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Managing critically endangered species: the Sumatran rhino as a case study

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The saving of critically endangered species is costly, and it is likely to conflict with other societal objectives. Methods are needed for clarifying and resolving such conflicts. In this chapter we will discuss an analytical tool called decision analysis (Raiffa, 1968). Decision analysis provides an explicit framework for identifying species in immediate danger of extinction, defining cases that may require intervention, evaluating the risks and benefits of alternate management strategies, and assessing whether or not the management efforts required to prevent a species' extinction can be justified in terms of their costs to society.

Why is an explicit framework needed? Conservation biology is essentially a crisis discipline (Soulé, 1985); neither time nor abundant economic resources are on its side. Difficult choices often must be made, usually in the absence of adequate data. When the outcomes of alternate actions are uncertain, it is hard to anticipate intuitively which one will be best. Furthermore, there are often several criteria for evaluating outcomes, such as minimizing costs versus maximizing protection; one action may seem to be best under the former criterion but a second far more desirable under the latter. Decision analysis provides a means of evaluating alternatives in a logical and repeatable manner; it is also a useful tool for communicating alternate management plans to others so that they can be persuaded to endorse one or more of them.

We will show how probabilities of extinction (pE) can be estimated and how these pE can be used to identify critically endangered species. Then we will suggest management procedures (interventions) that might be used to decrease pE in different situations and show how decision analysis can be used to choose the best alternative when the outcomes of these interventions are uncertain. We will illustrate the use of decision analysis with a specific example: assessing the status of the Sumatran rhinoceros *Dicerorhinus sumatrensis* and evaluating several management alternatives which recently have been proposed to save it from extinction.

Identifying species of concern

For the purpose of this chapter, we will call a species extinct when no breeding pairs remain. We will define the extinction probability for a species as the probability that habitat and/or population trends will result in no breeding pairs within two or three generations. We emphasize that two or three generations is a very short time horizon, and we do not advocate its use except for critically endangered species where options for longer-term management will be lost forever if short-term recovery programs are not successful.

A species' pE may be estimated from empirical studies, from analytical models of population processes, from computer simulations, or from subjective assessments by researchers or managers. In a few cases, empirical data on the loss of species diversity under various habitat and population trends are available; e.g., data on attrition from continental islands and from national parks (Wilcox, 1980).

Analytical models of population processes have been used to estimate pE (e.g., MacArthur and Wilson, 1967; MacArthur, 1972; Richter-Dyn and Goel, 1972; Leigh, 1981), but many of these have severe limitations (Brussard, 1985; Goodman, Chapter 2; Shaffer and Samson, 1985). Analytical models may provide a preliminary assessment of population trends under certain circumstances; examples in this book are those of Goodman (Chapter 2), and Ewens *et al.* (Chapter 4).

Simulation models can also be useful for estimating pE for particular species (e.g., Shaffer and Samson, 1985). These models may include both species-specific biological information (such as estimates of available habitat and potential rate of population increase) and sociological variables pertaining to the local human population. The latter might include estimates of poaching activity, type of agriculture, grazing impacts of domestic animals, and potential for catastrophes, disease, or habitat degradation. Unfortunately, data are scarce for building simulation models for many endangered species.

In many situations estimates of pE must be based on experience with related species and on general knowledge of local environmental and sociopolitical conditions. Methods of eliciting these subjective estimates are available (Behn and Vaupel, 1982; Spetzler and Stael von Holstein, 1975). Once determined, they can be used in the same way as probabilities obtained from empirical data or models.

Some factors influencing pE can be anticipated with appreciable certainty; others are fundamentally uncertain. For example, factors influencing pE for the Javan rhino *Rhinoceros sondaicus* over the next few generations include habitat destruction from timber harvests, increased mortality from poaching, and limitations on the area available for reserves. These can be estimated with some confidence. On the other hand, the impacts on pE of epidemic diseases, severe typhoons, fire, or volcanic eruptions are truly uncertain, but nevertheless important, especially as the range of the species shrinks. The recent outbreak of an undiagnosed fatal disease in Javan rhino populations (Oryx, 1982) and the loss of eight million acres of rhino habitat to fire in Borneo in 1983 (Geo, 1985) illustrate this point.

