Chapter 5

Elephant Economics, Ivory Trade and Poaching[1]

Rasmus Heltberg

[1] I wish to thank Edward Barbier, E.J. Milner-Gulland, Finn Tarp and Joachim von Braun for helpful comments on previous drafts of this paper. The usual disclaimer applies.

Abstract

The subject of this article is elephant poaching. After a brief review of the elephant economics literature, a simple model for open access elephant hunting is set up and used to derive empirical elasticity estimates, based on parameter values from Zambia. In order to analyse the likely impact of CITES' ivory trade ban, another model is formulated for world ivory demand and supply. It is shown that, when enforcement is imperfect, the effect of a trade ban on poaching is ambiguous. This is because under a ban legal ivory supplies and confiscations cannot be marketed, which may increase illegal market prices and poaching. However, when the model is evaluated empirically, it is shown that for a plausible range of parameter values, the ivory trade ban is likely to decrease poaching due to the moral effect of a ban in reducing demand.

1. Causes of African elephant decline

The African elephant population declined dramatically during the 1980s. It is estimated that the population of this important species fell from 1,2 million in 1981 to 600,000 in 1989 (ITRG, 1989). Though it is nowhere near extinction, the situation of the African elephant is cause for concern. The most important reason for the decline is illegal killings of elephants to supply the international market for ivory, at least in the short run. In the long run, land use changes - expansion of agriculture and livestock - are resulting in decreasing and fragmented elephant habitats.

The Convention for International Trade in Endangered Species (CITES) responded by adopting a trade ban in 1989 which prohibited all commercial international trade in elephant products. The CITES ivory trade ban has received substantial attention in international media. It is surrounded by considerable controversy (Concar and Cole, 1992; Favre, 1993). Countries in Southern Africa, whose elephant populations are well-managed and growing, lobbied for a partial lifting of the ban. In 1997, CITES granted limited quotas for Botswana, Zimbabwe and Namibia to export ivory to Japan.

Environmentalists claim that the ban is necessary to curb an unsustainable level of elephant offtake (EIA, 1992) and argue that managed trade is an unrealistic option. Prior to 1989 issuance of CITES ivory export permits was poorly controlled, and smuggled ivory tended to find its way into the legal trade through various loopholes in trade regulations. Opposed to this, the arguments in favour of allowing ivory exports are, first, that Southern African elephant populations are increasing and that culling is necessary for ecological reasons[1], and that ivory from culling programs should be marketed to generate funds for conservation purposes. Second, by lowering returns to wildlife the ban may reduce incentives to invest in wildlife protection and law enforcement (Swanson, 1993; Khanna and Harford, 1996). In Zambia, for example, prior to the ban law enforcement patrols recovered about 40% of operating costs by selling seized ivory (Leader-Williams et al. 1990). Third, the CITES trade ban does not address the long-run causes of elephant habitat decline, and efforts to involve local people in wildlife management through revenue sharing are adversely affected by the ban (Pitman, 1992), as discussed in the following.

Elephants and humans compete for the same scarce water and land resources, and this conflict has intensified in recent years as a growing number of people claim bush for agriculture and livestock production. The elephant is extremely demanding in terms of land area, water and vegetable material, and roaming elephants damage crops and occasionally cause loss of human life. Therefore, continued human population expansion in Africa is bound to conflict with present elephant stocks. In biological terms, this process amounts to competitive exclusion (Parker and Graham, 1989). Trade bans on wildlife products may aggravate this conflict by reducing returns from unconverted land when wildlife products cannot be marketed (Swanson, 1993). The magnitude of induced land conversions depend on the importance of income from consumptive utilisation relative to non-consumptive income, as well as on the proportion of revenue trickling down to local land use decision makers. Income from sales of elephant ivory, hides and meat prior to the CITES ban constituted a significant proportion of wildlife revenue in Zimbabwe (Brown et al. 1993), but less so in most other African countries. Ivory export revenue amounted to US$ 50-60 million for Sub-Saharan Africa as a whole (Barbier et al. 1992). However, the share of elephant revenue accruing to the owners, occupiers and users of elephant rangelands was modest, at best, even prior to the ban. Yet, Zimbabwe marks an exception as its wildlife revenue sharing programs like CAMPFIRE provide villagers with a stake in local wildlife resources (Kiss, 1990). In Zimbabwe, the ban hurts local communities. In Kenya, on the other hand, tourism is the dominating source of wildlife revenue and the share from ivory trade is relatively less important.

