SOIL LOSS AND RUN-OFF IN UMFOLOZI GAME RESERVE AND THE IMPLICATIONS FOR GAME RESERVE MANAGEMENT

VOLUME 1

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Submitted in partial fulfilment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

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March 1988

ABSTRACT

Two management blocks were set aside in Umfolozi Game Reserve (UGR) to compare different management approaches. In one, the non-cull block, a noninterventionist policy was followed and no large mammals were removed, while the other, the cull block was subjected to the same game removal treatment as the remainder of the reserve. The main objectives of this study were to determine the relationships between vegetation, soil surface variables and both soil loss and rainfall run-off, to derive predictive models for run-off and soil loss based on vegetation and soil surface variables, to determine the relationship between different levels of soil erosion and the production potential of soils, and to determine the relationship between different levels of soil erosion and herbaceous species diversity.

Rainfall simulator trials and natural run-off plots were used to collect quantitative data on soil loss and run-off. Bivariate scattergrams showed that the relationship between soil surface and vegetation variables plotted against soil loss was curvilinear. "Susceptibility to erosion" showed the highest positive correlation, and "surface cover" the highest negative correlation with soil loss. The relationship between annual run-off and both the soil surface and vegetation variables was also curvilinear, with "soil capping" showing the highest positive and "litter cover" the highest negative correlation with run-off. Using multiple regression analysis it was found that "susceptibility to erosion" and "surface cover" were the best predictors of annual soil loss. "Soil capping" and "percentage contribution of forbs" were the best predictors of annual run-off.

No clear relationship between either soil loss and stocking rate, or run-off and stocking rate was apparent in the two experimental blocks, and the differences in soil loss and run-off could not be explained by differences in stocking rate alone. There were however defects in the experimental design which invalidated the

assumption that the stocking rate differential between the two management blocks would increase with time.

Because of the above deficiency, an alternative study area on the western boundary fence, which allowed for paired sampling sites on either side of the fence, was chosen. Gerlach troughs were used to measure soil loss. The greatest variability in soil loss was explained by the position of the plots on the slope rather than whether the plots were in UGR or in adjacent KwaZulu. Similarly, differences in topography, rather than differences in landuse, exerted an overriding effect on A-horizon depth, herbage accumulation and grass species richness. Considering the results obtained, the opinion that a noninterventionist policy would lead to a decline in vegetation productivity and to a long-term reduction in species diversity appears to be unfounded.

Finally, based on the data collected and on a review of current scientific literature, changes to the Natal Parks Board soils policy and objectives are suggested, and the objectives are translated into operational management goals.

To Michele, Carly and Kirsty
for their help, support and patience

DECLARATION

This thesis is the result of the author's original work, unless specifically stated to the contrary in the text.

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CONTENTS

CHAPTER 1 : INTRODUCTION

1.1 BACKGROUND TO STUDY

In the past decade, international concern regarding the culling of over-abundant mammal species has grown rapidly (Jewell & Holt, 1981), and there are two distinct schools of thought regarding the management of populations of large mammals in conservation areas. The first is a noninterventionist approach where emphasis is placed on little or no intervention with ecological processes. Proponents of this approach maintain that it is more prudent for conservation purposes to rely on the powerful self-regulatory and self-stabilising processes of intact or nearly intact bio-ecosystems than to rely on continuous intervention (Ricklefs, Naveh & Turner, 1984). The second school of thought holds that active and often continuous intervention is necessary, and emphasis is placed on the conservation of genetic material and species diversity (Ferrar, 1983). *u* Failure in the past to distinguish between these two approaches has led to conflicting reserve management objectives being formulated, even within the same conservation area (Walker *et al.*, 1987). For example, the primary nature conservation objective for Natal Parks Board reserves is "to conserve the optimum number of appropriate indigenous (to the reserve) species and their habitats, maintain breeding populations and protect the specificity of these gene pools. Natural, physical and ecological processes will be allowed to operate without interference except under imperative circumstances" (Grobler, 1984). The first objective clearly emphasises the maintenance of species and genetic diversity and implies an interventionist approach. The second objective emphasises the conservation of processes and inclines towards a noninterventionist approach.

An extensive area of land comprising two management blocks has been set aside to compare these differing management approaches in Umfolozi Game Reserve. In the one management block a noninterventionist policy has been followed, and no large mammals have been removed since 1978 by either game capture or culling. The other block has been subjected to the same game removal treatment as the remainder of the reserve, and an interventionist policy has been followed.

One of the dilemmas facing reserve managers is whether a higher priority should be placed on protecting large mammal populations from human intervention (by, for example, adopting a no-culling approach) instead of culling large herbivores, and by this maintaining a vigorous vegetation cover as a defense against soil erosion (Owen-Smith, 1983). The Natal Parks Board has the following conservation policy regarding soils: "to ensure that accelerated erosion is attended to and that, in the long term, soil loss equals soil genesis" (Grobler, 1983). From an ecological perspective, accelerated soil loss may lead to a long-term reduction in species diversity, which contradicts the primary nature conservation objective of conserving "the optimum number of appropriate indigenous species and their habitats" (Grobler, 1984). However, soil erosion is also considered to be a natural process which should be allowed to operate without interference according to the primary nature conservation objective. Here, the conflict in objectives is clear-cut, particularly when it is appreciated that there is no objective way of distinguishing between natural and accelerated erosion, and so this study was started in March 1982 to determine the effect of a noninterventionist policy on soil erosion when compared to an interventionist policy. The study fell under the auspices of the National Weather, Climate and Atmosphere Research Programme of the Council for Scientific and Industrial Research (CSIR).

1.2 GENERAL REVIEW OF LITERATURE ON EROSION

The threat of soil erosion to long-term crop production has been well documented by Larson, Pierce & Dowdy (1983), and the effects of soil erosion on productivity in rangeland environments by Gifford & Whitehead (1982). Wilson & Tupper (1982) have stated that physical soil erosion is the most serious manifestation of a decline in range condition because of its long-lasting and progressive effect on production attributes, and recently research on the erosion-soil productivity relationship has been recognised as one of the highest research priority needs (Rosewell, 1986). In a global perspective, Riquier (1982) points out the urgent need to conserve and protect the productive soil remaining, since soil degradation is already widespread and population pressures continue to mount.

Stocking (1981) states that the most important environmental problem in Africa is soil erosion caused by overgrazing. Accelerated erosion is probably the most serious and least reversible form of land degradation, particularly in the tropics (El-Swaify, 1981). Because of their generally sparse vegetation cover, semi-arid regions experience high rates of erosion, even without the influence of man (Dunne, Dietrich & Brunengo, 1978). Brown (1981) concluded that if soils continue to deteriorate in Africa, then the decade-long decline in food output per person could become chronic.

Soil conservation in its broadest context has been covered in an excellent review by Hudson (1981), and soil erosion has been reviewed by Morgan (1979). The problems of accurate prediction of soil loss have been well documented for arable lands, but these same problems have not enjoyed similar attention in rangeland conditions (Blandford, 1981). Barnes & Franklin (1970) pointed out that it is important to know the acceptable limits of cover reduction by grazing and browsing animals under different conditions and on different soil types, and they concluded that critical studies of cover in relation to run-off and erosion would be of great value. Blair-Rains (1981) suggested that if sedimentation rates are to be reduced on grazing land, then it is essential to ensure correct stocking levels based on the grazing capability of the most seriously degraded areas. Wischmeier (1975) has stressed the importance of cover in undisturbed (i.e. nonarable) areas. The effects of manipulating the cover factor (by grazing) on potential erosion have been discussed by Johnson, Schumaker & Smith (1980), and the state of knowledge of the impact of grazing on watersheds has been reviewed by Blackburn, Knight & Wood (1982). Gifford & Hawkins (1978) critically reviewed the hydrological impact of grazing on infiltration, and Gifford & Springer (1980) produced an excellent annotated bibliography on the hydrological effects of grazing. The importance of ground cover in the prevention and control of soil erosion under natural grazing conditions has been recognised by various authors (Stewart, 1965; Lang, 1979; Moore *et al.*, 1979a; Dadkhan & Gifford, 1980; Nyamapfene, 1982). Other authors have pointed out the need to maintain a good grass cover in the catchment area (Scott, 1981), and the impact of grazing intensity on infiltration rates (Gifford & Hawkins, 1979). Walker (1979) found that in semi-arid savannas it is the seasonal soil moisture regime that directly influences the annual primary production, and that this soil moisture is in turn influenced by the rate of water infiltration and the water-holding capacity of the soil.

Wischmeier & Smith (1978) produced a standard reference manual for the prediction of rainfall erosion losses and a guide to conservation planning, and in an earlier paper (Wischmeier, 1976) cautioned against the misuse and misinterpretation of the Universal Soil Loss Equation (USLE) in soil erosion prediction and control. Lastly, the use of the soil resource base (with particular emphasis on crop production) has been widely applied by various authors (Carter, 1977; Barlow, 1979; Larson, 1981; Mannering, 1981).

Although the above literature survey is by no means exhaustive, it does show that most of the erosion literature is directed towards the agricultural sector, with particular emphasis on the effects of erosion on the production potential of cropland. Except for some research work undertaken in North America, very little work has been done on soil loss and run-off from natural veld. That which has been done on natural veld deals with domesticated herbivores, primarily cattle, and to the best of this author's knowledge no research work has been published on erosion from natural veld utilised by wild large herbivores.

It is also clear from the literature survey that plant cover is an important factor influencing the rate of soil erosion, and in a game reserve context it is the only variable in the USLE which can be extensively manipulated, either directly through burning, or indirectly through the control of herbivore numbers. For this reason, the present study emphasised the effect of vegetation cover on soil loss and run-off in a game reserve situation.

1.3 OBJECTIVES OF STUDY

Brooks & Macdonald (1983) divided the management history of the Hluhluwe and Umfolozi Game Reserves (H&UGR) into several management periods, the most recent being the data-based period which started in 1978. During this period, the "single species" research approach was replaced by multidisciplinary integrated research programmes, which included research into fire and its effects; vegetation monitoring; animal censussing; herbivore-plant interactions and plant cover-soil erosion interactions (Brooks & Macdonald, 1983). The current study focused on the plant cover-soil erosion interactions in Umfolozi Game Reserve (UGR) and had eight major objectives:

a) the determination of rainfall erosivity values for spatially diverse sites;

b) the determination of the soil loss and run-off potentials of fixed sites within key vegetation types;

c) the determination of the relationship between vegetation, soil surface variables and soil loss;

d) the determination of the relationship between vegetation, soil surface variables and rainfall run-off;

e) the derivation of predictive models for run-off and soil loss based on vegetation and soil surface variables;

f) a comparison of the accuracy of two vegetation monitoring methods in predicting run-off and soil loss;

g) the determination of the relationship between different levels of soil erosion and the production potential of soils; and

h) the determination of the relationship between different levels of soil erosion and herbaceous species diversity.

The study area was located in western UGR, and in the following chapter a detailed description of UGR, and in particular the study area, will be given.

1.4 CHAPTER SUMMARY

a) There are two distinct schools of thought regarding the management of populations of large mammals in conservation areas. The first is a noninterventionist approach where emphasis is placed on little or no intervention with ecological processes. The second school of thought holds that active and often continuous intervention is necessary, and emphasis is placed on the conservation of species diversity.

b) Failure in the past to distinguish between these two approaches has led to conflicting reserve management objectives being formulated, and in March 1982 the current study was started to determine the effect of a noninterventionist policy on soil erosion when compared with an interventionist policy.

c) A general review of literature on erosion indicated that very little work had been done on soil loss and run-off from natural veld in Africa, and no references could be found on erosion from natural veld utilised by wild large herbivores.

d) It was also evident from the literature review that plant cover was an important factor influencing the rate of soil erosion, and since it is the only variable in the USLE which can be extensively manipulated in a game reserve context, the present study emphasised the effect of vegetation cover on soil loss and run-off.

CHAPTER 2 : DESCRIPTION OF STUDY AREA

2.1 GEOGRAPHICAL AND HISTORICAL SETTING

The Hluhluwe and Umfolozi Game Reserves (H&UGR) are situated in central Zululand between 28° 00′ and 28° 26′ south latitude and 31° 43′ and 32° 09′ east longitude (Figure 2.1). These two reserves are joined by a corridor of state-owned land and are fenced as a unit which is 96 453 ha in extent. The area of Umfolozi Game Reserve (UGR) is 47 753 ha and for this thesis UGR is defined as that area of the game reserve located south of the Black Umfolozi River (Figure 2.1).

Umfolozi Game Reserve was first proclaimed as the Umfolozi Junction Reserve in the Zululand Government Notice no. 12, gazetted on April 30, 1895, and later as the Umfolozi Game Reserve on 27 April 1897 in terms of Government Notice (2) no. 16 of 1897 (Anon., 1985a). This reserve, together with Hluhluwe Game Reserve (HGR), was established because of concern felt by both officials and members of the public at the reduction in the numbers of game animals in Zululand, particularly square-lipped rhinoceros *Ceratotherium simum*.

The study area is located in the western part of UGR and comprises two management blocks (Figure 2.2). The non-cull block is 6 278 ha and the cull block 4 634 ha in extent. In the non-cull block no ungulates have been removed by either game capture or culling since 1978. In contrast, in the cull block the game removal treatment has been identical to that of the remainder of UGR.

2.2 CLIMATE

Temperature and rainfall data are available from the third-order weather station based at Mpila, and rainfall data are available from Mbhuzana outpost which is located in the non-cull block (Figure 2.2).

2.2.1 Temperature

The mean maximum and mean minimum monthly air temperatures measured at Mpila are given in Table 2.1. The hottest months are December to February, with the highest monthly mean maximum temperature of 32,9°C recorded in February, and the coldest months are June to August, with the lowest monthly mean minimum temperatures of 13,2°C recorded in June and July (Table 2.1).

2.2.2 Rainfall

The maximum, minimum and mean monthly rainfall figures for Mpila are given in Table 2.2. The two wettest months are January and February, and the driest month is June. The wet season is mainly confined to the summer months, and during the six-month period from October to March, 73% of the mean annual rainfall of 693,3mm (n = 27) is received. There have been two extended periods of below-average rainfall since the inception of rainfall data collection at Mpila (Figure 2.3). The first drought period began in the 1966/67 rainfall year and lasted for five years, and the second drought period began in 1977/78 and lasted for six years. During the final year of the 1977/78 drought period, less than half of the mean annual rainfall was recorded. In the following rainfall year (1983/84) almost double the mean annual rainfall was received, primarily because of Cyclone Domoina, during which 513mm of rainfall was recorded over a five-day period. The non-cull/cull experiment was initiated during the 1977/78 rainfall year, at the beginning of an extended below-average rainfall period.

At Mbhuzana, which is the only weather station in the study area, rainfall measurements were begun in April 1981 and the annual rainfall is given in Table 2.3. The mean annual rainfall is 684,9mm (n = 5) which is considerably lower than the 724,7mm measured at Mpila over the same five-year period.

2.3 TOPOGRAPHY, GEOLOGY AND SOILS

The topography is generally hilly with the altitude ranging from 40m above sea level at the confluence of the Black and White Umfolozi Rivers to 580m along the western boundary fence (Figure 2.1). Flat areas are mainly confined to the flood plains of the larger rivers and streams. Generally, the steepest slopes predominate in the west of the reserve along the boundary fence and in the central part of the reserve, although the topography is extremely varied (Figure 2.4). In the study area, the topography is also diverse, and all five slope classes are present.

There are two major rock formations, viz. sedimentary and volcanic rocks. Quartzites and sandstones of the Natal Group form the oldest outcrop in the reserve (Figure 2.5). Dwyka tillite of the Karoo Supergroup is the next oldest rock. Shales of the Pietermaritzburg Formation of the Karoo Supergroup are next in sequence. The shale is overlaid by the Vryheid Formation which comprises layers of sandstones interspersed with layers of mudstones and shales, and this formation covers the most extensive area of the reserve. The youngest sedimentary rocks in the reserve are sandstones and shales of the Beaufort Group (Downing, 1980). Dolerite outcrops are distributed throughout UGR, but the most extensive outcrop occurs in the south western section of the reserve (Figure 2.5). In the study area, sandstones and shales of the Vryheid formation predominate. In the central part of the study area, shales and thin sandstones of the Volksrust formation are well represented, while dolerite outcrops are scattered throughout both the cull and non-cull blocks (Figure 2.5).

The soils of the reserve are diverse, and 14 different soil forms or associations have been identified (Figure 2.6). The predominant soil forms are Swartland, Mispah and Shortlands, and the predominant association is Swartland/Mispah. The Shortlands soil form is derived from dolerite, and the other three are derived from sedimentary rock comprised primarily of mudstones and shales. Valsrivier and Dundee soil forms are also abundant and are confined to the smaller streams and watercourses. Again, it is evident from Figure 2.6 that the distribution of soils in the reserve is heterogeneous. Except for the Bonheim and Milkwood soil forms, all the soil forms and associations mapped in Figure 2.6 are present in the study area.

2.4 VEGETATION TYPES

Whateley & Porter (1983) described the woody vegetation communities in the reserve, based on physiognomy and dominant woody species. They recognised two riverine forest, eight woodland and two thicket communities. The distribution of these broad physiognomic communities is shown in Figure 2.7. Open and closed woodland communities predominate throughout the reserve, while thicket and wooded grassland communities are largely confined to the western part of the reserve. Grassland communities are scarce and are primarily found on the crests of high ridges along the western boundary fence. Because of the heterogeneity of the topography and soil, it is hardly surprising that the vegetation is also heterogeneous.

Generally, the *Ficus sycamorus* - *Schotia brachypetala* riverine forest is confined to the banks of the larger rivers and their major tributaries, while the *Spirostachys africana* - *Euclea schimperi* riverine forest occurs along seasonal watercourses. The *Spirostachys africana* woodland covers extensive areas in bottomland situations. On rocky hillsides, *Combretum apiculatum* woodland is found, and has a restricted distribution. *Euclea divinorum* woodland is usually found on gently sloping ground and is scarce in the reserve. *Acacia nilotica* woodland is also scarce, and is usually found below the 300m contour. *Acacia burkei* woodland occurs mainly on broad, flat ridges at lower altitudes, and *Acacia gerrardii* woodland covers large gently undulating areas of shales. In contrast, *Acacia nigrescens* woodland is nearly always found on hillsides having a dolerite substrate. *Acacia tortilis* woodland occurs throughout the reserve, while *Acacia caffra* thicket is found in upland situations, usually above the 300m contour, on hillsides and ridges and has a limited distribution. *Acacia karroo* - *Dichrostachys cinerea* induced thicket is found at all altitudes throughout the reserve (Whateley & Porter, 1983). Except for *Euclea divinorum* woodland, all the above woody vegetation communities are found in the study area.

2.5 FAUNA

The vertebrates of H&UGR have been listed by Bourquin, Vincent & Hitchins (1971), and for reptiles, amphibians and mammals the above authors have also included distributional data. The birdlist was updated by Macdonald & Birkenstock (1980), Owen-Smith (1980) and Johnson & Gerber (1986); and the carnivore checklist by Whateley & Brooks (1985).

2.5.1 Invertebrates

Only a checklist of butterflies recorded in UGR exists (Anon., 1985a). Eight families were recognised and 112 species positively identified.

2.5.2 Fish

A total of 12 fish species has been recorded for UGR (Bourquin *et al.*, 1971).

2.5.3 Amphibians

Bourquin *et al.* (1971) recorded 17 amphibian species from UGR.

2.5.4 Reptiles

Sixteen lizard, one crocodile, two tortoise, two terrapin and 21 snake species have been recorded from UGR by Bourquin *et al.* (1971).

2.5.5 Birds

Macdonald & Birkenstock (1980) listed 326 indigenous bird species which definitely have occurred or do occur in H&UGR, and an additional six alien species which have been introduced or have immigrated into the reserves. A further eight indigenous species were later added to the checklist of birds of UGR (Johnson & Gerber, 1986). Owen-Smith (1980) made a quantitative assessment of the avifauna present in one vegetation type in UGR.

2.5.6 Mammals

A total of 64 mammal species has been recorded for UGR, including three shrew, 11 bat, two primate, 16 carnivore, 20 ungulate, two hare and eight rodent species (Bourquin *et al.*, 1971). Whateley & Brooks (1985) have summarised information collected on carnivores between 1973 and 1982.

2.5.7 General

A total of 585 animal species, including four which have recently been re-established, has been recorded in UGR to date. This high species richness is due to the high habitat diversity found in the reserve, which in turn is attributable to the extremely varied topography, geology and soils of the reserve.

In attempting to preserve this habitat and species diversity, reserve managers need to know what effect different management approaches have on soil erosion and, in turn, what influence soil erosion has on habitat and species diversity. For this reason, rainfall simulator trials were carried out in various woody vegetation communities, in both the non-cull and cull blocks, to determine the influence of two different management approaches on soil erosion. These rainfall simulator trials will be discussed in detail in the following chapter.

2.6 CHAPTER SUMMARY

a) The study area was located in western UGR and comprised a non-cull block, 6 278 ha in extent, and a cull block of 4 634 ha.

b) In the non-cull block, no ungulates have been removed since 1978, while in the cull block the game removal treatment has been identical to that of the remainder of UGR.

c) The topography, geology and soils of UGR are diverse, resulting in a diverse floral and faunal component.

CHAPTER 3 : RAINFALL SIMULATOR TRIALS

3.1 INTRODUCTION

This chapter, and the remainder of the thesis, addresses the wider key question: "What effect did the noninterventionist approach, adopted in the non-cull block, have on soil loss and run-off rates when compared with the interventionist approach adopted in the cull block?" To try to answer the above question, this part of the study had four aims:

a) to determine the soil loss and run-off potentials of fixed sites within key woody vegetation communities;
b) to determine the interrelationships between vegetation and soil surface variables and both soil loss and run-off;
c) to derive and validate predictive models for run-off and soil loss based on vegetation and soil surface variables; and
d) to compare the ability of two different vegetation monitoring methods to predict accurately soil loss and run-off.

Whateley & Porter (1983) recognised 12 woody vegetation communities in UGR, and several of these communities were used as the basic sampling units for the placement of Walker transects (Walker, 1976) to monitor trends in both the woody and herbaceous vegetation in the non-cull and cull blocks. Some of these communities have a higher potential for soil loss and run-off than others. This potential may be linked to the inherent erodibility of their constituent soil types, their topographical positions and/or the preference that herbivores show for them. The ideal would have been to quantify soil loss and run-off from each of the woody vegetation communities, but this was not possible with available time, funds, manpower and equipment. Instead, key communities were chosen and rainfall simulator trials were limited to fixed sites within these communities.

The rainfall simulator was chosen as an experimental tool because, using it, it was possible to simulate several storms of standardised intensity and duration at various sites over a short period. Because storm intensity and duration, as well as data collection, were strictly standardised, comparative data between sites could be collected. Also, because each site was permanently marked, comparative data over a period of years could be collected at each site.

Although the non-cull/cull experiment started in 1978, rainfall simulator trials began only in 1982. However, vegetation monitoring data based on Walker's (1976) method had been collected in the study area since 1979, and a large data base on the condition and trend of the vegetation existed. Because of this data base, it was decided to continue the use of the Walker (1976) method to monitor vegetation and soil surface variables at each rainfall simulator site. If a significant correlation could be found between soil surface and/or vegetation variables and soil loss and run-off, then a retrospective analysis of soil loss and run-off could be carried out at each Walker vegetation monitoring site. For this reason, rainfall simulator trial sites were placed near existing Walker transects.

Lastly, as part of a larger Natal Parks Board study to review the vegetation monitoring programme currently in use, the Walker (1976) method was compared to the USLE method (Wischmeier, 1975) to see which of these two methods would give the more accurate prediction of soil loss and run-off.

3.2 SOIL LOSS AND RUN-OFF MEASUREMENTS

3.2.1 Materials and methods

Woody vegetation communities which covered an extensive area of the reserve, and which were susceptible to erosion and/or preferred by grazing herbivores, were identified, and eight rainfall simulator sites were located in these key communities.

Five of these sites were located in the non-cull block and three in the cull block and all, except site 284, were near existing Walker vegetation monitoring sites (Figure 3.1), and in close proximity to tourist roads or good management tracks, to allow easy access for the simulator which was mounted on a caravan chassis. Because access was confined to roads and tracks, it was not possible to place the sites randomly and a representative sampling strategy had to be used.

The simulator sites covered a wide variety of soil types and slopes (Table 3.1). Trials were started at these sites in March 1982, and repeated on exactly the same sites in April 1983 and March 1984 (Table 3.2). Rainfall simulator trials were also undertaken in 1985, to collect an independent data set. Three of the original sites were retained and, except for these, the 1985 trials were carried out on new sites (Figure 3.2).

Individual sites were permanently marked with concrete blocks, allowing rainfall simulator trials to be repeated on exactly the same sites every year. At each site, corrugated iron sheeting was used as shuttering to demarcate a total of four plots, two of which were laid out on each side of the simulator (Figure 3.3). Each plot was 10,67m long and 1,80m wide. These are the standard dimensions for plots when a Swanson rotating-boom rainfall simulator (Swanson, 1965) is used. At the downward slope end of the plots, collection troughs were installed to aid in the collection of sediment samples and run-off data. Simulated rainfall was applied at a fixed intensity of 63mm/h and three storms, each lasting 36 minutes and applying about 36mm of rainfall, were simulated annually per site. The purpose of the first simulator storm on each site was merely to standardise the soil moisture content, making between-site comparisons possible. Under normal circumstances, the duration of this storm and the second storm would be one hour each, but because none of the sites were near permanent water, a water tanker was used to supply water for each simulator storm, and only enough water was available for a storm lasting 36 minutes. For this reason, three storms of 36 minutes each were simulated instead of the normal two storms lasting 60 minutes each. Once water had started running off these plots, each plot was alternately sampled, at two-minute intervals, for the rate of run-off and amount of sediment produced. Run-off rate was measured by timing how long a container of known volume took to fill. Also, samples of run-off water were collected in 1-litre glass screw-top jars and taken to a field laboratory where concentrated hydrochloric acid was added to facilitate settling of the sediment. After the sediment had settled, the excess water was siphoned off and the sediment samples were then oven-dried and weighed. The technique is described in detail by McPhee *et al.* (1983).

A computer program was used on the Dept. of Agriculture and Water Supply mainframe terminal at Cedara to compute the following from the input data:

total run-off (litres) per plot and per storm;
percentage run-off per plot and per storm; and
total soil loss (grams) per plot and per storm.

Total run-off (litres) for each plot was computed for each storm as follows:

$$\text{Total run-off during storm 2} = \sum_{k=1}^{n} (\text{run-off})_k$$

where n = number of periods $_k$

$_k$ = intervals between successive measurements of flowrate. Each period $_k$ is therefore the period between two measurements of flowrate

$(\text{run-off})_k = B \times C$

where B = average flowrate (l/min) during period $_k$

$$= \frac{\text{flowrate at beginning of period k + flowrate at end of period k}}{2}$$

and C = interval represented by period $_k$ (min).

The flowrate (l/min) for any time during the rainfall simulator storm was calculated as follows:

$$\text{Flow rate} = \frac{\text{volume of container (l)} \times 60}{\text{time to fill container (s)}} - 0{,}189$$

The constant 0,189 represented run-off from simulated rain falling directly on the collection trough and not on the plot.

Total soil loss (g) for each plot was computed for each storm using the formula below:

$$\text{Total soil loss during each storm} = \sum_{q=1}^{n} (\text{sediment load})_q + H + I$$

where n = number of periods $_q$

$_q$ = intervals between successive measurements of sediment concentration. Each period $_q$ is therefore the period between two measurements of sediment concentration

(sediment load)$_q$ = sediment lost during period $_q$

= (D X E X F X G)

and D = average sediment concentration during period $_q$

$$= \frac{\text{sediment concentration at beginning of period q} + \text{sediment concentration at end of period q}}{2}$$

E = flow rate during period $_q$ (l/min)

= flow rate measured between the taking of samples at beginning and end of period $_q$

F = interval represented by period $_q$ (min)

G = correction factor

= ratio between flow rate calculated without, and flowrate calculated with, the correction factor of 0,189 l/min.

This correction factor accounts for the dilution effect of water falling directly onto the collection trough.

H = soil loss represented by the first sediment concentration sample

$H = J \times K \times L$

where J = interval between first flow rate measurement and first sediment sample (min)

K = first sediment concentration (g/l)

L = first flow rate (l/min)

I = soil loss represented by the last sediment concentration sample

$I = M \times N \times O$

where M = interval between last flow rate measurement and last sediment sample (min)

N = last sediment concentration (g/l)

O = last flow rate (l/min)

Lastly, the sediment concentration (g/l) at any particular time was calculated from the formula below:

$$\text{Sediment concentration} = \frac{\text{mass of sediment (kg)} \times 1000 \times G}{\text{mass of water (kg)}} - \text{concentration of sediment in water pumped through simulator (g/l)}$$

G = correction factor (see G above)

Again, the correction factor (G) accounts for the dilution effect of water falling directly onto the collection trough.

Results from the first simulator storm on each site were discarded on the basis that initial soil moisture content had not been standardised. A mean estimate of total run-off, percentage run-off and total soil loss was obtained for plots 1 and 2 (see Figure 3.3), based on the data collected during storms 2 and 3. The same applied to plots 3 and 4. Data from plots 1 and 2 were combined into one data set, and similarly, data from plots 3 and 4 were combined into another data set. These data were combined, firstly, to make the soil loss and run-off data more comparable with the natural run-off plot data, where the plot size was slightly more than double the size of individual rainfall simulator plots (see Chapter 5), and secondly, to increase the number of quadrats used in the assessment of vegetation and soil surface variables from ten to 20. This made the vegetation data more comparable with the standard Walker transect (Walker, 1976) which is based on a sample size of 25 quadrats. Because the percentage run-off is the amount of water that ran off expressed as a percentage of the total amount of water applied during the storm, this measurement was used in preference to the total run-off. The latter figure simply gives run-off in litres during the storm and is not as meaningful as the former.

Using the results from storms 2 and 3, the mean percentage run-off, the mean soil loss, and their standard errors and ranges were calculated for each site for each year that rainfall simulator trials were done on that site. Where more than one site was located within a particular vegetation type, the data for these sites were presented together.