The expected impact of an event on pE is a function of (1) the probability that the event will occur and (2) the consequences for population survival if it does. For example, if the probability of epidemic disease is 0.1, the probability of extinction (pE), if an epidemic occurs, is 0.95, and pE in the absence of an epidemic is 0.85, then E(pE), the expected value for pE, is:

$$E(pE) = [p(\text{epidemic})][pE(\text{epidemic})] + [p(\text{no epidemic})][pE(\text{no epidemic})] = (0.1)(0.95) + (0.9)(0.85) = 0.86. \quad (1)$$

Once calculated, this value can be used to determine if the probability of extinction for this species is unacceptably high.

'Acceptable' levels of pE vary among taxonomic categories, among social and economic groups in society, and among political entities. In developing countries where human demands on resources are already overwhelming, pEs that are acceptable to local governments may shock visiting scientists from industrial nations.

Criteria are also often biased taxonomically. For example, the US Forest Service is specifically charged with maintaining viable populations of native and desired non-native vertebrates on National Forest lands. There is no mention of invertebrates in current Forest Service regulations, although some, such as the spectacular Nokomis fritillary butterfly *Speyeria nokomis* are in more immediate danger of extinction than are many vertebrates. Similarly, current standards for placing species on official endangered lists emphasize mammals and birds over other taxonomic groups, although there is little biological justification for doing so. In the United States the bald eagle *Haliaeetus leucocephalus*, the national symbol, has received far more attention than other species equally threatened with extinction.

The social assessment of whether or not the pE for a particular species is unacceptably high often is made implicitly through budget decisions or the lobbying efforts of special interest groups. Fortunately, there are explicit procedures for assigning values in situations involving multiple interest groups and conflicting goals (Keeney and Raiffa, 1976) which provide a more deliberate method of evaluating pEs and deciding which species need intervention.

The process of debating alternatives for endangered species management can itself change social and governmental attitudes toward extinction. For example, international meetings to discuss management of the Sumatran rhino increased concern for the fate of this species among all participants, including both Malaysian and Indonesian authorities. Probabilities of extinction that seemed acceptable before the meetings are now considered unacceptable. There is now broad support for a species survival plan involving several governments as well as private conservation groups.

An analysis of Sumatran rhino management

We will illustrate the use of decision analysis in the management of critically endangered species with a case study of the Sumatran rhinoceros, evaluating its current status using probability of extinction. Our estimates of pE are subjective, but represent a synthesis of opinion from rhino biologists and managers. We then will analyze management interventions that might be used to improve the species' status, with a consideration of random events that could influence the outcome of those management strategies. Finally, we will use two criteria for choosing among alternatives: probability of extinction and financial cost. The purpose of the analysis is to identify management plans with the best combination of low probability of extinction and low cost.

Current status

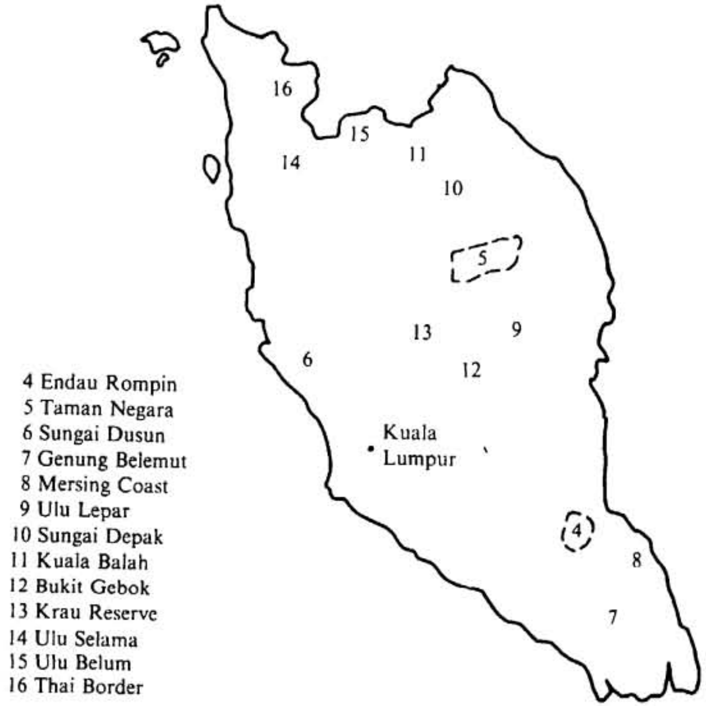
The Sumatran rhino persists in small, isolated subpopulations in increasingly fragmented habitat (Van Strien, 1974; Mittermeier and Konstant, 1982) (Figure 8.1). Unprotected habitat is threatened by timber harvest, human resettlement, and hydroelectric development.