Illegal ivory constituted 80% of all ivory traded prior to the CITES ban (ITRG, 1989). It is important to understand the determinants of poaching, especially its links to legal and illegal trade in ivory. The aim of this article is to apply economic logic to better understand how the

CITES ivory trade ban affects elephant poaching. This is done, first, by analysing the incentives for poaching in a simple open access model, using plausible empirical parameter estimates. Second, a model for the international ivory market is formulated incorporating imperfect enforcement, illegal markets and poaching. It is shown that the net effect of a trade ban on poaching theoretically is ambiguous, but evaluating the model with plausible parameter estimates indicates that the trade ban is likely to reduce poaching.

The remainder of this article is organised as follows. In Section 2 the literature on elephant economics is briefly reviewed. In Section 3 a model for poaching under open access and imperfect law enforcement is set up and evaluated using empirical parameter estimates from Zambia. The international market for ivory is modelled in Section 4 focusing on the interaction between trade, ivory prices and poaching, and the impact of a trade ban on poaching is gauged using a range of plausible parameter values. Results and conclusions are discussed in section 5.

2. Review of the elephant economics literature

Elephant economics is an interesting topic both because of the fascination with which most people view this large mammal, and because it demonstrates that economics can supply important arguments to conservation policy discussions traditionally within the spheres of biology and ecology. Elephant economics originated from the contribution by a group of environmental economists to the Ivory Trade Review Group, a panel of experts set down to advise CITES on ivory trade and elephant conservation. While the report generally recommended the ivory trade ban, adopted shortly after in 1989, the economists dissented (ITRG, 1989). The book volume by Barbier et al. (1992) elaborate the position of these economists, basically that an ivory trade ban, by reducing wildlife revenue, may undermine incentives for conservation. The book also documents and analyses ivory trade flows and demand patterns.

Following this, a small and diverse body of elephant economics literature emerged. Milner-Gulland and Leader-Williams (1992) construct a model for poaching incentives under various assumptions about penalty structure and access. They provide detailed and unique data on poachers' costs, ivory prices received and law enforcement based on a Zambian case study. Their work inspires the model presented in Section 3 below. Burton (1994) employs some of the parameter values from Milner-Gulland and Leader-Williams (1992) in a reworking of the elephant population projections of Caughley, Dublin and Parker (1990). He shows that when poaching effort is modelled as a standard open access resource utilisation problem governed by average expected profit, the African elephant will not become extinct, as predicted in the Caughley et al. model, which assumes exponentially growing poaching effort. A contribution by Milner-Gulland (1993) reworks the ivory demand econometrics of Barbier et al. (1992).

Three contributions assess aspects of CITES' ivory trade ban. Anderson (1992) discusses the welfare economics of trade bans, identifying the (uncompensated) welfare losses in ivory producer and consumer countries from a trade ban and the welfare gains accruing to conservationists. Khanna and Harford (1996) use a Nash-Cournot model of optimal enforcement to analyse the incentives for individual states to comply with a trade ban by supplying enforcement effort. They conclude that, for a trade ban to achieve its objective of protecting endangered species, conservationist countries should provide incentives for enforcement in producer countries. That is, monetary compensation for the ban is called for. Finally, Bulte and Kooten (1996) model the effect of the ivory trade ban on optimal elephant stocks, as perceived by a national government, arguing that optimal stocks ultimately guide actual stocks through enforcement decisions.

The present study takes a different approach. It focuses on poaching in an open access context, and as such employs no concept of optimal stock. It is inspired by the literature on regulation of open access resources when enforcement is imperfect, including Sutinen and Andersen (1985) and Anderson and Lee (1986). The main mechanisms through which a trade ban affects poaching are demonstrated in static models of open access exploitation and trade, and the models are quantified using plausible parameter values. The justification for using static models is that under open access exploitation, the equilibrium population level is a function of prices and technology, not the biological growth function. Also, there is evidence to suggest that heavily poached elephant populations may not grow. In any case, the purpose of the empirical estimates is to present quantitative examples exploring the likely orders of magnitude implied by the models. The models are not intended to forecast elephant populations.

3. A model for poaching

In this section, a simple model is formulated to analyse how incentives for poaching are shaped by ivory prices and deterrence in the form of law enforcement. Leader-Williams, Albon and Berry (1990) provide detailed data on the organisation of elephant and rhinoceros poachers, ivory prices received by them, law-enforcement, detection and penalties gathered from anti-poaching patrols in Luangwa Valley, Zambia, covering 1979-85. Milner-Gulland and Leader-Williams (1992) use this data to impose realistic parameter values in a microeconomic model of poaching incentives. The model presented in this section is quite similar, based on the same stylized facts and parameter values, but corrects an important error in their paper. This correction leads to different conclusions, especially with regard to the effect of penalty structure on elephant mortality.