3.2.2 Results

3.2.2.1 Soil loss potential of fixed sites within key woody vegetation communities

Soil lost from each site during the rainfall simulator trials is shown in Figures 3.4 to 3.7. The general trend in those plots sampled over three or four years is that the highest soil losses were recorded in 1983, and this shows there is a large degree of temporal variability in the data. The reasons for this variability are discussed in Section 3.5.2. Site 206 showed a far higher soil loss in 1983 than site 241 (Figure 3.5). These sites are both on the same soil form and series, and have similar slopes (Table 3.1). Also, both are located in the non-cull block near to one another (Figure 3.1). This emphasises the inherent spatial variability in the system, and this, together with the temporal variability, leads to the conclusion that the explanation for the variability in the data lies with time and spatially dependent variables instead of variables independent of these two aspects (Section 3.3.2.3).

Excluding sites 206 and 295, which were monitored for only two years, the lowest mean soil loss over a period of three years (1982 - 1984) was measured from site 280 and the highest from site 279 (Table 3.3). These sites were both located in *Acacia tortilis* woodland, but in different management blocks and on different soil forms (Table 3.1). The large ranges and standard errors indicate a high degree of variation around the mean over the duration of the trials. This high temporal variability emphasises that time dependent, instead of time independent, variables have a dominant effect on the soil loss potentials of the sites.

3.2.2.2 Run-off potential of fixed sites within key woody vegetation communities

The mean percentage run-off from each site is illustrated in Figures 3.8 to 3.11 and there is obviously a high degree of temporal variability within any particular site over the duration of the simulator trials. In *Acacia nigrescens* woodland sites, there is a general increase in mean percentage run-off between 1982 and 1983, and then a decrease between 1983 and 1984, which is continued in 1985 in site 241 (Figure 3.8). This trend is shown irrespective of whether the plots are in the non-cull or cull blocks (Table 3.1). However, in *Acacia tortilis* woodland, site 279 shows the opposite trend to the other two sites (Figure 3.9), and yet when compared to site 296, which is also located on a Shortlands soil form (Figure 3.11), the trends are very similar. Site 294 follows the general trend with a peak in mean percentage run-off in 1983 (Figure 3.10). The only exceptions to this trend are site 280, which shows a higher run-off in 1982 and 1984 than in 1983, and site 284 where the run-off remained relatively constant between 1983 and 1984 (Figure 3.9). The greatest variability about the mean is also shown by these two sites (Figure 3.9), which are both located in *Acacia tortilis* woodland on the Swartland soil form.

The lowest mean percentage run-off measured over three years was recorded from site 241 and the highest from site 296 (Table 3.4). This excludes data from sites 206 and 295 which were monitored for only two years. Both sites 241 and 296 are located in the non-cull block, but occur on different soil types and woody vegetation communities (Table 3.1), and this illustrates that no consistent trend in run-off was evident from the two experimental blocks.

3.3 RELATIONSHIP BETWEEN VEGETATION AND SOIL SURFACE VARIABLES AND RUN-OFF AND SOIL LOSS

3.3.1 Vegetation and soil surface variables

3.3.1.1 Materials and methods

A $1m^2$ quadrat was laid down systematically at ten points within each rainfall simulator plot, and two semi-quantitative methods were used to assess vegetation and soil surface variables. Using the Walker (1976) method, the following variables were assessed in each $1m^2$ quadrat:

a) herbaceous canopy cover (%), which was defined as the percentage of herbaceous, aerial cover;

b) litter cover (%), which represented the percentage of dead, non-woody plant material in contact with the soil surface;

c) soil capping (%), which was the percentage of the total surface area of the quadrat which was capped;

d) susceptibility to erosion (%), which represented the percentage of the total surface area of the quadrat which was bare ground;

e) rock (%), which was the percentage of the total surface area of the quadrat which was covered by rock;

f) forb (%), which was defined as the percentage contribution that forbs made to the total above-ground living herbaceous biomass within the quadrat;

g) Aizoon (%), which was defined as the percentage contribution that the forb *Aizoon glinoides* made to the total herbaceous biomass within the quadrat. This forb covered extensive areas of the study area in 1982;

h) maximum grass height (cm), which was the height above ground level of the highest piece of live grass material, straightened against a metre rule; and

i) mean grass height (cm), which was the average height above ground level of the grass occurring within each quadrat.

The data from plots 1 and 2 were combined, and those from plots 3 and 4, so that the sample size in each case was 20. Variables a) to e) were ranked on an eight-point scale, with uneven class intervals, given below :

RANK	CLASS INTERVALS (%)
0	0
1	1-10
2	11-25
3	26-50
4	51-75
5	76-90
6	91-99
7	100

Variables f) and g) were ranked on the same eight-point scale depending on the contribution that they made to the total above-ground living herbaceous biomass within each quadrat.

The mean values for variables a) to g) were determined by the formula:

$$\sum_{i=1}^{6} (n_i \times c_i) \ / \ N$$

where n_i = number of quadrats with rank i

c_i = class mid-point of rank i

N = total number of quadrats.

Maximum and mean grass height were expressed as the mean values (in cm) for the 20 quadrats.

The following variables were assessed in each $1m^2$ quadrat using the USLE method (Wischmeier, 1975):

a) herbaceous canopy cover (%), which was defined as the percentage of the total surface area of each quadrat that could not be hit by vertically falling raindrops because of the herbaceous canopy cover;

b) surface cover (%), which was the percentage of the total surface area that could not be hit by vertically falling raindrops because of the canopy and any ground cover;

c) woody canopy cover (%), which represented the percentage of the total surface area that could not be hit by vertically falling raindrops because of the woody canopy cover; and

d) woody canopy height (cm), which was the mean height of the woody canopy above ground level measured to a maximum height of 4,0m. The canopy of any woody plant less than 0,5m in height was not measured.

Variables a) to c) were ranked on a six-point scale, with even class intervals, given below :

RANK	CLASS INTERVAL (%)
0	0
1	1-20
2	21-40
3	41-60
4	61-80
5	81-100

The data from plots 1 and 2 were combined into one data set, and those from plots 3 and 4 were combined into another data set. Based on the two data sets, the mean values for variables a) to c) were determined using the following formula:

$$\sum_{i=1}^{6} (n_i \times c_i) / N$$

where n_i= number of quadrats with rank i

c_i= class mid-point of rank i

N = total number of quadrats.

The mean woody canopy height (variable d) was calculated using the following formula:

$$n_1 + n_2 \ldots\ldots n_n / N$$

where n = woody canopy height in each quadrat in which woody plants occur

N = total number of quadrats in which woody plants occur.

Mean values for vegetation and soil surface variables measured at each rainfall simulator site were calculated for every year that rainfall simulator trials were done at that site.

3.3.2 Relationship between vegetation and soil surface variables and run-off and soil loss

3.3.2.1 Materials and methods

Correspondence analysis, a multivariate analysis technique which can handle large matrices of multivariate data, was used to find the best objective fit to the association between vegetation and soil surface variables and soil loss and run-off. This provided a graphical display of the interrelationships in the data matrix which made data summarisation and interpretation easier.

This technique was recently evaluated by Beardall, Joubert & Retief (1984) for the analysis of herbivore-habitat selection and showed promising results because it illustrated animal species associations based on the selection of similar habitat factors. It also identified those habitat factors most favoured by each species. Because the data collected in the above study were similar to those collected during the current study, and because of the promising results obtained, it was decided to use this method of analysis.

The theory and applications of correspondence analysis have been described in detail by Greenacre (1984). Briefly, this technique takes a large data matrix, arranged in rows and columns, which is essentially a set of points in multidimensional space and reduces this set of points to an approximate low-dimensional display. The end product is a two-dimensional graph in which similar entities are grouped together and dissimilar entities are relatively far apart. It is obviously not possible to transcribe all the data onto such a two-dimensional graph, and so there is a trade-off between ease of interpretation and completeness of description (Gauch, 1982). The usefulness of an ordination technique like correspondence analysis is that the gain in interpretability far exceeds the loss in information (Greenacre, 1984).

To examine the interrelationships between soil loss, run-off and various soil surface and vegetation variables, the data from the 1982, 1983, 1984 and 1985 rainfall simulator trials were analysed separately. In each case, the data matrix was drawn up using the format outlined in Table 3.5. The mean value for each independent variable within a particular soil loss or run-off class was calculated using the formula:

$$\frac{(n_1 + n_2 \ldots + n_n)}{N}$$

where n_1, n_2 ... n_n = value of particular independent variable occurring in a specific soil loss or run-off class

N = total number of observations occurring within that specific soil loss or run-off class

The codes used in the text, in Figures 3.12 - 3.15, in Table 3.5 and in Tables 3.7 - 3.10 are explained in Table 3.6.

Besides correspondence analysis, bivariate scattergrams of soil loss and run-off against specific independent variables were plotted from the combined 1982, 1983 and 1984 rainfall simulator trial data. In many cases, the relationship between soil loss and these independent variables was curvilinear, and therefore various transformations were tried to normalise the data relationship. A logarithmic transformation produced the closest linear relationship, and a cross products correlation matrix including both the dependent and independent variables was drawn up using a statistical program* on an Apple IIe microcomputer. Those variables which were correlated significantly with either mean percentage run-off or $\log_e$ of mean soil loss were then presented as bivariate scatter plots.

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* Statpro, Wadsworth Electronic Publishing Co., California.

3.3.2.2 Relationship between run-off, soil loss and vegetation and soil surface variables as determined by correspondence analysis

The results of the 1983 rainfall simulator trials, obtained from correspondence analysis, are presented in Table 3.7 and Figure 3.12. To assist in the interpretation of Table 3.7, an explanation of each column heading will be given before the significance of the data in the table is discussed. The values in the QLT column summarise the quality of representation of the points in two-dimensional subspace and indicate how well the multidimensional points are transcribed onto a two-dimensional graph. A high QLT value shows that the point is almost on the flat, two-dimensional plane of the paper and there is hardly any error in its display, as is the case with SUTE on the first principal axis (Table 3.7). These QLT values are the sums of the COR columns, which in turn indicate the axis most important to a particular point. The high values for SOCA and SUTE (in the first COR column) show that the first principal axis represents these points far better than the second principal axis. In contrast, MSL3 is better represented by the second principal axis. The values in the CTR columns are the absolute contributions that the points make to the total variability of the respective axis, and this indicates which points contribute most to the variability of the particular axis; for example, the highest contributions to variability along the first principal axis are made by SOCA and MSL5 with CTR values of 361 and 308 respectively, and the lowest contributions by ROCK and MPR3. This means that SOCA and MSL5 contribute the most to explaining the variability along the first principal axis, while ROCK and MPR3 contribute scarcely anything to explaining this variability. The values in the K=1 column give the co-ordinate of the points for the first axis, and in the K=2 column for the second axis. The sign shows whether the points are positively or negatively "correlated" with the respective principal axis (Beardall *et al.*, 1984).

Table 3.7 is interpreted in two ways. Firstly, for each principal axis, the column headed CTR is inspected to interpret which points contribute most to the variability. Those points contributing the most to the first principal axis are SOCA, SUTE, MSL5 (i.e. very high soil loss) and MPR5 (i.e. very high run-off), and those contributing the most to the second principal axis are MEGH, SURC, MSL3 (i.e. moderate soil loss), MPR1 and MPR2 (i.e. very low to low run-off). Secondly, for each point, the values in the two COR columns are examined to identify which axis represents the point well (Greenacre, 1984). In SOCA, SUTE, MSL5, MPR1, MPR2 and MPR5 these points are best represented by the first principal axis, while the second principal axis best represents MSL3. Table 3.7 supports the interpretation of the graphical output of the correspondence analysis (Figure 3.12).

The angle formed between lines joining any two points and the origin (i.e. where axis 1 and axis 2 intersect) gives a good indication of their similarity or dissimilarity. If the angle so formed is small, then the points are similar, and highly "correlated" with one another. The larger the angle, the more dissimilar are the points, and when the angle approaches 180°, the points are strongly dissimilar. If the position of a point is very close to the origin such interpretation becomes weaker, since the point probably does not "correlate" highly with the subspace depicted by the first two principal axes (Beardall *et al.*, 1984).

The first principal axis (axis 1 in Figure 3.12) distinguishes between very low to low run-off (MPR1 & MPR2) on the left-hand side of the axis and very high amounts of soil loss (MSL5) and run-off (MPR5) on the right. Along this profile, soil capping and susceptibility to erosion are correlated positively with very high run-off. Because the angles between litter cover, the origin and susceptibility to erosion and soil capping respectively approach 180°, these latter two variables are correlated negatively with litter cover. This relationship is to be expected, because as litter cover increases, so the other two variables will decrease. Axis 2 contrasts very low run-off (MPR1) with moderate amounts of soil loss (MSL3) and low run-off (MPR2). Along this profile, mean grass height is correlated positively with the former, and surface cover with the latter two. Axes 1 and 2 explain 96,5% of the total variability.

In the 1982 rainfall simulator trial data, axis 1 distinguishes high soil loss (MSL4) and very high run-off (MPR5) from low run-off (MPR2)(Figure 3.13). Soil capping is correlated positively with the former two variables, and litter cover with low run-off. Again, the angle of the origin between litter cover and soil capping is almost 180°, showing that these two variables are correlated negatively along this specific profile. Because SUTE made a high contribution to the CTR column of axis 2 in the original analysis, it played an overwhelming role in determining the second principal plane and in effect swamped the second principal axis. It is preferable to remove the influence of this obvious and isolated feature from the display, so that the more subtle multidimensional patterns can be investigated. In the subsequent analysis SUTE was treated as a supplementary point (Table 3.8). The second principal axis distinguishes low run-off (MPR2) from moderate run-off (MPR3) and low soil loss (MSL2). Maximum grass height and woody canopy cover are correlated positively with the former, and herbaceous canopy cover (HCC1), measured using the Walker (1976) method, with the latter. This herbaceous canopy cover is also correlated negatively with woody canopy cover and maximum grass height. The relationship between woody and herbaceous canopy cover can be explained by the fact that as the woody canopy cover increases, so the herbaceous canopy cover decreases because of the shading effect. The negative correlation between herbaceous canopy cover and maximum grass height, along this profile, is not as easy to explain as one would expect these two variables to be positively, instead of negatively, correlated. Axes 1 and 2 explain 79,2% of the variability.

In the 1984 rainfall simulator trial data, MSL4 (high soil loss) has been treated as a supplementary point because of its excessively high contribution to the first principal axis. This axis separates moderate to high soil loss (MSL3 & MSL4) and moderate run-off (MPR3) from low run-off (MPR2)(Figure 3.14). The contribution that forbs make to the herbaceous biomass is correlated positively with moderate to high soil loss and moderate run-off. The second axis distinguishes very high run-off (MPR5) from high run-off (MPR4). Along this profile, woody canopy cover, susceptibility to erosion and soil capping are correlated positively with very high run-off (Table 3.9). This correlation is to be expected because with bush-encroached conditions (high woody canopy cover), and high amounts of bare ground and soil capping, one would expect increased run-off. Of the total variability, 90,0% is explained by axes 1 and 2.

Lastly, the 1985 rainfall simulator trial data contrasts moderate soil loss (MSL3) and high run-off (MPR4) with very low run-off (MPR1)(Figure 3.15). In this analysis, MPR2 has been treated as a supplementary point so that the more subtle multidimensional patterns along the second principal axis can be investigated (Table 3.10). Mean grass height is correlated positively with very low run-off, and soil capping and susceptibility to erosion with moderate soil loss and high run-off. An increase in soil capping and bare ground will lead to higher levels of run-off and soil loss, while an increase in mean grass height should lead to reduced run-off levels. This is because the capacity of grasses to intercept rainfall is proportional to the product of mean height and percentage surface cover (Crouse, Corbett & Seegrist, 1966). Along axis 2, very low soil loss (MSL1) and run-off (MPR1) are contrasted with low soil loss (MSL2) and low (MPR2) to moderate (MPR3) run-off. Forb and woody canopy cover are correlated positively with the higher levels of soil loss and run-off, while herbaceous canopy cover, based on Walker's (1976) method (HCC1), is correlated positively with very low levels of run-off and soil loss. Axes 1 and 2 explain 97,3% of the variability.

In summary, correspondence analysis on the rainfall simulator trial data contrasts low run-off and soil loss with high levels of these two variables. Soil surface variables such as soil capping and susceptibility to erosion are invariably correlated positively with high levels of run-off and soil loss, while vegetation variables, particularly mean grass height and litter cover, are correlated positively with low levels of run-off. Except for the 1982 rainfall simulator trial data, 90,0% or more of the total variability was explained by a two-dimensional display of the data.

3.3.2.3 Interpretation of scattergram results

Mean percentage run-off showed the highest negative correlation with litter cover, and the highest positive correlation with soil capping, while mean soil loss showed the highest positive correlation with susceptibility to erosion, and the highest negative correlation with surface cover (Table 3.11). The three variables that were not significantly correlated with either mean percentage run-off or mean soil loss were either time independent variables, such as rock, or variables which only changed a small amount over the study period, such as woody canopy cover and the percentage contribution that the forb *Aizoon glinoides* made to the total herbaceous biomass.

With the exception of litter cover, all the variables which were correlated significantly showed a higher correlation with mean soil loss than with mean percentage run-off. Because some relationships with mean soil loss were curvilinear, this variable was transformed using a $\log_e$ transformation. Except for soil capping, all the significant variables showed a higher correlation with the transformed data (Table 3.11), but because the transformation did not change the statistical significance of any relationship, the untransformed data were used in the bivariate scattergrams.

The statistically significant correlations are shown in Figures 3.16 - 3.23 for mean soil loss and Figures 3.24 - 3.31 for mean percentage run-off. Only percentage susceptible to erosion (Figure 3.17) and percentage soil capping (Figure 3.18) are correlated positively with mean soil loss, and for soil capping this relationship is linear; all the other variables are correlated negatively with soil loss, and their relationships are curvilinear (Figure 3.16, Figures 3.19 - 3.23). There is a higher correlation between soil loss and herbaceous canopy cover, measured by Walker's (1976) method, than between soil loss and herbaceous canopy cover, measured by the USLE method (Wischmeier, 1975)(Figures 3.16 and 3.22 respectively).

Only susceptibility to erosion and soil capping are correlated positively with run-off (Figures 3.25 and 3.26); the remainder of the variables are correlated negatively, with litter cover showing the highest negative correlation (Figure 3.27). Although there is a large amount of scatter, all variables appear to show a linear rather than a curvilinear relationship with run-off. Except for litter cover, all variables show a higher correlation with soil loss (Figures 3.16 - 3.23) than with run-off (Figures 3.24 - 3.31). Again, run-off is more highly correlated with herbaceous canopy cover, as measured by Walker's (1976) method, than with the same variable measured by the USLE method (Wischmeier 1975) (Figures 3.24 and 3.30 respectively).

In summary, the relationship between run-off and the measured vegetation and soil surface variables is linear, with mean percentage run-off showing the highest positive correlation with soil capping, and the highest negative correlation with litter cover (Table 3.11). Except for soil capping, the relationship between soil loss and all the variables is curvilinear, and a linear relationship is best obtained by the logarithmic transformation, to the base e, of the mean soil loss data. On the transformed data, soil loss shows the highest negative correlation with surface cover and maximum grass height, and the highest positive correlation with susceptibility to erosion (Table 3.11).

3.4 DERIVATION OF PREDICTIVE MODELS FOR SOIL LOSS AND RUN-OFF BASED ON VEGETATION AND SOIL SURFACE VARIABLES

3.4.1 Materials and methods

Multiple stepwise regression analysis of the 1982-84 rainfall simulator trial and vegetation monitoring data was used to derive predictive models for soil loss and run-off. Two predictive models were constructed for soil loss and run-off respectively:

a) predictive models based on the variables measured by Walker's (1976) method only, so that a retrospective analysis of soil loss and run-off could be carried out on vegetation monitoring data collected in the study area since 1979; and

b) predictive models based on variables measured by both Walker's (1976) and the USLE (Wischmeier, 1975) methods, for incorporation into the current vegetation monitoring programme for UGR (Figure 3.32).

Multiple regression analysis was also carried out on the 1982-85 data set, excluding a 20% random subsample of this data set. As was the case with the 1982-84 data set, two predictive models were constructed for soil loss and run-off respectively (Figure 3.32).

Because the independent variables were correlated in many cases, the partial regression coefficients were examined at each stage of the stepwise regression to ensure that they did not change in sign, or significantly in size. The significance of each partial stepwise regression coefficient was tested using analysis of variance and the F-test (Parker, 1976). Plots of residual values were examined for heteroscedascity to ensure that the regression equation was not biased (Freund & Minton, 1979). These predictive models were then tested against an independent data set collected during the 1985 rainfall simulator trials and against a 20% random subset of the entire 1982-85 data set (Figure 3.32). Using bivariate scattergrams, the actual and predicted values for run-off and soil loss were plotted and the points were visually examined for nonrandom deviations from the 1:1 line of perfect agreement. The coefficient of determination (D) was used to measure the degree of association between actual and predicted values (Aitken, 1973). However, although D is a good measure of the degree of association between actual and predicted values, it does not reveal the existence of systematic errors (Aitken, 1973), and the presence of such errors was determined by the coefficient of efficiency (E) which is defined below:

$$E = \frac{\sum(\bar{q}_c - q_c)^2 - \sum(q_c - q_e)^2}{\sum(\bar{q}_c - q_c)^2}$$

where q_c = actual soil loss or run-off values

$\bar{q}_c$ = mean of actual soil loss or run-off values

q_e = predicted soil loss or run-off values.

The difference between D and E, known as the error function (F_1), gives an indication of the systematic error in the model. The closer F_1 is to zero, the less systematic error occurs in the model (Hope, 1980). The most suitable model was defined as that model which had the highest value for D and the lowest value for F_1.

3.4.2 Results

3.4.2.1 Predictive models of soil loss

Using the 1982-84 rainfall simulator data, and the variables measured by the Walker (1976) method only, the following multiple regression equation was derived for the prediction of soil loss:

$$\text{Log}_e \text{ SOIL LOSS (g)} = 8{,}9141 - 0{,}0266 \text{ MAGH} - 0{,}016 \text{ HCC1} \quad \text{(Equation 3.1)}$$

This equation explained 73,7% of the variation in the data set (Table 3.12). When the variables measured by the USLE method (Wischmeier, 1975) were also included in the regression analysis, the equation explained 78,5% of the variability (Table 3.12). This predictive equation became:

$$\text{Log}_e \text{ SOIL LOSS (g)} = 9{,}0603 - 0{,}0301 \text{ SURC} - 0{,}0371 \text{ MEGH} \quad \text{(Equation 3.2)}$$

The regression analysis of the 1982-85 data was based on 54 sets of variables, because a 20% random subsample of 14 sets of variables had been removed from the data set. Of the total variation in the data, 78,6% was accounted for by the equation derived from the Walker (1976) method variables only (Equation 3.3), and 80,9% by Equation 3.4 which was derived from both methods (Table 3.13). The two equations are given below:

$$\text{Log}_e \text{ SOIL LOSS (g)} = 5{,}9383 + 0{,}03 \text{ SUTE} - 0{,}0211 \text{ MEGH} \quad \text{(Equation 3.3)}$$

$$\text{Log}_e \text{ SOIL LOSS (g)} = 9{,}0025 - 0{,}0319 \text{ SURC} - 0{,}0238 \text{ MEGH} \quad \text{(Equation 3.4)}$$

The regression equation which explained the highest percentage of the variability in the data was derived from the 1982-85 data set using the vegetation and soil surface variables measured by both methods (i.e. Equation 3.4). Also, a higher percentage of the variability was explained when independent variables measured by both methods were included in the regression analyses, irrespective of which data set was used.

3.4.2.2 Predictive models of run-off

Based on the 1982-84 data, the following equations were derived for the Walker (1976) method and for both methods respectively:

RUN-OFF (%) = 36,1376 - 0,6493 LITC + 0,6796 SOCA + 0,3626 MAGH
(Equation 3.5)

RUN-OFF (%) = -2,0964 - 0,485 LITC + 0,9995 SOCA + 0,7323 SURC
+ 2,3597 WOCC (Equation 3.6)

Equation 3.5 explained 59,3% of the total variability and Equation 3.6 explained 70,4% (Table 3.14).

The multiple regression analysis on the 1982-85 data using Walker's (1976) method, but excluding a random 20% subset of the data, explained 55,3% of the total variability (Table 3.15). Using both methods, 59,5% of the variability was explained. The equations are given below:

RUN-OFF (%) = 56,0545 + 0,544 HCC1 - 0,4 LITC - 10,0248 ROCK
(Equation 3.7)

RUN-OFF (%) = 5,5126 + 0,9777 SOCA + 0,5852 SURC - 0,4692 LITC
(Equation 3.8)

The model derived from the 1982-84 data set, using both methods to measure soil surface and vegetation variables (i.e. Equation 3.6), best explained the variability in the data, but in all cases less of the variability was explained by the run-off models than by the soil loss models (Tables 3.12 - 3.15).

3.4.2.3 Testing of soil loss models

Jeffers (1982) pointed out that if modelling is to be included as a part of scientific research, then it is essential to test the validity of the model by using it to make a prediction about the real-word system. Snee (1977) listed several methods to determine the validity of regression models, including the collection of new data to check model predictions, and data splitting or cross-validation in which a part of the data is used to estimate the model coefficients and the remainder of the data used to measure the prediction accuracy of the model. He concluded that data splitting was an effective method of model validation when it was not practical to collect new data to test the model.

There was no significant correlation between the predicted values for $\log_e$ soil loss, derived from Equation 3.1, when compared with the actual values measured during the 1985 rainfall simulator trials (r=0,434; d.f.=18). Predicted values from Equation 3.2 were however significantly correlated at the 5%, but not at the 1%, probability level (r=0,465; d.f.=18). Predicted values from both Equations 3.3 and 3.4 were highly significantly correlated (r=0,826 and 0,824 respectively; d.f.=12) with measured soil loss values in the reserve data subset. Although Equation 3.4 had a slightly lower D value than Equation 3.3 (Table 3.16), it was considered to have less systematic error than Equation 3.3 because its F_1 value was considerably lower than that of Equation 3.3. For this reason, Equation 3.4, which was derived from variables measured by both Walker's (1976) and the USLE (Wischmeier, 1975) methods, was considered to be the most accurate predictor of soil loss.

The distribution of soil losses measured during the 1985 rainfall simulator trials differed when compared with the 1982-84 trials, with the 1985 independent data set consisting entirely of soil losses of less than 1 000g, with a strong bias towards those of less than 250g (Figure 3.33). This meant that instead of testing the predictive equations through their entire range of soil loss, the 1985 independent data set tested the predictive ability of the equations only in the low to moderate soil loss range. In contrast, the 20% random subset of the 1982-85 data was more evenly distributed over the whole range of soil loss, and it was only soil losses of more than 5 000g which were not represented in this subset (Figure 3.34). This subset therefore tested the predictive ability of the equations over the low-to-high soil loss range and, for this reason, it was felt that this data subset gave a better validation of the predictive models. Based on the above, and on the respective values of D and F_1, Equations 3.3 and 3.4, which were derived from the random data subset, were considered to be the best predictors of soil loss, with Equation 3.4, which used variables from both the Walker (1976) and USLE (Wischmeier, 1975) methods, being marginally better at predicting soil loss.

3.4.2.4 Testing of run-off models

There was no significant correlation between predicted values from any of the four equations and actual values (Table 3.17). The correlation coefficient for the relationship between values predicted by Equation 3.8 and actual values, was just not significant at the 5% probability level (r=0,523; d.f.=12), and because this equation had the highest D and lowest F_1 value it was considered to be the best model (Table 3.17). Neither Equation 3.5 nor Equation 3.7, which were based on the Walker (1976) method variables only, showed high values for D or low values for F_1, and consequently no attempt was made to undertake a retrospective analysis of run-off at Walker vegetation monitoring sites in the study area.

3.5 RETROSPECTIVE ANALYSIS OF SOIL LOSS AT WALKER VEGETATION MONITORING SITES IN THE STUDY AREA

3.5.1 Materials and methods

There were 36 Walker transects located in the study area (Figure 3.35), but only those transects in woody vegetation communities which were sampled using the rainfall simulator were included in the retrospective data analysis. This amounted to 27 sites, ten in the cull block and 17 in the non-cull block (Table 3.18).

A $1m^2$ quadrat was laid down at 25 points at each Walker transect (Figure 3.36) and the following variables were assessed in each $1m^2$ quadrat using the Walker (1976) method:

a) Herbaceous canopy cover (%)

b) Litter cover (%)

c) Soil capping (%)

d) Susceptibility to erosion (%)

e) Rock (%)

f) Forb (%)

g) Aizoon (%)

h) Maximum grass height (cm)

i) Mean grass height (cm).

The definitions of these variables are given in Section 3.3.1.1. The mean values for variables a) to g) were determined using the same formula given in Section 3.3.1.1, and maximum and mean grass height were expressed as the mean values (in cm) for the 25 quadrats.

Equation 3.3 was used to calculate the $\log_e$ mean soil loss for each Walker vegetation monitoring site. These $\log_e$ values were then converted to mean soil loss and expressed in grams. The mean soil losses, their standard errors and ranges for all sites located in the cull block in 1979 were calculated. This was also done for those sites located in the non-cull block in 1979, and the procedure was repeated for both experimental blocks from 1980 to 1986.

