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The Elephant Problem in Ruaha National Park, Tanzania*

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ABSTRACT

This review paper attempts to provide a comprehensive picture of the elephant over-population problem in Ruaha National Park by drawing together data on elephant numbers, elephant feeding behaviour, and the effects of elephant browsing on woodlands.

Elephant numbers appear to have increased since about 1946 following a change in human distribution and a simultaneous change in the rainfall pattern. Elephants have caused a dramatic reduction in woodland density and there is no regeneration. The arguments for and against six management options are discussed: providing artificial water supplies, improved fire control, reducing human pressures, culling, non-interference, and poaching.

INTRODUCTION

The tragedy of the African elephants *Loxodonta africana* Blumenbach is that their total numbers are falling but at the same time conservationists

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argue about culling elephants in national parks. Cases of elephant over-population have been recorded in 22 national parks that contain elephants (Bell, 1973). They all follow the same pattern: an increase in elephant numbers, decline in woody vegetation, and a debate about culling elephants. In nine out of 22 cases elephants have been culled (Bell, 1973). In some the debate has never been settled or no decision has been made. In recent years a final act has been written to the elephant tragedy: poaching gangs have stepped in and (illegally) culled the elephants (Douglas-Hamilton, 1979; Eltringham & Malpas, 1980; Malpas, 1981).

A good example of an elephant problem is Ruaha National Park in south-central Tanzania (Fig. 1). This park follows the now traditional

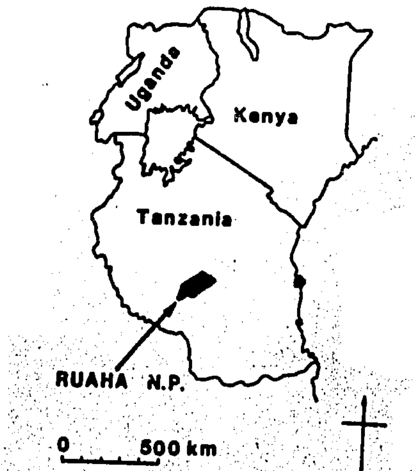

Fig. 1. Map of East Africa showing location of Ruaha National Park.

pattern, save perhaps for the final stage. It provides a case history from which we may learn how to cope with over-population problems in the future, whether of elephants or of other large herbivores.

The Saba River Game Reserve was established in the early years of the century (Greenway, 1969). In 1946 this wilderness, approximately 20 000 km^2 of *Brachystegia*, *Acacia*, and *Commiphora* dominated woodlands and bushlands, was regazetted the Rungwa Game Reserve. The south-eastern half was declared the Ruaha National Park in 1964 (Bjornstad, 1976). There was already evidence of over-browsing by elephants (Savidge, 1968*a*). The first Park Warden reported that trees in all parts of the Park were being damaged, that regenerating trees were

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being killed, and that the rate of tree damage was increasing (Savidge, 1968*b*). He recommended that elephants should be culled immediately (Savidge, 1968*b*). Aerial counts in 1965 and 1966 showed that the elephant density in the worst affected areas was about $1{\cdot}5\,km^{-2}$ (Savidge, 1968*a*; re-analysed by Barnes & Douglas-Hamilton, 1982). Vesey-FitzGerald (1973) surveyed this part of the Park in 1970 and argued that the loss of adult trees allowed regeneration to succeed and so the woodlands re-established themselves. Therefore he believed the culling would solve nothing. But in the following year Bjornstad (1971) found little regeneration and reported a high rate of tree loss. Aerial sample counts in 1972 and 1973 showed that the elephant density over the whole Park was $1{\cdot}6\,km^{-2}$ (Norton-Griffiths, 1975).

The Tanzania National Park authorities felt that no management decisions could be made until more information could be collected. The purpose of the Ruaha elephant project was to answer four key questions:

1. What is the trend in the elephant population?
2. What vegetation components are the elephants eating?
3. What effects are the elephants having upon the vegetation?
4. What management options are open?

The apparent trend in elephant numbers, their feeding, their effects on three tree populations, and the simulated effects of culling have been described in separate papers (Barnes, 1980; 1982*a*,*b*; Barnes & Douglas-Hamilton, 1982). In this review paper I wish to draw the threads together to give an integrated picture of the problem. I will then describe the possible management options, impartially presenting the arguments for and against each option. In doing this, I adopt the position that it is the role of the scientist to provide the technical information and set out the management alternatives. It is then the role of the manager to choose the most appropriate management policy for the Park (Bell, 1980, 1981, in press *c*).