Single gang

Output from hunting is described by a linear function, assuming constant returns to hunting effort

$$q(x) = axB \tag{1}$$

In equation (1), q is output defined as number of tusks, x is hunting time (number of expeditions), B is the hunted elephant population[2] and a, catchability, is a constant. The joint probability of detection and conviction once detected, $\grave{E}$, is assumed proportional to effort

$$\grave{E} = bx, \qquad 0 < b < 1. \tag{2}$$

Once detected and convicted, the penalty is assumed to be increasing in output (*i.e.* the severity of the offence) and to include the confiscation of one trophy (since often some gang members manage to escape arrest with remaining trophies):

$$F = rq + p \tag{3}$$

where r is penalty per unit of output, p is the value of confiscated output and F is total cost to poachers of conviction (in which the cost of trophy confiscation dominates). This formulation assumes that sentences, whether fines or prison, can be expressed in equivalent monetary units (Milner-Gulland and Leader-Williams, 1992; Becker, 1968). The expected loss from getting caught is hence $\grave{E}F = bx[rq+p]$.

The research in Zambia revealed two kinds of poachers. There are small groups of hunters from local village communities hunting mostly for meat and using low-effective weapons. And there are well-organised gangs, employed by urban-based smugglers and middlemen, equipped with effective modern weapons (Leader-Williams et al. 1990). The analysis in the following is focused on the organised gangs, presumably the main suppliers of illegal ivory to international markets. Assume the decision-maker is a middleman/smuggler, who employs the gang and bears the cost of fines by bailing out convicted gang members. Profit maximization for a risk-neutral single firm entails

$$\begin{aligned} Max_x \ E\eth &= pq(x) - wx - c - \grave{E}F \\ &= paxB - wx - c - bx[rq + p] \end{aligned} \tag{4}$$

where *Eð* is expected profit, *w* is variable costs (mostly salary to gang members) per unit of hunting time and *c* is fixed costs associated with organising a gang of poachers, including equipment and training. Optimal short-run hunting input for an individual gang is:

$$x* = \begin{cases} \dfrac{paB-w-bp}{2braB} & if \ paB-w-bp > 0 \\ 0 & otherwise \end{cases} \tag{5}$$

Industry equilibrium

In order to derive conclusions regarding the hunting mortality of entire elephant populations, it is necessary to move beyond the individual gang model by accounting for the reaction of all poachers (‘industry equilibrium’). If individual gangs of poachers over extended periods operate at positive levels of profits, other units are likely to start exploiting this *de facto* open access resource. The error in the paper by Milner-Gulland and Leader-Williams (1992) mentioned above occurs because their analysis intermingles a short-run individual gang situation with long-run open access equilibria. This gives rise to some confusion about the impact of penalty structure. In the long run, gangs will enter into poaching until elephant populations are reduced to a level where expected profit, net of all costs, is zero, that is

$$E\eth = paxB - wx - c - bx[rq + p] = 0 \qquad (6)$$

Inserting the expression for optimal input of hunting effort, (5), the equilibrium (surviving) elephant population can be found implicitly as the *B** that solves

$$\begin{aligned} &[paB^* - w - bp]^2 - 4cbraB^* = 0 \\ s.t. \quad &paB^* - w - b[raB^* + p] \geq 0 \end{aligned} \qquad (7)$$

In other words, under open access poaching will drive elephant populations down to a level where middlemen perceive the expected returns from an additional hunting expedition to equal its costs, including the expected value of penalties. Because of open access, the equilibrium elephant population depends on prices, costs, law enforcement and hunting technology, but not on assumptions about the biological growth function of elephants (Gordon, 1954).