The stocking rate differential between the non-cull and cull blocks was calculated from helicopter, fixed-wing and total ground counts of the study area from 1979 to 1986. The stocking rates of grazing herbivores (waterbuck *Kobus ellipsiprymnus*, buffalo *Syncerus caffer*, square-lipped rhinoceros *Ceratotherium simum*, blue wildebeest *Connochaetes taurinus* and zebra *Equus burchelli*) and mixed feeders (nyala *Tragelaphus angasi* and impala *Aepyceros melampus*) were calculated for each count for the non-cull and cull blocks separately using the following formula:

$$\text{Stocking rate (kg/ha)} = \frac{\sum [(n_1 \times m_1)(n_2 \times m_2)\ldots(n_n \times m_n)]}{Z}$$

where n_1 = number of species 1 counted

n_n = number of species n counted

m_1 = mean body mass of species 1 (kg)

m_n = mean body mass of species n (kg)

Z = area of experimental block (ha)

Mean body masses of the species listed above were obtained from figures published by Coe, Cumming & Phillipson (1976). The stocking rate differential was calculated for each count by dividing the stocking rate of the cull block into the stocking rate of the non-cull block. Where more than one count was done in any particular year, the mean stocking rate was also calculated. Because more animals are counted from a helicopter than from a fixed-wing aircraft (Knott & Brooks, 1986), data from fixed-wing counts were included only when no helicopter counts had been conducted in a particular year. Since the species composition of grazers and mixed feeders in the two blocks was similar, no attempt was made to standardise the stocking rate by converting the mean body mass of individual species into metabolic mass equivalents.

3.5.2 Results and discussion

Conceptually, one would expect that because of a lack of population control, herbivore density would increase over time in the non-cull block, while in the cull block it would remain relatively constant (Figure 3.37). These trends are shown by lines AB and AC respectively. With an increase in herbivore density in the non-cull block, the vegetation would be more heavily utilised, leading to a reduction in maximum and mean grass heights, herbaceous canopy cover and litter cover and to an increase in susceptibility to erosion and soil capping. This should result in a higher soil loss and increased run-off as time progresses. Because vegetation variables are also affected by the amount of precipitation received during the growing season, annual values for these variables will differ and will fluctuate about the AB trend line. These fluctuations will also cause soil loss and run-off to fluctuate about the trend line. In the cull block, because the grazer stocking rate presumably remains relatively constant, the utilisation remains constant and so does the soil loss and run-off, except for fluctuations about the AC trend line resulting from seasonal differences in rainfall, and hence, vegetation variables. Conceptually, when the experiment is started (at point A) the soil losses and run-off rates from both blocks should be similar, but as time progresses, the soil loss and run-off differentials between the two blocks should become progressively larger, as represented by points B and C. This latter concept is important, because when the actual results for both experimental blocks are interpreted, the progressive increase in the soil loss differential should be considered, rather than year-to-year fluctuations in soil loss.

The predicted mean soil losses from Walker transect vegetation monitoring sites are shown in Figure 3.38, and the mean, range and standard errors are given in Table 3.19. Exactly the opposite trend to what was expected conceptually is visible, with the soil loss differential decreasing, and not increasing, as time progresses (Figure 3.38). Two conclusions can be drawn from Figure 3.38, viz. either that an increase in stocking rate leads to a decrease in soil loss over time or that one or more of the assumptions made in the conceptual model were incorrect.

Actual stocking rate data from counts of the study area are summarised in Table 3.20 and it is clear from these data that the grazer and mixed feeder stocking rate in the non-cull block was never more than 40% higher than the cull block. The highest stocking rate differential between the two blocks was recorded in 1981 and then this differential declined until 1986 (Table 3.20). Although there were fluctuations about this declining trend, the stocking rate differential between the two blocks did not increase as time progressed, but rather peaked and then declined.

The assumption in the conceptual model of an increasing stocking rate differential with time was thus invalid.

The relationship between the predicted soil loss differential and the actual stocking rate differential is shown in Figure 3.39, and no clear relationship between these two variables is discernible. The stocking rate is essentially a measure of the average density of animals in the entire experimental block, on a maximum of three occasions, in any particular year. It gives no indication of habitat preference, and is an extensive instead of an intensive sampling method. In contrast, the Walker transects sampled only a minute part of the study area, and even in this small portion the predicted soil loss varied widely within sampling sites as seen in the large ranges and standard errors in Table 3.19. Also, the Walker transect vegetation monitoring sites were subjectively, and not randomly, placed (A.J. Wills - *pers. comm.*). The initial differences between the mean predicted soil losses in the two experimental blocks in 1979 strongly suggest that sites showing signs of heavy herbivore utilisation were selected in the non-cull block, and that the converse was true in the cull block. This means that the 1979 starting points of the trends shown in Figure 3.38 should be interpreted with caution, but it does not invalidate the observation made that the differential between the two blocks declined with time. To determine the relationship between stocking rate and soil loss, either more extensive sampling of the vegetation in the study area would be necessary, or more intensive sampling of the stocking rate. Without these data, it can be concluded that the stocking rate in the non-cull block did not increase relatively constantly over time, as one would expect in the absence of game removal, but peaked in 1981 and then declined to levels below that of the adjacent cull block. Because there was no consistent increase in the stocking rate differential between these two blocks over time, the soil loss differential did not increase as predicted and, when the relationship between stocking rate and soil loss is examined, no clear relationship is discernible, leading to the conclusion that the soil loss differential cannot be explained by a differential in stocking rate alone. Rather, it is postulated that there is a complex interrelationship between rainfall, vegetation cover, herbivore stocking rates and soil loss.

3.6 GENERAL DISCUSSION

The rainfall simulator trials on fixed sites located in key vegetation types showed a large degree of temporal variability within each site, in both soil loss and run-off, over the experimental period. Not only was there this temporal variability within sites, but sites located in the same experimental block on similar slopes, identical soil types and in the same woody vegetation community showed vastly different soil loss and run-off rates, indicating a high degree of spatial variability also. The two experimental blocks coincided with two game removal blocks (Figure 2.2). These are the smallest units of animal population control management, because culling and live removal figures for large herbivores are given per game removal block, and not on any smaller scale. The diversity of soil and vegetation types in the study area was emphasised in Sections 2.3 and 2.4 and is obvious from Figures 2.6 and 2.7 respectively. Superimposed on the spatial variability at this level is a spatial variability in soil loss and run-off *within* each soil and vegetation type as shown by the rainfall simulator trials. Again, superimposed on this level of spatial variability is a temporal component where soil loss and run-off varies *at the same site* with time. Overgrazing and high herbivore stocking rates have been put forward as key factors influencing soil loss and run-off rates in UGR, and large-scale game removals have been recommended to reduce excessive soil loss (Smuts, 1980). Given that game removals can be implemented practicably only at the game removal block level, and given the high degree of variability within any game removal block, the question that needs to be asked is whether the manipulation of the stocking rate through herbivore population control lowers significantly the rates of soil loss and run-off, *given that semi-arid savanna areas are characterised by marked fluctuations in annual rainfall*. In Chapter 7 the rationale of setting stocking rates with a drought situation in mind (Smuts, 1980) will be examined in depth, but for the present the high degree of spatial and temporal variability in soil loss and run-off suggests that the explanation for this lies in variables that fluctuate markedly from year to year (such as rainfall) instead of in variables which are less time dependent (such as herbivore stocking rate).

In this study, the relationship between vegetation, soil surface variables and soil loss was found to be curvilinear and best described by an exponential curve. Snyman, van Rensburg & Opperman (1985), using a Swanson rotating boom rainfall simulator on natural veld in the Orange Free State, also found a highly significant curvilinear relationship between soil loss and basal cover, which was best described by a logarithmic function. Trieste & Gifford (1980) used rainfall simulator data to evaluate the USLE in rangeland conditions and suggested that the equation could be optimised with certain exponents to account for more of the variability in the sediment yields. They found that the cover-management (C) factor was related to sediment yields by a power equation with a negative exponent. Wischmeier (1975) also showed a curvilinear relationship between his Type II cover (i.e. mulch and close-growing vegetation) and soil loss. The significance of a curvilinear, exponential relationship between soil loss and both vegetation and soil surface variables, is that as a variable, such as surface cover, halves from 50% to 25%, the soil loss does not double as it would with a linear relationship, but trebles. This has obvious implications for soil loss, especially at the lower end of the cover scale.

There was a statistically significant linear relationship between run-off and the soil surface and vegetation variables measured.

This does not agree with Snyman *et al.* (1985) who found a significant curvilinear relationship between percentage basal cover and percentage run-off. They did however measure cover differently by using basal instead of canopy cover, and their range of basal cover varied from 1 - 5%, while the range of herbaceous canopy cover measured in the current study varied from 5 - 93%. No other cover and run-off data are available from rainfall simulator trials on natural veld. However, there was a highly significant linear relationship between annual run-off and percentage bare soil measured on 17 range watersheds in western Colorado (Blackburn *et al.*, 1982).

When, in the present study, the four predictive models for soil loss were validated against a reserve data set, Equations 3.3 and 3.4 were considered to be the best predictors of soil loss. Because Equation 3.3 was based on independent variables measured by Walker's (1976) vegetation monitoring method, it could be used for a retrospective analysis of soil loss on permanently marked sites in the study area. The predicted soil losses from Walker transects in the cull and non-cull blocks showed a decreasing instead of increasing soil loss differential as time progressed, which was contrary to the conceptual trend. Closer examination of ungulate count data showed that the stocking rate did not increase with time in the non-cull block, but rather peaked in 1981 and then decreased. There was, however, no clear relationship between stocking rate and soil loss. Other studies have shown that such a relationship does exist; for example, Durnford (1954) found that grazing utilisation by herbivores substantially affected soil loss when more than half of the herbage was removed, and Smith (1967) found that the average erosion rates (kg/m^2) of surface run-off on heavily grazed rangelands were about four times that on lightly or moderately grazed rangeland. It is suggested in the current study that deficiencies in the experimental design, including nonrandom placement of vegetation monitoring sites, low sampling intensity and collection of stocking rate data at too coarse a level, confounded the relationship between stocking rate and soil loss. Nevertheless, the high degree of temporal variability in the data, irrespective of the grazer stocking rate, indicated that differences in soil loss could not be explained by differences in stocking rate alone, as there are complex interrelationships between rainfall, herbaceous vegetation variables and grazer stocking rates which affect soil loss.

From the rainfall simulator trials it can be concluded that the effect of the noninterventionist approach, adopted in the non-cull block, on soil loss rates was negligible when compared to the cull block over the same period. However, critics of the rainfall simulator technique may question this conclusion because rainfall simulators only simulate natural rainfall. The conclusion drawn in this chapter is based on two simulated storms each lasting 36 minutes and applying 36 mm of rainfall per site per annum. This poses the question whether it is valid to use simulated erosion events to build predictive models of soil loss and run-off, and then use these predictive models to draw conclusions about a natural system, when natural erosion events may have far different rainfall intensities and durations. For this reason, natural run-off plots were laid out in the study area.

3.7 CHAPTER SUMMARY

a) A rainfall simulator was used because it can simulate storms of standardised intensity and duration at various sites over a short period.

b) The results from the rainfall simulator trials showed high temporal and spatial variability, and the explanation for the variability in the data lay with time and spatially dependent, instead of independent, variables.

c) Correspondence analysis showed that soil capping and susceptibility to erosion were correlated positively with high levels of run-off and soil loss, while mean grass height and litter cover were correlated positively with low levels of run-off.

d) Bivariate scattergrams of soil loss against soil surface and vegetation variables showed that the relationship between the above variables and soil loss was curvilinear. Surface cover and maximum grass height had the highest negative correlation with mean soil loss once this latter variable had been transformed logarithmically. Susceptibility to erosion showed the highest positive correlation.

e) Four predictive models for soil loss based on two different data sets and on two different methods to measure soil surface and vegetation variables were derived, using stepwise multiple regression analysis, and all four models explained more than 70% of the variability in the data set.

f) The regression equation derived from the 1982-85 data set and using the vegetation and soil surface variables measured by both methods (Equation 3.4) explained the highest percentage of the variability in the data set.

g) A higher percentage of the variability was explained when independent variables measured by both methods were included in the regression analysis irrespective of which data set was used.

h) The predictive models were tested against an independently collected data set and a reserve data set, but it was felt that the reserve data set gave a better validation of these models.

i) When validated against the reserve data set, Equations 3.3 and 3.4 were considered to be the best predictors of soil loss.

j) The predicted soil losses from Walker transects in the cull and non-cull blocks showed a decreasing instead of increasing soil loss differential as time progressed, which was exactly the opposite trend to what was expected conceptually. A critical examination of the stocking rate showed that it did not increase with time in the non-cull block, but rather peaked in 1981 and then declined.

k) No clear relationship between soil loss and stocking rate was apparent, and the high degree of temporal variability in the data, irrespective of the grazer stocking rate, indicated that the differences in soil loss could not be explained by differences in stocking rate alone, and there was an interrelationship between rainfall, herbaceous cover, herbivore stocking rate and soil loss.

l) Bivariate scattergrams showed that the relationship between run-off and soil surface and vegetation variables was linear, and a correlation matrix indicated that soil capping had the highest positive correlation with run-off, and litter cover the highest negative correlation.

m) When validated against either an independently collected or a reserve data set, none of the four predictive models derived for run-off gave predicted values which correlated significantly with measured run-off values. For this reason, no retrospective analysis of run-off in the study area was undertaken.

n) In all cases, less of the variability in the data sets was explained by the run-off models than by the soil loss models.

CHAPTER 4 : NATURAL RUN-OFF PLOTS

4.1 INTRODUCTION

This part of the study had the following five aims:

a) to determine rainfall erosivity values for spatially diverse sites;

b) to determine the soil loss and run-off potentials of fixed sites within key woody vegetation communities;

c) to determine the interrelationships between vegetation and soil surface variables and soil loss and run-off;

d) to derive and validate predictive models for run-off and soil loss based on rainfall, vegetation and soil surface variables; and

e) to compare the ability of two different vegetation monitoring methods to predict accurately run-off and soil loss.

Soil loss depends on a combination of rainfall erosivity (i.e. the power of rain to cause erosion) and soil erodibility (i.e. the ability of soil to withstand the rain). Because there is a close association between erosion and rainfall intensity (Hudson, 1981), autographic raingauges were installed to measure rainfall intensity at various natural run-off plots.

Although it would have been ideal to measure soil loss and run-off from all 12 of the woody vegetation communities recognised in UGR by Whateley & Porter (1983), logistical constraints, particularly financial limitations, prevented the placement of natural run-off plots in all except four key communities. Unlike the rainfall simulator trials, where the aim was to cover as wide a variety of vegetation and soil types as possible, an attempt was made initially to locate comparative paired natural run-off plots in both experimental blocks. These run-off plots were paired with respect to slope, aspect, vegetation and soil type.

Because of the large amount of vegetation monitoring data which had been collected in the study area since 1979, an attempt was made to correlate measured soil loss and run-off with the different variables assessed using the Walker (1976) vegetation monitoring method. The ultimate aim of the exercise was to derive predictive models for soil loss and run-off based on variables measured using the Walker (1976) method, so that trends in both soil loss and run-off since the beginning of the non-cull/cull experiment in 1979 could be obtained.

Lastly, to assist in upgrading the present Natal Parks Board vegetation monitoring programme, other vegetation variables not measured by Walker's (1976) method were assessed, to see which variables gave the most accurate prediction of soil loss and run-off.

4.2 AUTOGRAPHIC RAINGAUGE DATA

4.2.1 Materials and methods

Measurements of rainfall amount and intensity were made using autographic raingauges which were installed at six sites in the study area adjacent to natural run-off plots (Figure 4.1). Data collection started in July 1983 and continued until June 1986. Each raingauge traced successive increments of rainfall as a cumulative total on a clock-driven chart which was changed at the end of every month. The data recorded on these charts were processed using the digitising system of the Department of Agricultural Engineering, University of Natal, Pietermaritzburg, which ran on a Hewlett-Packard 9816S microcomputer (Dent & Schulze, 1987). This procedure involved the digitising of the charts at breakpoints in the slope of the trace, and all the information recorded on the charts was stored, in digital form, as computer data files (Schulze & Arnold, 1980). For this study, the following variables were calculated from the autographic raingauge charts:

total monthly rainfall (mm);

total monthly kinetic energy of rainfall (J/m^2); and

EI_{30} index (erosivity units).

Total monthly rainfall was simply the cumulative total, in millimetres, of rainfall recorded on the chart in any particular month. The kinetic energy was calculated using the equation below which relates rainfall kinetic energy and rainfall intensity (Wischmeier & Smith, 1978):

$$E = 11{,}8975 + 8{,}7319 \log_{10} I$$

where E = kinetic energy ($J/m^2/mm$ rainfall)

I = rainfall intensity (mm/h).

Wischmeier, Smith & Uhland (1958) examined a large data set from natural run-off plots, and showed that the best estimator of soil loss, related to rainfall intensity, was the product of the kinetic energy of the storm and the 30-minute intensity. The latter is the highest average intensity, expressed in mm/h, measured in any 30-minute period during the storm, and is computed from the autographic raingauge data files (Hudson, 1981). This measure of rainfall erosivity is called the EI_{30} index, and the subscript 30 denotes that the greatest rainfall intensity was measured over a 30-minute period. The EI_{30} index was computed for individual breakpoint data using the formula below, and then summed to give monthly values for erosivity:

$$\text{Monthly } EI_{30} = \frac{\sum \text{daily } (E \times I_{30})}{1\ 000}$$

where E = kinetic energy (J/m^2)

I_{30} = maximum average intensity in any 30-minute period during a particular storm (mm/h).

Values of total rain, rainfall kinetic energy and EI_{30} for individual months from all sites were summarised and expressed as the range, mean and standard error. Monthly mean values for the above variables were also summed to give annual mean figures for individual rainfall years (July to June). Data were collected over three rainfall years and were tested statistically, using single classification Model II analysis of variance (Sokal & Rohlf, 1981), to establish whether there were any significant between-year differences.

4.2.2 Results

A monthly mean rainfall of more than 200mm was recorded in only one month during the study period, and this was linked to Cyclone Domoina which occurred in January 1984 (Figure 4.2). In a further seven months a monthly mean rainfall exceeding 100mm was measured, in all cases between the months of October and March.

It was only during Cyclone Domoina that the monthly mean rainfall kinetic energy exceeded 10 000 J/m^2, but in 11 other months it exceeded 1 000 J/m^2 (Figure 4.3). These high values always occurred between October and March, but there was a general peak in monthly mean rainfall kinetic energy values in January and February (Figure 4.4). The distribution of the monthly mean EI_{30} values is very similar to that of kinetic energy, with an EI_{30} value exceeding 100 000 erosivity units in January 1984, and EI_{30} values exceeding 10 000 erosivity units on 14 other occasions between the months of September and March (Figure 4.5). Unlike kinetic energy, there were peaks in monthly mean EI_{30} values in December, January and February in different years (Figure 4.6). The above figures show there is a large degree of temporal variability in these three variables within any individual rainfall season. There was sometimes also a high degree of spatial variability between sites as shown by the large ranges and standard errors (Tables 4.1 - 4.3). When annual mean values for the various rainfall variables were compared (Table 4.4), there were significant differences between rainfall years for each of these variables, indicating a high degree of temporal variability, not only within rainfall years but also between years.

4.3 SOIL LOSS AND RUN-OFF MEASUREMENTS

4.3.1 Materials and methods

Three key woody vegetation communities, each covering an extensive area, which were preferred by grazing herbivores and were susceptible to erosion because of their mid-slope position on the catena, were identified. These communities were *Acacia nigrescens* woodland, *Acacia tortilis* woodland and *Acacia nilotica/A. gerrardii* woodland. Natural run-off plots were located in these key communities, and before the 1983/4 rainfall season, 12 plots were installed at six different sites; three in the non-cull block and three in the cull block (Table 4.5). A paired plot approach was adopted where, once a natural run-off plot site had been selected in the non-cull block, a similar site was located in the same woody vegetation community, on the same soil form and preferably also the same soil series, and with a similar slope, in the cull block. This approach met with limited success, because while it was relatively easy to match up sites with regard to vegetation community and slope, it was difficult to match them up with soil form and particularly with soil series (Table 4.5). For this reason, a different sampling approach was adopted before the 1984/5 rainfall season, and a further eight plots were installed at four different sites in the non-cull block (Figure 4.7). These sites were located along a catena and extended from the upper mid-slope position to the bottom slope position on this catena. Three of the four sites were located in *Acacia tortilis* woodland, and the bottom slope site was located in *Acacia grandicornuta* woodland near a major drainage line (Table 4.5).

Two natural run-off plots were installed at each site to take replicate samples. Flat iron sheeting was used as shuttering to demarcate each plot which was 1,83m wide and 22,13m long (Figure 4.8). The significance of these dimensions was that each plot enclosed an area of one-hundredth of an acre, these being the standard dimensions for run-off plots in the United States of America (USA) (Wischmeier, 1955). The run-off from a plot during a rain storm was channelled via a collection trough to a sedimentation tank where most of the coarse soil particles were deposited (Figure 4.9). This sedimentation tank had a capacity of 150 litres and if the run-off from an erosion event was less than 150 litres then the entire soil and run-off sample was collected in this tank. Before installation each sedimentation tank was calibrated so that, by measuring the water level, run-off volume could be determined. One gram of powdered alum was then added to the collected run-off to speed up sedimentation, and the excess run-off was drained off. The remaining soil and some run-off were collected in 20-litre buckets with sealed lids and transported back to HGR where each sediment sample was air dried and weighed. When run-off from an erosion event exceeded 150 litres, the water flowed from the outlet of the sedimentation tank through a filtration unit, where the finer soil particles were filtered out, and then to waste via a flowmeter which measured the volume of excess run-off (Figure 4.9).

The natural run-off plots were monitored in the middle and at the end of each month and the following data were collected if an erosion event or events had taken place during the previous two weeks:

total run-off (litres) per plot; and
total soil loss (grams) per plot.

Because rain falling on the natural run-off plots is measured in millimetres, the total run-off, which was measured in litres, was converted to millimetres of run-off using the following formula:

$$\text{total run-off (mm)} = \frac{\text{total run-off (l)}}{40{,}4979}$$

After each rainfall year the data were summed for individual plots and then expressed as:

total annual run-off (litres) per plot; and
total annual soil loss (grams) per plot.

Total annual run-off in litres was also converted to total annual run-off in millimetres, using the formula given above.

Because run-off generated by Cyclone Domoina far exceeded the capacity of the collection apparatus, the data could not be used, and all soil loss and run-off results presented exclude this major erosion event. The mean annual run-off and mean annual soil loss were calculated for each site, and when more than one site was located in a particular vegetation type, the data for these sites were presented together.

4.3.2 Results

4.3.2.1 Soil loss potential of fixed sites within key woody vegetation communities

Of those sites which were monitored over three years, the highest mean annual soil loss was measured at site 402 located in *Acacia tortilis* woodland in the non-cull block, and the lowest soil loss was measured at site 411 located in the cull block in *Acacia nilotica/Acacia gerrardii* woodland (Table 4.6). In the above two woody vegetation communities, and in sites monitored over three years, higher mean annual soil losses were measured from sites in the non-cull block (402 and 294) than from sites in the cull block (284 and 411). However, in *Acacia nigrescens* woodland the mean annual soil loss was higher in the cull block site (403) than in the non-cull block site (241) (Table 4.6). From these data no consistent trend can be detected for three annual measurements of soil loss from the two experimental blocks.

The data in Table 4.6 are simply presented as baseline data to indicate the soil loss potentials from natural rainfall at fixed sites, measured over a period of 2-3 years. Because there was a significant difference between annual totals of rainfall, kinetic energy and EI_{30} values measured during the three rainfall years (Table 4.4), no attempt has been made to present graphically the annual soil loss data based on individual rainfall years, as between-year comparisons of soil loss would be invalid.

4.3.2.2 Run-off potential of fixed sites within key woody vegetation communities

The highest mean annual run-off, monitored over three years, was measured from site 284 located in *Acacia tortilis* woodland in the cull block. The lowest mean annual run-off, monitored over the same period, was recorded in *Acacia nigrescens* woodland in the cull block at site 403 (Table 4.7). Comparison of sites in the same woody vegetation community, which were monitored over a period of three years, showed the mean annual run-off was highest from sites 284, 241 and 411. Two of these three sites were located in the cull block (284 and 411) and there appeared to be no consistent trend in run-off when sites from both experimental blocks were compared.

No between-year comparisons of annual run-off were made, because rainfall amount and intensity varied significantly between years (Table 4.4).

4.4 RELATIONSHIP BETWEEN RUN-OFF, SOIL LOSS AND VEGETATION AND SOIL SURFACE VARIABLES

4.4.1 Vegetation and soil surface variables

4.4.1.1 Materials and methods

A $1m^2$ quadrat was laid down systematically at 20 points within each natural run-off plot, and two semi-quantitative methods were used to assess vegetation and soil surface variables. Each plot was assessed three times per rainfall year: in the early growing season (October), the mid-growing season (January) and late growing season (March). The following variables were assessed in each $1m^2$ quadrat using the Walker (1976) method: herbaceous canopy cover (%); litter cover (%); soil capping (%); susceptibility to erosion (%); rock (%); forb (%); *Aizoon* (%);

maximum grass height (cm), and mean grass height (cm). Based on a sample size of 20 quadrats per plot, mean values for the above variables were calculated using the formulae given in Section 3.3.1.1. The results for the early, mid and late growing season assessments were pooled and a mean value for the above variables was obtained, per natural run-off plot, for individual rainfall years. Plots were assessed over three rainfall years, starting in the 1983/4 growing season.

The following variables were also assessed in each $1m^2$ quadrat using the USLE method (Wischmeier, 1975): herbaceous canopy cover (%); surface cover (%); woody canopy cover (%), and woody canopy height (cm). Mean values, based on 20 quadrats per plot, were calculated for each of the above variables using the formulae given in Section 3.3.1.1. Mean values for vegetation and soil surface variables measured at each natural run-off plot were calculated annually from the early, mid and late growing season data.

4.4.2 Relationship between vegetation and soil surface variables and run-off and soil loss

4.4.2.1 Materials and methods

Correspondence analysis, along with other multivariate analysis techniques, can summarise large, multivariate data sets into a low-dimensional space such that similar entities are close by and dissimilar entities are far apart. Because of this summarisation, the comprehension of the data set is made easier, and thus the interpretation of the data is facilitated and the results can be communicated more effectively (Gauch, 1982).

In the present study, correspondence analysis was used to find the best objective fit to the association between vegetation and soil surface variables and soil loss and run-off. To examine the above interrelationships, the data from the natural run-off plots were analysed separately for the 1983/4, 1984/5 and 1985/6 rainfall years. The data matrix was drawn up with the independent (vegetation and soil surface) variables as rows and the dependent (annual soil loss and annual run-off) variables as columns. The structure of a typical multivariate data matrix was shown in Table 3.5. The mean for each independent variable falling in a particular soil loss or run-off class was calculated using the formula given in Section 3.3.2.1, and an explanation of the codes used in the text, Figures 4.10 - 4.12 and Tables 4.9 - 4.11, is given in Table 4.8. In some of the original analyses, a single variable played an overwhelming role in determining the second principal plane and in effect swamped the second principal axis. In subsequent analyses the influence of this obvious and isolated feature was removed from the display so that the more subtle multidimensional patterns could be investigated, and the variable was treated as a supplementary point.

Apart from correspondence analysis, data from all three rainfall years were combined and bivariate scattergrams, plotting a particular independent variable against soil loss and run-off, were drawn and examined. It was evident from this examination that the relationship between the independent and dependent variables was curvilinear, and thus both annual soil loss and annual run-off values were transformed logarithmically to the base e. Because of this transformation, the relationship between the independent and dependent variables became more linear, which is a prerequisite for the construction of a product-moment correlation matrix (Sokal & Rohlf, 1981). Those independent variables which were correlated significantly with either annual soil loss or annual run-off were then presented as bivariate scatter plots. A statistical program running on an Apple IIe microcomputer was used to produce the scattergrams and the cross correlation matrices.

4.4.2.2 Relationship between run-off, soil loss and vegetation and soil surface variables as determined by correspondence analysis

In the 1983/4 natural run-off plot data, the first principal axis (axis 1 in Figure 4.10) distinguished very low annual soil loss and run-off on the left hand side of the axis from very high annual soil loss on the right. Soil capping and susceptibility to erosion were correlated positively with the very high soil loss, while litter cover was correlated positively with very low soil loss and run-off. This relationship was to be expected, because high levels of soil capping, and particularly susceptibility to erosion, would lead to very high levels of soil loss due to the lack of vegetation cover which protects the soil from the erosive effect of rainfall (Wischmeier, 1975). In contrast, with high levels of litter cover, very low levels of soil loss and run-off could be expected because these high levels would not only protect the soil surface from raindrop splash erosion (Wischmeier, 1975) but also facilitate infiltration (Blackburn *et al.*, 1982). Because the angles between LITC, the origin and SOCA and SUTE respectively were large, the latter two variables were not correlated closely with LITC along this profile (see Table 4.8 for explanation of codes). In axis 2, ASL2 (low annual soil loss) was treated as a supplementary point because of its excessively high contribution to the second principal axis. This axis distinguished between low levels of run-off and soil loss at the bottom of the axis, and high levels of soil loss at the top (Figure 4.10). Forb was correlated positively with low levels of run-off and soil loss; and woody canopy cover and, to a lesser extent, litter cover were correlated positively with high soil loss levels (Table 4.9). The relationships in this axis were not as easy to interpret as in axis 1. With high levels of woody canopy cover one would expect a shading effect and a reduction in herbaceous canopy cover and, consequently, high levels of soil loss. However, the relationship between high levels of litter cover and high soil loss levels could not be explained, particularly when in axis 1 LITC was correlated with very low soil loss. Similarly, the relationship between FORB and low soil loss and run-off was difficult to explain, because presumably a high forb component in the herbaceous vegetation would have resulted from heavy utilisation of the grass component and under such conditions one would not expect low levels of soil loss and run-off. Alternatively, the high forb component might have indicated a sward with a high species richness and in this case it would have been possible for high forb levels to be correlated with low levels of soil loss and run-off. Axes 1 and 2 explained 93,3% of the variability (Figure 4.10).

In the 1984/5 natural run-off plot data, axis 1 contrasted low annual run-off on the left side of the axis to very low, as well as high, annual run-off values (Figure 4.11). The profile along this first principal axis does not make ecological sense because there appears to be no theoretical reason why the axis distinguishes between low annual run-off as opposed to very low and high annual run-off. In most of the other analyses the distinction has been made between low soil loss and run-off levels on one side of the axis and high levels on the opposite side of the axis. Litter cover was correlated positively with very low and high run-off. The correlation between litter cover and very low run-off was expected, as a high litter cover would facilitate rainfall infiltration (Blackburn *et al.*, 1982), but there was no theoretical basis for a relationship between high litter cover and high run-off levels. Forb and, to a lesser extent, woody canopy cover were correlated positively with a moderate soil loss and a low run-off. The correlation between moderate soil loss and high values for both FORB and WOCC was expected, because in a situation where there is a high woody canopy cover (indicative of bush encroachment) and a high forb component (indicative of heavy herbivore utilisation) at least moderate levels of soil loss could be expected. The relationship between low run-off levels and WOCC and FORB could not be explained. FORB and LITC were almost 180° apart and were correlated negatively along this axis. Since high litter cover levels are indicative of low herbivore utilisation and high forb levels indicative of high herbivore utilisation, this relationship was to be expected.