RUAHA NATIONAL PARK

Ruaha is Tanzania's second largest national park. It covers $10\,300\,km^2$, and the Rungwa and Kizigo Game Reserves on its northern boundary cover another $15\,000\,km^2$.

Most of the Park lies on Precambrian rocks of the Tanzanian granitic

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shield, but it also includes the sedimentary deposits of the rift valley. The mean annual rainfall at Park HQ is 580 mm; 94% of the year's rain falls between December and April. There is a gradient of increasing rainfall from east to west, with the eastern part of the Park falling in Pratt & Gwynne's (1977) ecoclimatic zones V (arid), and the western part in ecoclimatic zones IV (semi-arid) and III (dry humid to semi-arid). The Park lies on the ecotone between the *Brachystegia* (miombo) woodlands of south and west Tanzania and the *Acacia* and *Commiphora* woodlands and bushlands of north and east Tanzania. Bjornstad (1976) recognised four principal vegetation communities. The miombo covers the western half of the Park. In the far western mountainous corner is an evergreen upland/submontane forest dominated by *Drypetes gerrardii*. The drier eastern half of the Park is a *Commiphora-Combretum* dominated zone of woodland and bushland. The far north-eastern corner of the Park is covered by *Acacia*-bushed grasslands and bushlands.

Over 1600 plant species (Bjornstad, 1976) and over 370 bird species have been recorded within the Park. Both greater and lesser kudu *Tragelaphus strepsiceros* Pallas and *T. imberbis* Blyth are found in the Park, and also roan *Hippotragus equinus* Desmarest and sable antelope *H. niger* Harris and Lichtenstein's hartebeest *Alcelaphus lichtensteini* Peters.

THE PROBLEM

The trend in elephant numbers

Barnes & Douglas-Hamilton (1982) compared aerial counts of the rift valley sector of the Park made in 1965/66 and in 1977, and also aerial counts of the whole Park made in the dry seasons of 1972 and 1977. These comparisons showed an apparent increase in elephant density of 8–10% per annum between 1965 and 1977. Elephants were the only large herbivore to show a significant increase between 1972 and 1977. The apparent increase could not be accounted for by differences in observer efficiency, and Barnes & Douglas-Hamilton (1982) attributed it to two causes: human pressures and rainfall.

There was an inverse relationship between elephant density and human pressures (settlement and hunting) over the 31 500 km^2 census zone. The highest elephant densities were in the National Park where there was least

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poaching and no human settlements. The highest densities within the Park were in the rift valley sector, which is the most frequently patrolled part of the ecosystem (Barnes & Douglas-Hamilton, 1982). These results suggest that elephants are driven into the areas of lowest human density and at the same time their numbers outside the Park are probably reduced by hunting.

Before 1946 settlements were scattered throughout the area which is now covered by the National Park (Savidge, 1968*a*). They were close to water so in the dry season elephants were denied access to many potential water sources. The people lived by growing crops in the wet season and hunting in the dry (W. B. Summay, pers. comm.). Elephants were harassed to keep them away from crops (W. B. Summay, pers. comm.) and they were also hunted (Savidge, 1968*a*). Thus elephants and humans were probably distributed at low density throughout the whole ecosystem. After 1946 the people were resettled outside the Park boundaries (Savidge, 1968*a*). Human pressures were then concentrated around the edges of the ecosystem. This would have encouraged elephants to migrate into the Park where there were no human pressures. The unrestricted access to water may have resulted in a change in juvenile survival (Laws, 1969).

Rainfall records for Madabira, about 60 km from Park HQ, date from 1924. They show a 15-year period of higher rainfall between 1955 and 1970 (Barnes & Douglas-Hamilton, 1982). Since primary production is proportional to rainfall (Rosenzweig, 1968), the probable improvement in the food supply would have resulted in higher conception rates, a shorter mean birth interval, a lower age of puberty, and possibly higher juvenile survival (Laws, 1968; Laws & Parker, 1968).

The higher rainfall coincided with the last stage of the resettlement programme and the declaration of the National Park in 1964. Therefore it is possible that the higher rate of reproduction plus the increased immigration acted together to cause an accelerated increase in elephant numbers after 1964.

The woodlands

The 1977 aerial census (Barnes & Douglas-Hamilton, 1982) provided a large-scale picture of the extent of tree damage over an area of 31 500 km^2. The census zone covered Ruaha National Park, the Rungwa and Kizigo Game Reserves on the Park's northern boundary, and part of

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the Game Controlled Area to the east and south. The general pattern of tree damage corresponded with the distribution of elephants. Woodlands in the National Park and the southern sector of the Game Reserve were suffering the highest rates of loss. There were no untouched woodlands in the National Park. The rate of tree mortality fell away towards the edges of the census zone, i.e. closer to areas of human disturbance.