In the case where fixed costs are zero, equation (7) simplifies to

$$B^* = \frac{w + bp}{pa} \qquad (8)$$

which, unfortunately, implies that an infinite number of gangs operate at very low levels of effort each. Assuming instead an integer number of expeditions, and setting $x^*=1$, yields

$$B^* = \frac{w + bp}{pa - bra} \qquad (9)$$

An additional variant of the model assumes fines are constant, i.e $F=r+p$. In this case, the open access equilibrium population (for the case with zero fixed costs) is given as

$$B^* = \frac{w + b[r + p]}{pa} \qquad (10)$$

Empirical estimates

The size of the equilibrium elephant population can be calculated using empirical parameter values provided by Milner-Gulland and Leader-Williams (1992). The elephant population of Luangwa Valley was estimated to be 25323 in 1985. Variable costs of poaching amount to K 503.2 per expedition (all monetary units are in 1985 Zambian Kwacha (K)). The mean price received by poachers is 285 K/kg × 4.8 kg/tusk = 1368 K/tusk, which is also the cost of trophy confiscation when detected. Probability of detection is estimated in two different ways, both giving an estimate around 0.05. The average penalty for a poacher appearing in court is K500, and with four gang members caught, this gives K2000 as the penalty per gang if caught and convicted. Catchability, *a*, is set at 2.26×10^{-4}, corresponding to 3.54 kills per expedition and 1.88 tusks per kill. Milner-Gulland and Leader-Williams do not give any data on fixed costs. In the absence of any plausible estimates, fixed costs are set arbitrarily to 1, 5 and 10 times marginal costs.

Results are presented in Table 1. For the model with variable fine and fixed costs, parameter values imply that, in the long run, poaching will reduce elephant populations by somewhere between 72-89% of their initial 1985 size, depending on the value of the fixed costs. The models with zero fixed costs predict elephant mortality of approximately 93% for both variable and constant fine structure. Thus, once account is taken of entry into the poaching industry, assumptions about the shape of penalty structure do not strongly affect model conclusions. Assumptions about fixed costs have a larger effect on elephant mortality. The findings underscore the severe threat which poaching gangs, equipped with modern weapons and supplying active international markets, pose to all but the best-guarded elephant populations. They raise serious concerns about the effectiveness of prevailing conservation strategies.

The model can be used to calculate empirical elasticities with respect to the different policy variables. Elasticities are expressed in terms of the percentage change in surviving elephant populations for a one percent change in the parameter value, and are measured in the vicinity of the (low) open access equilibrium population level. Elasticities, reported in Table 1, are calculated numerically in the fixed cost models, while the models with zero fixed costs permit reduced-form expressions for elasticities.

Table 1 here

Although the elasticity estimates depend on assumptions about fixed costs and penalty structure, it is interesting that the price of ivory universally has the largest impact on poaching. Depending on assumptions, a 10% increase in price will reduce long-run elephant populations by 9-14% from the open access equilibrium. The variable cost of hunting, *w*, has a large effect on poaching only for low levels of fixed costs. For high levels of fixed costs, the law enforcement variables become more important in elasticity terms. The likelihood of detection and conviction has a larger elasticity than fine because of the deterrence caused by trophy confiscation. Whereas in the models with zero or low fixed costs law enforcement appears of little use in deterring poaching, this conclusion will not hold if fixed costs are relatively high. In order to determine which model specification is best, more data is necessary, including data on poachers' cost structure.

The likelihood of detection is the only variable which is directly under the control of wildlife authorities. It is shaped by patrolling and prosecution, both of which are relatively costly. Penalties are decided by courts, whose practise is not easily changed by other authorities, and in the Zambian case not even by new national legislation (Leader-Williams et al. 1990). Poaching costs *w* depend on salaries in other sectors, and hence are not a policy instrument available to wildlife authorities. Illegal market ivory prices depend on supply and demand for ivory and on international regulation of the ivory trade, including customs effort. Therefore, the repercussion of ivory trade regulations on illegal market prices have important consequences for poaching incentives. No data are available to make cost-benefit assessments of the different policy options. It does seem, though, that regulatory measures that depress illegal ivory prices are likely to be very cost-effective, especially compared to patrolling. This important conclusion motivates the analysis in the following section on the impact of a trade ban on illegal prices and poaching.

4. Global ivory trade, poaching and CITES' trade ban

The subject of this section is the effect of an ivory trade ban on elephant killings. A simple model is presented below that captures the interaction between trade ban, illegal price and supply of ivory to world markets. The model pinpoints a few unintended side effects of the trade ban which tend to be neglected in conservation debates.