Moderate annual run-off (ARO3) made a disproportionately high contribution to the second principal axis in the original analysis and was thus treated as a supplementary point in the subsequent analysis. Axis 2 contrasted moderate run-off (ARO3) and low soil loss (ASL2) with very low run-off (ARO1) and moderate soil loss (ASL3)(Figure 4.11). Soil capping was correlated positively with moderate run-off and to a lesser extent with low soil loss, while woody canopy cover was correlated positively with moderate soil loss and to a smaller degree with very low run-off (Table 4.10). It could be expected that high levels of soil capping would result in at least moderate levels of run-off, because soil capping reduces infiltration, thereby enhancing run-off (Dadkhan & Gifford, 1980). The relationship between high soil capping and low levels of soil loss had been observed in the field where some sites, particularly those located on the Swartland soil form, showed high levels of soil capping which facilitated run-off. However, despite the high run-off levels, very little soil was lost from the hard, impermeable soil surface. A good example of this phenomenon was site 284, where the mean annual run-off was 1 709 litres, while the mean annual soil loss was only 1 327 grams (Table 4.6). The positive correlation between WOCC and moderate soil loss was also emphasised in axis 1 and the explanation for this relationship has already been provided. A total of 89,8% of the variability was explained by axes 1 and 2 (Figure 4.11).

Axis 1 of the 1985/6 natural run-off plot data contrasted very low run-off (ARO1) and very low soil loss (ASL1) on the left-hand side of the axis to moderate soil loss (ASL3) and moderate run-off (ARO3) on the right (Figure 4.12). Litter cover was correlated positively with the very low run-off and soil loss levels, while soil capping and susceptibility to erosion were correlated positively with moderate levels of soil loss and run-off. LITC was correlated negatively with SUTE and with SOCA along this axis. The positive correlation between high levels of litter cover and very low levels of soil loss and run-off was expected and the reasons for the relationship have already been discussed. The positive correlation between high levels of soil capping and susceptibility to erosion and moderate levels of soil loss and run-off was also to be expected, because soil capping would impede infiltration and facilitate run-off, while moderate levels of run-off, and especially soil loss, were to be expected with high susceptibility to erosion values. This latter variable was essentially a measure of the percentage of bare ground, and the higher this percentage, the greater the surface area of the soil which would be exposed to raindrop splash erosion. Axis 1 explained 87,0% of the variability in the data set (Figure 4.12).

Axis 2 was anomalous because the COR values for this axis, and particularly for the dependent variables, were low (Table 4.11). The significance of these low COR values was that the second principal axis did not graphically represent the variables very accurately, particularly ASL2, ARO1 and ARO2 (Table 4.11). For this reason, axis 2 was not interpreted.

In summary, the correspondence analyses on the natural run-off plot data showed that high levels of soil capping and susceptibility to erosion were correlated positively with moderate to very high levels of soil loss and run-off. In contrast, very low levels of soil loss and run-off were correlated positively with high levels of litter cover. Of all the independent variables used in the correspondence analyses, the above three consistently explained the greatest variability in the data sets.

4.4.2.3 Interpretation of scattergram results

Log$_e$ annual soil loss showed the highest positive correlation with susceptibility to erosion and soil capping , and the highest negative correlation with surface cover and litter cover (Table 4.12). Log$_e$ annual run-off also showed the highest negative correlation with litter cover, and the highest positive correlation with soil capping (Table 4.12). Except for rock and woody canopy height, all the variables which were correlated significantly showed a higher correlation with log$_e$ annual soil loss than with log$_e$ annual run-off.

The statistically significant correlations with log$_e$ annual soil loss and log$_e$ annual run-off as dependent variables are presented in Figures 4.13 - 4.21 and 4.22 - 4.31 respectively. Log$_e$ annual soil loss was correlated negatively with herbaceous canopy cover, litter cover, maximum and mean grass height, and surface cover (Figures 4.13 - 4.14 and 4.18 - 4.21). Herbaceous canopy cover, as measured by the USLE method (Wischmeier, 1975), had a marginally lower coefficient of correlation with log$_e$ annual soil loss than herbaceous canopy cover measured by the Walker (1976) method (Figures 4.20 and 4.13 respectively), and surface cover, which was measured by the USLE method, showed the highest correlation with soil loss (Figure 4.21). Soil loss was correlated positively with soil capping, susceptibility to erosion and the contribution of the forb *Aizoon glinoides* to herbaceous biomass (Figures 4.15 - 4.17 respectively). In the case of *Aizoon glinoides*, the positive correlation could be explained by the fact that this forb covered extensive areas of UGR only at the height of the drought in 1982 and 1983 (see Section 2.2.2). Under these drought conditions there was very little plant cover, and high soil losses were experienced, as was measured on the rainfall simulator trial sites in April 1983 (Figures 3.4 - 3.7). It was also during these below-average rainfall years that *Aizoon glinoides* flourished. The drought was broken by Cyclone Domoina in January 1984 and the grass sward recovered, while *Aizoon glinoides* declined. As a result of the recovery of the grass sward, herbaceous canopy cover increased and soil loss levels decreased, as shown by the rainfall simulator results in 1984 and 1985 (Figures 3.4 - 3.7).

Only soil capping and susceptibility to erosion were correlated positively with $\log_e$ annual run-off, with soil capping showing the highest correlation coefficient overall (Figures 4.24 and 4.25). Log_e run-off was correlated negatively with herbaceous canopy cover, litter cover, rock, maximum and mean grass height, surface cover and woody canopy height (Figure 4.22 - 4.23 and 4.26 - 4.31). The significant negative correlation between $\log_e$ run-off and rock (Figure 4.26) should be interpreted with caution because the range in percentage rock is low, with the highest value being 4,1%. Litter cover, which was measured by the Walker (1976) method, had the highest correlation coefficient (Figure 4.23) and when the two methods of assessing herbaceous canopy cover were compared, the measurement using the Walker (1976) method showed a marginally higher correlation coefficient with $\log_e$ run-off (Figures 4.22 and 4.29 respectively).

In summary, the relationships between the soil surface and vegetation variables and annual soil loss and annual run-off were curvilinear. When the latter two variables were transformed logarithmically, then soil loss showed the highest positive correlation with susceptibility to erosion, and the highest negative correlation with surface cover. Log_e annual run-off showed the highest positive correlation with soil capping, and the highest negative correlation with litter cover. Of those variables correlated significantly with either soil loss or run-off, only rock and woody canopy height showed a higher correlation with $\log_e$ annual run-off than with $\log_e$ annual soil loss. This significant correlation shown by rock should however be interpreted with caution because of the low range of percentage rock values.

4.5 DERIVATION OF PREDICTIVE MODELS FOR SOIL LOSS AND RUN-OFF BASED ON RAINFALL, VEGETATION AND SOIL SURFACE VARIABLES

4.5.1 Materials and methods

4.5.1.1 Rainfall variables

Autographic raingauge records were kept at six sites in the study area during the 1983/4, 1984/5 and 1985/6 rainfall years (Section 4.2), and based on these records the following variables were calculated:

a) total annual rainfall (mm), which was defined as the total rainfall measured at a particular site during a rainfall year;

b) total wet season rainfall (mm), which was the amount of rainfall measured at a particular site during the period October to March; and

c) total effective rainfall (mm), which was defined as the cumulative annual total of rainfall from rainfall events where 10mm or more of rain fell over a 24-hour period starting at 08h00.

The cutoff point of 10mm for effective rainfall was determined by plotting those erosion events where less than 25g of soil was lost against total rainfall measured during the particular event (Figure 4.32), and also by plotting total rainfall against run-off events where less than 20 litres of run-off was measured (Figure 4.33). In both cases, when less than 10mm of rain fell, no run-off or soil loss was recorded.

Occasionally, the clock mechanisms of the autographic raingauges were subject to mechanical failure, and rainfall data were not collected over these periods. Missing data values for rainfall recorded during an erosion event were synthesised using double mass analysis (Schulze, 1975). In this technique, all the rainfall values recorded during erosion events for a particular autographic raingauge, but excluding missing values, were plotted as a cumulative total against the mean rainfall values for each corresponding erosion event. These mean values, obtained from all the autographic raingauges which were functioning during the specific erosion event, were also plotted as a cumulative total (Figure 4.34). The final ratio between the two cumulative totals was then used to correct for missing values using the following formula:

$$\text{Synthesised value} = \frac{M}{R}$$

where M = mean value for particular erosion event

$$R = \frac{\text{final mean cumulative total}}{\text{final cumulative total for specific autographic raingauge}}$$

4.5.1.2 Construction and validation of predictive models

A data matrix consisting of the 1983/4, 1984/5 and 1985/6 natural run-off plot data, the vegetation monitoring data and rainfall data was drawn up. A 25% random subsample of this data matrix was removed and held in reserve to measure the prediction accuracy of the models (Snee, 1977), and multiple stepwise regression analysis on the remainder of the data matrix was used to derive predictive models for soil loss and run-off. These analyses were undertaken on an Apple IIe microcomputer using a statistical program. Two predictive models were developed for both soil loss and run-off:

a) predictive models based on the variables measured by the Walker (1976) method only, which enabled a retrospective analysis of soil loss and run-off to be carried out on vegetation monitoring data collected in the study area since 1979; and

b) predictive models based on combined variables measured by both the Walker (1976) and USLE methods (Wischmeier, 1975) in order to determine which variables gave the most accurate prediction of soil loss and run-off, so that the monitoring of these vegetation and soil surface variables could be incorporated into the current vegetation monitoring programme for UGR.

Because the independent variables were frequently correlated, those variables used in the regression model were checked for autocorrelation using the Durbin-Watson statistic (Freund & Minton, 1979). The significance of each partial regression coefficient was tested using analysis of variance and the F-test (Parker, 1976), and plots of residual values were also examined for heteroscedascity (Freund & Minton, 1979). The predictive models were then tested against the 25% random subset of the data. Bivariate scattergrams of the actual and predicted values were plotted and examined visually for deviations from the 1:1 line of perfect agreement. The coefficient of determination (D) was used to measure the degree of association between actual and predicted values, and the coefficient of efficiency (E) was used to determine the presence of systematic errors. The difference between D and E, known as the error function (F_1), gives an indication of the systematic error in the model. The closer F_1 is to zero, the less systematic error occurs in the model (Hope, 1980). The predictive model which had the highest value for D and the lowest value for F_1 was considered to give the most realistic predictions. The relevant formulae are given in Section 3.4.1.

4.5.2 Results

4.5.2.1 Predictive models of soil loss

Based on the variables measured by the Walker (1976) method only, the following multiple regression equation was derived for the prediction of annual soil loss:

$$LOG_e \text{ ANNUAL SOIL LOSS (g)} = 2,8715 + 0,0582 \text{ SUTE} + 0.0036 \text{ TOTAL RAIN}$$

(Equation 4.1)

This equation explained 51,2% of the variability in the data set (Table 4.13). When the variables measured by the USLE method (Wischmeier, 1975) were also included, the equation explained 51,7% of the variation, which was only marginally better than Equation 4.1 (Table 4.13). The predictive equation was:

$$\mathrm{LOG_e\ ANNUAL\ SOIL\ LOSS\ (g)} = -3{,}6286 + 0{,}1346\ \mathrm{SUTE} + 0{,}0953\ \mathrm{SURC} \quad \text{(Equation 4.2)}$$

4.5.2.2 Predictive models of run-off

The following equation was derived for the Walker (1976) method:

$$\mathrm{LOG_e\ ANNUAL\ RUN\text{-}OFF\ (l)} = -2{,}1043 + 0{,}0254\ \mathrm{SOCA} + 0{,}0244\ \mathrm{FORB} + 0{,}025\ \mathrm{WET\ SEASON\ RAIN} - 0{,}01\ \mathrm{EFFECTIVE\ RAIN} \quad \text{(Equation 4.3)}$$

This equation explained 61,4% of the variability in the data set (Table 4.14). Equation 4.3 was converted to millimetre equivalents by dividing by $\log_e$ 3,7012, which yielded the following equation:

$$\mathrm{LOG_e\ ANNUAL\ RUN\text{-}OFF\ (mm)} = -1{,}5969 + 0{,}0254\ \mathrm{SOCA} + 0{,}0244\ \mathrm{FORB} + 0{,}025\ \mathrm{WET\ SEASON\ RAIN} - 0{,}01\ \mathrm{EFFECTIVE\ RAIN} \quad \text{(Equation 4.4)}$$

When the variables of the USLE method (Wischmeier, 1975) were also included in the multiple regression analysis, exactly the same equation as Equation 4.3 above was derived; therefore none of the USLE variables were included in the predictive equation (Table 4.14).

4.5.2.3 Testing of soil loss models

There was a highly significant correlation between predicted $\log_e$ soil loss values, derived from both Equations 4.1 and 4.2, and actual soil loss values held in the reserve data subset (Table 4.15). Equation 4.2, which had the highest D value and the lowest

F_1 value, was considered to give the most accurate predictions of soil loss (Table 4.15). This equation was derived from variables measured by both the Walker (1976) and USLE methods (Wischmeier, 1975).

4.5.2.4 Testing of run-off models

Since no variables measured by the USLE method were incorporated in the predictive models for run-off (Section 4.5.2.2), only Equation 4.4 was tested against the reserve data subset. The predicted run-off values were correlated significantly with actual values although both the D and F_1 values indicated that this model did not predict run-off as accurately as Equations 4.1 and 4.2 predicted soil loss (Table 4.15). This equation was used in the retrospective analysis of run-off at Walker vegetation monitoring sites.

4.6 RETROSPECTIVE ANALYSIS OF SOIL LOSS AND RUN-OFF AT WALKER VEGETATION MONITORING SITES IN THE STUDY AREA

4.6.1 Materials and methods

A total of 27 representatively placed Walker transects, ten in the cull block and 17 in the non-cull block, was included in the retrospective data analysis of soil loss and run-off (see Figure 3.35). The variables assessed at each Walker transect and details of the Walker (1976) method are given in Sections 3.3.1.1 and 3.5.1.

Three rainfall variables were calculated from daily rainfall records, kept at Mpila camp and at Mbhuzana outpost, for each rainfall year. These variables were:

a) total annual rainfall (mm);

b) total wet season rainfall (mm); and

c) total effective rainfall (mm).

The above variables are defined in Section 4.5.1.1. Because rainfall data collection was started at Mbhuzana outpost only in April 1981, rainfall variables for the study period before this (i.e. the 1978/79, 1979/80 and 1980/81 rainfall years) were calculated from daily rainfall records kept at Mpila (Table 4.16).

Equation 4.1 was used to calculate the $\log_e$ annual soil loss for each Walker vegetation monitoring site. These values were converted to annual soil loss values and expressed in grams. Annual run-off values were also calculated by converting the $\log_e$ values, predicted by Equation 4.4, to annual run-off values expressed in millimetres. The mean annual soil loss and mean annual run-off, and their standard errors, 95% confidence limits and ranges were calculated separately for Walker vegetation monitoring sites located in the cull and non-cull blocks in 1979. The procedure was repeated for both experimental blocks from 1980 to 1986.

The predicted values for soil loss and run-off were tested for normality using the Q-Q plot correlation coefficient (Johnson & Wichern, 1982) and once it was determined that the data were distributed normally, the mean soil loss and run-off and their 95% confidence limits were plotted for each experimental block for each year separately over the period 1979 to 1986.

4.6.2 Results and discussion

4.6.2.1 Retrospective analysis of soil loss

Not only did the predicted annual soil loss vary between rainfall years in each experimental block, but there was also a high degree of variability within any individual rainfall year at different Walker transects. This was obvious from the extremely wide ranges and large standard errors in individual rainfall years, particularly in the non-cull block (Table 4.17).

Between-year fluctuations in annual soil loss were to be expected, because the predictive equation took into account the total rainfall received during any particular rainfall year, and this variable varied widely from year to year (Table 4.16). The within-year fluctuations in predicted soil loss from different

Walker transects emphasised the spatial variability in the system, and when the mean annual soil losses and their 95% confidence limits were compared between the cull and non-cull blocks, significantly higher mean soil losses occurred on only three occasions in the non-cull block, viz. during the 1978/79, 1979/80 and 1982/83 rainfall years (Figure 4.35). The general trend was towards a decrease in the soil loss differential between the experimental blocks as time progressed (Figure 4.35). Because of the large fluctuations in mean annual soil loss values, the differential between the two experimental blocks was more meaningful than the annual values themselves, and this differential was compared to the actual stocking rate differential to see if there was any relationship between these two variables. The assumed relationship was that an increase in the stocking rate differential would lead to an increase in the soil loss differential, and that a decrease in the stocking rate differential would have the opposite effect. When these two differentials were compared no consistent relationship was evident, with an increase in the stocking rate differential being accompanied by a decrease in the soil loss differential in some years, and the opposite being evident in other years (Figure 4.36). The problem of trying to relate two variables which were sampled at different levels of intensity, the one (soil loss) being very site-specific and the other (stocking rate) representing an average figure for an extensive area of land, was discussed in Section 3.5.2. There are other flaws in the experimental design which will be discussed in Section 4.6.2.3.

4.6.2.2 Retrospective analysis of run-off

The predicted annual run-off did not show as much variability about the mean as the predicted values for soil loss, and in most cases the standard errors of the mean and the ranges were small (Table 4.18). Generally, the trend in both blocks was very similar, with significantly higher predicted annual mean run-off values recorded in the non-cull block on only two occasions, viz. in the 1978/79 and 1983/84 rainfall years (Figure 4.37). Unlike the trend in the annual mean soil loss, there was no decrease in the run-off differential between the two experimental blocks as time progressed (Figure 4.37). The extremely high mean annual run-off value for the non-cull block during the 1983/84 rainfall year was primarily because of two outlier data points. The predicted run-off exceeded 250 mm from both points, and these high predicted values were a result of exceptionally high forb levels at the two sites. Without these two data points, the predicted mean annual run-off would have decreased by 37,4 mm, from 111,9 mm to 74,5 mm (Table 4.18).

There was no consistent relationship between the actual stocking rate differential and the predicted run-off differential (Figure 4.38). A rise in the stocking rate differential between 1980 and 1981 was accompanied by a decrease in the predicted run-off differential, while an increase in the stocking rate differential between 1983 and 1984 was accompanied by a corresponding rise in the predicted run-off differential (Figure 4.38).

4.6.2.3 Critical evaluation of the sampling design

Soil loss, run-off and stocking rate differentials were calculated to establish the extent to which the noninterventionist approach in the non-cull block had an impact on soil loss and run-off rates, when compared with the interventionist approach adopted in the cull block. Results in this section showed that the soil loss and run-off differentials either decreased with time or remained relatively constant, despite no animals being removed from the non-cull block. Results presented in Section 3.5.2 showed that the stocking rate differential peaked and then declined, instead of increasing constantly with time as expected. When the stocking rate differential was compared with the soil loss and run-off differentials, no apparent relationship was discernible, and the conclusion drawn from these results was that stocking rate *per se* was not an over-riding factor influencing soil loss or run-off. Nevertheless, it has been recognised for over 75 years that heavy continuous grazing accelerates erosion and run-off (Rich, 1911) and data from natural run-off plots in South Africa, collected approximately 25 years ago, showed that run-off and soil loss from heavily grazed veld was higher than that from moderately grazed veld (du Plessis & Mostert, 1965). In the light of the extensive volume of literature pointing to the fact that stocking rate does influence soil loss and run-off (Moore *et al.*, 1979b; Gifford & Springer, 1980; Blackburn *et al.*, 1982), the experimental design leading to the conclusion made in this study needs to be considered.

One of the ten principles that Green (1979) listed for optimal sampling design was that randomly allocated samples for each control variable should be taken, and he pointed out that the placement of sampling sites in "representative" locations was not random sampling. The Walker transects were not randomly placed, nor were they placed using some form of stratified random sampling design (Section 3.5.2). Another important principle in sampling design is that areas with a large-scale environmental pattern need to be subdivided into homogeneous subareas, and samples allocated to each subarea in proportion to its size (Green, 1979). The heterogeneity of UGR, including the vegetation, was emphasised in Section 2.4, and samples were allocated to homogeneous subareas, viz. woody vegetation communities. Unfortunately, these communities were not sampled in proportion to their sizes. Over one half of the sites sampled in the non-cull block were in *Acacia grandicornuta/Spirostachys africana* woodland (see Table 3.18), although this community was less extensive in the non-cull block than the *Acacia tortilis* community (Whateley & Porter, 1983). By using these sampling sites to predict soil loss and run-off from the entire block, the predictions may have been biased. Also, the non-cull block contained 70% more sampling sites than the cull block, although it is only 35% larger than the latter (see Table 3.18). There was thus a disproportionately higher sampling intensity in certain woody vegetation communities and in the non-cull block, which may have led to biased predictions of soil loss and run-off from the study area.

Because of the extended period over which the vegetation monitoring was undertaken, at least five different observers estimated the various soil surface and vegetation variables at the Walker transects (A.M. Whateley - *pers. comm.*). A recent study on the effect of observer bias (Gotfryd & Hansell, 1985) showed that observers differed significantly in their measurements of 18 out of a total of 20 vegetation variables, including two of the important variables measured in this study, viz. canopy cover and ground cover. However, because of the long-term nature of this experiment, staff changes were inevitable and the impact of observer bias was therefore unavoidable. This bias should nevertheless be considered when interpreting the data.

The main flaw in the experimental design was that the expected increase in the stocking rate differential between treatments did not materialise with time. To test whether stocking rate had an effect on soil loss and run-off, sampling needed to be done both where the effect of culling was expected to be present (i.e. in the cull block) and where it was absent (i.e. in the non-cull block) but where all else was the same. An effect can only be demonstrated by comparison with a control (Green, 1979), and in this experiment the stocking rate in the non-cull block was never more than 40% higher than that of the control (cull) block, and in some years it was below that of the control (see Table 3.20). The conclusion is that because of the less than optimal sampling design, the small differential in stocking rate between two experimental blocks, and the large variability in run-off and soil loss rates within each experimental block, no cause and effect relationship between stocking rate and both run-off and soil loss could be demonstrated.

4.7 GENERAL DISCUSSION

A higher mean annual soil loss was measured on two of the three paired natural run-off sites in the non-cull block when compared with the sites in the cull block. However, in *Acacia nigrescens* woodland, the soil loss was higher in the cull block site than the non-cull block site. Both the highest and lowest mean annual run-off values were measured at paired sites in the cull block, and there was no consistent trend in soil loss and run-off in the two experimental blocks. This was primarily because of the high degree of temporal and spatial variability in the data set.

The relationship between soil surface and vegetation variables and both annual soil loss and annual run-off was curvilinear, and an exponential curve best fitted the data. Snyman & van Rensburg (1986), using a rainfall simulator on natural veld, also found curvilinear relationships between canopy cover, soil loss and run-off. Such a curvilinear relationship between canopy cover, ground cover, percentage bare soil and soil loss has been recognised by many authors (Wischmeier, 1975; Wischmeier & Smith, 1978; Dissmeyer, 1982; Gebhardt, 1982). When annual soil loss values were transformed logarithmically to the base e, then susceptibility to erosion showed the highest positive correlation, and surface cover the highest negative correlation, with soil loss. Since a good vegetation cover helps to arrest the erosive energy of rainfall (Stocking, 1973), it is not surprising that high levels of surface cover give rise to low soil losses; conversely, sites having high levels of bare ground (and hence high levels of susceptibility to erosion) will be more vulnerable to rainfall erosivity and this will lead to high soil losses. Correspondence analysis also showed that susceptibility to erosion was correlated positively with moderate to very high levels of soil loss. This agrees with the findings of McCalla, Blackburn & Merrill (1984) who listed total vegetation cover and bare ground as two of the six variables which significantly influenced sediment production from grass-dominated communities in Texas.

Log$_e$ annual run-off showed the highest positive correlation with soil capping, and the highest negative correlation with litter cover. Correspondence analysis also indicated a positive correlation between soil capping and moderate to very high levels of run-off, and a positive correlation between very low run-off levels and litter cover. This finding corresponds with that of Packer (1963) who showed that run-off increased as vegetation and mulch (litter) cover decreased, or as bare soil increased.

Predictive models for annual soil loss and run-off, which were based on variables measured using the Walker (1976) method, were validated against a reserve data subset. The predicted values were correlated significantly with actual values, and the two models were used in a retrospective analysis of soil loss and run-off from the study area. Because there were large fluctuations in annual soil loss and annual run-off values, the differential in soil loss and run-off between the non-cull and cull block was used instead of annual values. The retrospective analysis showed a decrease in the soil loss differential between the two blocks as time progressed, but the run-off differential remained relatively constant and, except for two rainfall years, there was no significant difference in the differential between the two blocks. When the predicted soil loss and run-off differentials were compared with the actual stocking rate differential, no consistent relationship was detected. Previous studies, as reviewed by Blackburn *et al.* (1982), showed that soil loss and run-off differentials increased as the stocking rate differential increased, and it was concluded that flaws in the sampling design, and particularly in the experimental design (in that a stocking rate differential between blocks could not be maintained during the entire study period), prevented the determination of relationships between stocking rate, soil loss and run-off.

The major drawback with natural run-off plots is that they depend on natural rainfall which is always erratic and usually deviates from the normal. An important advantage of the rainfall simulator is that research is greatly accelerated because the results are no longer dependent on natural rainfall (Hudson, 1981). A potential disadvantage is that results are based on simulated instead of natural rainfall and, if these results are to be extrapolated to natural systems, the conclusions drawn may be invalid. In the next chapter the two techniques will be compared in an attempt to resolve the above dilemma.

4.8 CHAPTER SUMMARY

a) Rainfall intensity data collected over three rainfall years showed that a rainfall kinetic energy exceeding 10 000 J/m^2 was measured on only one occasion, but on 11 other occasions it exceeded 1 000 J/m^2. These kinetic energy values were always measured between October and March, with peak values measured in January and February.

b) Of those natural run-off sites which were monitored over three years, the highest mean annual soil loss was measured at site 402 located in *Acacia tortilis* woodland in the non-cull block, and the lowest mean annual soil loss was measured at site 411 situated in the cull block in *Acacia nilotica/Acacia gerrardii* woodland.

c) Correspondence analysis showed that high levels of soil capping and susceptibility to erosion were correlated positively with moderate to very high levels of soil loss and run-off. In contrast, high levels of litter cover were correlated positively with very low levels of soil loss and run-off.

d) The relationship between annual soil loss and both the soil surface and vegetation variables was curvilinear, and when annual soil loss was transformed logarithmically, susceptibility to erosion showed the highest positive correlation, and surface cover the highest negative correlation, with soil loss. Except for woody canopy height, all the variables which were significantly correlated showed a higher correlation with $\log_e$ annual soil loss than $\log_e$ annual run-off.

e) Two predictive models were derived for soil loss; one based on the Walker (1976) method only, and the other on the Walker and the USLE (Wischmeier, 1975) methods. The former model explained 51,2% of the variability in the data set, and the latter 51,7%.

f) The predicted values for $\log_e$ annual soil loss from the above two equations were correlated significantly with actual soil loss values when compared with a reserve data subset. Equation 4.2, which was derived from both methods, had the highest D and lowest F_1 values and was thus considered to give the most accurate predictions of soil loss.

g) The general trend in predicted annual mean soil loss values was towards a decrease in the soil loss differential between the two experimental blocks as time progressed, but when the soil loss and stocking rate differentials were compared no consistent relationship was evident.

h) The highest mean annual run-off, monitored over three years, was measured from site 284 located in *Acacia tortilis* woodland in the cull block. The lowest mean annual run-off, monitored over the same period, was recorded in *Acacia nigrescens* woodland in the cull block on site 403.

i) The relationship between annual run-off and both the soil surface and vegetation variables was also curvilinear, with $\log_e$ annual run-off showing the highest positive correlation with soil capping, and the highest negative correlation with litter cover.

j) The predictive model based on the Walker (1976) method explained only 61,4% of the variability in the data set. When variables measured by the USLE method (Wischmeier, 1975) were added, none of these variables were included in the predictive equation, and only one predictive model was derived.

k) When the above model was tested against the reserve data subset, the predicted values for $\log_e$ annual run-off were correlated significantly with the actual values.

l) Unlike the trend in annual mean soil loss, the predicted run-off trend did not show a decrease in the run-off differential as time progressed, and except for two rainfall years there was no significant run-off differential between the two blocks.

m) When the predicted run-off differentials and the actual stocking rate differentials were compared, no consistent relationship was found between these two variables.

n) The conclusion is that no cause and effect relationship could be demonstrated between stocking rate and run-off, or between stocking rate and soil loss, possibly because of defects in the experimental and sampling design.

CHAPTER 5 : COMPARISON OF RAINFALL SIMULATOR TRIALS AND NATURAL RUN-OFF PLOTS

5.1 INTRODUCTION

In this chapter, the results obtained from the rainfall simulator trials (Chapter 3) are compared with those from the natural run-off plots (Chapter 4) to assess whether simulated erosion events provide a reasonable representation of naturally occurring erosion events. This part of the study had the following three aims:

a) to compare the soil loss and run-off potentials of rainfall simulator and natural run-off plots at paired sites;

b) to compare the interrelationships between vegetation, soil surface variables, soil loss and run-off as determined by rainfall simulator trials and natural run-off plots; and

c) to compare predicted soil loss and run-off trends from the non-cull and cull blocks as determined by rainfall simulator trials and natural run-off plots.