Figure 8.1. (a) Current range of the Sumatran rhinoceros. Numbers indicate areas where one or more rhinos are presumed to persist. Boldfaced numbers indicate populations probably preservable in the wild if actively managed and adequately protected.

a

Figure 8.1 *continued* **(b) Detailed map of West Malaysia. Locality 4 is a 1000 km² reserve, proposed as a national park. Locality 5 is a national park, but under pressure. See text for further details. (c) Detailed map of Sabah.**

b



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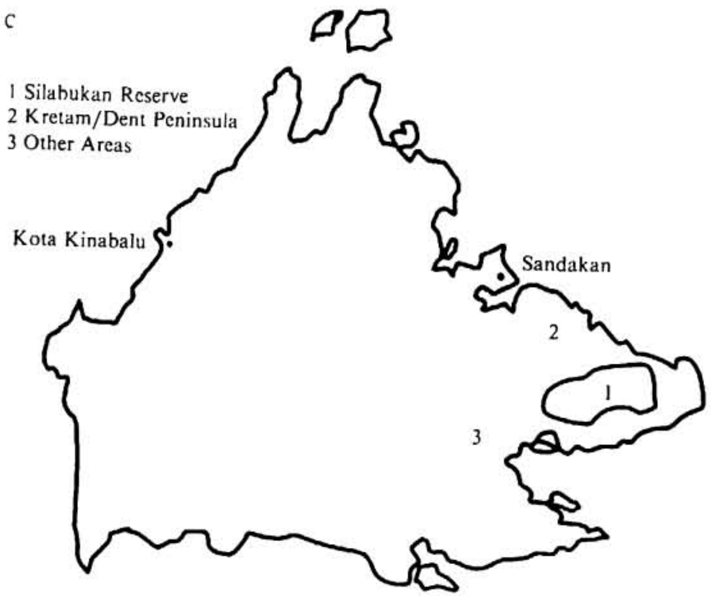


There are only a few designated reserves, and even these are subject to exploitation. Poaching takes a heavy toll, and this pressure can only increase as the human populations rise. Disease, such as the recent epidemic in the Javan rhino population, or a catastrophic storm could wipe out most of the remaining wild populations. Only two Sumatran rhinos are currently held in captivity. The rapidity of both ecological and political change in Southeast Asia argues for quick action if any is to be effective.

Known wild populations of Sumatran rhinos fall under three political jurisdictions: Sabah (East Malaysia), Indonesia, and West Malaysia; small remnant populations also may persist in Thailand and Burma. We evaluated pE over a period of 30 years, which is about two generations for this species. With current management practices, and in the absence of an epidemic, we estimate pE for rhinos in Sabah to be very close to unity (say 0.99). For the Indonesian population the probability of extinction within 30 years is probably about 0.95, only because some isolated subpopulations are in remote and inaccessible terrain. The West Malaysian population is the smallest but best protected, with a pE of about 0.9. Combining these estimates gives the species as a whole a pE of $(0.99)(0.95)(0.9) = 0.85$ within the next three decades. The choice between *status quo* management and intervention is shown on the decision tree in Figure 8.2 as two branches emerging from a square (a convention in decision analysis indicating a decision point).

Many unpredictable or random processes also influence pE and the outcome of management actions. These include natural events, such as disease, unusual weather, and unpredictable human actions, such as changes in government attitudes toward habitat protection and control of poaching. To weigh the impact of these random events on pE, one must estimate their probabilities of occurrence. For some events, objective probabilities expressed as theoretical expectations or long-term frequencies are available; for example, weather records help predict the expected frequency of severe typhoons in rhino habitat. For other factors, subjectively estimated probabilities are the only option.

Under *status quo* management perhaps the random event with the greatest potential impact on pE for the Sumatran rhino over the next 30 years is epidemic disease. We estimate the probability of this event to be at least 0.1. The events 'epidemic' or 'no epidemic' appear in Figure 8.2 as branches emerging from a circle, indicating a random process that rhino management cannot influence. The probability of each event appears on the corresponding branch. We estimate that an epidemic would increase the pE to at least 0.95. (The pE for each outcome is shown at the end of its corresponding branch in Figure 8.2.) Following equation (1), we use the probability of each random event (epidemic disease or not) to weight pE for each action/event combination, to get E(pE) for current management: (0.1)(0.95) + (0.9)(0.85) = 0.86, which we list under the column headed E(pE) in Figure 8.2. Since maintaining *status quo* management involves no increase in cost over what is currently being expended, we list this as $0 under the dollar cost criterion.

