THEORY AND PRAGMATISM IN THE CONSERVATION OF RHINOS

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ABSTRACT

If rhinos are to survive for the next couple of hundred generations, theoreticians suggest that it is best to conserve large populations. Equally, if effective population size is small, theoreticians wish to interchange genetic material between wild and captive populations. In an ideal world, it would be highly desirable to plan conservation activities over fifty or a hundred generations. However, major conservation crises and successes for rhinos have occurred over much shorter time frames. Thus 95% of Africa's large black rhino populations have been lost, and southern white and Indian rhino populations have been re-built, over a few decades. Initial successes for certain black rhino populations, and even for northern white and Javan rhinos, have materialised within a decade (or one rhino generation). A model of incentives to kill black rhinos in Zambia suggests that wildlife managers should aim to increase their success at detecting poachers. By analogy with this model, most of the successes or failures in rhino conservation *in situ* can be attributed to the provision of, or lack of, adequate protection. Despite complaints that providing adequate protection is expensive, it is more cost-effective on a per-animal basis than measures which involve zoos and captive breeding. Therefore, the pragmatic option to achieve success in rhino conservation over the next few decades appears to be to build up small populations *in situ* and maintain them in natural habitat with minimum interference.

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INTRODUCTION

Concern among scientists and conservationists at the increasing loss of the world's biological diversity and natural resources has led to the establishment of conservation biology as a distinct discipline (Soulé 1986a). A casual survey of the theory of conservation biology suggests that it dominated by two main, but interconnected, themes that bear upon reducing the risk of extinctions. First, the theory of island biogeography propounds the ideal of conserving large populations of endangered species in large protected areas (PAs). Second, given that many populations and PAs are already small, the theory underlying the concept of minimum viable populations (MVP) assesses requirements for endangered species to retain demographic vigour and genetic variablity, usually through a population viability analysis (PVA). This body of theory has given rise to a set of options that could be aspired to in an ideal world. However, it is also worth examining how successful the theory of conservation biology is in relation to achieving conservation objectives in a real world that tends not to operate to a set of ideals (see Deshmukh 1989).

One of the species groups that is now of great concern to conservationists is the rhinos, and their conservation requirements provide an ideal opportunity to examine the interface between theory and practice. Of the five extant species, two species in Asia (Javan rhinos, *Rhinoceros sondaicus*, and Sumatran rhinos, *Dicerorhinus sumatrensis*) and one sub-species in Africa (northern white rhinos, *Ceratotherium simum cottoni*) teeter on the edge of extinction and are critically endangered. Over the past two decades, the formerly numerous black rhino, *Diceros bicornis*, has plummeted from an estimated 65000 to 3000 and has become locally extinct over large areas of Africa. By contrast the southern white rhino, *C.s. simum*, is currently well conserved in limited areas of southern Africa, as is the Indian rhino, *Rhinoceros unicornis*, in India and Nepal, with total world populations in the low thousands for both species. In this review I aim to examine the following in relation to the conservation of rhinos: the role that theoretical conservation biology has attempted to play; the role that theoreticians and zoos aspire to play; and, most importantly, what has and has not actually resulted in practical conservation success stories.

LARGE PROTECTED AREAS AND LARGE POPULATIONS

Theory. Protected areas (PAs) aimed at conserving endangered species like rhinos have, in effect, been increasingly established as islands in a sea of humanity. The size and shape of PAs have been determined largely by social and political factors, and also by topographical criteria convenient for siting of boundaries (reviewed in Leader-Williams et al., 1990b). Theoreticians argued that size and shape of PAs based on such criteria may not be ideal and can result in loss of biological diversity. Scientific debate on the ideal size and shape of PAs began when it was realised that the theory of island biogeography provided a framework for studying PA design in terms of species richness. This theory as applied to PA design stressed the need for big reserves, but has since been questioned for a variety of theoretical reasons (reviewed in Soulé and Simberloff 1986). Most importantly, perhaps, the theory of island biogeography cannot predict accurately which extant species, say black rhinos or wild dogs in an African reserve, are likely to go extinct, or at what rate extinctions will occur.

As PAs are only successful to the degree that their contents retain their integrity, the emphasis has now switched to considering what size PAs should be to support minimum viable populations (MVP) of threatened species, using the more testable theories of population biology (Soulé 1986b). Extinction is a matter of risk and time to extinction is likely to be shorter for smaller populations. Intrinsic and extrinsic factors are believed to contribute to extinctions in small populations. Intrinsic factors include: (i) demographic stochasticity (the increased probability that all individuals in a particular generation will, say, be male in populations of less than 50 individuals) and (ii) loss of genetic diversity (heterozygosity or variability, resulting in inbreeding, increased mortality and reduced long-term adaptability). External factors such as disturbances, contagious diseases, environmental variables and catastrophes can also have serious consequences, especially in single, small populations.