Natural run-off plots, in contrast to rainfall simulator trials, measure soil loss and run-off caused by natural, instead of simulated, rainfall, and because of this their results can be used with greater confidence to make predictions about soil loss and run-off in natural systems. Natural run-off plots do, however, have two major disadvantages. Firstly, data can be collected only after a naturally occurring erosion event, and it may take a considerable amount of time to collect enough data for meaningful interpretation (Hudson, 1981). For example, over a three-year period in the current study a total of 23 erosion events was monitored from the natural run-off plots while, during one month, 30 erosion events were simulated on the rainfall simulator trial plots. Secondly, it is not possible to control the intensity of natural rainfall, thus making between-site comparisons within a particular rainfall year risky because different rainfall intensities and amounts may have fallen on various sites during any particular storm. It was also shown in Section 4.2.1 that differing amounts and intensities of natural rainfall from year to year made between-year comparisons invalid. Comparisons can be made only if the data are corrected for differences in intensity and amount, and not only is the instrumentation to measure rainfall intensity prohibitively expensive but these autographic raingauges are also prone to mechanical failure. A solution to these disadvantages would be to use a rainfall simulator, if it could be shown that this experimental technique provided results which adequately represent natural erosion events.

5.2 SOIL LOSS AND RUN-OFF MEASUREMENTS FROM RAINFALL SIMULATOR AND ADJACENT NATURAL RUN-OFF SITES

5.2.1 Materials and methods

Eight paired sites were located in the study area, five in the non-cull block and three in the cull block, where measurements were undertaken from rainfall simulator and from natural run-off plots (Figure 5.1). Details of the physical characteristics of these sites are given in Table 5.1. The layout of a typical site is shown in Figure 5.2 and detailed descriptions of rainfall simulator trials and natural run-off plots were given in Sections 3.2.1 and 4.3.1 respectively. These sites were installed to compare soil loss and run-off values calculated from the rainfall simulator trials with annual soil loss and run-off values obtained from the adjacent natural run-off plots. The purpose of these comparisons was to determine whether rainfall simulator trials could be used to predict which sites would have the highest and which sites would have the lowest annual soil loss and run-off values. In order to standardise the data between the rainfall simulator trials and natural run-off plots as closely as possible, the mean soil loss (g) and mean total run-off (l) were calculated from rainfall simulator storms 1, 2 and 3 at each site for every year that rainfall simulator trials were done on that site. This meant that results from the first rainfall simulator storm on dry soil were included, as this would be more representative of natural erosion events, where rain can fall on either dry or wet soil, depending on antecedent soil moisture conditions in the catchment. The methods and formulae used are detailed in Section 3.2.1. The mean values and ranges were calculated annually for each site and the results were presented in two forms: firstly as empirical data, and secondly as rankings from highest to lowest soil loss and run-off. With the natural run-off plot data, the total annual run-off (l) and total annual soil loss (g) were calculated per plot using the method outlined in Section 4.3.1. The mean values and ranges were calculated as well. These results were also presented empirically, and ranked from highest to lowest. Bivariate scattergrams were drawn, plotting mean annual soil loss (g) against mean soil loss (g), and mean annual run-off (l) against mean run-off (l). Because there was a small degree of curvilinearity, the data were also transformed logarithmically to the base e, and a product-moment correlation matrix (Sokal & Rohlf, 1981) was drawn up using a statistical program running on an Apple IIe microcomputer.

5.2.2 Results and discussion

5.2.2.1 Soil loss measurements

Based on the empirical data presented in Table 5.2, there was a highly significant correlation between mean annual soil loss, as determined by the natural run-off plots, and mean soil loss determined by the rainfall simulator trials (Figure 5.3). There was no improvement in the correlation coefficient when either of the variables was transformed logarithmically (Table 5.3). The soil loss values presented in Table 5.2 were also ranked from highest to lowest. Of the rankings obtained using the rainfall simulator several are identical, or similar, to those obtained from the natural run-off plots (Table 5.4). A total of 36,4% of the rainfall simulator rankings was within one rank of the rankings obtained from the natural run-off plots (Table 5.5).

From the data presented, it appears that rainfall simulator trials can be used to indicate which sites will yield higher and which sites will yield lower annual soil losses. Bearing in mind that the rainfall simulator rankings were based on two simulated storms lasting 36 minutes each, while the natural run-off plot rankings were based on 8-10 storms monitored during the entire rainfall year, the advantage of using a rainfall simulator for the rapid assessment of sites for erosion potential is obvious.

5.2.2.2 Run-off measurements

The actual run-off measurements are presented in Table 5.6 and there was a highly significant correlation between mean annual run-off, which was determined by the natural run-off plots, and mean run-off as determined by the rainfall simulator trials (Figure 5.4). When mean annual run-off was transformed logarithmically the correlation coefficient increased, but when mean run-off was transformed logarithmically the correlation coefficient decreased (Table 5.3). The highest correlation coefficient was obtained when both the variables were transformed logarithmically to the base e (Table 5.3). When the run-off values were ranked, there was a closer agreement between these rankings (Table 5.7) than between the soil loss rankings (Table 5.4). A total of 54,6% of the rainfall simulator rankings was within one rank of the natural run-off plot rankings (Table 5.5).

The data above show that the rainfall simulator can also be used to give an indication of which sites will yield higher and which sites will yield lower annual run-off values. When the rainfall simulator rankings of soil loss and run-off were compared with the natural run-off plot rankings, the rainfall simulator was marginally better at ranking run-off values than soil loss values.

5.3 COMPARISON OF THE INTERRELATIONSHIPS BETWEEN VEGETATION AND SOIL SURFACE VARIABLES AND BOTH RUN-OFF AND SOIL LOSS

5.3.1 Vegetation and soil surface variables

5.3.1.1 Materials and methods

Two semi-quantitative methods, described by Walker (1976) and Wischmeier (1975), were used to assess vegetation and soil surface variables on all rainfall simulator and natural run-off plots. Details of these methods are described in Section 3.3.1.1 for the rainfall simulator sites and in Section 4.4.1.1 for the natural run-off plots. Mean values for the above variables, measured at each rainfall simulator site, were individually calculated for every year that rainfall simulator trials were done at that site. The natural run-off plots were assessed annually in the early, mid and late growing season, and mean values for vegetation and soil surface variables measured at each natural run-off plot were calculated annually from the three growing season measurements.

5.3.2 Interrelationships between vegetation and soil surface variables and both run-off and soil loss

5.3.2.1 Materials and methods

A product-moment correlation matrix (Sokal & Rohlf, 1981) which included vegetation and soil surface variables as the independent variables and, in the case of rainfall simulator trials, mean soil loss and mean run-off as the dependent variables, was drawn up using a statistical program on an Apple IIe microcomputer. The mean soil loss and run-off data from the rainfall simulator trials were calculated from storms 2 and 3 only. A similar correlation matrix, including the above independent variables and annual soil loss and annual run-off for the dependent variables, was also drawn up from the natural run-off plot data. These two matrices were based on data from all the natural run-off plots and the rainfall simulator trials over the entire data collection period. In all cases, the soil loss and run-off values were logarithmically transformed to the base e. The independent variables which were significantly correlated with either $\log_e$ run-off or $\log_e$ soil loss were tabulated and then ranked from highest to lowest value for the correlation coefficient. The data were then tested, using the Wilcoxon matched-pairs signed-ranks test (Siegel, 1956), to determine whether the ranks of the natural run-off plot data differed significantly from the rainfall simulator trial ranks.

5.3.2.2 Comparison of correlation matrix results for soil loss

$\log_e$ mean soil loss, as determined by the rainfall simulator trials (RST), and $\log_e$ annual soil loss which was determined by the natural run-off plots (NRP) showed the highest positive correlation with susceptibility to erosion and the highest negative correlation with surface cover (Table 5.8). The significance of these results is that, in spite of two different experimental approaches being used, soil loss showed the highest positive and negative correlation with the same independent variables. When the correlation coefficients were ranked from highest to lowest there was no significant difference between the rankings obtained from the rainfall simulator trials and those obtained from the natural run-off plots with regard to soil loss (Table 5.9).

5.3.2.3 Comparison of correlation matrix results for run-off

The rainfall simulator trial data showed that the independent variables were correlated more highly with mean run-off than with $\log_e$ mean run-off (Table 5.8). This implies that there is a stronger linear than curvilinear relationship between run-off and the independent variables. In contrast, the relationship between annual run-off (NRP) and the independent variables is clearly curvilinear (Section 4.4.2.3). This basic difference is probably because the mean run-off values were calculated from the second and third rainfall simulator storms, and thus the moisture content of the soil was always at or near field capacity. In the natural run-off plots the soil moisture content could vary from dry to field capacity, depending on how recently rain had fallen in the catchment.

$\log_e$ annual run-off (NRP), $\log_e$ mean run-off and mean run-off (RST) all showed the highest negative correlation with litter cover and the highest positive correlation with soil capping (Table 5.8). This emphasises the high correlation between run-off and both litter cover and soil capping, despite the two different techniques used to measure run-off.

When the correlation coefficients for the independent variables were ranked from highest to lowest, there was no significant difference between the rankings for $\log_e$ mean run-off (RST) and $\log_e$ annual run-off (NRP) (Table 5.10). Similarly, there was no significant difference between the rankings for mean run-off (RST) and $\log_e$ annual run-off (NRP)(Table 5.10).

In summary, the rainfall simulator trials identified those independent variables which showed the highest positive and negative correlation with run-off on natural run-off plots. The same was true for soil loss. There was also no significant difference between the rainfall simulator trial rankings and the natural run-off plot rankings when the independent variables, which were correlated significantly with either run-off or soil loss, were tabulated and then ranked from highest to lowest value according to their correlation coefficient. The conclusion from this section is that the rainfall simulator trials provided an effective representation of the interrelationships between vegetation and soil surface variables and natural erosion events.

5.4 COMPARISON OF PREDICTED TRENDS FOR SOIL LOSS AT WALKER VEGETATION MONITORING SITES IN THE STUDY AREA

5.4.1 Materials and methods

Because none of the predictive models for run-off, which were derived from the rainfall simulator trial data, gave predicted values that correlated significantly with measured run-off values, no retrospective analysis of run-off in the study area was undertaken (see Section 3.4.2.4). However a retrospective analysis of soil loss was done, using data from 27 of the 36 Walker vegetation monitoring sites located in the study area (see Figure 3.35). Based on the vegetation monitoring data, mean soil loss values were calculated separately for the non-cull and cull blocks using a predictive equation derived from the rainfall simulator trial data (Section 3.5). These soil loss values were calculated annually for the period 1979 to 1986. Based on the same vegetation monitoring data set, and on additional rainfall data, mean annual soil loss values were calculated using a predictive equation derived from the natural run-off plot data (Section 4.6). These values were also calculated separately for the non-cull and cull blocks over the period 1979 to 1986. The mean values for the cull and non-cull blocks were plotted separately for each year and the trends in soil loss, as determined by the rainfall simulator trials and the natural run-off plots, were compared.

5.4.2 Results and discussion

When the trend in predicted mean soil loss for both experimental blocks (Figure 5.5) is compared with the trend in predicted mean annual soil loss (Figure 5.6), these two trends are very similar. They both clearly show an initial large soil loss differential between the non-cull and cull blocks, which decreases with time. This agreement between the two trends is noteworthy, bearing in mind that Figure 5.5 shows what the predicted mean soil loss would have been at the various vegetation monitoring sites if two simulated storms of equal intensity had been applied per site, while Figure 5.6 shows the predicted mean annual soil loss at particular sites resulting from storms of differing rainfall amounts and intensities. The conclusion drawn from either the predicted values derived from the rainfall simulator trial data (Figure 5.5) or the predicted values derived from the natural run-off plot data (Figure 5.6) is that the soil loss differential decreases with time. The rainfall simulator trial data could therefore be used to predict adequately what the long-term trend in soil loss would be in a natural system.

5.5 GENERAL DISCUSSION

In 1980 the South African rainfall simulator programme for assessing soil loss and run-off was initiated by the Division of Agricultural Engineering, Department of Agriculture (McPhee *et al.*, 1983). One of the objectives of this national programme was to identify areas of high priority for increased effort in combatting soil erosion and, to achieve this, use was made of Swanson rotating-boom rainfall simulators. Simulated rainfall from these machines gave an energy value approximately 75% that of natural rainfall and the results obtained were considered valid for natural rain storms in the summer rainfall region of South Africa (McPhee *et al.*, 1983). The authors did not critically test this assumption, although the identification of areas of high erosion risk assumes that the soil loss caused by simulated rainfall can be equated with annual soil loss caused by natural rainfall. To the best of this author's knowledge, no work has been done in South Africa using rainfall simulators to confirm natural run-off plot results.

Results from the current study showed there was a highly significant correlation between mean annual soil loss (NRP) and mean soil loss (RST), and between mean annual run-off (NRP) and mean run-off (RST). The results also showed that the Swanson rotating-boom rainfall simulator could be used to rank sites from highest to lowest soil loss and run-off, and that these rankings were similar to those from adjacent natural run-off plots. The rankings for run-off did however show a greater similarity than those for soil loss.

The rainfall simulator was also used to identify which soil surface and vegetation variables were most closely correlated with soil loss and run-off, and it correctly identified those variables which showed the highest positive and the highest negative correlation with both annual soil loss and annual run-off. When the vegetation and soil surface variables, which were correlated with soil loss and run-off, were ranked from highest to lowest there were no significant differences between the rankings for the rainfall simulator trials and the natural run-off plots.

The predicted trend in soil loss from both experimental blocks, as determined from rainfall simulator data, was very similar to that determined from natural run-off plot and rainfall data. Both trends clearly showed a decreasing soil loss differential with time. Rainfall simulator trial data could thus be used in the prediction of long-term trends in soil loss, as these predicted trends did not differ materially from those based on natural erosion events. The assumption made by McPhee *et al.* (1983) that results using simulated rainfall were valid when compared to natural erosion events appears to be sound.

Dunne, Dietrich & Brunengo (1980) stressed that land management decisions in remote grazing lands of Africa would have to be based on data from controlled experiments under artificial rainstorms, rather than on long-term plot studies of run-off and erosion under natural rainstorms. Several authors (Costin & Gilmour, 1970; Hudson, 1981; Platford, 1982) have also stressed the advantages of rainfall simulators, viz. portability, controlled rainfall intensity and a short data collection period.

These advantages, together with the fact that in this study rainfall simulator results were comparable to natural erosion event results, lead to the conclusion that the rainfall simulator should be used in preference to natural run-off plots as an experimental tool in developing countries and/or on natural grazing lands where funds are limited and the data base is small or nonexistent.

5.6 CHAPTER SUMMARY

a) There was a highly significant correlation between mean annual soil loss, determined by natural run-off plots, and annual soil loss which was determined by rainfall simulator trials.

b) A comparison between the correlation coefficients of independent vegetation and soil surface variables and soil loss, derived from the rainfall simulator and natural run-off plots, showed that the same independent variables had the highest positive and negative correlation with soil loss.

c) When the correlation coefficients were ranked from highest to lowest, there was no significant difference between the rankings obtained from the rainfall simulator trials and those of the natural run-off plots with regard to soil loss.

d) The trend in predicted mean soil loss from Walker transect vegetation monitoring sites, which was derived from the rainfall simulator data, was very similar to the trend in predicted annual mean soil loss which was derived from natural run-off plot and annual rainfall data.

e) There was a highly significant correlation between mean annual run-off, which was determined by the natural run-off plots, and mean run-off, as determined by the rainfall simulator trials.

f) When the rainfall simulator rankings of soil loss and run-off were compared with the natural run-off plot rankings, it was evident that the rainfall simulator was marginally better at ranking run-off than soil loss.

g) The rainfall simulator trials identified the same independent vegetation and soil surface variables that showed the highest positive and negative correlation with run-off as the natural run-off plots.

h) When these independent variables, which were correlated with run-off, were ranked from highest to lowest value for the correlation coefficient, there was no significant difference between the rainfall simulator trial rankings and the natural run-off plot rankings.

CHAPTER 6 : FENCE-LINE CONTRAST STUDY

6.1 INTRODUCTION

In Sections 3.5.2 and 4.6.2.3 reasons were given why the experimental design in the cull and non-cull blocks was suboptimal. Briefly, because the boundary between these two blocks was unfenced, uncontrolled movement of game was possible, and this made it impracticable to maintain a long-term stocking rate differential. Since there was no clear fence-line contrast between these two blocks, it was difficult to identify paired sites which were located on comparatively similar slopes, in similar woody vegetation communities and on the same soil form and series (see Section 4.3.1). Finally, the sampling sites for both the rainfall simulator trials and the natural run-off plots were not randomly placed and because of constraints on time, funds, manpower and equipment, sampling was at a low intensity (Sections 3.2.1 & 4.3.1).

For the above reasons, another study area on the western boundary fence of Umfolozi Game Reserve (UGR) at Cengeni gate was chosen in July 1984 (Figure 6.1). The game-proof boundary fence bisected the study area and no movement of game between UGR and the adjacent KwaZulu (KWZ) area was possible. Because there was a distinct fence-line contrast, sampling sites could be located 25m or closer to the fence on either side of it, which facilitated a comparative study. Lastly, the sampling sites were randomly placed and sampling was at a relatively high intensity, which greatly facilitated the statistical analysis of the data.

The aims for this part of the study were:

a) to determine the short-term and long-term levels of soil erosion adjacent to the boundary fence in both KwaZulu and Umfolozi Game Reserve;

b) to confirm the relationship between vegetation and soil surface variables and soil loss which had already been established in the rainfall simulator trials and natural run-off plots;

c) to determine the relationship between different levels of soil erosion and the production potential of the remaining soil;

d) to determine the relationship between different levels of soil erosion and grass species richness and diversity; and

e) to compare two vegetation monitoring methods used to determine herbaceous species composition by bulk contribution to herbaceous biomass.

6.2 DETERMINATION OF SOIL EROSION LEVELS

6.2.1 Short-term soil loss measurements

6.2.1.1 Materials and methods

Forty-eight Gerlach troughs (Morgan, 1979) were located at 24 different sites in the study area. Twelve of these sites were located in KWZ and 12 in UGR (Figure 6.2). The study area extended 25m on either side of the boundary fence and stretched from the top of Sabokwe hill to a drainage line approximately 2km downhill. It can be seen from the contour lines in Figure 6.2 that this fence ran along a ridge crest which divided the study area into two catchments.

At each site a pair of Gerlach troughs was installed along the contour line (Figure 6.3). The run-off during a rainstorm was collected in the collection trough and channeled via a hosepipe to a sealed plastic bucket (Figure 6.4). This bucket had a capacity of 25 litres and if the run-off event was less than 25 litres then the entire soil and run-off sample was collected in the bucket. One gram of powdered alum was added to the collected run-off to speed up sedimentation, and the excess run-off was siphoned off. The remaining soil and some run-off were collected in 1-litre plastic containers with sealed lids and transported to HGR where each sediment sample was oven-dried to a constant mass and weighed. When run-off from an erosion event exceeded 25 litres, the water flowed from the outlet of the bucket through a filter element where the fine soil particles were filtered out, and then to waste (Figure 6.4).

The Gerlach troughs were monitored in the middle and at the end of each month over two rainfall years, and the soil loss (grams per trough) was recorded if an erosion event or events had taken place during the previous two weeks. After each rainfall year the data were summed for individual plots and then expressed as:

annual soil loss (grams per trough).

The mean annual soil loss (g) for the 1984/5 and 1985/6 rainfall years was calculated separately for each site from data collected from the two adjacent Gerlach troughs. In addition, data from sites in either KWZ or UGR, which were located in the upper, mid or lower slope positions on the catena, were pooled and expressed as the mean, standard error and range. Annual soil loss data from the 1984/5 and 1985/6 rainfall years were combined and tested for normality using the Q-Q correlation coefficient (Johnson & Wichern, 1982). These data were then tested using single classification analysis of variance (ANOVA)(Sokal & Rohlf, 1981) to establish whether there was any significant difference between the population means from KWZ and UGR, and to test for differences between the upper, mid and lower slopes. Data were also tested using two-level nested ANOVA (Sokal & Rohlf, 1981) to establish whether there were significant differences between upper, mid and lower slopes after the average UGR versus KWZ difference had been removed.

6.2.1.2 Results

The physical characteristics of the Gerlach trough sites are given in Table 6.1 and it is evident that the sites in the upper slope position on the catena were located on the steepest slopes. In all cases, the annual mean soil loss was higher from the KWZ sites than the UGR sites, irrespective of their position on the slope (Tables 6.2 & 6.3). When the annual soil loss data from both years were combined, there was no significant difference between the population means of the UGR and KWZ sites (Table 6.4). There was however a significant difference when the soil losses from the upper, mid and lower slopes were compared, but when the mean UGR versus KWZ difference was accounted for there was no significant difference in soil losses measured from the upper, mid and lower slopes (Table 6.4). Gerlach troughs are used as a plotless sampling method for point estimates of soil loss, and because the size of the microcatchments above each Gerlach trough may vary, there is the potential for large between-trough variability in soil loss measurements. This is evident from the analysis of variance results in Table 6.4 and from the large ranges and standard errors, particularly in the KWZ sites, in the 1985/6 data (Table 6.3).

To summarise, there is no significant difference in soil loss from UGR plots when compared with KWZ plots, or when UGR plots in the upper, mid and lower slopes are compared with KWZ plots on the same slopes. There is a significant difference in soil loss from plots on the upper, mid and lower slopes when data from the UGR and KWZ plots are combined. This indicates that it is the position on the catena, rather than the positioning of the plots in either UGR or KWZ, that accounts for most of the variability in soil loss.

6.2.2 Long-term erosion levels

6.2.2.1 Materials and methods

The reason for collecting A-horizon depth data was to use these data to give an indication of long-term erosion levels, with shallower A-horizon depths suggesting higher long-term erosion levels. The depth of the A-horizon was measured, to the nearest centimetre, at 48 randomly selected sites, 24 of which were in KWZ and 24 in UGR. A hole approximately 50 cm in diameter and 75cm deep was dug at each site and four measurements of the

A-horizon were taken at each hole to determine the mean A-horizon depth at each site.

This mean depth was calculated separately for all of the sites located in KWZ and in UGR. Also, A-horizon depth data from sites in either KWZ or UGR which were located in the upper, mid or lower slope positions were pooled and expressed as the mean, standard error and range. Mean A-horizon depth data were tested using single classification ANOVA to establish whether the population means from KWZ and UGR differed significantly, and whether population means from the upper, mid and lower slopes differed significantly. The same data set was also tested using two-level nested ANOVA (Sokal & Rohlf, 1981) for significant differences between the upper, mid and lower slopes in KWZ when compared with the same slopes in UGR.

6.2.2.2 Results

The mean A-horizon depth for sites in KWZ was always less than that for UGR sites (Table 6.5). There is also a clear trend in both KWZ and UGR showing the shallowest A-horizons on the upper slope, and the deepest A-horizons on the lower slope (Table 6.5). There was no significant difference between the population means for the A-horizon depths measured in KWZ when compared with UGR, but there was a highly significant difference ($p<0.01$) between the depths measured on the upper, mid and lower portions of the catena (Table 6.6). When the mean UGR versus KWZ difference was accounted for, there was no significant difference in A-horizon depths measured on the upper, mid and lower slopes (Table 6.6).

The assumption in collecting these data was that if there were no long-term differences in soil erosion levels between KWZ and UGR, then there would be no statistically significant difference in the depth of the A-horizon. Given the result in Table 6.6, this assumption is true, and the real differences in A-horizon depth are dependent on position on the catena rather than on differences in erosion levels between KWZ and UGR.

To summarise this section, there are no statistically valid reasons for concluding that either the short-term soil loss rate (measured over two rainfall years) or the long-term erosion level differs significantly between KwaZulu and Umfolozi Game Reserve. Rather, the differences in both short-term and long-term erosion levels appear to be determined by catenal position. A cautionary note is however needed here. Although not statistically significant, there was a consistent trend showing that both short- and long-term erosion rates were higher in KWZ than in UGR (Tables 6.2, 6.3 & 6.5), and if these rates continue then in the longer term there may be a significant difference in erosion levels.

6.3 RELATIONSHIP BETWEEN VEGETATION AND SOIL SURFACE VARIABLES AND SOIL LOSS

In Section 5.3.2.3 it was shown that the rainfall simulator trials identified the same independent variables which exhibited the highest positive and negative correlation with $\log_e$ soil loss as did the natural run-off plots. The significance of these results was that, in spite of two very different experimental approaches being used, the same independent variables, which were highly correlated with soil loss, were identified. The purpose of this current section was to use the Gerlach trough results as another independent confirmation of those independent variables which showed the highest correlation with soil loss, and to confirm that this relationship was curvilinear rather than linear.

6.3.1 Physical vegetation measurements

6.3.1.1 Materials and methods

A $1m^2$ quadrat was laid down systematically at ten positions above each Gerlach trough (Figure 6.5), and the Walker (1976) and USLE (Wischmeier, 1975) methods used to assess vegetation and soil surface variables. Each site was assessed annually in January, and the variables assessed and their definitions are given in Section 3.3.1.1. The data for each site were pooled and, based on a sample size of 20 quadrats per site, mean values for all the variables were calculated using the formulae outlined in Section 3.3.1.1.

6.3.2 Relationship between vegetation and soil surface variables and soil loss

6.3.2.1 Materials and methods

Annual mean soil losses were calculated for each site by taking the mean of the annual soil loss (g) measured at both adjacent troughs (see Figure 6.3). Data from both rainfall years were pooled and bivariate scattergrams, plotting mean annual soil loss against a particular independent vegetation or soil surface variable, were drawn and examined visually. From this examination it could be seen that the relationship between the dependent and independent variables was curvilinear, and so mean annual soil loss was transformed logarithmically to the base e to normalise the data relationship. The normality of the data were tested using the Q-Q correlation coefficient, and a product-moment correlation matrix (Sokal & Rohlf, 1981) was constructed using a statistical program running on an Apple IIe microcomputer.

6.3.2.2 Results

Except for rock, all the independent variables showed a higher correlation coefficient with $\log_e$ mean annual soil loss than with mean annual soil loss (Table 6.7). Log_e mean annual soil loss showed the highest positive correlation with susceptibility to erosion and the highest negative correlation with surface cover (Table 6.7). These are the same two variables which showed the highest correlation with mean annual soil loss, as determined by the natural run-off plots (Section 4.4.2.3), and with mean soil loss, as determined by the rainfall simulator trials (Section 3.3.2.3).

In Section 6.2.1.2 it was shown that the soil loss measurements from the upper, mid and lower slopes differed significantly, and from this it can be inferred that slope steepness may have affected soil loss. There was, however, no significant correlation between percentage slope and $\log_e$ mean annual soil loss (Table 6.7).

The significance of these results is that again they illustrate a curvilinear relationship between soil loss and both vegetation and soil surface variables, which is best described by an exponential curve. They also illustrate that $\log_e$ soil loss shows the highest positive correlation with percentage susceptibility to erosion and the highest negative correlation with percentage surface cover.

6.4 RELATIONSHIP BETWEEN DIFFERENT LEVELS OF SOIL EROSION AND THE PRODUCTION POTENTIAL OF SOILS

The perceived management problem is that if the long-term soil loss rates exceed those of soil genesis, then the productivity of the system will ultimately decline and this will lead to a decrease in the carrying capacity of the system. Although it was not possible to demonstrate a significant short-term difference in soil loss between the KWZ and UGR plots, there was a consistent trend indicating that erosion rates may be higher in KWZ than in UGR. Assuming there is a higher long-term soil loss rate in KwaZulu areas adjacent to UGR than in UGR itself, then a fence-line contrast study should show whether there is any difference in the production potential of the remaining soils which have been subjected to different levels of soil erosion.

It was felt that the amount of herbage accumulated over a growing season would be a good indication of the production potential of the soil, assuming the soil type, aspect, vegetation type and rainfall on each side of the fence-line were identical.

6.4.1 Materials and methods

The study area was divided into an upper, mid and lower slope position and one paired sampling site was randomly located within each of these positions along the catena (Figure 6.6). Each paired site consisted of two 10m x 10m exclosure plots, one located in KWZ and one in UGR, and two adjacent 10m x 10m control (grazed) plots (Figure 6.7). The exclosure plots were fenced with game-proof fencing which excluded herbivores of grey duiker *Sylvicapra grimmia* size and larger, while the adjacent control plots were unfenced, with only creosoted poles marking the plot borders (Figure 6.7).

The exclosure and control plots were constructed in November 1984 and the herbage accumulated on all the plots was clipped immediately after construction was completed. Initially the herbage was clipped to standardise conditions between plots and to provide baseline measurements of herbage accumulation. The plots were reclipped after each growing season in March 1985, March 1986 and March 1987. Since there was generally no growth in the dry winter season, seasonal production could be regarded also as annual production. In each plot, herbaceous plants were clipped to a height of 1cm and the clipped material collected and transported to HGR where it was oven dried to a constant mass and weighed. These clippings represented the standing crop of herbaceous material and included the above-ground biomass (attached live green material) and the above-ground necromass (attached dead material) (Grossman, 1982). Data were expressed as mass of herbage accumulated (kg dry matter) per plot.

All the plots and the adjacent veld were burnt annually, in late June or early July, as part of the firebreak burning programme in UGR. The advantage of this annual burning was that it made it possible to measure biomass and necromass increments more accurately than on unburnt areas, as residual biomass and necromass were largely removed by burning (Grossman, 1982).

Data from the exclosure plots were used to compare herbage accumulation between KWZ and UGR in the absence of grazing, and data from the control plots were used to compare herbage accumulation in the presence of grazing. The distinction was made between grazed and ungrazed plots because McNaughton (1979) found that in Serengeti National Park, Tanzania, the net annual above-ground primary production could differ considerably between grazed and ungrazed swards. Because the amount of rainfall received during any particular growing season influenced herbage production over that growing season, the data were standardised by calculating the percentage contribution that each exclosure plot made to the total annual dry-mass of accumulated herbage in the exclosure plots. The same calculation was performed for control plots. After the data had been tested for normality, single classification ANOVA (Sokal & Rohlf, 1981) was used to test whether the population means differed significantly between KWZ and UGR, both in the absence of grazing (exclosure plots) and in the presence of grazing (control plots). Single classification ANOVA was also used to establish whether there was any significant difference in herbage accumulation, in the absence and in the presence of grazing, between the upper, mid and lower slope positions. Herbage accumulation data from control and exclosure plots were tested separately using two-level nested ANOVA to examine possible differences between the upper, mid and lower slopes in KWZ when compared with the same slopes in UGR.