Management interventions

If an E(pE) of 0.86 for the Sumatran rhino under current management conditions is judged to be unacceptably high, what might be done to reduce it? Table 8.1 lists examples of intensive management strategies for endangered animal populations, both wild and captive. Several of these have been proposed for recovering the Sumatran rhino. They are shown in Figure 8.2 and include: (1) increasing control of poaching in existing reserves; (2) doubling the size of one national park;

Figure 8.2. Decision tree for management of the Sumatran rhino. Squares indicate decision points; circles indicate random events. Probabilities of random events are estimated for a 30-year period; pE = probability of species extinction within 30 years; E(pE) = expected value of pE for each alternative. Costs are present values of 30-year costs discounted at 4% per year; M = million.

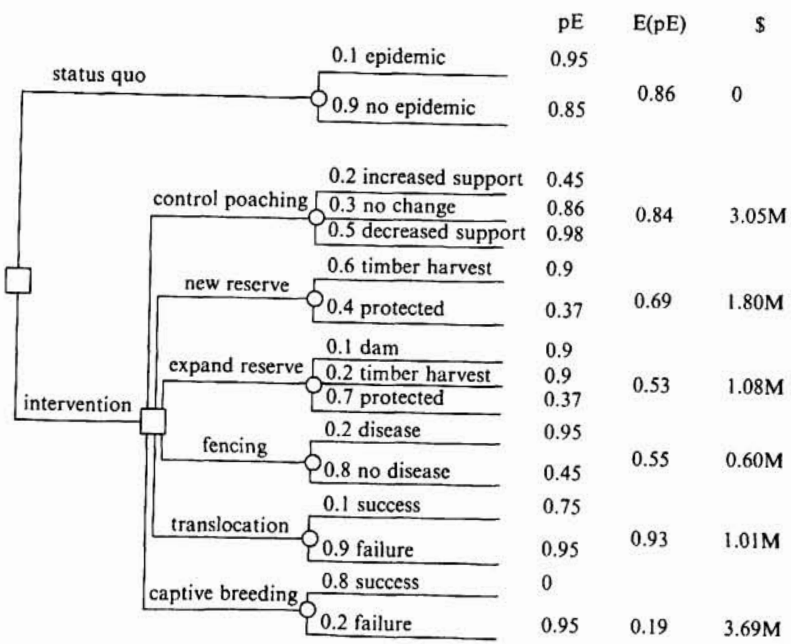


Table 8.1. *Examples of management interventions for critically endangered animal species*

Wild populations and habitat only
Translocating individuals or genetic material
Raising carrying capacity (e.g., artificial feeding)
Restricting dispersal (e.g., fencing)
Fostering and cross-fostering young
Reducing mortality (e.g., vaccination; parasite, predator, poaching control)
Culling
Preserving habitat
Restoring habitat
Captive populations only
Maintaining captive breeding populations for reintroduction and/or perpetual captivity
Genetic and demographic management
Maintaining gametes or embryos in 'miniature zoos' (i.e., freezers)
Captive and wild populations
Reintroduction of captive-reared individuals or genetic material to occupied or unoccupied habitat
Continued capture of wild individuals or genetic material for captive propagation

(3) creating a new national park; (4) fencing a large area of prime habitat, managing the enclosed population with supplemental feeding and veterinary care, and translocating isolated rhinos into the enclosure; (5) translocating rhinos among wild subpopulations to restock depleted habitats and to maintain gene flow among subpopulations; and (6) capturing wild rhinos to form captive breeding populations in at least four separate institutions in four countries. The captive populations would serve both as a reservoir of genetic material and as a source of animals to bolster populations in currently or previously occupied habitat.

To choose among management alternatives, assessments of each option must include both their respective expected improvements in the species' status and their costs. We now will examine each strategy for recovery of the Sumatran rhino (Figure 8.2), including uncertainties affecting the expected probability of extinction for each plan and an estimate of its financial cost.