A simple general model

Poaching is assumed to depend positively on trophy price. In this section, the influence of law enforcement (patrolling, conviction) on poaching is ignored - it is assumed the likelihood of detection and conviction is unchanged by the ban. The amount of smuggled ivory that can be intercepted in customs is allowed to vary with the imposition of a trade ban, though. Africa-wide poaching offtake is written

$$
\begin{aligned}
Q_0^I &= Q^I(p), \qquad \frac{\partial Q^I}{\partial p} > 0 \qquad (pre-ban) \\
Q_1^I &= Q^I(p[1-t]), \qquad 1 > t \geq 0 \qquad (post-ban)
\end{aligned}
\tag{11}
$$

where Q^I is illegal supply in kilogram of ivory, p is market price and t is the additional share of smuggled ivory that can be intercepted in customs because of the ban; subscripts 0 and 1 are used to distinguish pre-ban from post-ban values. Equation (11) takes account of the fact that a trade ban facilitates customs control by closing the loopholes, including ways through which poached ivory might acquire legal status. Expected price to the smuggler-cum-poacher is therefore p before the ban and $p[1-t]$ after.

The supply of ivory, S, from legal and illegal sources reaching overseas consumer markets is

$$
\begin{aligned}
S_0 &= Q^I + \bar{Q}^L \qquad (pre-ban) \\
S_1 &= [1-t]Q^I \qquad (post-ban)
\end{aligned}
\tag{12}
$$

where Q^L denotes legal ivory production stemming from problem animal control and culling programs, assumed to be unaffected by the ban. Prior to the ban, the range states were able to market all stocks and confiscations. Under CITES' trade ban no ivory whatsoever can be legally marketed. Not even stockpiles originating from confiscations and culling programs. Supply therefore reduces to that part of the illegal component which passes customs, with the remaining ivory piling up in government stocks. Total offtake is

$$
Q = Q^I(\cdot) + \bar{Q}^L \tag{13}
$$

in both cases.

Global demand for unworked ivory is assumed to depend negatively on price and on the imposition of a ban

$$
\begin{aligned}
D_0 &= D_0(p), \qquad \frac{\partial D_0}{\partial p} < 0 \\
D_1 &= D_0(p) - M, \qquad M \geq 0
\end{aligned}
\tag{14}
$$

where M is the demand reduction stemming from a trade ban. This captures the 'moral impact' of a ban; historically, a large part of the international ivory market disappeared in response to the ban and to (western) consumers' concern over the rapid decline of elephant populations. Market equilibrium implies

$$\begin{aligned} D_0(p) &= Q^I(p) + \bar{Q}^L && (pre\text{-}ban) \\ D_0(p) - M &= [1-t]Q^I(p[1-t]), && (post\text{-}ban) \end{aligned} \tag{15}$$

The model is illustrated in Figure 1. Pre-ban market supply $S_0 = Q_0^I + Q^L$ is the sum of illegal and official production. Where it intersects the demand curve D_0 gives the pre-ban price level, and the corresponding poaching level can be found at the Q_0^I-curve at *f*. By increasing confiscations the ban changes the shape of the poaching function, which becomes steeper. Official ivory production as well as that part of smuggled ivory which is intercepted in customs is withheld from consumers. The market supply curve, S_1, therefore now lies to the left of the poaching function. Market price is where the post-ban demand curve, D_1 , which is lower due to the moral effect, *M*, intersects with market supply, S_1. The level of killing corresponding to this price is *g*. This may be more or less than *f* depending on the relative size of the different effects. The figure illustrates the case where ivory offtake is higher under a trade ban.

Figure 1 here

The model is static and, in contrast to the model in Section 3, it ignores the influence of the elephant population level on poaching incentives. It is therefore best to be seen as short-run and valid for a single point in time. It may nevertheless be useful for indicating the direction of change in poaching implied by the CITES ban.

Summing up, the main effects of a trade ban on legal and illegal markets are as follows. One, the moral impact of a trade ban (and associated attention surrounding the plight of the African elephant) is to reduce demand, depressing prices and poaching. Second, official production is withheld from overseas ivory markets. This draws up illegal prices and induces poaching. Third, to the extent a trade ban facilitates additional customs interceptions of smuggled ivory, this has an ambiguous effect on price and poaching incentives. Interceptions reduce smugglers' expected price, and hence the price they offer poachers. However, under a trade ban confiscated goods cannot be marketed and confiscations therefore work to increase ivory prices in consumer countries. Fourth, outside the presented model, a trade ban may negatively affect funding for conservation, including patrolling. The net effect of the different effects of a trade ban cannot be known *a priori*. Yet, the present analysis, by showing that increased killings of elephants during a ban is a theoretical possibility, highlights the need for CITES to consider more carefully the interactions between trade bans and illegal trade in endangered species in cases where enforcement of bans is imperfect.