6.4.2 Results

The total yield was always higher in UGR exclosure plots than in exclosure plots in KWZ (Table 6.8). Another trend was that exclosure plots on the lower slope generally produced higher yields than plots on the mid or upper slope, irrespective of whether these plots were located in KWZ or in UGR (Table 6.8). A similar trend was also evident in the control plots located in UGR, but not in those located in KWZ (Table 6.9). Also, except for March 1985, the total yield in UGR control plots was higher than in KWZ control plots (Table 6.9).

When the initial data (November 1984) were compared, there was no significant difference between herbage accumulated in exclosure plots in KWZ and that accumulated in KWZ control plots (paired t-test = 0,31; d.f. = 2), nor between exclosure plots in UGR and UGR control plots (paired t-test = 0,35; d.f. = 2). This nonsignificant difference between the exclosure and control plots was to be expected because the exclosure plots had just been erected and insufficient time had passed to allow for differences between the grazed (control) and ungrazed (exclosure) plots to manifest themselves. also firebelts had removed veg?

Using single classification ANOVA on the pooled 1985, 1986 and 1987 data, there was no significant difference between herbage accumulated in KWZ when compared with UGR in the absence of grazing (exclosure plots), but there was a significant difference ($p<0.05$) in the presence of grazing (Table 6.10). This difference is presumably due to different amounts grazed. There was a highly significant difference between herbage accumulated in the absence of grazing, when the upper, mid and lower slope plots were compared (Table 6.10), and there was also a significant difference in the presence of grazing, i.e. in the control plots. A two-level nested ANOVA on the exclosure plot data showed that after the mean difference between the UGR and KWZ plots had been removed, there was a highly significant difference between herbage accumulation in the upper, mid and lower slope plots (Table 6.10). There was also a significant difference between herbage accumulation in the upper, mid and lower slope control plots after the mean difference between the UGR and KWZ plots had been removed (Table 6.10).

To summarise this section, in the absence of grazing there was no statistically significant difference in herbage accumulation between plots in KwaZulu and Umfolozi Game Reserve. There was however a significant difference in the presence of grazing but this may be merely a reflection of differing grazing intensities on each side of the fence. What does affect herbage accumulation significantly, both in the presence and absence of grazing, is the position of the plots on the catena. This is evident not only when the plots on each side of the fence are combined, but also when the KWZ plots are separated from the UGR plots, both in the absence and in the presence of grazing.

6.5 RELATIONSHIP BETWEEN DIFFERENT LEVELS OF SOIL EROSION AND GRASS SPECIES RICHNESS AND DIVERSITY

The perceived management problem in this section is that accelerated soil loss may lead to a long-term reduction in species diversity, which contradicts the primary nature conservation objective of the Natal Parks Board, viz. "to conserve the optimum number of appropriate indigenous species and their habitats" (Grobler, 1984). Assuming there is a higher long-term soil loss rate in KwaZulu areas adjacent to UGR than in UGR itself, and that higher soil losses lead to reduced species diversity, then a fence-line contrast study should indicate differences in species diversity. Ideally, the species diversity of all components of the biota should be measured but, given the small size of the study area and the rudimentary state of knowledge of South African invertebrate taxonomy, it was decided to measure the species richness and diversity of a segment of the biota, viz. grasses. The reasons for choosing grasses were that their taxonomy is well researched and they are relatively easy to identify, especially when flowering. Also, they form an important component of the diet of herbivorous ungulates, microherbivores and seed-eating birds. Because forbs were also present in the grass sward, they were included in the measurement of species richness and diversity. However, since forbs were more difficult to identify, especially when not flowering, they were lumped as one species on the assumption that they all had a similar ecological function. Another assumption made in this section was that an increase in herbaceous species diversity was indicative of an increase in the biotic diversity of the system.

6.5.1 Materials and methods

6.5.1.1 Herbaceous species frequency

Herbaceous species frequency data were collected in the exclosure and control plots in January 1985 using a nested quadrat technique. A total of 80 nested quadrats was randomly placed within each plot, and the presence of individual grass species within each of the nested quadrats was noted. Forbs were lumped as one species. The sizes of the nested quadrats were 20 x 20cm; 20 x 40cm and 40 x 40cm. Diversity, in the sense of richness of species, was measured by S, which is the number of species in a sample of a standard size and taxonomic inclusion from a given community (Whittaker, 1977). In the current study, S was expressed as the total number of herbaceous species occurring within the 80 quadrats for the three different nested quadrat sizes. Using single classification ANOVA, the total species richness (S) was compared between KWZ and UGR plots, and between the upper, mid and lower slopes. Two-level nested ANOVA was used to compare the KWZ upper, mid and lower plots with the same plots in UGR. Finally, three-level nested ANOVA was used to detect significant differences between grazed (control) and ungrazed (exclosure) plots on the upper, mid and lower slopes in KWZ and UGR.

The species frequency data were also expressed as percentage frequency of occurrence, which was obtained using the following formula:

$$\text{frequency of occurrence (\%) of species A} = \frac{a}{N} \times \frac{100}{1}$$

where a = number of quadrats in which species A occurs
N = total number of quadrats.

The percentage frequency of occurrence was calculated for each species for each quadrat size. Species which occurred in the 40 x 40cm nested quadrat in each plot were ranked from highest to lowest contribution to percentage frequency, and the data were presented separately for grazed (control plot) and ungrazed (exclosure plot) conditions. An index of similarity (S_i) (Odum, 1971) was used to compare the similarity in species composition between the exclosure and control plots on the various slope positions in UGR with adjacent plots in KWZ. The formula for this index is:

$$S_i = \frac{2C}{A + B}$$

where A = number of species in sample A

B = number of species in sample B

C = number of species common to both samples

The higher the value of S_i, the greater the similarity between the two samples.

6.5.1.2 Herbaceous species composition by bulk contribution to biomass

Whittaker (1965) considered the best single measure of the species' importance in the community to be its productivity, which could be expressed by either the dry mass of organic matter produced or the energy bound per unit area per time unit. The former expression of productivity was used in this study, and in November 1984 and January 1986 a 1m^2 quadrat was located randomly at twenty points within each exclosure and control plot, and the Walker (1976) and Barnes, Odendaal & Beukes (1982) methods were used to determine the grass species composition of the herbaceous layer. Both methods employed a dry-mass ranking technique to determine those herbaceous species contributing the most bulk to the above-ground living herbaceous biomass.

In each quadrat, those herbaceous species contributing the greatest, the second greatest and the third greatest amounts of herbage on a dry-mass basis were identified, and the ranks of 1, 2 and 3 were allocated respectively to these species. Again, forbs were lumped as one species. In the Walker (1976) method if there were only two species in the quadrat then the heavier species was ranked 1 and the lighter species 2; if there was only one species in the quadrat it was ranked 1. In the Barnes *et al.* (1982) method if there was only one species in the quadrat it was ranked 1, 2 and 3; if there were two species in the quadrat then the heavier species was ranked 1 and 2, and the lighter species 3. This method had two other conditions which applied to the ranking of species. Firstly, if there were four or more species in the quadrat with two being equally heavy, then these two species were equally ranked 1 (i.e. species A and species B); the next by weight (species C) was ranked 2; and the next by weight (species D) was ranked 3. Secondly, if species A comprised more than 85% of the dry-mass herbage yield, then it was ranked 1 and 2, and the next species by weight (species B) was ranked 3.

In the Walker (1976) method, the herbaceous species composition by bulk contribution to biomass was determined for each plot in which all quadrats had three or more species, using the procedure outlined below:

Firstly, for each species, the proportions of quadrats in which it was ranked first, second and third was determined. Secondly, the percentage contribution to biomass was determined, using the constants of Mannetje & Haydock (1963), by multiplying the proportion of quadrats where species A was ranked first by 70.2; multiplying the proportion of quadrats where species A was ranked second by 21.1; and multiplying the proportion of quadrats where species A was ranked third by 8.7. The percentage contribution to biomass of species A was the sum of these three amounts.

If some quadrats had less than three species, then calculations were carried out on the number, and not the proportion, of quadrats. The procedure was as follows:

Firstly, for each species, the number of quadrats in which it was ranked first, second and third was determined. Secondly, the sum of each species was calculated by multiplying the number of quadrats where species A was ranked first by 8.0; multiplying the number of quadrats where species A was ranked second by 2.4; and multiplying the number of quadrats where species A was ranked third by 1.0. The sum for species A ($\sum a$) was the sum of these three amounts. Thirdly, the sum of all species was added up to give a total ($\sum T$). Fourthly, the percentage contribution to biomass for species A was calculated using this formula:

$$\% \text{ contribution to biomass of species A} = \frac{\sum a}{\sum T} \times \frac{100}{1}$$

In the Barnes *et al.* (1982) method, the data were tabulated to give the proportion of quadrats in which each species was ranked first, second or third. Where a rank was allocated equally to two or more species in a quadrat, the quadrat value of unity for the rank concerned was divided by the number of species to which the same rank was allocated, before summation of numbers of quadrats by ranks and species. The proportionate values for ranks 1, 2 and 3 were respectively multiplied by constants derived by Mannetje & Haydock (1963). These constants were 70.2, 21.1 and 8.7 respectively. The products for each rank were summed to give the percentage contribution to biomass for each species (Barnes *et al.*, 1982).

The data were presented separately for November 1984 and January 1986 and for the Walker (1976) and the Barnes *et al.* (1982) methods. An index of similarity (S_I) (Odum, 1971) was used to compare the similarity in species composition between the upper, mid and lower slope plots in KWZ with that in adjacent plots in UGR.

6.5.1.3 Correspondence analysis

Correspondence analysis was carried out on the 1984 and 1986 data sets to determine the best objective fit to the association between herbaceous species composition on the one hand and the exclosure and control plots on the other. The 1984 and 1986 data sets derived from the Walker (1976) and Barnes *et al.* (1982) methods were analysed separately. Details of this multivariate analysis technique have been given in Sections 3.3.2.1 and 4.4.2.1, and the interpretation of the output tables has been described in Section 3.3.2.2. In each analysis, the data matrix was drawn up using the format outlined in Table 6.11. The codes used in the text, in Figures 6.8 - 6.11, in Table 6.11 and in Tables 6.21 - 6.24 are explained in Table 6.12.

6.5.2 Results

6.5.2.1 Herbaceous species frequency

The total herbaceous species richness was higher in the upper and mid-slope exclosure and control plots in UGR than in the corresponding plots in KWZ (Table 6.13). However, in the exclosure and control plots on the lower slope, the species richness was higher in the KWZ plots than in the UGR plots (Table 6.13). There was no significant difference between the species richness (S) measured on plots in KWZ and UGR, but there was a highly significant difference when the data from the KWZ and UGR plots were combined and species richness was compared between upper, mid and lower slopes (Table 6.14). There was still a significant difference in the species richness between upper, mid and lower slope plots, even when the average KWZ and UGR differences, as well as the average control and exclosure plot differences, had been removed (Table 6.14).

To summarise, it is evident from these results that there was not a significant difference in the alpha species diversity (i.e. the species diversity within individual plant communities) (Whittaker, 1965) between KWZ and UGR plots, but there was a highly significant difference when the upper, mid and lower slopes were compared.

When the paired plots were compared under grazed conditions, the index of similarity (S_I) increased from 0,60 in the upper slope plots to 0,92 in the lower slope plots (Table 6.15). This indicated that the grass swards in KWZ and UGR became more homogeneous with respect to species composition as plots further down the catena were sampled. The following grass species occurred in the KWZ plots only: *Diplachne eleusine*, *Panicum natalense*, *Sporobolus nitens* and *Sporobolus pectinatus* (Table 6.15). *Rhynchelytrum repens*, *Hyparrhenia filipendula*, *Heteropogon contortus*, *Sporobolus smutsii* and *Urochloa panicoides* were the only species found exclusively in the UGR plots. Thirteen herbaceous species occurred in plots in both KWZ and UGR (Table 6.15).

A comparison of the paired plots under ungrazed conditions also showed an increase in the index of similarity from the upper to the lower plots (Table 6.16), again emphasising the difference in the species composition of the grass sward in the upper KWZ plot and the adjacent UGR plot. The UGR and KWZ exclosure plots also shared 13 herbaceous species, with *Diplachne eleusine*, *Sporobolus nitens* and *Sporobolus pectinatus* occurring only in KWZ plots and *Panicum deustum* and *Heteropogon contortus* occurring in UGR plots only (Table 6.16).

To summarise, the grass swards in the upper slope plots in KWZ and UGR were less similar in species composition than the swards in the mid and lower slope positions. This relationship was evident, irrespective of whether the plots were grazed or not.

6.5.2.2 Herbaceous species composition by bulk contribution to biomass

In both KWZ and UGR, forb species, *Digitaria swazilandensis* and *Themeda triandra*, made the highest mean percentage contribution to herbaceous biomass on the upper, mid and lower slope plots respectively (Table 6.17). These measurements were made in November 1984 using the Walker (1976) method. When the Barnes *et al.* (1982) method was used, similar results were obtained (Table 6.18). The index of similarity (S_i) increased from 0.58 on the upper slope plots to 0.82 on the lower slope plots, irrespective of the method used to determine the species composition (Tables 6.17 & 6.18). This shows there is a larger difference in the grass species composition when the upper KWZ plots are compared with the adjacent UGR plots, than when the mid or lower slope plots are compared. When the same exclosure and control plots in KWZ were remonitored in January 1986, forb species, *Digitaria swazilandensis* and *Eustachys paspaloides*, contributed the most to the herbaceous biomass on the upper, mid and lower slopes respectively (Tables 6.19 & 6.20). In the UGR plots, the highest contributions were made by forb species, *Digitaria argyrograpta* and *Themeda triandra*. Similar results were obtained irrespective of the vegetation monitoring technique used. Again, the index of similarity increased from the upper slope to the lower slope plots (Tables 6.19 & 6.20).

In summary, there is a marked difference between the species composition of the upper slope plots in KWZ and UGR. This difference becomes progressively less pronounced in the mid and lower slope plots. The above relationship is evident irrespective of which of the two vegetation monitoring methods was used and regardless of the year in which the vegetation was monitored.

6.5.2.3 Correspondence analysis

In the November 1984 herbaceous species composition data, which was determined using the Walker (1976) method, the first principal axis (axis 1 in Figure 6.8) distinguished the upper and mid-slope exclosure plots in KWZ (plots 413.2 and 415.2 respectively) from the lower slope exclosure and control plots (plots 419.3 and 419.4 respectively) in UGR. The grass *Themeda triandra* was correlated positively with plots 419.3 and 419.4, and *Digitaria swazilandensis* was correlated positively with plots 413.2 and 415.2. This axis contrasts upper and mid-slope exclosure plots in KWZ with bottom slope exclosure and control plots in UGR based on the presence of *Digitaria swazilandensis* and *Themeda triandra*. Axis 2 contrasts the upper slope exclosure and control plots in UGR, and the adjacent exclosure plot in KWZ (plots 413.3, 413.4 and 413.2) with the mid-slope control plot in UGR (plot 415.4). This latter control plot is correlated positively with *Digitaria swazilandensis* and the upper slope plots are correlated positively with forb species and *Diplachne eleusine* (Figure 6.8 & Table 6.21). Axes 1 and 2 explain 60,0% of the variability in the data set.

When the Barnes *et al.* (1982) method was used to determine the herbaceous species composition in 1984, the first principal axis in Figure 6.9 contrasted the mid-slope control and exclosure plots in KWZ and the adjacent exclosure plot in UGR (plots 415.1, 415.2 and 415.3 respectively) with the bottom slope exclosure and control plots in UGR (plots 419.3 and 419.4). Again, *Themeda triandra* was correlated positively with the bottom slope plots in UGR and *Digitaria swazilandensis* with the mid-slope plots in KWZ and the adjacent exclosure plot in UGR (Figure 6.9 & Table 6.22). The second principal axis showed that forb species were correlated positively with plot 413.4 and to a lesser extent 413.3, and it also showed the positive correlation between *Diplachne eleusine* and plots 413.1 and 413.2 (Figure 6.9). A total of 61,4% of the variability in the data set was explained by these two axes.

In summary, the above two correspondence analyses primarily contrasted mid-slope plots in both UGR and KWZ with bottom slope plots in UGR, by the presence of *Digitaria swazilandensis* in the mid-slope plots and *Themeda triandra* in the bottom slope UGR plots.

In the original 1986 correspondence analysis, based on the Walker (1976) method, *Digitaria argyrograpta* played an overwhelming role in determining the second principal axis and it was treated as a supplementary point in the subsequent analysis so that the less obvious multidimensional patterns could be examined (Table 6.23). In axis 1 *Chloris gayana*, *Panicum maximum* and *Diplachne eleusine* were correlated positively with the upper slope KWZ plots (413.1 & 413.2) and *Themeda triandra* was correlated negatively with the same plots (Figure 6.10). Axis 2 distinguished the lower slope exclosure and control plots in KWZ (plots 419.1 and 419.2) from the upper slope exclosure and control plots in UGR (plots 413.3 and 413.4). These upper slope plots were correlated positively with *Themeda triandra* and the lower slope KWZ plots with *Digitaria argyrograpta* and *Eustachys paspaloides* (Figure 6.10). The first and second principal axes accounted for 56,0% of the total variability.

Using the Barnes *et al.* (1982) method (Figure 6.11), the first principal axis contrasted the mid-slope exclosure and control plots in UGR (plots 415.3 and 415.4) with the adjacent KWZ plots (plots 415.1 and 415.2) on the basis of *Digitaria swazilandensis* being correlated positively with the KWZ plots and *Digitaria argyrograpta* with the UGR plots. Axis 2 contrasted the mid-slope exclosure and control plots in KWZ and the adjacent control plot in UGR (plots 415.1, 415.2 and 415.4) with the upper slope exclosure and control plots (plots 413.1 and 413.2) in KWZ (Figure 6.11 & Table 6.24). These upper slope plots were correlated positively with forb species and *Chloris gayana*, while the mid-slope plots were correlated positively with *Digitaria swazilandensis* and *D. argyrograpta*. Axes 1 and 2 explained 54,9% of the variability in this data set.

In summary, correspondence analysis of the 1986 data set contrasted the different positions on the slopes instead of contrasting UGR and KWZ plots on the same slopes. It was only in the first principal axis of Figure 6.11 that mid-slope plots in KWZ were contrasted with mid-slope plots in UGR. Also, the correspondence analysis did not contrast grazed (control) and ungrazed (exclosure) plots on the same slope position. The results of the analyses above show that it is the position on the slope, and not the siting of the plots in either KWZ or UGR or whether they are grazed or ungrazed, that accounts for the highest degree of variability in the data.

6.6 COMPARISON OF TWO VEGETATION MONITORING METHODS USED TO DETERMINE HERBACEOUS SPECIES COMPOSITION

6.6.1 Materials and methods

The Walker (1976) and Barnes *et al.* (1982) methods were compared to establish whether there was any statistically significant difference in the percentage contribution of each species to herbaceous biomass. The two methods were outlined in Section 6.5.1.2, and the experimental design of the exclosure plots in Section 6.4.1. Since the data on species composition were collected at the ordinal scale, in that the most important species in each quadrat were ranked instead of measured, nonparametric statistics were used to analyse the data (Siegel, 1956). Because the data comprised two related samples, the Wilcoxon matched-pairs signed-ranks test (Siegel, 1956) was used to detect any significant difference. The null hypothesis (H_o) was that there was no significant difference between the two methods, and if the calculated T values were less than or equal to the critical T values, then H_o was rejected (Siegel, 1956).

6.6.2 Results

There was no significant difference between the Walker (1976) and Barnes *et al.* (1982) methods when the percentage contributions to herbaceous biomass of various herbaceous species were compared (Table 6.25). This was so, regardless of whether the herbaceous species composition was determined in UGR or KWZ, or in 1984 or 1986 (Table 6.25). If the relevant columns in Tables 6.17 and 6.18 are examined, it is apparent that there is very little difference in percentage contribution to biomass of plots in KWZ assessed using either the Walker (1976) method (Table 6.17) or the Barnes *et al.* (1982) method (Table 6.18). The same is evident for UGR plots and for the 1986 data set (Tables 6.19 & 6.20).

Since there is no significant difference between the two methods, it can be inferred that each is equally good at determining percentage contribution to biomass. The Barnes *et al.* (1982) method is more flexible in the ranking of herbaceous species (Section 6.5.1.2) because it makes allowance for situations where two species are equally heavy. Here, the two species are both ranked 1, while in the Walker (1976) method a decision has to be made to determine which species is the heavier as there is no provision for shared rankings. Also, one species sometimes contributes the bulk of the biomass in a quadrat and the remaining two contribute very little. Using the Walker (1976) method, these species are ranked 1,2 and 3, while in the Barnes *et al.* (1982) method if a species contributes more than 85% to the herbaceous biomass it is ranked 1 and 2. For the reasons given above, the author found it easier to use the Barnes *et al.* (1982) method in the field, and it is recommended that in future this method should be used in preference to the Walker (1976) method to monitor herbaceous species composition in UGR.

6.7 GENERAL DISCUSSION

Soil loss and soil depth data were collected in KWZ and UGR to determine whether there were any differences in short-term and long-term erosion levels arising from differences in land use. The results showed that the largest degree of variability in the data was explained by differences in position on the slope, rather than by differences in land use. Examination of the soil loss data did show however that short-term soil loss levels were consistently, but not significantly, higher in KWZ than in UGR. The same conclusion was reached regarding long-term soil erosion levels when the A-horizon depth data were examined.

Given this disparity in erosion rates, which in the longer term may differ significantly, the perceived management problem in following a noninterventionist policy (which is essentially the policy that has been followed in KWZ regarding large herbivore numbers) is the following. There is the danger that the soil loss rates will exceed those of soil genesis and this will lead to a decline in vegetation productivity and ultimately to a decrease in the carrying capacity of the game reserve. As early as 1948, Smith & Whitt noted that the ultimate objective of soil conservation was to maintain soil fertility, and hence crop production, indefinitely. This concept of the maintenance of soil fertility and crop production was formalised by Wischmeier & Smith (1978) in their definition of "soil loss tolerance", which denotes the maximum amount of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely. From their definitions it can be seen that soil loss tolerances were based on the long-term maximisation of crop production, and the application of the soil loss tolerance concept to rangelands is of questionable value (Wight & Lovely, 1982). This is also true in a game reserve situation where emphasis is placed on the maintenance of species diversity and ecological processes (Grobler, 1984) rather than on the maximisation of long-term productivity. The herbage accumulation results in this study show that in the absence of grazing there is no statistically significant difference in herbage accumulation in KWZ and UGR. What did affect herbage accumulation significantly, both in the presence and absence of grazing, was the position of the plots on the slope. The perceived management problem in following a noninterventionist policy appears to be unfounded; both conceptually, when considered in the light of Natal Parks Board objectives for game reserves under their control, and practically, when the herbage accumulation results are considered.

The second perceived management problem is that, given a noninterventionist policy, there is a danger that accelerated soil loss may lead to a long-term reduction in species diversity which would conflict directly with the primary nature conservation objective of the Natal Parks Board, viz. to maintain species diversity (Grobler, 1984). Conservation biology points out that the proper objective of conservation is the protection and continuity of entire communities and ecosystems, and that the long-term viability of natural communities usually implies the persistence of diversity with little or no help from humans (Soulé, 1985). The objective of the maintenance of species diversity will be discussed in detail in Chapter 7; suffice it to say that it is considered to be an important and valid nature conservation objective (Soulé, 1985). If accelerated soil loss resulting from a noninterventionist policy can be shown to effect a reduction in species diversity, then this would contradict the primary objective of UGR and such an approach could not be adopted as a management option. Based on the assumption that an increase in herbaceous species diversity was indicative of an increase in the biotic diversity of UGR, the results showed there was no significant difference in herbaceous species diversity (in the sense of richness of species) between KWZ and UGR. There was however a highly significant difference when the grass species richness was compared between the upper, mid and lower slopes. This significant difference persisted even after the average variability accounted for by UGR versus KWZ, and by the exclosure versus control plots, was removed. Using an index of similarity, it was evident that the grass swards on the upper slope in UGR and KWZ were less similar in species composition than the swards in the mid and lower slope positions. The overriding effect of slope position on soil loss, A-horizon depth, herbage accumulation and grass species richness has been evident in all the results in this chapter.

The influence of topography on soil properties and soil moisture regimes has been described by Young (1976). These topographical influences are reflected in a regular sequence of different soil profiles which develop along a gradient from hilltop to adjacent valley bottom, and the changes in soil properties and moisture regimes in turn cause zonal patterns in the composition, structure, productivity and quality of vegetation (Bell, 1981; Tinley, 1982). In the current study, these zonal patterns exerted a consistently stronger influence on soil erosion, herbage accumulation and species richness than did differences in land use between UGR and KWZ. Thus, the factor having the greatest influence on species richness was found to be topography and not land use. Again, the perceived management problem in following a noninterventionist policy appears to be unfounded and, as there is not a conflict with the primary objective of maintaining species diversity, such a noninterventionist approach can be considered for implementation as a management option. It should however be remembered that species diversity can be defined on several scales from a point or microenvironment to a whole region (Whittaker, 1977), and before such a noninterventionist policy is implemented it should be established that this does not lead to a decrease in gamma diversity either on a reserve scale or on a regional scale. Gamma diversity is diversity of vegetation patterns and landscapes resulting from both alpha diversity (diversity within individual communities) and beta diversity (relative extents of differentiation of communities along topographical gradients) (Whittaker, 1965).

In conclusion, when a noninterventionist policy which was followed in KWZ was compared to an interventionist policy in UGR, there was no significant difference in the relationship between inferred different levels of soil erosion and the production potential of the remaining soil, nor in the relationship between inferred different levels of soil erosion and grass species richness and diversity. Significant differences were however obtained when the positions of the plots on a topographical gradient were considered.

6.8 CHAPTER SUMMARY

a) Because of deficiencies in the experimental design in the non-cull and cull blocks, an alternative study area on the western boundary fence was chosen in 1984. The experimental design here allowed for paired sampling sites on either side of the boundary fence.

b) The sampling sites were placed randomly and the sampling intensity was relatively high when compared with the non-cull/cull experiment. These factors facilitated the statistical analysis of the data.

c) There was no significant difference in soil loss from UGR and KWZ plots, but when soil loss on the upper, mid and lower slopes was compared there was a significant difference. This shows that it is the position on the slope, and not whether the plots are in UGR or KWZ, which explains the greatest variability in soil loss.

d) There was no significant difference between the A-horizon depths measured in KWZ and UGR, but there was a highly significant difference between the depths measured on the upper, mid and lower slopes. Like soil loss, the real differences in A-horizon depth were dependent on topographical position and not on differences in erosion levels between KWZ and UGR.

e) Although no statistically significant differences between either short-term or long-term erosion rates in KWZ and UGR could be shown, there was a consistent trend indicating that both these rates were higher in KWZ than in UGR.

f) Using soil loss and vegetation measurements from the Gerlach troughs, it was possible to confirm that a curvilinear relationship, best described by an exponential equation, exists between soil loss and both vegetation and soil surface variables.

g) The variable showing the highest positive correlation with $\log_e$ soil loss was percentage susceptibility to erosion, and the variable showing the highest negative correlation was percentage surface cover. This confirms the results obtained from both the rainfall simulator trials and the natural run-off plots.

h) There was no significant difference between herbage accumulated in KWZ and UGR in the absence of grazing, but there was a significant difference in the presence of grazing. This may be merely a reflection of differing grazing intensities on each side of the fence.

i) There was a highly significant difference in herbage accumulation on the upper, mid and lower slope plots in the absence of grazing and a significant difference in the presence of grazing. This indicates that the position of the plots on the slope influences herbage accumulation significantly, both in the presence and absence of grazing.

j) There was no significant difference between the grass species richness measured on plots in KWZ and UGR, but there was a highly significant difference when the grass species richness was compared between upper, mid and lower slopes.

k) The grass swards in the upper slope plots in KWZ and UGR were less similar in species composition than the swards in the mid and lower slope plots, and this relationship was evident regardless of whether the plots were grazed or not.

l) When herbaceous species composition was determined by bulk contribution to biomass, there was a marked difference between the species composition of the upper slope plots in KWZ and adjacent UGR plots. This difference became progressively less pronounced in the mid and lower slope plots.

m) The relationship in l) above was evident despite the two vegetation monitoring methods used and in spite of the year in which the vegetation monitoring was done.

n) Correspondence analysis on the 1984 herbaceous species composition data set primarily contrasted mid-slope plots in both UGR and KWZ with bottom slope plots in UGR, based on the presence of *Digitaria swazilandensis* in the mid-slope plots and *Themeda triandra* in the bottom slope UGR plots.

o) The results of the 1984 and 1986 correspondence analyses show that it is the position on the slope, and not whether the plots are in KWZ or UGR, or are grazed or ungrazed, that accounts for the highest degree of variability in the data.

p) There was no significant difference between the results obtained using the Walker (1976) method and the Barnes *et al.* (1982) method, regardless of whether the species composition was determined in UGR or KWZ, or in 1984 or 1986.

q) Because there was no significant difference between the results obtained using the above two methods, and because the Barnes *et al.* (1982) method is easier to use in the field, it is recommended that this method should be used in future vegetation monitoring programmes in UGR.

r) The major conclusion drawn in this chapter was that differences in topography, rather than differences in land use, exerted an overriding effect on soil loss, A-horizon depth, herbage accumulation and grass species richness.

s) Based on the results presented in this chapter, the perceived management problems of a noninterventionist policy leading to a decline in vegetation productivity and to a long-term reduction in species diversity appear to be unfounded.

