIs, and how this minimum level compares to actual scores in key countries. Nevertheless, it seems plausible to argue that many countries are currently not up to the task to implement and guide a full-fledged tradable quota or tax scheme to regulate the use of wildlife species.

This is not to say that nothing can be done to greatly improve the efficiency and sustainability of wildlife harvesting. While full-fledged implementation of EIs is cumbersome and expensive, it is often possible to make substantial improvement by making small steps forward. Specifically, by defining and protecting property rights to land and wildlife (be it at the level of the individual or a well-defined group of users, depending on the context), resource harvesters will be able to reach the privately optimal level of resource harvesting and conservation. While inferior to the socially optimal level of harvesting and conservation, it arguably represents a significant improvement over the unregulated open access outcome that eventuates when property rights do not exist. To complement the management scheme, other instruments can be applied after property rights have been established. This would internalize any external effects. However, whether making this additional step is warranted from a cost-benefit perspective is something that has to be assessed on a case-by case level.

We believe, but have not analyzed, that the scope for using complementary EIs in regulating harvest levels may be modest (but there clearly will be cases where this is not true and where the gains from implementing EIs to regulate harvesting are large). The efficiency gains from EIs may be modest, for example, because harvesting technologies are sometimes fairly homogenous, suggesting little scope for gains from trade. The greatest perspective for implementing EIs, we believe, is with respect to land use and habitat conversion. Specifically, it seems advisable to closely consider the scope of implementing an international transfer system from North to South to compensate for transboundary spillover benefits from conservation. Current transfer systems are rather ad hoc, and certainly incomplete. Whether the political will exists in the North to fund such an effort, and whether the institutional capacity exists in the South to manage the subsidy flows, are relevant matters that must be faced.

To reiterate our caveat from the introduction: the current study has been written under great time pressure, and is therefore certainly incomplete, possibly even in key respects. We would therefore like to suggest some useful directions for follow-up research, aimed at closing the gap between what is known today and what needs to be known before EIs can be usefully implemented.

1. Gains from implementing EIs: what are the potential efficiency gains from tax or tradable quota systems? How much heterogeneity exists among harvesters for key wildlife species (say: crocodiles)? How do the gains from complementary instruments compare to the gains from securing property rights?

2. Costs of implementing EIs: What are minimum institutional requirements to successfully implement EIs? Do many countries currently have the ability in terms of social capital and institutions to do so (and if not, what are the associated costs of establishing such an institutional infrastructure)?

3. Costs and benefits: how do the costs and benefits of implementing a system of EIs compare for key wildlife species? Are there general lessons to be drawn?

4. What are the prospects for using EIs in global trade in species? How does a wildlife quota system work across countries? What institutions are required and how many countries satisfy institutional requirements for implementing EIs? How do dynamics affect the usual "optimal tariff" or "optimal quota" result?

5. How important are international positive effects of wildlife conservation at the margin (compared to domestic benefits of sustainable wildlife management)? What is the

scope for capturing such benefits to promote habitat conservation in the South, and how should this be organized?

6. Operational issues: how can one define and allocate property rights, and how can one implement a tax or tradable quota scheme? How high should the tax be (or how large the total allowable catch) in light of many real-life uncertainties? Are there many parallels with ITQ experiences in fisheries in developed countries, and if so: how can we exploit them? How much income should be allocated to wildlife management? Is there a role for eco-labeling?

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