The effectiveness of efforts to control poaching is critically dependent on government support for conservation. If protection against poaching is increased by four additional rangers in Sabah and six in Indonesia, with a vehicle for every two rangers, pE might be reduced to roughly 0.45 over the next 30 years. This additional effort would cost about $3.05 million (in this analysis all annual costs were discounted at 4% per year). However, the probability that government support for conservation would both be uninterrupted and high enough to permit this level of success is only about 0.2. If the *status quo* is maintained (probability about 0.3), we expect pE to remain at about 0.86. If government support for rhino conservation is seriously eroded, especially in Sabah and Indonesia (probability about 0.5), control efforts would be wasted; and pE could increase to at least 0.98. The expected value of pE for control of poaching is thus (0.2)(0.45) + (0.3)(0.86) + (0.5)(0.98) = 0.84.

Designating a new national park for rhino protection would support approximately 25 animals; this action would reduce pE to about 0.37. The cost of acquiring and maintaining the new national park would be about $1.8 million, not counting loss of revenue from timber harvest or agricultural use. However, even if the area is designated as a park, pressure to exploit timber within it will be extreme, with perhaps a 0.6 probability of timber harvest in the reserve within the next 30 years. We estimate that pE would rise to about 0.9 in that case because of habitat loss and poaching by loggers.

The third alternative is to double the size of an existing park, which also would support about 25 additional animals. The same site has been suggested for a hydroelectric dam. If installed, this project would eliminate a large proportion of prime rhino habitat. Although the hydroelectric project has been deferred for the next five years, it has not been permanently abandoned. We estimate the probability of the dam being built within the next 30 years to be about 0.1. Even if the dam is not built, timber interests threaten the integrity of the expanded reserve, with perhaps a 0.2 chance of harvest. If the reserve were maintained intact, pE for the species as a whole might drop from 0.86 to about 0.37; if either dam-building or timber harvest occurs, pE would rise to about 0.90. The combined probabilities of these events result in an E(pE) of 0.53. The cost of acquiring and maintaining the expanded reserve over the next 30 years is estimated at about $1.08 million, not counting income lost from reserving the timber and not building the dam.

Another option is to fence an area in an existing or new reserve, managing the resulting high density of rhinos with supplemental feeding and veterinary care, as in the successful South African rhino ranches (Martin, 1984). Most problems, such as food shortage or nutrient imbalance, probably can be detected and remedied quickly enough to avoid heavy mortality.

Disease is a major risk associated with this plan. We estimate the probability of such a disease outbreak to be about 0.2. If an epizootic occurs, pE for the population as a whole would rise to about 0.95. This increase from the *status quo* level of 0.86 is due to the transfer of animals from isolated subpopulations to the fenced area: isolation, *per se*, provides some assurance of escape from exposure to the pathogen. On the other hand, if the fenced population can be maintained successfully, pE would decline to about 0.45. Together, these alternatives result in an E(pE) of 0.55. The fenced area would cost about $60 000 to establish and about $18 000 per year to maintain, for a total 30-year cost of about $0.60 million.

The fifth management intervention illustrated in Figure 8.2 is the translocation of rhinos among isolated subpopulations. Many remaining rhinos exist in groups of two or three individuals, and random mortality and reproductive failure (demographic stochasticity) are big threats to the viability of such small groups. Subdivided populations that are as small and as isolated as these easily slip below the level for long-term viability (see Goodman, Chapter 2). In addition, habitat fragmentation reduces the chance that subpopulations which go extinct will be recolonized by migrants from other areas.

Deliberately moving rhinos among subpopulations to compensate for mortality and lack of natural recolonization is one way to assure survival of these populations, but there are hazards at each phase of the operation. Capture and transport have a high probability of death or injury. Furthermore, the chances are rather high that translocated animals will disperse from the new site or not be accepted into the social hierarchy and, hence, will fail to reproduce. An intensive effort to adapt rhinos to their new locations could cost $100 000 per rhino. A translocation program with no follow-up after release might have about a 0.1 chance of success and would cost about $20 000 per animal. If animals were translocated among subpopulations at the rate of seven per year for the first three years of the program and two per year for the remaining 27 years, the program might reduce pE over the next 30 years to about 0.75. On the other hand, if translocation results in death or injury to several rhinos, the pE for the species as a whole may increase to 0.95. These alternatives result in an E(pE) of 0.93 and a total cost of $1.01 million.