Ground surveys showed in more detail what the elephants were doing to the vegetation. In the wet season grasses formed 60–70 % of the plants eaten by elephants but in the dry season they fed mainly on trees and shrubs (Fig. 2a). In the 1975 dry season more leafy than woody browse was eaten, but a third of the diet still consisted of woody plant parts (Fig. 2b).

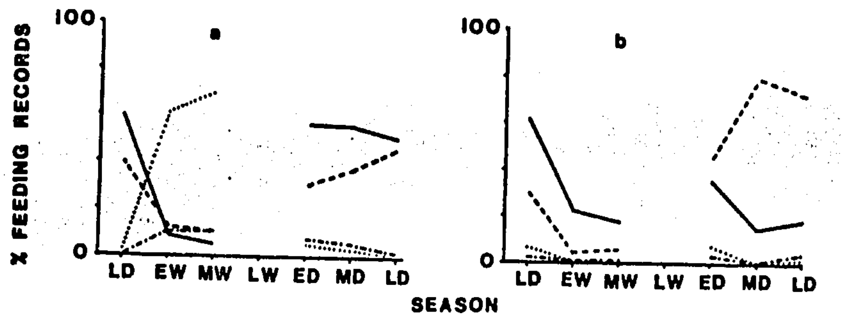

Fig. 2. The composition of elephants' diet at different times of year. Data from Barnes (1982*b*), bulls and cows combined. EW, early wet season; MW, mid wet season; LW, late wet season; ED, early dry season; MD, mid dry season; LD, late dry season. No data were collected in the late wet season. (a) Composition of the diet by plant types: (.........) grasses; (–·–·–·–) herbs; (– – – – –) shrubs; (———) trees. (*b*) Composition of the diet by plant parts: (———) browse leaves; (– – – – –) wood; (–·–·–·–) bark; (.........) fruits.

In the harsh dry season of 1976 woody plant parts accounted for up to four-fifths of the diet. It seems unlikely that elephants were doing much damage in the wet season when they were feeding mainly on grass and on the green leaves of trees and shrubs (that is, the production of that particular season). But in the dry season when they fed on woody plant parts they were removing the product of many years' growth. To use a financial analogy, they were feeding on the vegetation interest in the wet season and on the vegetation capital in the dry season. The message is clear: most damage to the woodlands was caused during the dry season.

The age distribution of the baobab tree *Adansonia digitata* L. population indicated that elephant feeding pressure fell heavily on the

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younger trees (Barnes, 1980). A survey of the *Acacia albida* Del. population revealed that the stratum with the lowest browsing mortality had some regeneration, whereas the other strata, suffering greater elephant damage, had none. Similarly the size distribution of the *Commiphora ugogensis* Engl. woodlands showed no young trees at all (Barnes, 1982*a*). Wing & Buss (1970), Laws *et al.* (1975), Caughley (1976*a*), and Olivier (1978) have also described elephants' preference for young trees.

Forty per cent of the *Acacia albida* trees had perished. All but a small fraction had been killed by elephants. Sixty-seven per cent of the *Commiphora ugogensis* trees had been killed over a 6-year period (Barnes, 1982*a*). The scale of woodland decline is illustrated by Bjornstad's (1971) estimate that the tree density of untouched *Commiphora ugogensis* woodlands (before elephants had a significant effect) was about 250 ha^{-1}. In 1977 the mean tree density of the woodlands was 9 ha^{-1} (Barnes, 1982*a*).

The estimated rate of loss of baobab trees was considerably less than that of *Commiphora ugogensis* and *Acacia albida*—3 % killed in one year. But the pattern of baobab loss may be one in which each elephant kills a constant number of trees each year. If this hypothesis is correct, then the baobab population will experience a rapid decline to zero, even if the elephant density does not increase further (Barnes, 1980). If the elephant population continues to increase, then the baobabs will suffer an ever-increasing rate of decline and plunge to extinction (Barnes, 1982*a*).

The high loss rates of adult trees and the lack of regeneration show that all three species are in decline. The same is probably true of all the other tree species. However, the pattern of tree mortality caused by elephants varies between tree species. A simulation model showed that *Acacia albida* is more likely to survive than *Commiphora ugogensis*, which is predicted to disappear completely (Barnes, 1982*a*). The pattern of tree mortality experienced by the baobab tree population means that it will disappear very rapidly (Barnes, 1982*a*), as has already happened in Tsavo National Park (Leuthold, 1977).