The exact size of an MVP has not been determined and, indeed, is likely to be unique for each species in each location. However, to avoid any significant loss in genetic diversity over 50 or more generations, current consensus is that an MVP should consist of at least 500 genetically effective individuals (see Figure 1a), which translates into a total population of around 1000 individuals to include juveniles and other non-breeders (Soulé 1986b). With fewer genetically effective individuals, the rate of loss of genetic diversity increases so that with an effective population of 25 individuals over 50% of genetic diversity would be lost in 50 generations.

Three steps are necessary to convert the concept of an MVP to PA size: (1) identify a flagship species, such as a species of rhino, whose disappearance would significantly decrease the value or the species diversity of the PA; (2) determine the minimum number of individuals in a population needed to guarantee a high probability of survival; (3) use known densities or ranging patterns to estimate the size of the area necessary to sustain the minimum number. As a result, the ideal size of PAs will vary relative to the key species under protection, but reserves holding say, 1000 black rhinos at their normal densities of 0.4 rhinos per sq km, will need to be several thousand sq km in size. Therefore, whether using island biogeography or MVPs, the basic recommendation of theoreticians is that large populations of endangered species ought ideally to be conserved in large PAs. Thus the approach of focusing on flagship species complements the goal of conserving whole ecosystems in that the persistence of flagship species is critical to the survival of ecosystems.

Practice. Naturalists and conservationists have expressed concern for the long-term future of large mammals like African and Asian rhinos for several decades. However, a greater public awareness in the 1980s demanded action when it was realised that the formerly numerous black rhino was disappearing at a rate that was unprecedented for any large mammal. With the raising of funds to help conserve black rhinos, the African Rhino and Elephant Specialist Group followed the principles of theoretical conservation biology and gave the highest priority to protection of the large populations within the large PAs, primarily in the Selous in Tanzania and the Luangwa Valley in Zambia (AERSG 1984). The situation in the Selous was not studied in detail, but surveys suggest that many rhinos were lost during the 1980s. The situation in Luangwa Valley was monitored closely (Leader-Williams and Albon 1988; Leader-Williams et al., 1990a) and it was clear that over 90% of rhinos disappeared in less than half a rhino generation (Figure 1b). The difference in terms of time, measured both in years and in rhino generations on the horizontal axes, between the theory and practice of conservation is striking. However, it should be noted that the vertical axes on the upper and lower graphs in Figure 1 differ. In Figure 1a it is measured in terms of genetic diversity and in Figure 1b in terms of an index of rhino numbers, and I have not attempted to calculate how much genetic diversity is lost in such a rapid population decline (see Frankel and Soulé 1980).

The loss of rhinos in Luangwa Valley and of other large populations in Africa has been due in large part to shortage of resources in national conservation authorities (Cumming et al., 1984; Parker and Graham 1989). In Luangwa Valley, there was a direct relationship between the levels of effort expended by law enforcement patrols upon different areas, and the rates of decline of rhinos in those areas. Hence, patrol effort did deter poaching but, at an initial average scout density of one man to patrol 760 sq km, the effort was very *thinly* spread (Leader-Williams 1985). Even when staff were concentrated in more heavily patrolled core areas, they could not prevent the decline of rhinos, and results from the study suggested that a staff density of around one man per 10-20 sq km would be necessary for this task (Leader-Williams et al., 1990a). More men cost more money, and therefore it was not too surprising to find that there was a direct relationship between the rates of decline of rhinos and national conservation budgets across Africa in the 1980s (Figure 2). Low spending countries with large populations of rhinos experienced high losses, but countries spending over a threshold of $230 per sq km in 1980 were more successful at conserving their rhinos (Leader-Williams and Albon 1988). This result has led to comments that successful *in situ* conservation is very expensive to achieve, but it is worth putting the amount of money spent on conservation in Africa in context. In 1980 a total of around $75 million was spent by the national wildlife authorities of all the countries in sub-Saharan Africa (Bell and Clarke 1986). In contrast, the San Diego Zoological Society, which is so generously hosting this conference as part of its 75th anniversary celebrations, has an annual budget of around $70 million. This is said, not to detract from the well-merited success of San Diego, nor to plead for a budget cut and donation of the balance to Africa. Instead, the point is made merely to put the sums devoted to *in situ* conservation in context.

Unfortunately information on patrol effort and conservation agency budgets for Asia is less detailed than for Africa. Indonesian conservation budgets of around $2-10 per sq km in 1989 and 1990 (K. MacKinnon, pers. comm.) suggest that national wildlife authorities in Asia also generally underinvest in their PAs. A survey of rhino sub-species shows a similar lack of success in protecting large, widespread populations of Asian rhinos living in rainforest (Figure 3). Most efforts at conserving the formerly widespread sub-species of Sumatran and Javan rhinos have not met with success. Indeed, one sub-species of Javan rhino, that formerly living in India, Burma and Thailand, has probably gone extinct (Nardelli 1988).