The last management alternative for the Sumatran rhino is the establishment of a captive breeding program. Success or failure depends on a series of random factors, including capture of wild animals, shipping, behavioral adaptation to captivity, breeding success, disease, and cooperation among institutions. Animals captured for captive propagation would come from small isolates under heavy pressure from poaching and timber harvest and, therefore, with little potential to contribute to long-term population growth. Nevertheless, loss of these long-lived animals from the wild population might increase pE to about 0.95 if the captive breeding program fails. However, if the captive program is successful, and breeding populations are established at several facilities in Malaysia, Indonesia, the United States, and Great Britain, the pE for the species would be reduced to zero over the next 30 years.

To estimate the probability that the proposed captive breeding program will be successful, we can draw on previous experience with capturing and breeding white *Ceratotherium simum*, black *Diceros bicornis*, and Indian *Rhinoceros unicornis* rhinos with judgements about differences between these species and the Sumatran rhino. We can also use subjective information. For example, the likelihood of successful breeding by the existing captive Sumatran rhino female has been assessed by biologists familiar with other rhinocerus species. Those who have observed her unusually calm and 'friendly' behavior and obvious adjustment to captivity feel confident that breeding will be no problem, provided an appropriate mate can be found. Based on previous experience with captive breeding of other rhino species and on the current levels of political support for the program, we believe the chances of success to be about 0.8, resulting in an E(pE) of 0.19.

The costs of a captive breeding operation will be high. Development of facilities and propagation techniques in Malaysia and Indonesia will cost about \$1.25 million for the first three years plus \$30 000 per year for maintenance, or \$2.06 million over 30 years. Costs of maintaining captive rhinos in the United States and Great Britain are about \$3000 per animal per year. The original populations of about eight animals in the US and eight in Great Britain are expected to quadruple in the 30-year period. The total cost for captive populations in these two countries, not counting expansion of physical facilities, is at least \$1.63 million, bringing the total to \$3.69 million.

Evaluating management alternatives

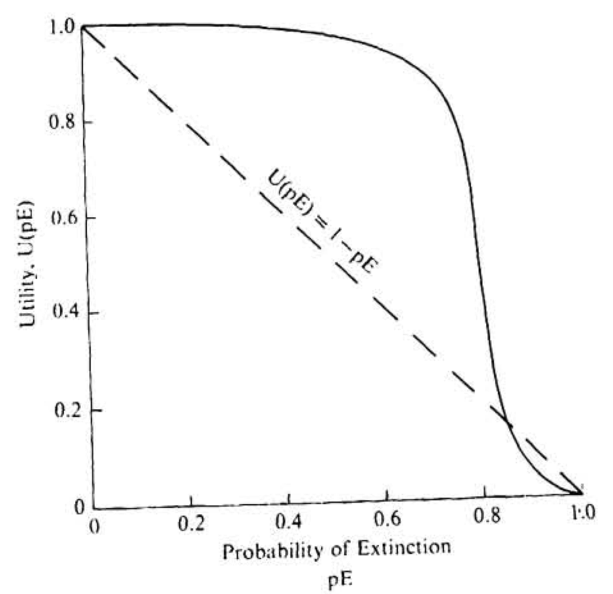
For each of the six proposed management interventions, and for the *status quo*, we have estimated the expected probability of extinction and financial costs, listed on the right in Figure 8.2. Given that the purpose of rhino management is to minimize the probability of extinction, preferably also with minimum cost, which alternative is best?

Translocating animals among wild subpopulations is far too risky to be recommended. Its E(pE) is even higher (0.93) than for the status quo (0.86). The chances of success and attendant benefits to the populations are not high enough to outweigh the loss of translocated animals if the program fails.

Captive breeding is the most promising option in terms of minimizing the expected probability of extinction for the species, with an E(pE) of 0.19. Even if removal of rhinos to captivity raises pE for the wild population, the chances of successful captive breeding are high enough to justify this option. However, many conservationists are wary of emphasizing captive programs, feeling that survival of the species in captivity is a poor second to survival in the wild and fearing that captive programs will divert attention and resources from the conservation of wild populations. These concerns could be incorporated into this analysis by assigning lower values to survival in captivity than to survival in the wild, as described by Maguire (1986). However, in cases like the Sumatran rhino, where extinction in the wild seems likely even with strong conservation efforts, we feel that the most pressing issue should be survival of the species in any form.