The picture is further complicated by fire. The tree surveys were made in a part of the high elephant density stratum which was completely protected from fire. Between a quarter and a third of the Park may be burnt each year (Barnes, 1979). The combined effects of fire and elephants on woodlands have been described by Buechner & Dawkins (1961) and Laws *et al.* (1975). The rate of tree loss will be even higher in the areas of

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the high elephant density stratum where fires are frequent. Even in the lower elephant density stratum the rate of tree loss will be high because of the combined effects of elephants and fire.

The mutilated appearance of most shrubs and their importance in the elephants' diet (Fig. 2*a*) indicate that shrubs must also be suffering from the heavy elephant browsing pressure. Shrub seedlings are probably just as vulnerable to elephants and fire as are tree seedlings. But established shrubs may well be able to withstand heavy browsing: R. H. V. Bell (pers. comm.) has pointed out that many miombo species respond to heavy elephant browsing by coppicing leading to an increase in browse production.

The large herbivores

Diversity is a measure of both the number of species and the abundance of each species. In order to compare the diversity indices of the 1973 and 1977 counts (Norton-Griffiths, 1975; Barnes & Douglas-Hamilton, 1982), in which different numbers of species were counted, the same eight species were used to calculate the diversity index H and the evenness, or equitability of dispersion, index J (Zar, 1974):

$$H = -\sum p_i \ln p_i$$

$$J = \frac{H}{\ln k}$$

where p_i is the proportion of the ith species in the sample and k is the number of species.

Between 1973 and 1977 there was a decrease in large herbivore diversity (Table 1a). This, and the drop in the equitability of dispersion, could well be due to the greater abundance of elephants in the sample. But when these indices were computed without elephants (Table 1b), both H and J were still lower in 1977. This indicates that the composition of the whole herbivore community may have changed.

Barnes (1979) found some evidence to suggest that there may have been a change in the Park's large herbivore community: the biomasses of the larger species were higher than expected while those of the smaller species were lower than expected. Cobb (1976) presented density estimates for the Tsavo area in the early years of this century, when no elephants were seen, and also for the 1970s, when elephants dominated the biomass.

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TABLE 1
Indices of Diversity (H) and Evenness (J) for the Ruaha National Park (RNP), Rungwa and Kizigo Game Reserves (GR), and the Game Controlled Area (GCA) (Only species that were counted in both the 1973 and 1977 aerial censuses (Norton-Griffiths, 1975; Barnes & Douglas-Hamilton, 1982) were included in the calculations. k is the number of species.)

Index	*1973*	*1977*		
	RNP	*RNP*	*GR*	*GCA*
(*a*) *Including elephants* ($k = 8$)				
H	1·65	1·49	1·39	1·51
J	0·79	0·72	0·67	0·73
(*b*) *Excluding elephants* ($k = 7$)				
H	1·48	1·36	1·10	1·30
J	0·76	0·70	0·56	0·67

Although the total biomass density was the same, he suggested that the presence of elephants had depressed the numbers of most other species.

As the elephant density rises, the more palatable plant species can be expected to disappear as the vegetation deteriorates as a result of the heavy browsing and grazing pressure. This would place the smaller herbivores, which need to maintain a higher proportion of rich plant parts in their diets (Bell, 1971; Jarman, 1974) at a disadvantage. Another explanation for the change in the herbivore community structure is suggested by Bell (in press *a*): elephants modify the habitat to the detriment of the grazing herbivores, particularly those of the miombo, by coppicing miombo and preventing grass production.

Any conclusions drawn from population estimates must be viewed with some caution as aerial censusing is open to many errors and biases (Pennycuick & Western, 1972; Caughley, 1974; Caughley & Goddard, 1975; Norton-Griffiths, 1976). There are three main problems in comparing the 1973 and 1977 aerial counts: (a) The issue of differential observer efficiency was discussed by Barnes & Douglas-Hamilton, 1982); (b) the low densities and clumped distributions of most species cause large sampling errors; (c) visibility varies with season, particularly in miombo (R. H. V. Bell, pers. comm.), which covers half the Park. In the wet season elephants and buffalo tend to be more visible compared with the smaller species. So in the dry season of 1977, if herbivore numbers had remained unchanged since 1973, one would have expected to see relatively more of the smaller herbivores than in the wet season count of 1973. In fact, out of the five smaller species, only one appeared to have increased (Barnes & Douglas-Hamilton, 1982), which tends to reinforce the earlier conclusion that there has been a shift in relative numbers. This shift is in the direction to be expected (Bell, in press *a*).