Implementing the model with a specific functional form

In order to provide a quantitative assessment of the trade ban, the model can be implemented with realistic parameter estimates. This requires choice of a functional form. Cobb-Douglas functions were chosen for ivory demand and poaching supply. The model can be written in compact form as

$$\begin{aligned} D_0 &= kp^{-ã} = S_0 = lp^{ä} + \bar{Q}^{L}, & k,l,ã,ä \geq 0 \\ D_1 &= kp^{-ã} - mD_0 = S_1 = [1-t]l(p[1-t])^{ä}, & 1 > t,m \geq 0. \end{aligned} \qquad (16)$$

where, in addition to the already introduced variables, *ã* and *ä* are the price elasticities of demand and poaching supply, *k* and *l* are constants and *m* is the share of pre-ban demand that falls away due to the moral effect.

The model is calibrated using parameters estimated for global ivory trade in the 1979-88 period, where relatively good data exist. From 1989 there is no data available on international ivory trade due to the CITES ban. Choice of parameter values is described in the following. First, an equation for global ivory demand was estimated econometrically, using data for implicit raw ivory prices and net imports by major consumer countries from 1979-88, presented in Barbier et al. (1990; Table 2.2 and 1.3). All values refer to kilogram of ivory and US$ per kilogram. The regression is plotted in Figure 2, giving *ã* = 1.7 and $k = \exp(20.7) = 9.85\times10^{8}$. This demand elasticity appears rather high for a luxury good such as ivory, so complimentary sensitivity analysis was carried out with a lower *ã*, as explained below.

Figure 2 here

Second, for supply, Q^L was fixed at 20% of the 1979-87 mean ivory trade flow, or 161,500 kg, in accordance with ITRG (1989) estimates. For poachers' supply elasticity, *ä* = 0.9, in the lower range of the values calculated in Section 3, was used. No good estimate exist, however, for the constant in the supply equation, *l*, as well as for the moral demand-reduction effect, *m*, and for additional interceptions, *t*. This problem was addressed in the following manner.

The constant term, l, was used to calibrate the model to historical trade flows. That is, l is adjusted so that at the mean 1979-87 price, predicted supply equals predicted demand.

The lack of estimates for m and t make it impossible to use the model to produce a plausible 'forecast' or point estimate for the level of post-ban poaching. If, for example, $(m,t) = (0.1, 0.1)$ the model predicts a poaching increase of 1.2%, while for $(m,t) = (0.5, 0.4)$ the model implies 33% less poaching. Clearly, the model is very sensitive to choice of parameter values for m and t. Because these parameters relate to illegal transactions, there is no obvious way to estimate them. However, the model can be used to chart 'iso-poaching' curves, that is, combinations of (m,t) that imply same, more and less poaching than before the ban. Through this kind of sensitivity analysis, it is possible to explore the implications of the model and to pinpoint the key factors necessary for a trade ban to successfully curb poaching.

The results are plotted in Figure 3. The curves plot pairs of (m,t) that imply same level of poaching as in the pre-ban period, 5% more poaching, 20% less poaching and 40% less poaching, respectively. From these 'iso-poaching' curves, it is seen that poaching is decreasing in both m and t. For these parameter values, a trade ban will result in less poaching unless customs interceptions and consumer demand remain largely unaffected by it. If demand falls by more than 20% due to a ban ($m > 0.2$) poaching decreases irrespective of what happens to interceptions. If at least 30% of smuggled goods can be intercepted after a ban ($t \geq 0.3$), poaching will decrease (as long as demand does not grow). Assume, for example, that the ban caused demand to cease altogether in North America and Europe, as evidence indicates. These countries imported around 34% of world ivory production prior to 1988. If $m=0.34$, the model predicts less poaching for the entire range of t. It seems likely that the reaction in many consumer countries, caused by the awareness surrounding the plight of the African elephant, has been to reduce or eliminate demand sufficiently to make a trade ban result in less poaching.[3] The mechanism for this is through lower prices received by poachers.

Figure 3 and 4 and Table 2 here

Table 2 reports elasticity values for the supply and demand parameters, calculated as the percentage increase in post-ban poaching caused by a 1% increase of the parameters. Elasticities are evaluated at two different arbitrary base cases of (m, t), but the choice of base case does not seem to exert a large influence on the estimates. Poaching is decreasing in (the absolute value of) the price elasticity of demand and increasing in the price elasticity of supply as well as in the constants of the demand and supply equations.