But all is not doom-and-gloom in the field of rhino conservation. Two situations, involving one species and one sub-species of rhino, that were close to complete failure, have been converted to practical conservation success stories (Figure 4). As a result of overhunting early this century, both Indian and southern white rhinos were reduced to very low numbers. The subsequent increases seen in Indian rhinos, both in Chitawan in Nepal (see Dinerstein 1992 in this volume) and Kaziringa in India, and in the southern white rhino in the Umfolozi and Hlulhulwe complex in South Africa (Figure 4), have been achieved by protecting small remnant populations in small areas. The recovery of both these species has been helped because they are the most gregarious species of rhinos and live at high densities in alluvial floodplains and grasslands, respectively (see Laurie 1982). Thus, more rhinos can be protected in smaller areas than for the other solitary species of rhinos. Indeed, high levels of protection are maintained even today. In Kaziringa in the 1970s 200 park staff patrolled 425 sq km, at densities of 1 man per 2 sq km (Bradley Martin 1983). Chitawan is today even more heavily protected, by 700 men of the Nepalese army and 45 park staff in 907 sq km, at densities of 1 man per 1.2 sq km (E. Bradley Martin, pers. comm.).

Besides these actual successes, we have the beginnings of success stories even for solitary species of rhinos. The approach used for the East African sub-species of black rhino in Kenya, for the desert subspecies of black rhino in Namibia, and for the Central African sub-species of black rhino in South Africa, are all discussed elsewhere in this volume (Brett 1992; Brooks 1992). The general formula for success has involved rounding up stragglers, concentrating resources on small areas, and once the population has built up sufficiently making translocations to unoccupied habitats in areas of former natural range (see also the establishment of translocated populations of Indian rhinos discussed in this volume by Sinha 1992; Shrestha 1992). Even two of the world's most endangered sub-species of rhinos, the northern white rhino (discussed in this volume by Hillman-Smith et al., 1992) and the Javan rhino (Figure 5) have shown promising increases when given better protection. To date, therefore, affording good protection of small populations of rhino *in situ* remains the only proven method of achieving success in rhino conservation.

In order to understand the mechanism underlying the success of better protection, it is necessary to turn to the theory of economics, and to consider the incentives to undertake any form of illegal activity, whether this be robbing a bank or poaching a rhino (Milner-Gulland and Leader-Williams 1992). Rhinos are poached because their horn has a high economic value (see Bradley Martin 1992 in this volume). In order to decrease the incentive to poach a rhino, one or other of three different parameters must be altered. The first possibility is to increase the opportunity cost of crime through improved wages elsewhere. That is to say, reduce the earnings from the illegal activity relative to earnings to be made from some form of legitimate activity, either by providing the opportunity to earn legal wages in deprived areas or by raising wage rates through a general improvement in the national economic climate. The second possibility is to increase the probability or severity of punishment, and the third possibilty is to decrease the actual profit from the crime. Law enforcement can affect incentives in both these ways. Studies of various crimes, including of incentives to poach in Luangwa Valley (Milner-Gulland and Leader-Williams 1992), show that a high penalty is less of a deterrent than improving the rates of detection and capture. Indeed, this conclusion is re-inforced by another study discussed in this volume, which shows the importance of reducing time to detection in Zimbabwe's rhino conservation strategy (Martin 1992).

Therefore, had economists been willing to advise conservationists (or the conservationists been willing to listen!) in the 1980s, there would have been more lessons on successful conservation of rhinos to be learned from the theory of economics than of conservation biology. In only one country, South Africa, has it been possible to couch a rhino management plan in terms of the theory of conservation biology (Brooks 1989) and, at this stage, be fairly certain of its success. And that is because South Africa presently has the economic resources to build up its black rhino numbers (and to conserve its large population of white rhinos) *in situ* (see Figure 2). While the theory of conservation biology may not be intrinsically incorrect, it has been of little help in the practice of conserving large populations in most countries. And now many rhino populations are small, and according to theoreticians too small to be viable.

MINIMUM VIABLE POPULATIONS AND CAPTIVE BREEDING

Theory. As once large populations become reduced, and as human expansion cuts down the availability of corridors between PAs, individual populations that were once able to interchange genetic material become increasingly isolated. Furthermore, as the population becomes smaller, the risk of losing genetic diversity (Figure 1a) and, therefore, of extinction becomes greater. To date the evidence to support this theory is largely restricted to models (Lacy 1987) and empirical data are generally lacking. Where there are empirical data, the situation is often contradictory, and one of the best examples of this comes from a study of Indian rhinos (Dinerstein and McCracken 1990). As noted earlier, both the Chitawan and Kaziranga populations have been through recent population bottlenecks which, according to theory, should result in loss of genetic diversity. Clearly, genetic material was not available from before the bottleneck for comparison, but somewhat surprisingly Indian rhinos in Chitawan are one of the most genetically variable mammals studied to date (see also McCracken 1992, in this volume).