The suggestion made here is that the elephant problem does not involve just interactions between elephants and the vegetation, or even interactions between human pressures, elephants, and the vegetation. It involves interactions between human pressures, all large herbivores (including elephants), and the vegetation.

Other ecosystem components

There are no data on the trend in the herb layer. The fact that elephants may feed on grass in the wet season, when it is super-abundant, leads one to guess that their grazing effect may not be important. On the other hand, heavy grazing in certain areas may reduce the frequency and intensity of fires by leaving less grass to burn in the dry season.

There are also no data on the trends in the insect, bird, and small mammal populations. It is reasonable to suppose that changes in their food, host, and cover plants, combined with the microclimatic changes, would cause changes in their population levels.

Nothing is known about changes in the Park's soil conditions or ecosystem nutrient stores. However, the decline in the Park's woody vegetation cover can be expected to have had important implications. Each woodland tree alters its environment by modifying the microclimate beneath it (Geiger, 1965; Cousens, 1974). Changes in the plant species result in a different type of ground litter and therefore in the nutrient input to the soil. Microclimatic and nutrient changes also affect soil microorganisms. The combined effect of these changes must be alterations in the chemical and physical properties of the soil.

A large proportion of the community's inorganic nutrients is locked up in the tree biomass (Cousens, 1974). This is especially true on poor soils, such as those on which the miombo woodlands stand. After death a tree's nutrients may be taken up by other plants or leached away (Cousens, 1974). They are more likely to be leached away if the tree is burnt (Wein, 1978). Cousens (1974) believes that when large trees are replaced by shrubs and grasses the ecosystem nutrient capital is run down. In Africa a high proportion of the nitrogen-fixing plants are trees, e.g. *Acacia* spp. (Harris & Fowler, 1975). Nitrogen is volatilised by fire (Harris & Fowler, 1975; Wein, 1978) and therefore frequent bush fires combined with the loss of the main nitrogen-fixing plants and their nutrient stores would lead to impoverishment of the ecosystem.

Woody vegetation influences the water content of the subsoil (Walker *et al.*, 1981) and so the loss of woody cover may lead to greater soil water deficits in the dry season which may further affect the vegetation communities. Reduced vegetation cover will bring the danger of soil erosion and greater sediment loads in the Great Ruaha River. These will be deposited behind the Mtera Dam which is being constructed lower down the river.

MANAGEMENT OPTIONS

Artificial water supplies

Bore holes could encourage elephants to spend less time near the river and therefore reduce tree damage in the riverine areas.

However, tree damage, although especially bad within 1 or 2 km of water, is serious throughout the Park (Barnes & Douglas-Hamilton, 1982). Bore holes would be unlikely to have a significant effect upon the distribution of tree damage except to make it worse within a 2 km radius of each bore. Further, if access to water was an important factor in the increase of elephants after 1946, as suggested above, then providing artificial water supplies could well exacerbate the problem. Sikes (1966) argued that artificial water supplies in the dry season were a major factor in the evolution of the Tsavo elephant problem.

Fire control

Fire control would reduce the mortality rate among saplings and damaged trees (Buechner & Dawkins, 1961; Laws *et al.*, 1975). Fewer fires would also reduce the loss of nutrients (e.g. nitrogen) from the system. Fire control is best effected by a network of fire-breaks and a higher patrolling frequency to deter the main causes of fires within the Park: poachers and illegal honey-gatherers. Pellew's (in press) simulations indicate that fire control could be the most effective management strategy for saving trees.

Fire control is unlikely to be effective in the Ruaha environment because of the lack of staff and vehicles and the difficulty of preventing illegal access to the Park. Patch burning in the early dry season, so as to minimise damage to young trees and remove tinder which would otherwise produce hot fires in the late dry season, may be a more effective strategy (R. H. V. Bell, pers. comm.).

Human pressures

The most cost-effective management strategy is likely to be the one that strikes at the root causes of the problem. Reducing human pressures around the Park would reduce elephant immigration into the Park (but only if there are still elephants left outside). This could only be done by resettling the people round about in order to create a buffer zone. But a buffer zone would only be a short-term solution, since the elephant population would gradually expand to fill it. Then the elephant problem would include the buffer zone as well as the Park.

Any further extension to the Park would be difficult to justify, even if the local people were willing to permit it, because (a) the Park and the two Game Reserves already cover a total area of approximately 25 000 km^2 (larger than Wales), and (b) Tanzania's human population, and therefore its demand for land and associated resources (e.g. firewood), is increasing.