CHAPTER 7 : A SYNTHESIS OF RELEVANT CONSERVATION MANAGEMENT PHILOSOPHY AND PRINCIPLES

7.1 INTRODUCTION

Management by objectives is a standard procedure for achieving goals in the business world (Miller & Child, 1983) and this approach has also been adopted by the Natal Parks Board (NPB) in the management of conservation areas under its control (Grobler, 1984). Not only is it important to identify major objectives before embarking on a conservation management programme (Ricklefs *et al.*, 1984), but these objectives also need to be re-evaluated constantly in the light of current knowledge (Miller & Child, 1983). Primary and secondary objectives were set for NPB areas in 1983 (Grobler, 1983), but conservation biology has recently supplied principles which can be used to achieve the objectives of nature conservation (Soulé, 1985). Concurrently, the objectives of the World Conservation Strategy (WCS) have been publicised (Allen, 1980; IUCN, 1980) and incorporated in the scientific literature (Siegfried & Davies, 1982; Hall, 1984; Ricklefs *et al.*, 1984). In the light of the extensive recent literature on the WCS and conservation biology, a decision was made to re-evaluate current NPB conservation policy before proposing changes to the NPB soils policy and soil reclamation objectives. It was felt that only once a policy had been determined could objectives be set to ensure that management practices were within the framework of such a policy (Grobler, 1984).

The three aims of this chapter are:

a) to synthesise relevant knowledge on the preservation of genetic diversity and the maintenance of essential ecological processes and life support systems;

b) to synthesise the relevant abiotic and biotic data obtained during the non-cull/cull and Cengeni fence-line experiments; and

c) to re-evaluate relevant Natal Parks Board conservation management policy and objectives in the light of the data synthesised above.

7.2 SYNTHESIS OF RELEVANT KNOWLEDGE ON THE PRESERVATION OF GENETIC DIVERSITY

7.2.1 Introduction

The WCS was drawn up to stimulate a more focused approach to the management and conservation of living resources. Its aim is to accomplish the three main objectives of living resource conservation which are:

a) to maintain essential ecological processes and life support systems on which human survival and development depend;

b) to preserve genetic diversity; and

c) to ensure the sustainable utilisation of species and ecosystems (IUCN, 1980).

Genetic diversity is defined in the WCS as the range of genetic material found in the world's organisms (IUCN, 1980), and the objective of preserving this diversity is by no means an academic exercise. If present rates of natural resource exploitation continue, about one million of the 5 - 10 million species on earth could be forced to extinction by the end of this century (Myers, 1984). Currently, more than 40% of the prescriptions filled each year in the USA contain a drug of natural origin, either as the sole active ingredient or as one of the main ingredients (Allen, 1980), and the potential value of these one million threatened species to the pharmaceutical industry, and to humanity, is obvious. Similarly, the current trend of degradation in savannas around the world, involving changes in composition and productivity, is affecting adversely the capacity of these systems to support humans and other organisms (Frost *et al.*, 1986). Degradation is not confined only to savanna areas, and this large-scale deterioration of the habitat and potential mass extinction of species is just one side of the coin. The other side is the cessation of significant evolution of new species of large plants and vertebrate animals because existing conserved habitats are too small to allow further speciation (Soulé & Wilcox, 1980).

7.2.2 Preservation of species diversity

Because efforts to save species are futile without preserving suitable habitats, many authors have emphasised the importance of habitat conservation in the preservation of species diversity (Eltringham, 1979; Diamond, 1982; Diamond, 1984; Prescott-Allen, 1984; Ricklefs *et al.*, 1984; Soulé, 1984). Botkin (1982; 1984) takes a broader perspective and argues that the conservation of species requires the conservation of ecosystems, but unfortunately such an ecosystem approach is not possible in Natal where the largest conserved areas represent only fragments of ecosystems (Hanks *et al.*, 1981). The important point is that conservation should be aimed at the protection and continuity of entire communities, as the long-term viability of natural communities usually implies the persistence of diversity (Soulé, 1985), and it is generally accepted that the maintenance of diversity should be the primary aim of conservation (Eltringham, 1979).

Diamond (1984) has listed the eight main determinants of species diversity. Firstly, if one compares unequal areas of similar habitat, species diversity increases with area. Secondly, the more isolated a habitat is from other patches of similar habitat, the fewer species will occur in the patch. It appears as though these two determinants are of relevance only when new land is acquired for conservation purposes but the following five determinants can be influenced, to a greater or lesser extent, by management within an existing conserved area.

Not only does species diversity tend to increase with diversity of habitat but, within a single habitat, it tends to increase with increasing complexity of the habitat's physical structure. Physical disturbances, such as drought, fire and flood, are important in increasing habitat complexity because, as a result of repeated disturbance, the habitat becomes a mosaic of successional patches in which there is a high species diversity instead of a few dominant species. There is now abundant evidence that without such disturbances many species would cease to exist (Sousa, 1984). It is important to note that with an increase in disturbance, species diversity may increase but it will eventually decrease with further increases in disturbance. This applies to predation and, in fresh-water environments, to productivity as well where species diversity increases over a certain range only (Diamond, 1984). Lastly, species diversity is also affected by history because habitats change with time, and species distributions and diversities do not adjust instantly to habitat changes.

To summarise, if one accepts that the primary objective of nature conservation is the maintenance of species diversity, then management should be aimed at maintaining, and if necessary increasing, habitat diversity both spatially and temporally. Physical disturbances and predation should be allowed to operate unhindered, unless they reach such a level that species diversity is reduced. These steps should be aimed at maximising the number of species whose populations can sustain themselves within reserve boundaries and which would be threatened with extinction in the absence of reserves (Diamond, 1984).

7.3 SYNTHESIS OF RELEVANT KNOWLEDGE ON THE MAINTENANCE OF ESSENTIAL ECOLOGICAL PROCESSES AND LIFE SUPPORT SYSTEMS

7.3.1 Conservation of ecological processes

The WCS considers the maintenance of ecological processes to be essential for the functioning of the biosphere because without them life would not be possible (Allen, 1982). The formation of soil is such an ecological process and, through protection of soil, the manager can ensure the long-term productivity of agriculture, forestry, pastures and rangelands. Since a major part of the earth's land surface will be transformed and modified to supply food to support the human population, the WCS stresses that management should ensure that such transformations and modifications do not create biological deserts, i.e. areas incapable of sustaining life-support processes in the long term because of artificially accelerated soil erosion, siltation and biotic extinctions.

Hopefully, such modifications and transformations will not occur in conservation areas, and here organisms and their habitats will be conserved in natural ecosystems or portions of ecosystems. Ricklefs *et al.* (1984) contend that as ecosystems, communities and habitats are dynamic entities, an understanding of basic ecological processes responsible for the origin and maintenance of organisms, habitats and landscapes is essential, and ecological processes must themselves be conserved. These ecological processes include all the physical processes as well as the plant and animal activities which influence the state of ecosystems and contribute to the maintenance of their integrity and genetic diversity, and thus to their evolutionary potential. Ecological processes may be broadly classified as biogeochemical cycles; primary and secondary production; mineralisation of organic matter in the soils and sediments; storage and transport of minerals and biomass; and lastly, regulation of the above processes, often by the activities of animals (Ricklefs *et al.*, 1984). The conservation of ecological processes and of ecosystems and their constituent populations (Botkin, 1984) go a step beyond species and habitat preservation because they consider the dynamics of the systems; however, they both share the common goal of the preservation of species diversity.

7.3.2 Soil erosion: an ecological perspective

Soil erosion is an ecological process which should be allowed to operate without interference. The rate at which it occurs should be the point of concern for managers, and not soil erosion *per se*. Rainfall affects the rate of erosion in two ways. Firstly, in regions of very low mean annual rainfall there can naturally be little erosion caused by rain while, at the other extreme, an annual rainfall of more than 1 000mm usually gives rise to dense forest vegetation which affords a protective cover to the soil and reduces erosion (Hudson, 1981). Between these two extremes of annual rainfall there is a peak in erosion corresponding to semi-arid conditions (Figure 7.1). Here the mean annual precipitation is not enough to sustain a complete vegetation cover throughout the year, but it is sufficient to cause erosion of the bare soil (Branson *et al.*, 1981). The importance of natural vegetation cover in reducing soil erosion above a mean annual rainfall of 400mm is evident when erosion rates from bare ground are compared with rates from undisturbed natural vegetation (Figure 7.1). Secondly, the rate of erosion is affected by rainfall intensity, which is usually much higher in the tropics than in the temperate zones, and the approximate limits of high intensity rainfall are latitudes 40° north and 40° south (Hudson, 1981). Generally, soil erosion by water can be expected to be most serious in areas between these latitudes where the annual rainfall is neither very high nor very low. In Africa this includes the entire continent except the dry deserts and equatorial forests (Figure 7.2).

Different land uses also affect the rate of erosion. Knott (1973) found that the average sediment yield was 65 times higher in cultivated land than in land which was protected by natural or re-established vegetation. Dunne (1979) concluded that erosion from rural roads and footpaths in Kenya may contribute significantly to the sediment yield of agricultural catchments and may cause an overestimation of the degradation of cultivated lands. The surface erosion rate from catchments with logging roads was 220 times (Megahan & Kidd, 1972) to 250 times higher (Fredricksen, 1970) than the erosion rate from adjacent and undisturbed watersheds. When compared with sediment production from cultivated lands, Wood & Blackburn (1981) concluded that, irrespective of the grazing intensity, sediment production from rangelands was extremely small. The mean sediment yield from grazed catchments was 4,7 times higher than that from adjacent ungrazed catchments, and the range in sediment yield varied from 1,3 times to 28,8 times (Table 7.1). The important point here is that heavy continuous grazing has far less influence on erosion rates than other forms of disturbance such as the construction of roads or the cultivation of land. The fact that an area is used for grazing land instead of cultivated land means that the rate of soil erosion will be appreciably less. This is illustrated by Wischmeier & Smith (1978) who showed that for every 100kg of soil lost from a ploughed field maintained in a bare fallow condition, only 45kg were lost from a comparable undisturbed piece of land with no ground or canopy cover.

Erosion rates from a natural environment undisturbed by man are considered to be geologic or normal. When it is proceeding at a rate greater than normal, reflecting man's activities at a site, erosion is considered to be accelerated (Branson *et al.*, 1981). Accelerated erosion has led to two major problems. Firstly, it threatens long-term crop production, and hence food supply, which is needed to sustain an increasing human population (Brown, 1981; Riquier, 1982; Larson *et al.*, 1983). In this context it also affects the productivity of rangelands, and thus their ability to produce animal protein for human consumption (Gifford & Whitehead, 1982; Wilson & Tupper, 1982). Secondly, sediment is the product of a selective erosion process in which the finer and lighter particles are preferentially removed from the soil. Sediment is recognised as a pollutant since these particles can adsorb pesticides and other chemicals, particularly persistent chlorinated hydrocarbons, which have a low solubility in water (Glymph, 1975). Sediment yield is of further significance because it reduces dam storage capacities, and recently it has been implicated in decreasing the depth of hippo *Hippopotamus amphibius* pools in the Kruger National Park, which has led to overpopulation in the remaining hippo pools and intraspecific fighting (Anon., 1987).

A third problem which is not covered in the literature but has been discussed in Sections 6.5 and 6.7 is that accelerated erosion may lead to a reduction in species diversity. This is an important problem if one accepts that the primary objective of nature conservation is the maintenance of species diversity. Not only is the rate of erosion affected by grazing intensity (Blackburn *et al.*, 1982), but very heavy grazing has also been shown to reduce grass species diversity in North American rangelands (Whittaker, 1977). The important point however, is that in semi-arid savannas (300 - 700mm rainfall per annum) the species composition of the herbaceous layer appears to be affected primarily by year-to-year and longer-term variations in rainfall, and to a lesser extent by grazing. The effects of grazing and fire become relatively more important as mean annual rainfall increases and its variability declines (O'Connor, 1985; Frost *et al.*, 1986). This point is crucial for the interpretation of the data from the non-cull/cull and Cengeni fence-line experiments, where the mean annual rainfall was less than 700mm per annum (Section 2.2.2), and for future management decisions on whether to cull large herbivores or not.

7.4 SYNTHESIS OF RELEVANT DATA FROM THE NON-CULL/CULL AND FENCE-LINE CONTRAST EXPERIMENTS

7.4.1 Rainfall, soil loss and vegetation data

Unfortunately, no long-term rainfall data are available from the study areas, but the long-term trend in annual rainfall at Mpila, UGR, is shown in Figure 7.3 and it is evident that the annual rainfall fluctuates widely about the mean. The coefficient of variation, which is a relative measure of dispersion about the mean (Fuller & Lury, 1977), is 31,6%. This is in accordance with the findings of O'Connor (1985) who concluded that there was an increase in the inter-seasonal variation in rainfall as the mean annual rainfall decreased, with the coefficient of variation ranging from about 10% at a mean annual rainfall of 1 000mm to >40% at 350mm.

There were more below-average rainfall years (n = 16) than above-average (n = 11), but only seven years were more than 25% above or below mean annual rainfall (Figure 7.3). The non-cull/cull experiment was started in the 1978/79 rainfall year, near the beginning of a 6-year drought period which culminated in the driest year on record when, during 1982/83, only 45% of the mean annual rainfall was recorded. The following year was the wettest year on record with the rainfall received being 92% above the mean annual rainfall (Figure 7.3).

The limited data set shows that the rainfall season begins with storms in October, November and December which have low values of erosivity, and it is only in January and February that high-erosivity storms occur (Figure 7.4). Soil loss from heavily utilised veld or recently burnt veld would be higher if the rainfall season began with high-erosivity storms because vegetation cover would be at its lowest at the end of the dry season. However, as lower-intensity storms occur during the earlier growing season, the vegetation cover has the opportunity to increase before the high-intensity storms in January and February, thus decreasing the amount of soil loss through raindrop splash erosion on bare ground.

In the rainfall simulator trials, the annual mean soil loss varied from 0,26 t/ha in the cull block to 0,88 t/ha in the non-cull block (Table 7.2). Snyman *et al.* (1985), using a similar rainfall simulator on natural veld in the Orange Free State, found that the mean soil loss varied from 1,45 t/ha on climax veld to 4,15 t/ha on pioneer veld. These values are considerably higher than those obtained in the current study, but they are based on only a single year's data. The only other published data on soil loss from natural veld in South Africa, which were based on natural run-off plots monitored over a period of 26 years, gave a mean annual soil loss of 0,75 t/ha on ungrazed veld and a maximum of 1,26 t/ha on grazed veld (Haylett, 1960). These values are also higher than those obtained from natural run-off plots in the current study, which ranged from 0,24 t/ha in the cull block to 0,74 t/ha in the non-cull block (Table 7.2).

To summarise, although on average approximately three times more soil was lost from plots in the non-cull block than from those in the cull block, the mean soil losses from plots in the non-cull block were lower than soil losses measured from plots on natural veld in other studies in South Africa.

7.4.2 Interaction between abiotic and biotic factors

Based on data collected at 27 Walker transects in the study area from 1979 to 1986 (Section 4.6.1), the herbaceous canopy cover in the cull and non-cull blocks was correlated significantly with the mean of the annual rainfall received in the current and previous growing season (Table 7.3). The response of both blocks to mean annual rainfall is similar, and rainfall is clearly an important driving variable in this system irrespective of whether an interventionist or noninterventionist management approach is followed (Figure 7.5). A similar interaction between rainfall and mean grass height was also evident in the two management blocks (Figure 7.6). Furthermore there was a highly significant correlation between these two variables in both management blocks (Table 7.3).

Mean soil loss was significantly correlated with herbaceous canopy cover, and $\log_e$ mean soil loss with both herbaceous canopy cover and mean grass height (Table 7.4). $\log_e$ mean soil loss was also significantly correlated with annual rainfall, which presumably influences soil loss through its effect on vegetation variables, since the mean soil loss was derived from rainfall simulator trials where both the amount and intensity of rainfall were standardised. Unless it can be shown that stocking rate also has a marked influence on vegetation variables, and hence on soil loss, then it must be concluded that rainfall, through its effect on vegetation, is the primary determinant of soil loss in this system, and that stocking rate plays only a secondary role.

During the period 1979 to 1986 there was no significant correlation between stocking rate and herbaceous canopy cover, nor between stocking rate and mean grass height, in either the cull or the non-cull block (Table 7.3). Nor was there any significant correlation between stocking rate and mean soil loss (Table 7.4). The vegetation variables were measured at 27 Walker transects in the study area (Section 4.6.1), and the relationship between stocking rate and vegetation variables is shown for the cull block and non-cull block in Figures 7.7 and 7.8 respectively. No clear relationship is evident between annual stocking rate and annual values for herbaceous canopy cover and mean grass height in either the cull or non-cull blocks. Obviously therefore stocking rate cannot be used to explain the between-year fluctuations in either of these vegetation variables, and hence in soil loss. There does however appear to be a longer-term relationship between stocking rate and the above vegetation variables, with a decrease in stocking rate from 1979 to 1986, being accompanied by an increase in herbaceous canopy cover and mean grass height over the same period in both the cull (Figure 7.7) and non-cull blocks (Figure 7.8). However, this long-term trend can also be explained by an increase in mean annual rainfall over this period (Figures 7.5 & 7.6), and again leads to the conclusion that rainfall and not stocking rate is the primary determinant of soil loss in this system, through its effect on vegetation cover.

The effect of rainfall on herbage accumulation, which is a crude measure of the carrying capacity of the system, is shown in Figure 7.9. The exclosure plots exclude large herbivore grazing and represent the extreme of population control, i.e. when all the large herbivores are removed. There is a significant difference in the mean herbage accumulation over the three growing seasons in these exclosure plots (F = 5.98, d.f. = 2,15), and this difference is related to rainfall (Figure 7.9). The influence of rainfall is evident in all the exclosure plots, where the highest yield was measured during the wettest rainfall year and the lowest during the driest rainfall year (Figure 7.9). These data, together with herbage accumulation data in the presence of grazing (Section 6.4.2), show that the abiotic effects of rainfall and slope position on yield are far greater than the effect of stocking rate. The results of 11 southern African studies which were reviewed by O'Connor (1985) all showed that rainfall variability had a primary effect on trends in yield, irrespective of grazing treatment.

To summarise, it appears as though rainfall, through its effect on vegetation cover and yield, is the primary determinant of soil loss in this system. Between-year fluctuations in yield are so large, due to the highly variable rainfall, that manipulation of the stocking rate by the removal of large ungulates will have a lesser influence on vegetation yield and cover, and hence on soil loss.

Proponents of one interventionist approach attempt to get around the question of rainfall variability by suggesting that stocking rates should always be set with a drought situation in mind (Smuts, 1980). The mean annual rainfall in below-average years in UGR is 554mm, which is 20% below that of the long-term mean annual rainfall of 693mm. To accommodate the "average" drought, stocking rates would have to be set approximately 20% lower than normal, and this is clearly a conservative approach. Smuts (1980) also recommended that the ungulate stocking rates should be based on suggested agricultural stocking rates, which encompass the concept of economic carrying capacity. This is the stocking rate where maximal animal production per unit area is achieved and it can be maintained only by an annual offtake of animals in perpetuity (Caughley & Walker, 1983). This contrasts with the concept of ecological carrying capacity which is the density at which a population will "stabilise" if it is left alone. This density is not a steady density, but will fluctuate from year to year (Caughley & Walker, 1983).

There are three basic problems in the use of agriculturally recommended stocking rates set with a drought situation in mind. Firstly, since nowhere is it stated that a NPB conservation management policy or objective is to maximise animal production per unit area, the concept of economic carrying capacity appears to be inappropriate in game reserves administered by the NPB. Secondly, by setting stocking rates to accommodate the average drought an attempt is made to stabilise the system and minimise the stress placed on it. Recent ecological theory on the properties of stability and resilience in savanna ecosystems and on disturbance regimes suggests that by maintaining a variable herbivory regime, and thus regularly exposing the species within the system to a wider range of conditions, managers can increase the resilience of the system to excessive herbivory and minimise the risk of sudden shifts in composition and production (Sousa, 1984; Frost *et al.*, 1986). Temporal and spatial variability in disturbances, such as floods, prolonged droughts and excessive herbivory, are important for the maintenance of species diversity and there is now abundant evidence that, without such temporal and spatial variability, many species would cease to exist (Sousa, 1984). Given that the primary objective of nature conservation is the maintenance of species diversity, the use of agriculturally recommended stocking rates set for drought years appears to be not only inappropriate in a game reserve context, but also inadvisable. Thirdly, the mere fact that stocking rate limits are set implies that the system, comprising the ungulate populations and their resources, is incapable of maintaining an equilibrium unaided, and therefore must be managed constantly (Caughley, 1981). Current evidence suggests that some large herbivore populations in savanna areas are regulated, and that the regulating mechanisms act through the availability of resources such as food and by interaction with other species (Sinclair, 1975; Melton, 1978). Sinclair (1974) showed that the density of buffalo *Syncerus caffer* in different areas of East Africa was correlated closely with rainfall which, in turn, was an index of grass production. Coe *et al.* (1976) were able to derive a predictive equation for biomass of large herbivores based on the correlation between herbivore biomass and rainfall from different areas in Africa. This correlation is obviously linked to the association between rainfall and primary production, and leads to the conclusion that in a complete functioning system large herbivore populations are regulated by resources such as food amount and availability.

Proponents of another interventionist approach suggest that, in H&UGR, fire should be used to halt woody plant invasion into grassland and thus slow down or stop the decline in large mammal species diversity (Brooks & Macdonald, 1982; 1983). They reasoned that this could be achieved by holding populations of large grazers at or below the estimated carrying capacity and thus increasing the frequency and intensity of fires. Although the authors do not define carrying capacity, it appears to be closer to the economic than to the ecological carrying capacity. The problem with such an approach is that the longer the density of herbivore populations is held below ecological carrying capacity, the lower the system's ability to absorb change, and the greater the resultant disruption if active control of density is relaxed (Caughley & Walker, 1983). An important assumption made by Brooks & Macdonald (1983) is that fire is the only determinant of grassland in H&UGR which can be manipulated. It will be argued later (Sections 8.2.2 and 8.3.2.1) that soil moisture balance is also an important determinant of certain grasslands and that it too can, and should, be manipulated.

In any event, the critical question appears to be whether the system can be guided to a dynamic equilibrium without the oscillations which *might* cause a change in the system with a consequent loss of genetic variability and excessive soil erosion (Hanks *et al.*, 1981). These concerns have been discussed in Chapter 6 and in this chapter, and appear to be largely unfounded.

As a result of two recent workshops, management in H&UGR has been based on an ecological carrying capacity approach with emphasis on the conservation of ecological processes (Densham & Wills, 1984; Anon., 1985b). This ecological management approach was initially proposed by Emslie (1985) and the following section and Section 8.3 will be devoted to the implications of this approach in the light of NPB conservation objectives and with particular emphasis on soil erosion.

7.5 RE-EVALUATION OF RELEVANT NATAL PARKS BOARD CONSERVATION MANAGEMENT POLICY AND OBJECTIVES

7.5.1 Introduction and historical perspective

The conservation policy of the NPB is to promote the wise use of natural resources in perpetuity and to prevent degradation of the environment (Grobler, 1984). More specifically, as far as the management of reserves is concerned, the primary objective is: "to conserve the optimum number of appropriate indigenous (to the reserve) species and their habitats, maintain breeding populations and protect the specificity of these gene pools. Natural, physical and ecological processes will be allowed to operate without interference except under imperative circumstances" (Grobler, 1984). Criticism of this objective and its implementation in reserves (Mentis, 1985) led to a revision of the primary objective for H&UGR which is: "to conserve a variety of habitats types and their associated indigenous species; to maintain breeding populations (of appropriate species); and to maintain their long-term genetic viability by management. In order to achieve this, ecological processes will be allowed to operate without interference except when:

a) these processes have been impaired;
b) there is a threat of a reduction in genetic diversity; and
c) the operation of an ecological process will jeopardise the future of the reserves due to socio-political circumstances" (Anon., 1985b).

As it stands above, this objective caters for the maintenance of species diversity and the conservation of ecological processes. It recognises that, for habitat diversity to be maintained, ecological processes which influence the dynamics of the habitats need to be conserved as well (Ricklefs *et al.*, 1984). It also recognises the important principle that species will disappear if their habitats disappear (Wilcox, 1980; Soulé, 1985). The terminology used in the above objective has been defined in a glossary of terms (Anon., 1985b).

The only qualification that the writer considers necessary in the above primary objective is to emphasise that the maintenance of species diversity should be aimed at those populations of species which would be threatened with extinction *in the absence of reserves* (Diamond, 1984).

7.5.2 Re-evaluation of current Natal Parks Board soils policy and soil reclamation objectives

The current soils policy is: "to ensure that accelerated erosion is attended to, and that in the long-term, soil loss equals soil genesis" (Grobler, 1983; Anon., 1985c). Although not defined in this policy statement, accelerated erosion is considered to be erosion proceeding at a rate which exceeds that of geologic erosion and reflects man's activities at a site (Branson *et al.*, 1981). No statement was made as to whether the influence of

Iron-Age man, who markedly modified the vegetation (Feely, 1980; Hall, 1979), was considered to be a natural phenomenon or whether his activities contributed to accelerated erosion. These are, however, minor points. The major problem with this policy is that it tries to balance the soil loss/soil genesis equation in the long term. Although this is a commendable goal, it is not possible to assess goal attainment because, even though soil erosion rates can be measured precisely, this is not possible with the measurement of soil genesis rates, and the relationship between the two in a natural situation cannot be determined. Very little data have been published on soil genesis rates in Africa, but Owens (1974) calculated that the average rate of soil formation from weathering granite in Zimbabwe varied from 0,15 to 0,4 t/ha/annum. The point here is that even when rates are known they still vary widely. The long-term geologic rate of erosion also varies markedly. Dunne *et al.* (1978) calculated that the geologic rate of erosion in semi-arid Kenya increased from about 0,22 t/ha/annum during most of the Tertiary period to between 0,77 and 2,0 t/ha/annum during the late Pliocene and Quartenary periods. In Kenya the calculated geologic erosion rate also varied widely between catchments, from 0,2 to 2,0 t/ha/annum (Dunne, 1979). This temporal and spatial variation in geologic erosion rates and the variation in soil formation rates make it very difficult, if not impossible, to attain the goal of balancing soil loss with soil genesis in the long term. The writer's conclusion is that this goal is not attainable and should be changed to an operational goal which is attainable. There appear to be two options.

The first possible goal is based on the concept of soil loss tolerance, which is the greatest amount of erosion that can be tolerated without productivity declining (Hudson, 1981). This concept has been used primarily in arable situations which is evident from its current definition, viz.: "the maximum rate of annual soil erosion that may occur and still permit a high level of crop productivity to be obtained economically and indefinitely" (Schertz, 1983), but recent attempts have been made to apply it to rangeland (Moldenhauer, 1982; Wight & Lovely, 1982; Williams, 1982). To date, these attempts have met with limited success and Gifford & Whitehead (1982) concluded that it is currently impossible to evaluate the impact on productivity of different rates of soil erosion on the western rangelands of the USA. In semi-arid savanna areas, where variable rainfall plays such an important role in primary production, it would be even more difficult to evaluate the effect of soil erosion on productivity, and the current study was not able to detect any significant effects (Section 6.4.2). Also, given that the primary objective is to maintain species diversity and not to maximise long-term productivity, it is suggested that another goal would be more appropriate. Briefly, this goal can be stated as the following policy: "To minimise the rate of accelerated erosion and thereby its effect on those populations of species which can sustain themselves within the reserves' boundaries and which would be threatened with extinction in the absence of the reserves."

The remainder of this section will deal with translating this policy into a broad objective and, where necessary, comparing it with current soil reclamation objectives. In this discussion the terms "goal" and "objective" are used synonymously, but recent literature (Miller & Child, 1983; Mentis, 1985) suggests that the word "goal" should be used in preference to "objective."

The current reclamation objective is as follows: "Erosion reclamation is aimed at preventing accelerated soil erosion from occurring and at minimising the deleterious effects of this erosion on habitat diversity in the reserve" (Anon., 1985c). As it stands, this objective covers the two facets of the proposed soils policy above. It does appear to be idealistic in its attempt to prevent accelerated soil erosion from occurring, since it is difficult to determine objectively what is geologic (natural) erosion and what is accelerated (man-induced) erosion. What constitutes accelerated soil erosion also needs to be clarified. The second aspect of minimising the deleterious effects of soil erosion on habitat diversity adequately covers the facet of the maintenance of species diversity, but needs to be stated more specifically so that progress towards attaining this goal can be measured.

To ensure that the soil reclamation objective does not conflict with the primary objective for H&UGR (Section 7.5.1) it is suggest that the soil reclamation objective should read as follows:

"Soil erosion is an ecological process which should be allowed to operate without interference unless:

a) the rate of erosion has been accelerated by the activities of recent man;

b) soil erosion, particularly donga incision, threatens a rare habitat type; and

c) accelerated soil erosion jeopardises the future of the reserves because of its socio-political consequences."

This objective adequately covers the conservation of ecological processes, the maintenance of habitat (and hence species) diversity, and the prescribed conditions which would warrant management intervention. What remains is the translation of this broad objective into management goals, and this will be covered in the next chapter.

7.6 CHAPTER SUMMARY

a) Current scientific knowledge indicates that the primary objective of nature conservation should be the maintenance of species diversity. Since this is dependent on the maintenance of habitat diversity, and because the dynamics of habitats are influenced by ecological processes, conservation management should be aimed at conserving ecological processes and maintaining habitat diversity, both temporally and spatially.

b) This management should be focused on maximising the number of species whose populations can sustain themselves within reserve boundaries and which would be threatened with extinction in the absence of reserves.

c) Measurements of soil loss showed that on average approximately three times more soil was lost from plots in the non-cull block than from those in the cull block, but the mean soil losses from plots in the non-cull block were still lower than soil losses recorded from plots on natural veld in South Africa during other studies.

d) In UGR it appears as though rainfall, through its effect on vegetation cover and yield, is the primary determinant of soil loss. Between-year fluctuations in yield are so large, due to rainfall variability, that the manipulation of the stocking rate by the removal of large ungulates will have only a secondary influence on vegetation yield and cover, and hence on soil loss.

e) Since the Natal Parks Board does not have as one of its conservation objectives the maximisation of animal production per unit area, the concept of economic carrying capacity appears to be inappropriate in a game reserve context. Such an economic carrying capacity may decrease temporal and spatial variability of disturbances, especially when the stocking rate is adjusted to drought years, and without such variability many species could disappear from the reserves. Given that the maintenance of species diversity is the primary objective of nature conservation, a management strategy which has the potential to reduce species diversity is clearly inappropriate in a game reserve context.

f) A more appropriate management strategy appears to be one based on the concept of ecological carrying capacity. This is the density at which a population will fluctuate about if it is left alone, *provided that* major ecological processes are functioning naturally or being adequately simulated, *and that* this density can be reached without severe oscillations which might cause a change in the system with a consequent loss of species diversity and excessive soil erosion.

g) The revised primary objective for H&UGR is: "to conserve a variety of habitat types and their associated indigenous species; to maintain breeding populations (of appropriate species); and to maintain their long-term genetic viability by management. To achieve this, ecological processes will be allowed to operate without interference unless i) these processes have been impaired; ii) there is a threat of a reduction in genetic diversity; and iii) the operation of an ecological process will jeopardise the future of the reserves because of socio-political circumstances."

h) This objective not only caters for the maintenance of species diversity and the conservation of ecological processes but it also recognises the importance of habitat diversity in the maintenance of species diversity.