Even though the exact size of a population too small to be viable is not known, and too much emphasis may have been placed on the erosion of genetic diversity attributable to bottlenecks, theoretical conservation biologists are becoming increasingly interested in the management of small populations as a way of helping the practice of conservation biology. An increasing number of species are being subjected to population viability analyses (PVA) that promote metapopulation management with proposals for interactive exchange between wild and captive populations (see Figure 6, and more of the same in Foose 1992 in this volume). Here, theoreticians have teamed up with the proponents of captive breeding in the zoo community to promote a concept of mutual interest. Not surprisingly, the plight of rhinos is attracting considerable attention in both camps. Indeed, an action plan for Asian rhinos has, like the plan for South Africa referred to earlier, been written within the framework of theoretical conservation biology (Khan 1989). But, in contrast to the South African plan, this has resulted in a number of proposals to take into captivity some of the most endangered species of rhinos. I now wish to examine some of the practicalities of these proposals.

Practice. In this discussion, I refer not to the taking of a small nucleus of rhinos for captive breeding from a large population, nor to the taking of further genetic material from populations such as Indian or southern white rhinos once they have recovered (see Figure 7). In either case, this simply spreads the risk should the population decline to crisis levels, and would have little effect on the source population in the wild. Instead, I refer to proposals (or failed proposals) to take a similar sized nucleus from the more critically endangered Sumatran, Javan and northern white rhinos (Seal and Foose 1990; Captive Breeding Specialist Group 1990). As can be seen from the wording on Figure 7, I believe this is an entirely inappropriate option for two reasons, one biological and the other economic. Other notes of caution such as the probably difficult task of successfully reintroducing animals from captive breeding programmes in 70-100 years time, are discussed elsewhere in this volume (Stanley Price 1992).

First, rates of increase in zoos are far lower than have been recorded in well-protected populations in the wild. The three browsing and solitary species, Javan, Sumatran and black rhinos, have very complex diets and probably very complex husbandry requirements. There have been no births to the few Javan rhinos that were kept in captivity in the past, and only 4 births to Sumatran rhinos since the 1890s, but all to animals caught when pregnant (Nardelli 1988). Rates of increase for the the zoo population of black rhinos is also considerably lower than the rates for wild populations. Even where evidence is available for Indian and white rhinos, the two grazing or mixed-feeding species with less complex dietary requirements, rates of increase are still lower than in the wild (Figure 8). Therefore, if the intention is to bring endangered populations of rhinos out of bottlenecks quickly (see Frankel and Soulé 1980), then all evidence to date indicates that this can be achieved more effectively by giving adequate protection to rhinos *in situ*.

Thus, I do not believe it is appropriate to "experiment" with captive breeding in critically endangered species like Javan rhinos when it is not a proven or effective method of achieving conservation success, or when it is not known if the species can be bred successfully in captivity. Equally, the proponents of captive breeding may argue that they need the material before they can be expected to get it right, but a counter-argument here is that captive breeding efforts need to be got right on related species first. The black rhino serves as a very good model for Javan and Sumatran rhinos, and it is clear that the former species still faces considerable husbandry problems in captivity, as discussed at length elsewhere in this volume (Miller 1992), besides still having a generally low breeding rate (Figure 8). With the benefit of hindsight, the situation of the northern white rhino may reinforce this point. In 1983 it was proposed that all 14 rhinos remaining in the wild should be removed to zoos, but this proposal was not accepted by the Zaire government, who wished to conserve the rhinos *in situ*. Since that time, there have been 16 births in Garamba, and only one birth to the captive northern white rhino population in Dvur Kralove (see Hillman-Smith et al., 1992; Spala 1992, both in this volume).

Second, conservation is a cause that has to compete for limited funds with other good causes, and the captive breeding option is cost-ineffective. As noted already, the costs of maintaining black rhinos on PA land in Africa worked out to be $230 per sq km (Leader-Williams and Albon 1988), and this figure has frequently raised the comment that *in situ* conservation is very expensive. However, using the common currency of costs per animal, the opposite is in fact true. At their usual densities of 0.4 rhinos per sq km, the costs of maintaining a black rhino in the wild was $575 in 1980, around three times less than in captivity (Leader-Williams 1990). It should also be noted that this difference is a very conservative estimate for two reasons. On the wild side, the sums spent per sq km look after much more biodiversity than simply rhinos. In the example worked out for Luangwa Valley, elephants and rhinos share the same conservation costs, and the difference increases to around fifty-fold between wild and captivity (Leader-Williams 1990). While not costed in the same way, other examples are given for Asian rhinos in this volume (Widodo et al., 1992). On the captive side, the data quoted are for adoption rather than actual total costs incurred by zoos. Again, I stress that this comparison is not presented to make a point that zoos cost too much or as a plea to reduce zoo budgets from central or local government sources. This would not be possible in any case, nor would it be desirable because zoos have perfectly valid educational functions that merit their budgets. Instead Figure 8 aims to illustrate how much more cost-effective *in situ* conservation could be per animal if funded to adequate levels from monies raised from adoption schemes.