Culling

Culling the elephant population is clearly the obvious way to reduce the rate of vegetation change. Ruaha is only part of an ecosystem that embraces the Game Reserves and the surrounding area. Therefore there is a case for drastic management when the natural regulatory mechanisms break down or are overloaded by artificially high rates of change in one or more ecosystem components. If human activities are the main cause of the Ruaha problem, then it can be argued that man ought to redress the ecological imbalances he has caused. Further, we know very little about tropical ecosystem processes, so we cannot predict the long-term consequences of the apparent imbalance between elephants and the

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vegetation. Therefore a low risk strategy would be to hold the herbivore population at a safe level by culling until we have learnt more about the system dynamics of national parks (Bell, 1981). Although culling would be expensive (man-power, vehicles, etc.) the sale of ivory means that it would be self-financing.

It is difficult enough to obtain an accurate estimate of the number of elephants within the Park, and even more difficult to assess how many of them should be killed to stabilise the vegetation changes. Coe *et al.*'s (1976) equation relating large herbivore biomass to rainfall predicts a large herbivore biomass of 3831 kg km^{-2} for Ruaha. The combined biomass of the other large herbivores in the Park was estimated to be 1263 kg km^{-2} (Barnes, 1982*a*). Subtracting this from 3831 kg km^{-2} gives a predicted elephant biomass of 2568 kg km^{-2}, which is equivalent to 1·49 elephants km^{-2}. To reduce the elephant density from the 1977 estimate of 2·41 km^{-2} (Barnes & Douglas-Hamilton, 1982) to 1·49 km^{-2} would require killing 9384 elephants. If an elephant density of 1 km^{-2} causes a *Commiphora ugogensis* mortality rate of 5% per annum (Barnes, 1982*a*), then it may be wise to bring the elephant density down to no more than 1 km^{-2}. This would mean killing 14 380 elephants.

Simple elephant-and-tree simulation models indicated that it may be necessary to kill 75% of the Ruaha elephant population to have a noticeable effect on the trend in tree numbers (Barnes, 1982*a*). This would mean killing 18 470 elephants.

These can only be approximate estimates. They give some idea of the scale of the killing that would be involved. Large-scale operations to cull more than 10 000 elephants have never been attempted. Such operations could only be conducted during the dry season, and even then the Ruaha terrain would make them very difficult.

A simple benefit/cost analysis during a simulation model suggested that by the time an elephant problem is recognised, the time of maximum cost-effectiveness of culling (measured as the numbers of trees saved per elephant shot) has already passed (Barnes, 1982*a*).

Thus an argument against culling is the sheer number of elephants that would have to be killed and the problems that would be encountered in killing them. The second argument is that the benefit/cost analysis indicates that culling the Ruaha elephants in the 1980s would save few trees for the number of elephants that would have to be shot. Non-biological questions associated with culling animals in national parks have been discussed by Barnes (1979) and Bell (1981, in press *c*).

Non-interference

It can be argued that man has no right to interfere in national parks. If it can be shown that an area is a natural ecosystem and that there are regulatory feedback mechanisms which will restore the equilibrium, or that the problem is part of a naturally-occurring cycle (in which case it is not a problem), then non-interference is the wisest policy. For instance, Phillipson (1975) argued that variations in elephant numbers at Tsavo were part of a climatic cycle. Therefore culling was not a solution and it was wiser to let nature take its course.

For a national park system that is short of money, non-interference has the considerable advantage that it is a cheap policy. But perhaps the most frequently used (but rarely stated) argument for non-interference is that the other management options (e.g. culling) require difficult decisions or are too dreadful to contemplate.

Non-interference does not necessarily mean that the elephant problem will conveniently go away. It relies upon the cycle theory: that periods of over-abundance will be followed by periods of low abundance. Yet evidence in favour of elephant cycles is weak. Phillipson (1975) argued that at Tsavo there was a 10-year rainfall cycle superimposed on a 43–50 year cycle. But his calculations were based on a short run of data—four years—of which two were the worst in Tsavo's history. Secondly, he calculated the probability of an event (drought) was 0·1 and therefore it would occur every ten years. It would have been more correct to say that the likelihood of the event occurring in any one of those years was 0·1. However his argument is supported by the occurrence of droughts in 1939–40, 1949–50, and 1960–61, and of major droughts in 1836, 1887, 1921, and 1970–71 (Phillipson, 1975). On the other hand, Cobb (1976) could find no evidence of a 10-year cycle when he applied a Fourier analysis to the Tsavo records. Even if rainfall cycles did exist, they could not explain why all elephant problems are in phase in different parts of Africa (Bell, 1973; Malpas, 1978), nor the magnitude of the elephant density fluctuations in Uganda (Malpas, 1978).