As mentioned, the estimate for elasticity of ivory demand, $\tilde{a}$, is relatively uncertain. In order to explore the sensitivity of the results, a different value for $\tilde{a}$ was tried. Barbier et al. (1990; ch 4) estimate a price elasticity of Japanese ivory demand of -0.7. Hence, the model was also run with $\tilde{a} = 0.7$ instead of 1.7. Results are not very different. In this case $(m,t) = (0.1, 0.1)$ increase poaching with 7.8%, and $(m,t) = (0.5, 0.4)$ lead to 24% less poaching. The iso-poaching curves for this parameter configuration are shown in Figure 4. Figure 4 reveals that when demand is inelastic, the level of t matters much less for poaching than m. This result complies with basic economic intuition that for an inelastic demand curve, changes in the supply curve (induced through t), generate relatively small changes in quantities traded. It is still found that a trade ban is likely to reduce poaching. Specifically, it continues to hold that as long as demand reduces by more than 20%, poaching will decrease irrespective of the level of interceptions.

5. Discussion

The purpose of CITES' trade bans on endangered species and products thereof is noble: to protect biodiversity by halting unsustainable exploitation of endangered species. Yet, trade bans are often imperfectly enforced and illegal consumer markets persist, as in the case of ivory, rhino horn, live parrots, orangutans and many other endangered species. One may find such black markets and the poaching that supplies them immoral, but ignoring their role by assuming them away may lead to misguided conservation policies.

In this paper elephant poaching has been analysed within two different models. In the first model, it was shown that ivory price is the most important determinant of poaching, much more so than enforcement. Hence, measures that can reduce the price paid to poachers or increase confiscations are likely to be effective means of halting poaching. This motivated the second model, intended to analyse the likely impact of CITES' trade ban on poaching. It was found to be ambiguous, depending on the magnitude of different effects.

A trade ban is likely to improve species protection if *(i)* it has a large moral demand-reducing effect, *(ii)* it facilitates interception of smuggled goods, *(iii)* there is little ivory from official production piling up, and *(iv)* it does not negatively affect law enforcement effort. Results indicate that, for plausible parameter values, the ivory trade ban is likely to have led to reduced poaching. Of course, all estimates presented in this study should be seen as empirical examples, not as population projections. A proper evaluation of CITES' ivory trade ban would require time-consuming and costly field studies of poaching, illegal markets and elephant population counts.

A key problem with the current CITES trade bans is that official stockpiles cannot be marketed, driving up illegal prices. Dublin, Milliken and Barnes (1994) report that nine African states in 1994 held a total of at least 100 tons of ivory in stock. This represents two-thirds of 1988 world ivory trade. By reducing wildlife-related revenue, trade bans may undermine incentives and budgets available for conservation. The ivory trade ban has led to a drastic reduction in elephant-related revenue in the Southern African range states, negatively affecting their capacity to enforce wildlife legislation (Khanna and Harford, 1996). Hence, unless conservationists provide compensation, trade bans may undermine themselves by eroding the range states' incentives for mounting costly enforcement.

The ivory trade ban has also had beneficial effects, but the question is whether these will persist. It has curbed demand through reduced availability, public awareness and appeal to the moral sense of consumers. Enhanced detection and confiscation of illegal items reduce the returns from poaching and smuggling. The reduction in demand was at its largest around 1989 when western ivory markets collapsed. Barbier *et al*. (1992) predicted that new ivory markets in the Far East gradually would emerge as smugglers moved into these markets. There is evidence to suggest that this has indeed happened (Dublin, Milliken and Barnes, 1994). Also, long established markets for ivory may only temporarily have stopped consumption.

The conclusion is neither that markets for products from threatened natural resources should be liberalised indiscriminately, nor that trade bans should form the core of conservation efforts. Rather, it should be investigated how legal marketing channels for official stocks can be set up and safeguarded, as in the current system with ivory export quotas from Botswana, Zimbabwe and Namibia. Due attention should be paid to developing secure techniques to distinguish legal from illegal supplies, for example through marking systems. The proceeds from ivory sales should be channelled back into wildlife conservation, for example by supplementing Wildlife Service budgets and providing local communities with incentives for wildlife conservation.

Bibliography

Anderson, K. (1992) The Standard Welfare Economics of Policies Affecting Trade and the Environment. Anderson K and Blackhurst R. The Greening of World Trade Issues. Hemel Hempstead. Harvester Wheatsheaf.

Anderson, L.G. and Lee, D.R. (1986) Optimal Governing Instrument, Operation Level, and Enforcement in Natural Resource Regulation: The Case of the Fishery. *American Journal of Agricultural Economics* **68**, 678-90.