If conservation in the wild is more cost-effective per animal than in zoos, what might be achieved if money used by zoos to set up captive breeding initiatives were instead directed to *in situ* conservation of rhinos? I present a hypothetical example, hedged around with considerable guesswork due to difficultes obtaining data, for Sumatran rhinos. Over the past five years, two zoos or zoo partnerships have apparently spent $2.5 million just to catch so called "doomed" Sumatran rhinos, with the aim of establishing captive breeding populations of Sumatran rhinos in zoos (see Nardelli 1988). This sum does not include the costs of maintaining the rhinos in zoos once captured. Losses of rhinos during capture (3 deaths) and particularly during post-capture (6 deaths) have been high. There is also some doubt that all were really "doomed" animals, because some captures were made very close to PAs containing rhinos, but this strays from my argument. To date (May 1991), 21 rhinos are in captivity and there have been no births, except to one female who was pregnant when captured. In contrast, at $230 per sq km, $2.5 million could effectively protect 700 sq km of prime rhino habitat, say in Gunung Leuser or Kerinci Seblat NPs in Sumatra, for nearly two decades. At their apparent normal densities of 0.1 rhino per sq km, an area of this size could hold a population of 70 Sumatran rhinos (see Nardelli 1988) which, with a rate of increase of 0.06 per year shown by all other rhino species given adequate protection, would be expected to give birth to 90 calves in this period. Apart from Gunung Leuser and Kerinci Seblat, no area holds 70 Sumatran rhinos (see Nardelli 1988), and many "doomed" animals do indeed need to be captured and re-located. But "doomed" animals could instead be termed "stragglers" and rounded up into sanctuaries following the Kenyan model (see Brett 1992 in this volume). Clearly, a proportion of the capture costs would then still be needed and capture would be associated with some mortality. However, rhinos could be released more quickly into a 700 sq km sanctaury of natural habitat, created at say Endau Ropmin in Malaysia, instead of being transported to zoos. Costs and mortality would be cut down considerably (the latter by two-thirds), still leaving money for protection of the area for 5 or 10 years and more founders, and more per births for the same financial outlay.

ROLE OF RESEARCH

At this conference we have heard, and will hear more, fascinating information on many aspects of "rhino-ology". For the rhino enthusiast and researcher alike, myself included, there are no end of possible research topics to enjoy. But what are the motives of the researcher in all this, and how much of the research carried out in the name of rhino conservation do rhinos actually need to enhance their chances of survival? All researchers, whatever their subject, undertake research because they enjoy it but many justify their research to themselves, to their audiences, and to possible funders as having some practical relevance, in this case to the conservation needs of rhinos. It would be interesting to know how much all the research carried out in the name of rhino conservation has cost, and compare that with the amount spent on *in situ* protection of rhinos. While basic knowledge clearly has its place, I suggest we are becoming over-cerebral in relation to management objectives and funds available for real conservation. The Serengeti in Tanzania is an example where this has been documented (Boshe 1990). As one of the world's best known ecosystems (Sinclair and Norton-Griffiths 1979), much of the research has been carried out in the name of the Serengeti's management needs. But in 1989/90, the researchers brought in research funds worth over $50 per sq km (and most of this research was irrelevant to conservation), while the mangement authority was able to spend only $20 per sq km for its protection (Boshe 1990). Hence the Serengeti has lost many of its economically valuable large mammals, including its black rhinos, to poachers. And I doubt if this is an isolated example of this phenomenon.

CONCLUSIONS

Rhinos are a source of major concern to the world's conservationists, and will continue to be so for many decades. In this review, I have attempted to show that the theory of conservation biology has played little part in the practical conservation of rhinos, and that instead the theories of economics probably better explain successes and failures in rhino conservation. All successes in rhino conservation to date have arisen from protecting small populations *in situ*. The pragmatic option to achieve further successes in rhino conservation over the next few decades appears to be to build up small populations *in situ* and maintain them in natural habitat with minimum interference and, once the population has built up, to translocate excess rhinos to unoccupied natural habitat. Zoos have teamed up with theoreticians to promote the role of captive breeding, but the most cost-effective role for zoos in rhino conservation appears to be in funding practical field research and in public education. This is clearly a very different statement to saying that zoos should cease efforts at captive breeding in order to maintain their own stocks.

Sadly, nothing I have said has any originality. Theodore Hubback was a naturalist of the old school and, in 1939, he wrote that the only hope of preventing the extinction of the Sumatran rhino is "to constitute inviolable sanctuaries in their own habitat where a suitable environment is known to exist. These sanctuaries must be properly guarded and freed from human interference." These words are as true today as when they were written over 50 years ago. Perhaps a hundred thousand or more rhinos of all species have been lost to poachers in that time. And I believe the challenge for conservationists more than ever remains taking the resources to the rhinos and their associated ecosystems and biodiversity, rather than taking some of the more spectacular elements of the biodiversity to the resources.