A stable limit cycle (Caughley, 1976*a*) requires a closed system with the elephant and tree populations being interdependent. Elephants can of course migrate away and they feed on a very wide range of plant species (Olivier, 1978). And as Caughley (1976*b*) has pointed out, large mammals are unlikely to show stable limit cycles because they are at risk at the low point of the cycle when random effects could lead to extinction. Further, there is little empirical evidence in support of an elephant/tree stable limit cycle. Caughley (1976*a*) recorded a baobab tree age distribution which he suggested was the result of an elephant cycle. But a simulation model shows that the same age distribution can be produced by an increasing elephant population feeding on a baobab population which starts the run with a stable age distribution (Barnes, 1979).

Noy-Meir (1975) and May (1977) have presented theoretical mathematical models which describe a non-cycling relationship between herbivore and vegetation populations. Their models make two predictions: (a) vegetation biomass declines slowly as the herbivore biomass rises, but a marked decrease in vegetation biomass can happen very suddenly; (b) marked changes in the condition of the herbivore population will only be seen *after* there has been a dramatic change in the vegetation. The break-point, where the sudden vegetation change occurs, can be brought about either by increasing herbivores, or by a drought, or by both. These theoretical conclusions are supported by reports of habitat over-use in the literature. Among reindeer *Rangifer tarandus* (Klein, 1968), deer *Odocoileus virginianus* and *O. hemionus* (Klein, 1970), Himalayan thar *Hemitragus jemlahicus* (Caughley, 1970), buffalo *Syncerus caffer* (Sinclair, 1977), and snowshoe hare *Lepus americanus* (Keith & Windberg, 1978), expanding populations have crashed following over-use of the food resource. After reviewing the literature on ungulate eruptions, Caughley (1970) concluded that rapid increases in ungulate numbers are generally terminated by habitat modification caused by the high ungulate densities. Even though homeostatic mechanisms, such as density-dependent variations in reproductive and mortality rates, may operate to slow the rate of population increase, these examples indicate that herbivore numbers do not start to decline until after serious vegetation changes have occurred. The question then is: will the vegetation recover? Following the reduction of the Tsavo elephant population and several years of high rainfall, the woody vegetation of Tsavo National Park is now regenerating. There is no evidence that there have been any changes in soil chemistry or structure (Douglas-Hamilton, 1979). But Noy-Meir (1975) cited evidence to show that following a herbivore crash the carrying capacity is set permanently at a lower level than before.

In summary, non-interference may lead to a population crash. This will involve both death by starvation for the elephants and other herbivores, as was seen at Tsavo in 1970–71 (Corfield, 1973), and vegetation degradation which may be permanent.

Another argument against non-interference is that national parks are

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rarely natural ecosystems. The creation of a national park introduces a number of artificial factors into the environment and these may lead to changes in the ecosystem which should be counter-balanced by management.

Poaching

I include poaching under the heading of 'management' because (a) it is a form of illegal wildlife management, and (b) it is the result of inefficient management by those appointed to enforce the wildlife conservation laws. This does not mean that the officers concerned are incompetent. It usually stems from the fact that they are not supplied with the tools for the job: vehicles, spotter planes, radios, rifles, or even boots.

Large-scale poaching is most likely to take place if a policy of non-interference is adopted.

Until the late 1970s Ruaha received little attention from commercial poachers, mainly because of its remoteness (no main roads or railways pass through it). In 1977 poaching for rhinoceros *Diceros bicornis* increased and this was followed by an upsurge in elephant poaching.

The main argument in favour of poaching is that it provides the final solution to the elephant problem. The high elephant density is reduced without administrators having to make any difficult decisions. In Uganda poachers have reduced the elephant populations of Rwenzori and Kabalega National Parks to 5% or less of their former levels (Malpas, 1981). Poachers reduced the elephant population of Kenya by about two-thirds between 1973 and 1980 (Anon., 1980). Bell (in press *c*) has noted that large-scale poaching has proved to be far more efficient than legal culling schemes in reducing elephant numbers.

An argument against poaching is that it is uncontrolled and so the elephants could be exterminated. But this seems unlikely because below a certain elephant density the benefit/cost ratio of operating in the difficult Ruaha terrain will be too low. Because poaching pressure is heaviest on the edges of a Park it tends to cause compression by driving elephants into the centre of the Park. This is exacerbated by the poaching methods which cause considerable disturbance to the population (in contrast to culling methods which are designed to cause the minimum disturbance; Laws *et al.*, 1975).