Beyond identifying the obviously best and worst management strategies (i.e., captive breeding and translocation for our rhino example), is it possible to evaluate objectively the remaining options within the middle range of E(pE)? These options can be treated more formally by constructing a utility function for pE (Figure 8.3), which

Figure 8.3. Utility function illustrating current social assessment of probabilities of extinction, where the most desirable pE has a utility of 1.0, and the least desirable has a utility of 0. The straight dashed line is the linear utility function U(pE) = 1 − pE.



reflects relative preferences for different values of pE. The best pE (0.0) is assigned a utility value of 1 and the least acceptable pE (1.0) a utility value of 0. We note that in the current social and political climate, intervention is often delayed until extinction seems imminent. Thus, the utility function we have drawn in Figure 8.3 (solid line) reflects our observation that society evidently considers high pE quite acceptable until it is very close to 1. For comparison, we include in Figure 8.3 a linear utility function (dashed line) where incremental improvements in pE are viewed equally enthusiastically over the entire range of pE.

When using utility to choose among alternatives, the best choice is the one that maximizes expected utility, rather than the one that minimizes pE. The curved utility function in Figure 8.3 (solid line) is used to assign utility values corresponding to each pE in Figure 8.2. For example, the utility of pE = 0.45 is about 0.98. Then expected utility, E(U), for each option is calculated using the probabilities of random events just as in calculating E(pE). The resulting E(U) values are compared to see which is highest.

Utility functions reflect individual preferences and are influenced by a person's current circumstances, including wealth, education, religious views, and so on. Decision analysts elicit an individual's utility function for an attribute such as pE by asking a series of questions about preferences (Keeney and Raiffa, 1976). Utility functions elicited at different times from the same individual tend to be similar, but they do change when the person's wealth, education, or other circumstances change (Officer and Halter, 1968).

Conservation education may be one means of altering a person's utility function for pE. Because utility functions reflect individual preferences, it may be difficult to develop utility functions for public decisions, where no single person is responsible for making the choice and where special interest groups may have divergent opinions. Keeney and Raiffa (1976) and Keeney (1977) have outlined a method for combining utility functions of different groups to form a composite useful for public decisions.

Because the utility curve in Figure 8.3 is only hypothetical, we will not calculate and compare E(U) for the rhino example, but will instead use E(pE) to evaluate management options. Our criteria for choosing among management alternatives include minimizing cost as well as minimizing pE. Which options are most cost-effective? Our estimates of costs are admittedly crude, and we have chosen to neglect the fact that random events will sometimes change the actual costs of a management activity. For example, reductions in government support for poaching control probably will result in reductions in the success of those efforts. Likewise,

our cost estimates for the establishment and expansion of reserves neglect potential revenues from timber harvest or other commercial exploitation.

Both the captive breeding and poaching control programs are very expensive, but captive breeding is clearly the preferred option because of its greater expected benefit to the species' security. Of the several management choices for wild populations with E(pE)s between 0.5 and 0.7, fencing looks most cost-effective, but we suspect that the costs of establishing and maintaining a fenced population have been underestimated. Methods for assessing the trade-offs between conflicting criteria, such as financial cost and species recovery, are described by Maguire (1986).

In many cases several options may be pursued at once, and the problem is to choose the mix of activities with the best impact on species recovery for the available budget. Multiobjective programming (Cohon, 1978) is a method of achieving this integration of population and cost criteria, but a detailed analysis of this is beyond the scope of this chapter. We simply point out that a number of the proposed interventions, including fencing and captive breeding, involve collecting animals from some of the remaining wild subpopulations and, therefore, compete for limited biological, as well as financial, resources. In addition, the impacts of the proposed interventions on population status may not be independent, and their interdependence must be expressed in any programming analysis.

As is common in endangered species management, the probability estimates in this analysis are themselves uncertain quantities, although they represent the best we can do with the information available. We need to assess how sensitive our ranking of management alternatives is to changes in the probability values used. If the selection of alternatives changes with small alterations in one of the probability estimates, we can focus research efforts on reducing uncertainty in that parameter.