POSTSCRIPT

This paper was presented in May 1991 but not actually written until some months later. Two points are now worth making that have some relevance to the arguments for and against captive breeding. First, two papers at the conference (this and Stanley Price 1992) caused some upset in the zoo community (Wachs et al., 1991), which has since provided evidence that a core group of zoos are achieving rates of increase of 0.04 per year in black and Indian rhinos (Foose et al., 1991) and evidence of a reassessment of captive breeding programmes (Wachs et al., 1991). Second, the conference was held in an atmosphere of some concern amongst field biologists over plans to move a large nucleus of Javan rhinos from Ujung Kulon to captive breeding facilities (Seal and Foose 1990). At a workshop in October 1991, the Indonesian management authority decided against this option and determined their first priority was the establishment of a second population in natural habitat. These all appear welcome developments arising from a very successful conference.

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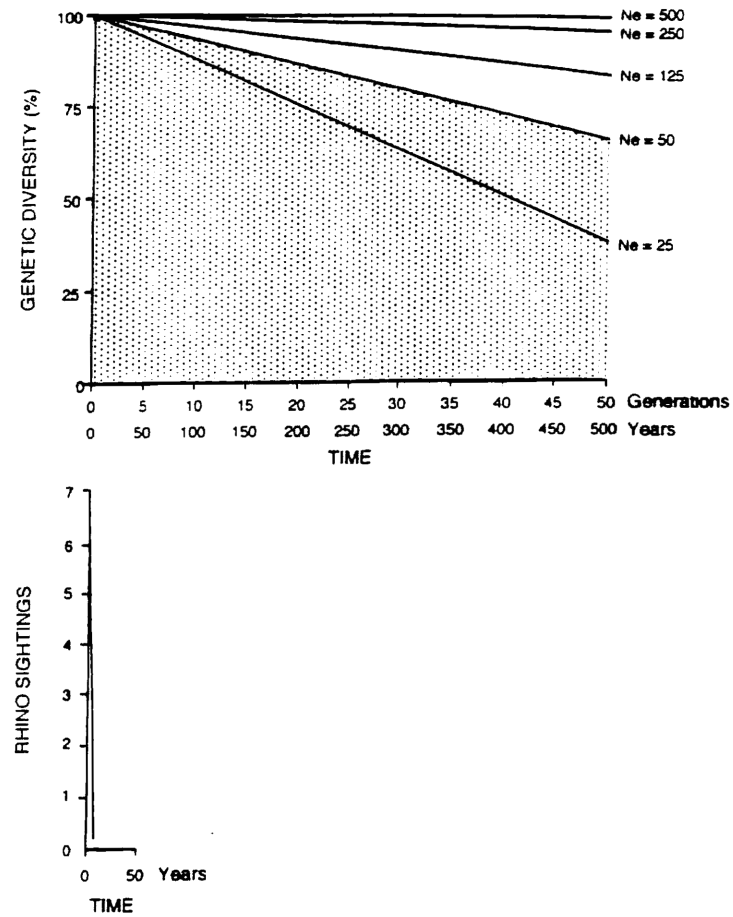

Figure 1. (a)-The proposed loss of genetic diversity in effective populations (Ne) of different sizes over numbers of generations (from Foose 1987), and converted to rhino chronological time with 1 generation = 10 years; (b) the actual losses of black rhinos from a large population in Luangwa valley, Zambia during 1980-1985 (from Leader-Williams et al. 1990a).

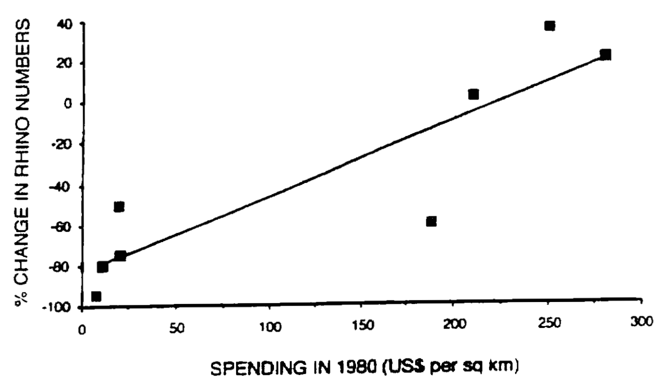

Figure 2. The success of protecting rhinos relative to spending on conservation areas in nine African countries with populations of over 50 rhinos in 1980 (from Leader-Williams and Albon 1988).