The other technical arguments against poaching are legal. The ivory taken by poachers is revenue lost to the National Parks system. If the local population sees the National Parks Ordinances openly flouted, their

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respect for the wildlife laws in general will be diminished. This is particularly important because national parks exist only because of the stout defences provided by the law. There is considerable pressure for access to national parks by a wide variety of interested parties (subsistence hunters, honey- and firewood-gatherers, fishermen, cultivators, pastoralists, charcoal-burners, timber extractors, etc.). Once the respect for the law goes, the parks will go too.

DISCUSSION

In East Africa the elephant over-population problem was largely a problem of the 1960s and 1970s. In Uganda and Kenya it has been solved by the poachers (Douglas-Hamilton, 1979; Anon., 1980; Eltringham & Malpas, 1980; Malpas, 1981). There are two reasons for discussing the elephant-and-woodland question further, even though for most parks the elephant problem will not be an issue of the 1980s. One is that excess populations of large herbivores will appear again and we may learn from past case histories how to deal with them. Second, throughout Africa human populations continue to increase while settlements, roads, cultivation, and deforestation reduce the available habitat for elephants. If elephants are worth conserving at all, it is vital to conserve the national parks and game reserves which will become their last sanctuaries. The state of a park in 10 or 20 years' time depends upon the events of today. In order to understand tomorrow's ecological and management problems we must study those that exist today.

Ruaha is one of the last great wildernesses. Its agricultural potential is low: the earlier inhabitants asked to be moved away because of famine (Savidge, 1968*a*). Its importance lies in the diversity of plants and animals it holds, and in its function as a water catchment area. Its elephant population has probably been increasing since at least 1946 when it was regazetted a game reserve. This increase seems to be the result of two events. One is the period of higher rainfall, a natural event. The other is the redistribution of human pressures in and around the Park. This latter factor means that any management policies carried out inside the Park could be counter-balanced by forces acting outside the Park. At the same time the rapid vegetation changes within the Park are likely to affect the management of a dam outside the Park. Thus the Park is influenced by, and may influence, events outside its boundaries.

Before considering the management alternatives, it may be helpful for

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the manager to first decide upon the goals of management. The present trend is from a stable community structure with a high density of mature trees, many small herbivores, and few elephant, towards an unstable community structure with a lower tree density, few small herbivores, and many elephants (Bell, in press *b*). The change in community structure is a consequence of the increasing elephant population.

What community structure is the manager aiming for, and what are the limits to community change that he will accept (Bell, in press *c*)? The manager must consider (R. H. V. Bell, pers. comm.): the rate of erosion, changes in ecosystem nutrient capital, whether the disappearance of certain species (because of habitat changes) is desirable, species diversity (both from the ecological and aesthetic points of view: tourists want to see a wide spectrum of animals), and the aesthetic effects of the loss of woodlands (for instance, the whole character of the Park will be changed by the loss of the baobab trees and *A. albida* groves). An important issue is that of public relations: what will be the public reaction towards culling, or if there is no culling, to habitat degradation and the diminished value of the area as a water catchment (Bell, 1981; Kombe, in press)?

The theory and practice of decision-making by wildlife managers has been discussed by Bell (1980, 1981, in press *c*). Decision-making is a two-stage process. The first stage is concerned with technical questions and the accumulation of facts. These lead to an array of possible management options. The second stage requires an aesthetic decision (or value judgement) to choose the most appropriate option. Bell's thesis is that the scientist should prepare a comprehensive technical briefing for the decision-maker, detailing the arguments for and against each management strategy. It is then the manager's job to choose between the options. But note that the technical information does not lead directly to a choice of management option (Bell, in press *c*): an aesthetic decision, which balances the technical aspects with less definable or less quantifiable issues, still has to be made.

Should the scientist, who is better acquainted than anyone else with the complexities of the problem, express his opinion on the most appropriate management policy? He will be as concerned as anyone else that the wisest course is adopted. So while the logic of the decision-making process requires that the scientist should hold his peace, he may express his opinion if he makes it clear that he is expressing a value judgement and not making a technical decision in his role as scientist (Bell, 1981, in press *c*).

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The issue of elephant management generates considerable emotion, with both sides (pro- and anti-culling) using scientific arguments to back their claims. In my view (this is a value judgement rather than a technical statement) it is important that the scientist should remain aloof from the clamour and be seen to be objective. This is because a wise management decision can only be made after carefully weighing the evidence, and the decision-maker must have confidence in the scientist's impartiality. I believe that the scientist should express his opinion on this issue only if asked to do so by the officer responsible for making the final decision.

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