What changes in probability estimates could change our ranking of the proposed interventions? Even if the probability of success of the captive breeding program were as low as 0.5, its E(pE) would still be lower (0.48) than any other alternative, although its cost might then be too high for the expected benefit. For the E(pE) of the translocation program to be lower than the value for the *status quo* (0.86), the probability of success for translocation would have to be at least 0.5. However, providing the follow-up care to ensure this level of success would be extremely costly. Note that the fencing and the captive breeding options differ only in pE for success (0.45 versus 0) and in cost ($0.60 million versus $3.69 million). If pE for a successful fencing option were as low as 0.2 or 0.3, it would become a more attractive option than captive breeding in zoos because of its lower cost. The sensitivity analysis shows that only large changes in probability estimates would alter our conclusions.

Implementation

A formal process for implementing an international conservation program for the Sumatran rhino was initiated by the Species Survival Commission (SSC) of the IUCN. A meeting in Singapore in October 1984 was attended by representatives of Indonesia, West Malaysia, Sabah, captive breeding groups from the UK and USA, the SSC, and researchers familiar with the status of the species in the wild.

An important feature of the meeting was the general agreement that the Sumatran rhino is a greatly endangered species, that its extinction will be a great loss, and that it is necessary to intervene if the species is to be saved. A basic set of goals was formulated, and tentative agreements for their implementation were drafted. The fundamental tenets of the agreements were that: (1) primary support would be given to conserving the Sumatran rhino as viable populations in sufficiently large areas of protected native habitat; (2) an educational program to enhance public awareness and support for the Sumatran rhino would be developed; and (3) a captive breeding program for preserving genetic diversity in the species would be developed in the countries of origin and elsewhere (USA and Europe), using animals with no hope of long-term survival in the wild.

Translating these agreements into working documents, budgets, a program, schedules, and a management and policy structure, required six months' activity after the meeting with numerous consultations among the parties. The entire exercise has been an important activity by the SSC/IUCN and will provide a working model for collaborative international intervention.

Conclusions

It is critical to identify species in imminent danger of extinction in order to focus conservation efforts on them. Methods for estimating the probabilities of extinction (pE) range from sophisticated analytical or simulation models to informed judgements. How high pE must be to be unacceptable is likely to depend on the species' position in the taxonomic hierarchy, its aesthetic or economic value, and the region of the world in which it is found. An extinction probability of unity for a small species of brown moth inhabiting the Falkland Islands is not likely to arouse much concern, while a pE of 0.25 or so for a large, well-known predator usually galvanizes considerable action.

Once a species has been designated as critically endangered, a number of management options must be considered, including no intervention. Decision analysis provides a framework for evaluating the efficacy of these options. In addition to facilitating identification of the best management alternatives, decision analysis helps pinpoint potential sources of uncertainty, provides a way of comparing costs and benefits, and permits a sensitivity analysis to assess the robustness of its conclusions.

While decision analysis provides a useful framework for discriminating among management alternatives, political processes play a large part in the implementation of any conservation program. Some elements of the political process can be included in a decision analysis *via* utility functions. However, political winds are notoriously shifty, and conservation biologists must be prepared to accept the fact that today's worst option may be tomorrow's first or only choice.

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9

The role of interagency cooperation in managing for viable populations

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The challenge of managing lands, resources, and people to sustain viable populations of large vertebrates and other taxa is enormous. Conservation biologists (e.g., Soulé, 1980; Frankel, 1983; 1984) have emphasized that few existing protected areas can provide this service for all desired species found within their bounds. However, most analyses of the ability of reserves to sustain viable populations (e.g., Soulé, 1980; Frankel and Soulé, 1981; Schonewald-Cox, 1983) consider each jurisdictional unit as a separate entity. This reflects a frequent lack of cooperation and the conflicting priorities and management practices that can exist in adjacent areas (Schonewald-Cox and Bayless, in press; Harris, 1984). The future of many species would not be nearly as bleak if managers who share species of concern would cooperate to minimize conflicts and reach mutual conservation goals.

The effectiveness of cooperation in attaining conservation goals has been demonstrated by The Nature Conservancy since its inception in 1951 (Jenkins, 1984). More recently, the value of cooperation was recognized in the first recommendation of the Terrestrial Animal Species Panel at the US Strategy Conference on Biological Diversity (US Dept. of State, 1982): '. . . identify, establish, and manage a worldwide system of national parks and other conservation areas to insure the perpetuation of all major ecosystem types and the diversity of organisms and processes they contain . . . in a way which promotes local economic development compatible with long-term ecosystem integrity and the sustained use of natural resources.' Current public interest in an 'ecosystem approach' to

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