FAILURES, SUCCESSES AND INITIAL SUCCESSES IN RHINO CONSERVATION

Species	Subspecies	Failure	Success	Initial success	Promising increase
Sumatran	*D.s. sumatrensis*	+			
	D.s. lasiotis	+			
	D.s. harrissoni	+			
Javan	*R.s. sondiacus*				(+)
	R.s. inermis	?E			
	R.s. annamiticus	+			
Indian			+		
Black	*D.b. longipes*	+			
	D.b. michaeli			+	
	D.b. minor	+		(+)	
	D.b. bicornis			+	
White	*C.s. cottoni*				+
	C.s. simum		+		

Figure 3. Failures, successes, initial successes and promising increases in rhino conservation, taken from a general survey of the rhino literature. ?E = likely extinction; D.b. minor has entries in two columns because it has not been successfully conserved over much of is range, but has shown the beginnings of a success story in one range state.

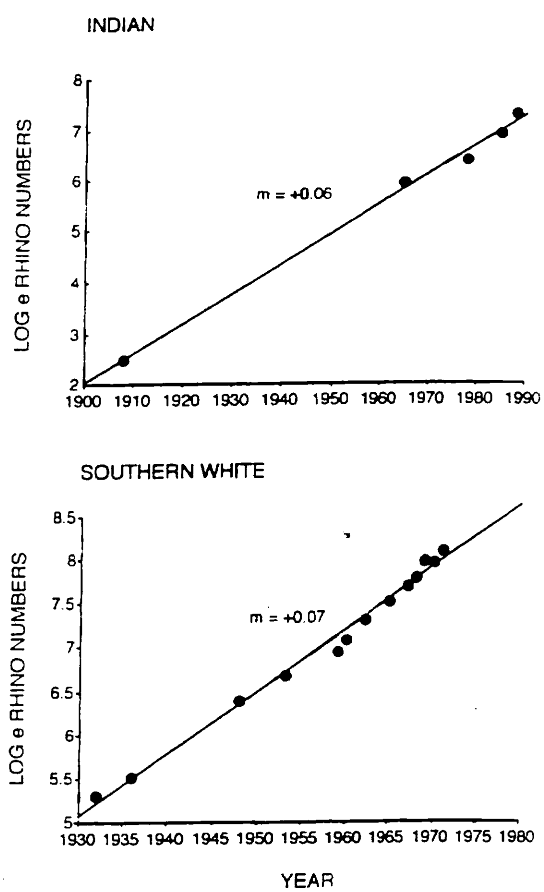

Figure 4. The rates of increase achieved by protecting small populations of Indian rhinos in Kaziranga and of southern white rhinos in Umfolozi/Hluhluwe, which have recovered to populations of around 1500 and 4000 respectively (data from Nardelli 1988; Vigne and Bradley Martin 1991; Owen-Smith 1981).

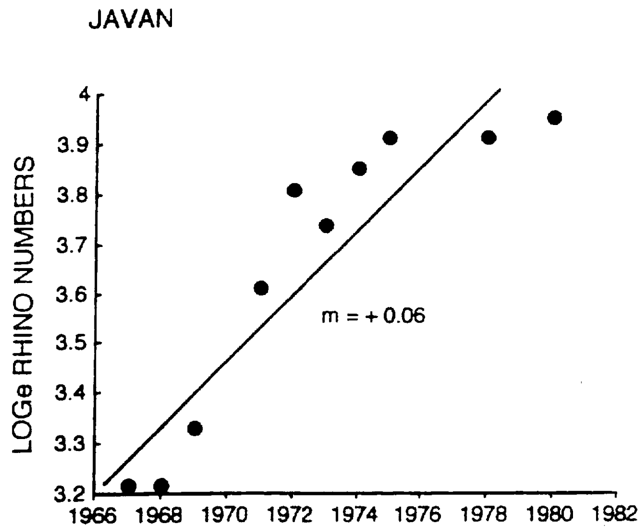

Figure 5. The promising increase shown by Javan rhinos in Ujung Kulon when given better protection (data from Amman 1985).