Barbier, E.B., Burgess, J.C., Swanson, T.M. and Pearce, D.W. (1992) *Elephants, Economics and Ivory*, London: Earthscan.

Becker, G. (1968) Crime and punishment: an economic approach. *Journal of Political Economy* **76**, 168-217.

Brown, K., Pearce, D.W., Perrings, C. and Swanson, T. (1993) *Economics and the Conservation of Global Biological Diversity*, Washington: GEF Working Paper 2.

Bulte, E.H. and Kooten, G.C. (1996) A Note on Ivory Trade and Elephant Conservation. *Environment and Development Economics* **1**, 433-443.

Burton, M.P. (1994) Alternative projections of decline of the African elephant. *Biological Conservation* **70**, 183-188.

Caughley, G., Dublin, H.T. and Parker, I.S.C. (1990) Projected Decline of the African Elephant. *Biological Conservation* **54**, 157-64.

Concar, D. and Cole, M. (1992) Conservation and the Ivory Tower. *New Scientist* **29 February**, 29-33.

Dublin, H.T., Milliken, T. and Barnes, R.F.W. (1994) *Four years after the CITES ban: illegal killing of elephants, ivory trade and stockpiles*, Nairobi: IUCN/SSC African elephant specialist group.

EIA (1992) *Under Fire: Elephants in the Front Line*, London: The Environmental Investigation Agency.

Favre, D. (1993) Debate within the CITES community: what direction for the future? *Natural Resources Journal* **33(4)**, 875-918.

Gordon, H.S. (1954) The Economic Theory of A Common-Property Resource: The Fishery. *Journal of Political Economy* **62**, 124-42.

ITRG (1989) *The ivory trade and the future of the African elephant* - *vol. 2*, Lausanne: Ivory Trade Review Group, CITES.

Khanna, J. and Harford, J. (1996) The Ivory Trade Ban: Is It Effective? *Ecological Economics* **19**, 147-155.

Kiss, A. (1990) *Living With Wildlife: Wildlife Resource Management With Local Participation in Africa*, Washington: World Bank Technical Paper 130.

Leader-Williams, N., Albon, S.D. and Berry, P.S.M. (1990) Illegal Exploitation of Black Rhinoceros and Elephant Populations: Patterns of Decline, Law Enforcement and Patrol Effort in Luangwa Valley, Zambia. *Journal of Applied Ecology* **27**, 1055-1087.

Leader-Williams, N. (1992) *The World Trade in Rhino Horn: A Review*, Cambridge: TRAFFIC International.

Milner-Gulland, E.J. (1993) An econometric analysis of consumer demand for ivory and rhino horn. *Environmental and Resource Economics* **3**, 73-95.

Milner-Gulland, E.J. and Leader-Williams, N. (1992) A model of incentives for the illegal exploitation of black rhinos and elephants: poaching pays in Luangwa Valley, Zambia. *Journal of Applied Ecology* **29**, 388-401.

Parker, I.S.C. and Graham, A.D. (1989) Elephant decline (part I): Downward trends in African elephant distribution and numbers. *International Journal of Environmental Studies* **34**, 287-305.

Pitman, D. (1992) *Zimbabwes Alternative to the Ivory Bans*, Harare: The Zimbabwe Trust.

Sutinen, J.G. and Andersen, P. (1985) The Economics of Fisheries Law Enforcement. *Land Economics* **61**, 387-397.

Swanson, T. (1993) Regulating Endangered Species. *Economic policy* 185-205.

Endnotes:

1. Zimbabwe claims its approximately 60,000 elephants are more than the land can sustain without serious degradation.

2. This assumes elephant populations are homogenous. In practice, things are more complicated since tusk size varies with age and sex and poachers select animals with large tusks. These complications are ignored for the purpose of this analysis.

3. The model explains the apparent failure of the long-standing (since 1979) CITES trade ban on rhinoceros horn to conserve rhinos, which are very close to extinction. The ban may not have affected consumer demand and interceptions much. First, rhino horn is demanded for its medicinal properties, supposedly fever reduction, in traditional Chinese medicine, making demand respond little to price and moral effects (Leader-Williams, 1992). Second, rhino horn is relatively small, easy to hide and extremely valuable and therefore difficult to intercept. The high price fetched for rhino horn make rhino poaching profitable, even at very low population densities, and especially as a joint activity with elephant poaching (Milner-Gulland and Leader-Williams, 1992).