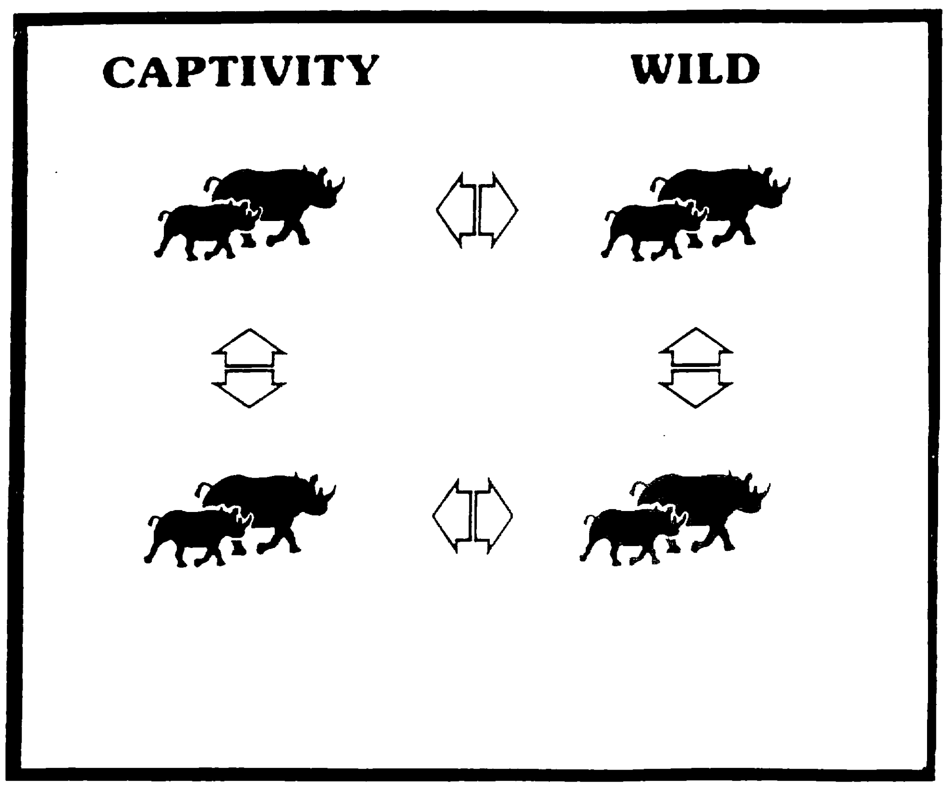

Figure 6. Metapopulation management with interchange between small populations <u>in situ</u> and <u>ex situ</u> (after Foose 1987, 1992).

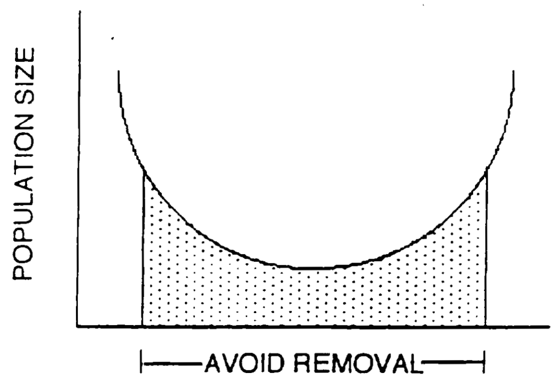

Figure 7. A scheme of the stage at which the removal of a large nucleus of rhinos (or other animals) for experimental attempts at captive breeding should be avoided.

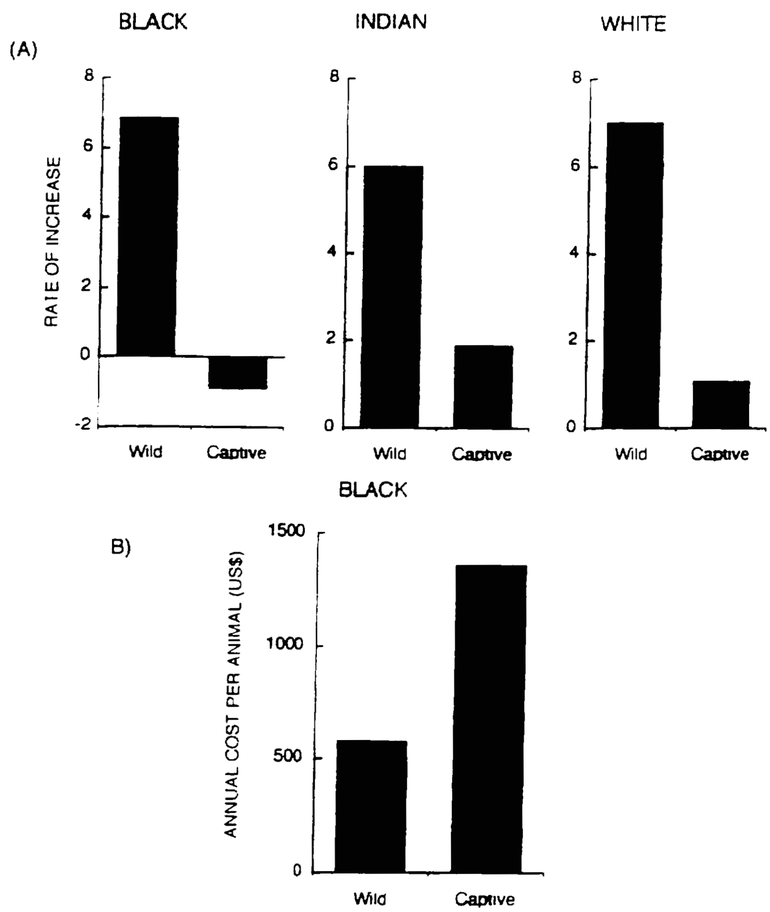

Figure 8. (a) rates of breeding of black, Indian and white rhinos in well protected sanctuaries and in captive populations (data from Brett 1992, Figure 4 and Anon. 1972-1989).
(b) cost of maintaining rhinos in the wild and in captivity (based on Leader-Williams 1990).