CHAPTER 8 : MANAGEMENT RECOMMENDATIONS

8.1 INTRODUCTION

Based on the data presented in Chapters 1 to 7, the aim of this chapter is to make management recommendations regarding:

a) conservation management policy and objectives;

b) future conservation management programmes; and

c) future monitoring programmes and research projects.

The background information in this chapter is in a condensed form and, where relevant, the pertinent section is quoted in the text to enable the reader to refer to the more detailed discussion and rationale.

8.2 CONSERVATION MANAGEMENT POLICY AND OBJECTIVES

8.2.1 Proposed future soils policy

Background: Accepting that the primary aim of nature conservation is to maintain the species diversity of populations which would be threatened with extinction outside the reserves, and that the maintenance of species diversity depends on the maintenance of habitat diversity and the conservation of ecological processes, I recommend that the current soils policy should be changed. Instead of trying to balance soil loss with soil genesis in the long term, greater emphasis should be placed on the maintenance of habitat diversity, and hence species diversity. The primary reason for changing the emphasis from the balancing of the soil loss/soil genesis equation is that, while it is possible to estimate soil loss accurately, it is difficult, if not impossible, to do the same for soil genesis. There is also evidence that the rates of soil genesis and geologic erosion differ both in space and in time (Section 7.5.2). Because of these difficulties, it would be impossible to measure progress towards the goal of ensuring soil loss equals soil genesis in the long term, and hence to establish whether any management action is succeeding in achieving this goal.

Recommendation: The soils policy should be changed to the following:

"To minimise the rate of accelerated erosion and thereby its effect on those populations of species which can sustain themselves within the reserves' boundaries and which would be threatened with extinction in the absence of the reserves."

8.2.2 Proposed future soil reclamation objectives

Background: The primary objective for the Hluhluwe and Umfolozi Game Reserves (H&UGR) is: "to conserve a variety of habitat types and their associated indigenous species; to maintain breeding populations (of appropriate species); and to maintain their long-term genetic viability by management. In order to achieve this, ecological processes will be allowed to operate without interference except when:

a) these processes have been impaired;
b) there is a threat of a reduction in genetic diversity; and
c) the operation of an ecological process will jeopardise the future of the reserves due to socio-political circumstances" (Anon., 1985b).

A number of terms used in the soils policy, the primary objective and in the soil reclamation objective are defined in the glossary below. Additional terms relevant to the primary objective are defined in various unpublished Natal Parks Board (NPB) documents (Anon., 1985b; Anon., 1985c).

Accelerated soil erosion: soil erosion proceeding at a rate greater than geological erosion because of man's activities at a site (Branson *et al.*, 1981).

Activities of recent man: those activities, excluding the activities of Iron-Age man (who is considered to form an integral part of the ecosystem) but including human activities, which have modified the environment directly (road construction, road and track drainage, quarrying, building construction), or which have resulted from the utilisation of the environment by humans since the proclamation of the reserves (footpaths, donkey and horse trails). The activities of recent man also include those human activities which have modified the environment indirectly (e.g. the erection of a boundary fence and consequent disruption of ecological processes such as migration, dispersion and immigration).

Ecological processes: all the physical processes and the plant and animal activities which influence the state of ecosystems and contribute to the maintenance of their integrity and genetic diversity, and thereby their evolutionary potential (Ricklefs *et al.*, 1984).

Rare habitat type: an association of species (usually floral), recognisable as a mapping unit in terms of its physiognomy and species composition, currently occupying less than 5% of the surface area of H&UGR (A.J. Wills - *pers. comm.*).

The primary objective emphasises that ecological processes should be allowed to operate unhindered unless certain conditions are encountered which justify management intervention. I recommend that the soil reclamation objective be slightly modified so that it follows the format of the primary objective, and so that the conditions which justify management intervention are clearly stated.

Recommendation: The soil reclamation objective should be changed to the following:

"Soil erosion is an ecological process which should be allowed to operate without interference unless:

a) the rate of erosion has been accelerated by the activities of recent man;

b) accelerated soil erosion, particularly donga incision, threatens rare habitat types; and

c) accelerated soil erosion jeopardises the future of the reserves because of its socio-political consequences."

The remainder of this section will translate the above broad soil reclamation objective into practical management goals. This will be done by considering separately the three prescribed conditions which warrant management intervention.

a) *The rate of erosion has been accelerated by the activities of recent man*

Background: The activities of recent man are broadly classified into those which directly modify the environment and those which have modified the environment indirectly. A higher priority should be placed on management intervention where accelerated erosion can be ascribed directly to the activities of recent man, as is the case in road construction.

There are many reports indicating that accelerated erosion occurs following road construction (Fredricksen, 1970; Megahan & Kidd, 1972; Megahan, 1977), and the resulting erosion from roads, tracks and mitre drains has been a source of concern for over 20 years in H&UGR (Stewart, 1965; Porter, 1972; Brooks, Macdonald & Whateley, 1980). White (1978) reported an annual soil loss from a road surface of 104 t/ha, and Dunne (1979) calculated a soil loss rate of 100 to 210 t/ha/annum from roads and tracks in an agricultural catchment in Kenya. He estimated that rural roads and tracks contributed between 15% and 35% of the total sediment yield from the catchment. These soil losses far exceed the soil losses measured in the non-cull block, which averaged 0,74 t/ha/annum (Section 7.4.1). Since erosion resulting from roads and tracks clearly falls into category a) above, and because of its disproportionate effect on soil loss, I recommend that erosion reclamation effort should be aimed at minimising such erosion, and that this should be given a high priority.

Recommendation: The practical management goal should be:

"to identify areas of active erosion resulting from road and track construction and if possible to prevent, or at least minimise, such erosion by the correct alignment and drainage of roads and tracks; and the closure and reclamation of nonessential management tracks and service roads."

A multidisciplinary workshop should be convened to draw up a technical manual detailing the principles involved in road planning, siting, construction and drainage, for use in NPB conservation areas. Since such a manual requires detailed roads engineering expertise in the form of standard designs for causeways and bridges, it should be produced and edited by a firm of consulting engineers on a contract basis (see Section 8.3.3).

b) *Accelerated soil erosion, particularly donga incision, threatens a rare habitat type*

Background: All too often it is not the species *per se* that is directly being destroyed, but its habitat. Lacking the essential support systems that the habitat once provided, the species then inevitably perishes (Hall, 1984; Soulé, 1985). There has been a widespread reduction in open grasslands in H&UGR in the recent past (Watson & Macdonald, 1983) and in particular, vlei areas and open grassland areas on ridge crests have decreased markedly in extent and currently cover only 0,2% and 1,3% respectively, of the area of H&UGR (Macdonald, 1982a). Because pure grasslands occur wherever there is poor site drainage, or where a shallow soil profile overlies an impermeable horizon (Tinley, 1982), and are thus maintained by hydrological processes, any factor which disrupts the hydrology of such systems will threaten these rare habitat types. The widespread reduction in vlei areas in H&UGR has been caused by donga incision and inadvertent drainage and damming through road and track development (Macdonald, 1982a). Geomorphologically, as a landscape achieves maturity and old age the rate of erosion slowly decreases, mainly because of a reduction in slope. If an existing drainage line is scoured out to form a donga, or if a donga forms as a result of an incorrectly sited management track or tourist road, a new base level is attained. This forces the surrounding landscape to re-equilibrate itself with this lowered base level (Goodman, 1984), with a consequent lowering of the water table (Figures 8.1 & 8.2). The result is a replacement of depression mesic grasslands with woody thickets and the subsequent local extinction in HGR of common reedbuck *Redunca arundinum*, African marsh harrier *Circus ranivorus*, marsh owl *Asio capensis* and grass owl *Tyto capensis*, all of which utilise depression mesic grasslands (Macdonald, 1984). Because this loss of hydrologically maintained mesic grassland habitat has resulted in a loss in species diversity within the reserves, which clearly conflicts with the primary objective of maintaining habitat and species diversity, it is obvious that erosion reclamation effort should be directed at preventing donga incision into these habitats. Such reclamation should also be given a high priority, because donga incision was probably initiated through misplaced roads and management tracks, and is thus a case where the rate of erosion has been accelerated by the direct activities of recent man.

Recommendation: Erosion reclamation effort should be directed at identifying those rare habitat types threatened by accelerated soil erosion, and particularly by donga incision into hydrologically maintained mesic grasslands. Once identified, eroding donga heads should be stabilised and, if possible, attempts should be made to reinstate hydrological processes by reconstructing the original base level of the local landscape and by removing invading woody plant species.

c) *Accelerated soil erosion jeopardises the future of the reserves because of its socio-political consequences*

Background: As socio-political circumstances and perceptions may change markedly with time, it is very difficult to predict which of these circumstances could jeopardise the future of the reserves. Only one scenario appears to be relevant at present. Briefly, in this scenario agricultural concepts and principles in range management are confused with the ecological concepts and principles of conservation biology. The objectives of agricultural rangelands and those for game reserves are different, and are based on different concepts of carrying capacity (Caughley & Walker, 1983). If these differences are not understood, then range managers and wildlife managers will be talking at cross purposes. This happened in UGR during the 1978 - 1983 below-average rainfall period (Brooks, 1983), when not only agriculturalists but also NPB staff questioned the advisability of maintaining the non-cull block because of perceived environmental degradation. The problem is that legislative control lies in the hands of agriculturalists in the form of the Soil Conservation Act of 1969 (Edwards, 1984). This Act makes provision for directives, such as the use of fire and the setting of limits on stocking rate, to be applied to misused land. Thus if the criterion of economic carrying capacity is applied to game reserves, then a situation where large herbivores are at or near ecological carrying capacity may be construed by agriculturalists as a misuse of land, paving the way to a directive being issued on the NPB. For instance, such a potential conflict could arise from the coordinated development of the Mfolozi catchment area. The Mfolozi catchment plan (Anon., 1985d) identified overstocking as a problem occurring throughout the key area of the catchment, and concluded that an important factor in improving land use practices was the reduction of livestock in both Natal and KwaZulu. The development of feedlots, as well as the planning and implementation of veld management schemes within the carrying capacity of the veld, was proposed to ease pressure on the grazing lands (Anon., 1985d). This is clearly an agricultural approach.

A recommendation emanating from a workshop on management strategies in H&UGR was that "international and popular articles to outline why there had been a change from an agricultural to an ecological approach in management were to be drafted" (Anon., 1985b). To date these articles have not appeared and yet they are important, not only to communicate the management approach in H&UGR but also to elicit critical scientific comment on this approach. For these reasons, the recommendation below is made:

Recommendation: Articles should be written for publication in the popular agricultural literature (Farmers Weekly) and in the popular wildlife literature (African Wildlife) which outline the conservation objectives of H&UGR, the rationale for an ecological approach to management, and the implications of such an approach. Scientific articles which outline the above points should also be written for publication in a national journal which has an agricultural bias (Journal of the Grassland Society of southern Africa), and for an International Journal (Biological Conservation or the Journal of Range Management) for the specific purpose of eliciting critical response to the ecological approach to management.

8.3 FUTURE MANAGEMENT PROGRAMME

The intention in this section is to furnish ecological principles and concepts which may assist in drawing up and implementing future management programmes. These concepts and principles are provided within the framework of the revised H&UGR primary objective which emphasises the maintenance of habitat and species diversity and the conservation of ecological processes.

8.3.1 Management of large herbivores with regard to minimising accelerated soil loss

The rationale given below is based on the assumption that the animal/resource system is capable of reaching a dynamic equilibrium between the rate of production of animals and the rate at which food resources are renewed. Coe *et al.* (1976) found a significant correlation between large herbivore biomass and rainfall from different areas in Africa, with biomass increasing as rainfall, and hence primary production, increased. This indicates that an interaction between large herbivore populations and their food supply exists, and that these populations can be regulated through the availability of resources. Caughley (1976) argues convincingly that the interaction of ungulates and vegetation typically leads to a stable equilibrium through eruption and dampened oscillations. Because such an equilibrium is the least disruptive to both the plants and the animals it is least likely to lead to extinction, and is thus in the evolutionary interest of both populations.

Whenever an ungulate population is faced with a standing crop of vegetation in excess of that needed for maintenance and replacement of the animals, an eruption and crash is the inevitable consequence. Although an eruption usually reflects the response of the population to abundant food, it sometimes reflects the relaxation of culling pressure (Caughley, 1981). It is during an eruption that displacement from an equilibrium state is extreme and, due to violent oscillations in herbivore and vegetation biomass, irreversible changes in floral composition and soil cover can occur at this stage (Caughley, 1976). Appropriate management can minimise the unwanted effects of these oscillations. This can be achieved by removing animals at an initial rate of approximately half the population's intrinsic rate of increase when animal density is well below its peak. The rate of removal is then reduced and maintained at a lower level. Such a removal programme will be adequate to flatten out most ungulate eruptions (Caughley, 1976).

To summarize, the management concern is that because of a relaxation in culling pressure in H&UGR since 1984, certain herbivore populations will erupt with a consequent reduction in vegetation biomass and hence an increase in soil loss. The violent oscillations accompanied by an eruption can be dampened by applying appropriate control measures.

Recommendation: The intrinsic rate of increase of populations of large herbivores, which a) are not currently being managed via dispersal sinks and b) whose densities have been artificially depressed through culling or game capture, should be dampened when they are in the exponential phase of population growth.

Population control should be implemented when animal densities are well below their peak, and should involve an initial removal equalling approximately half the actual annual increment. Control measures should be progressively reduced in order to dampen the violent oscillations associated with an eruption.

8.3.2 Management to minimise the threat of a reduction in genetic diversity

8.3.2.1 Fire management to minimise habitat loss through erosion

Background: The role of donga incision in lowering the landscape base level and the water table, and the effect of this on the loss of grassland habitat through woody plant invasion, together with the consequent reduction in species diversity, has been discussed in detail in Section 8.2.2. The important point is that to *reverse the trend* in woody plant encroachment, and the resulting loss of depression mesic grassland, the water table has to be restored to its previous level (Goodman, 1984). This requires the artificial reconstruction of the old landscape base level, which can be accomplished through the placement of key gabion structures in the drainage line. Headward cutting of dongas into depression mesic grasslands should also be halted by stabilising active donga heads with reno mattresses, and the raising of the water table should be facilitated by the removal of encroaching woody plants (Figure 8.3).

As part of an integrated erosion reclamation programme, the management goal should also be to maximise infiltration, and so minimise run-off, in the upper reaches of the treated catchment. Treated catchments, and those which have the potential to be treated, are extremely limited in extent and are confined to HGR and the corridor (Figure 8.4).

Infiltration of water into soil is primarily dependent on surface structure and soil hydraulic conductivity (Cass, Savage & Wallis, 1984). Surface structure is in turn sensitive to changes in the litter layer, vegetation cover and faunal activity. Kelly & Walker (1976) found that infiltration into soil completely covered by litter was up to nine times faster than into bare soil. In the current study a highly significant inverse relationship between mean run-off and litter cover was also established (Figure 8.5), and, through its destructive effect on litter, fire has been shown to reduce water absorption and infiltration rate (Cass *et al.*, 1984). Since back fires are generally more intense than head fires at ground level (Trollope, 1984) they should be avoided, and a low-intensity fire, which causes the least damage to the grass sward, should be used where practicable.

Since the amount of plant material incorporated directly into litter depends not only on fire but also on herbivory (Frost *et al.*, 1986), I suggest that in catchments where an integrated erosion reclamation programme has been undertaken to reclaim hydrologically maintained grasslands, the litter cover is monitored. The reason for monitoring litter cover is because there appears to be a threshold value of approximately 10% below which run-off increases markedly (Figure 8.5). The management concern is that as large herbivores reach ecological carrying capacity in these catchments, excessive run-off may result from a reduction in vegetation biomass and consequently litter cover. This may in turn lead to a new cycle of donga incision and a further loss of depression mesic grassland. Because it is being assumed that these grasslands may be threatened when large herbivores utilising these catchments are at ecological carrying capacity, I recommend that an adaptive management approach be adopted whereby rainfall, vegetation and herbivory are monitored to determine what large herbivore population density, if any, is unacceptable in terms of the goal of maximising infiltration in treated catchments.

It is clear from this discussion that a management and monitoring input will be necessary to achieve the management goal of maximising infiltration in these treated catchments. The relevant questions from a management point of view are firstly, whether such high management input is warranted and, secondly, whether it is practicable.

Dealing with the first point, over the period 1937 to 1975 tree and shrub cover increased by an average of 24,3% in areas sampled in HGR, and this trend appears representative of the entire HGR (Watson & Macdonald, 1983). This increase was at the expense of grassland, where fire has been used as a management tool to control woody plant invasion into the more open types of savanna vegetation (Ward, 1962; Porter, 1977; Brooks & Macdonald, 1983; Wills, 1984). Such a unidirectional successional change in vegetation conflicts with the primary objective of maintaining habitat diversity. Since fire itself has not been able to reverse this trend to date other ecological processes, such as the influence of precolonial man and the effect of elephants *Loxodonta africana*, have been simulated and reinstated respectively to maintain habitat diversity (Wills & Knott, 1985). Some mesic grasslands are maintained as grasslands by hydrological processes and not by fire, and these grasslands are recognised as threatened habitats in H&UGR and classified as biologically sensitive sites (Macdonald, 1982a). Given that the NPB conservation management policy is to identify any sensitive features in order to preserve them, and give them special attention where necessary (Anon., 1985c), high management input into these depression mesic grasslands is compatible with both this policy and the primary objective of the reserves.

The second point, whether such high management input is practicable, is largely beyond the scope of this chapter and is essentially a technical decision that should be taken by relevant NPB staff. The question as to whether it is possible to manage on the spatial scale of a treated catchment must be resolved.

Recommendation: An integrated management approach should be adopted to prevent the loss of hydrologically maintained grassland habitat. This should include the stabilisation of eroding donga heads using reno mattresses, and wherever possible the base level of the local landscape should be reconstructed using key gabion structures in the drainage line to raise the water table to its original base level (Figure 8.3). Invading woody plants should be removed to help in raising the water table, and fire management should be aimed at maximising infiltration into the treated catchments by ensuring that litter cover is not reduced below the critical level of 10%.

To achieve the management goal of maintaining litter cover above 10%, the upper reaches of these catchments, in the immediate vicinity of the erosion reclamation structures, should be burnt with a low intensity head fire. This can be done by burning when the air temperature is less than 20°C and relative humidity greater than 50%. These conditions frequently prevail before 11h00 and after 15h30 (Trollope, 1984).

An adaptive management approach should be adopted whereby rainfall, herbivory, litter cover and other relevant vegetation variables are monitored to determine what large herbivore density, if any, conflicts with the goal of maximising infiltration and minimising run-off in the treated catchments (see Section 8.4.1.2).

8.3.2.2 Management of large herbivores to minimise the threat of a reduction in genetic diversity

Background: Experience gained during the period stretching from 1977 to 1983, which was the longest and most severe drought period recorded in UGR since rainfall data collection was started in 1959, showed that the UGR system was resilient in that it apparently returned to its original state after the drought. Since the crossing of a threshold from one state to another may be accompanied by species extinctions (Walker & Goodman, 1983), a level of disturbance (like excessive herbivory) which causes such a change in the system could conflict with the primary objective of maintaining species diversity. Emslie (1985) documented the "recovery" of the study area after the recent drought period, and showed that two key grass species (*Themeda triandra* and *Panicum coloratum*) were able to recover from this severe drought and the high ungulate biomass, even after tuft mortality levels of around 90% were reached. In addition, no species extinctions of grasses (Emslie, 1985), rodents (Bowland, 1985), birds (Macdonald, 1982b), ungulates (Knott & Brooks, 1986), or carnivores (Whateley & Brooks, 1985) were noted in the study area as a result of this drought, and thus there did not appear to be any conflict with the primary objective of maintaining species diversity.

A recent review of the ecological consequences of a severe drought on four wildlife conservation areas in southern Africa (which included UGR), concluded that there was no evidence indicating that any of the systems would have undergone enduring change in the absence of large herbivore culling (Walker *et al.*, 1987). A decrease in large mammal species diversity was however noted in one of the areas, viz. Tuli, where sable *Hippotragus niger* and roan antelope *Hippotragus equinus* have disappeared in the last 30 years (Walker *et al.*, 1987), and such a decrease in species diversity would have conflicted with the primary objective for H&UGR.

The highest soil loss measurements were made in the non-cull block at the height of the drought in 1983, where the mean soil loss from rainfall simulator plots was equivalent to 2,2 t/ha, with 95% confidence limits giving a range of 1,25 to 3,15 t/ha. Despite this soil loss, the system apparently returned to its original state in the following above-average rainfall years. This indicates that the system can tolerate, in the short term, at least this level of soil loss without a loss in species diversity. However, there are populations of large herbivores in H&UGR which may be unable to stabilise at ecological carrying capacity because of the absence of one or more population regulation mechanisms, or which may stabilise at ecological carrying capacity, but at this density may depress the densities of other rarer species (Knott, 1985). I suggest that their populations should be regulated via managed removals if the long-term, predicted soil loss rate (see Section 8.4.1.1) should *exceed* 2,2 t/ha for more than one year *and* if it can be demonstrated that large herbivores are responsible for the reduction in vegetation cover. The animal populations in question are impala *Aepyceros melampus*, nyala *Tragelaphus angasi*, blue wildebeest *Connochaetes taurinus*, and square-lipped rhinoceros *Ceratotherium simum*.

The rationale here is that intervention management is warranted for those herbivore populations which may depress the densities of other rarer species when they themselves are at ecological carrying capacity (i.e. ecologically dominant species) and/or for those populations which are unable to stabilise at ecological carrying capacity because one or more of their population regulation mechanisms are absent. However, such intervention management should be implemented only *if* their populations increase to such an extent that, through their effect on vegetation cover, the predicted soil loss based on a two-year running mean exceeds that measured in the non-cull block at the height of the drought. The reasoning behind intervention management at this stage is that, without it, the system (or parts of the system) may be too highly stressed and may lose species diversity and its potential to recover. This is essentially a conservative strategy, since the system may be able to tolerate far more change than was observed in the non-cull block during the driest year on record, but until it can be shown that management can simulate adequately the missing population regulation mechanisms such intervention management appears to be prudent. An adaptive management approach is envisaged here where intervention management is undertaken experimentally in certain areas, but not in control areas. The effect of these two treatments should be monitored to detect any reduction in species diversity.

To summarise, because some population regulation mechanisms may not be simulated adequately by management, there is the potential for certain populations of large herbivores which are unable to stabilise at ecological carrying capacity and/or which are ecologically dominant, to erupt. If such an eruption causes a reduction in vegetation cover and soil loss in excess of that measured in the non-cull block in 1983, when herbivore biomass was high and rainfall extremely low, then intervention management is warranted on an experimental scale. The rationale for this intervention management is that the system (or parts of it) may not be able to tolerate such high soil loss rates in the longer term and there may be a loss in species diversity.

Recommendation: If the mean predicted soil loss for any reserve section exceeds 2,2 t/ha for two consecutive years *and* it can be demonstrated that large herbivores are primarily responsible for the reduction in vegetation cover, then removals of one or more of impala, nyala, blue wildebeest or square-lipped rhinoceros should be implemented to reduce the rate of soil loss. These removals should be implemented on an experimental basis, with control areas where no population reductions are undertaken, and should cease once predicted soil loss rates drop below 2,2 t/ha. The predictive equation is given in Section 8.4.1.1, and the data for this equation should be collected annually during the range condition evaluation.

8.3.3 Management to minimise accelerated soil erosion caused by road and track construction

Background: Since erosion resulting from road and track construction is clearly accelerated erosion caused by the direct activities of recent man, and because it has a disproportionate impact on soil loss (Section 8.2.2), the reclamation effort should be aimed at minimising such erosion. Accelerated erosion from roads and tracks can be minimised by their proper alignment and drainage, using under-road culverts, mitre drains, crossbanks or outsloping, and the closure and reclamation of nonessential management tracks and service roads.

A basic principle in minimising erosion from proposed new roads and tracks is that they should be properly sited on ridge crests wherever possible, as this disposes of a major problem in road maintenance, viz. drainage (Hudson, 1981). Where it is not possible to put a road on a crest, the next best alignment is on a gentle grade, close to the true contour (Hudson, 1981). Road traffic, especially on wet roads, can cause surface rutting which concentrates the flow of water along the road. Here cross-drainage measures are needed to interrupt this flow and divert it laterally before it has a chance to concentrate and cause erosion (Megahan, 1977). Cross-drainage can be done in three ways: by mitre drains, which are extensions of road drains leading away from the road on the contour at an angle of about 45° (Hudson, 1981); by crossbanks which are humps across the road (Marshall, 1982); or by outsloping which is the sloping of the road camber towards the downhill side of the road (Megahan, 1977). Crossing of natural drainage lines is best effected by using either fords where the stream channel bottom is stable and able to support vehicles, or under-road culverts where fords are impractical (Megahan, 1977).

Another basic principle in minimising the erosional impacts of roads is to control their total mileage and reduce the area of disturbance on those roads that are built (Megahan, 1977). Road mileage can be controlled by closing down nonessential tracks and service roads, and the area of disturbance on them can be reduced by grading only that part of the road which is used by traffic and encouraging the revegetation of graded road drains.

The above principles will help in minimising accelerated soil loss, but the siting and construction of road drains needs considerable technical expertise which is beyond the scope of this chapter and should be covered in the proposed technical manual.

Recommendation: A firm of consulting roads engineers should be contracted to plan the drainage and, if necessary, re-site main access roads which have high vehicle usage. The planning and implementation stages should be carried out in close liaison with the Roads Maintenance Unit and relevant management staff, so that the necessary expertise can be gained by NPB personnel.

Once NPB staff are familiar with the principles and techniques of road siting and drainage, the remainder of the roads and management tracks in H&UGR should be surveyed and recommendations made to minimise accelerated soil loss from roads, tracks and mitre drains.

Finally, included in the contract for the roads engineers should be the task of editing, and if necessary updating, a technical manual on road planning, siting, construction and drainage (see Section 8.2.2a).

8.4 FUTURE MONITORING PROGRAMMES AND RESEARCH PROJECTS

8.4.1 Future monitoring programmes

The specific monitoring actions detailed below should be seen in the context of a broader vegetation monitoring programme which is currently being designed in H&UGR (A.J. Wills - *pers. comm.*). Emphasis will be placed on separating the effects of climate, large herbivores and microherbivores on vegetation. Monitoring will thus be aimed not only at the detection of change, but also at determining which factors contribute to this change.

8.4.1.1 Monitoring of soil loss

Background: One of the aims of this study was to derive and validate predictive models for soil loss based on vegetation and soil surface variables, so that soil loss could be predicted from vegetation monitoring data obtained during the annual range condition evaluation, and long-term trends in soil loss could be established. Since the emphasis is on long-term trends, rather than on absolute estimates of soil loss, predictive equations derived from the rainfall simulator trials are the most appropriate. As rainfall amount and intensity are controlled in rainfall simulator trials, between-year comparisons are possible and, because 95% confidence limits can be calculated, it is also possible to show whether there is a significant increase or decrease in predicted mean soil loss with time. The trend in soil loss must not however be interpreted independently of annual rainfall patterns, since there is a significant correlation between the two-year running mean of rainfall and both herbaceous canopy cover and mean grass height. These two variables were in turn significantly correlated with mean soil loss (Section 7.4.2). The point here is that in below-average rainfall years, herbaceous primary production will be less, and hence vegetation variables such as mean grass height and herbaceous canopy cover will also have lower values. This will result in higher predicted values for mean soil loss which should not necessarily be of concern, given the reduced annual primary production, and therefore the inevitability of an increase in predicted soil loss values.

Recommendation: The measurement of surface cover and mean grass height, using Walker's (1976) method, should be included in the annual range condition evaluation and in any future objective vegetation monitoring programme. Using these variables, the soil loss can be predicted annually and the long-term trend in soil loss can be monitored. In these monitoring programmes emphasis should be placed on separating the effects of large herbivores, microherbivores, fire and climate on vegetation, and hence on soil loss.

Surface cover is defined as the percentage of the total surface area (of a 1m^2 quadrat) that cannot be hit by vertically falling raindrops because of the herbaceous canopy cover and any ground cover. Ground cover can consist of litter, rock or dead branches in contact with the surface of the ground. Mean grass height is the average height above ground level of the grass occurring within each quadrat, and is measured to the nearest centimetre.

Based on these two variables, the mean soil loss and 95% confidence limits should be calculated separately for the five reserve sections recognised by Wills (1985using the following predictive equations:

$$\log_e \text{soil loss (g)} = 9.0025 - 0.0319 \text{ surface cover} - 0.0238 \text{ mean grass height.}$$

The upper 95% confidence limit should be calculated as follows:

$$\log_e \text{soil loss (g)} = 9.0025 - 0.02372 \text{ surface cover} - 0.01257 \text{ mean grass height.}$$

Lastly, the lower 95% confidence limit should be calculated using the equation below:

$$\log_e \text{soil loss (g)} = 9.0025 - 0.04006 \text{ surface cover} - 0.03494 \text{ mean grass height.}$$

These natural logarithms should then be converted to their corresponding numbers and expressed as soil loss in t/ha using the following formula:

$$\text{soil loss (t/ha)} = \frac{\text{soil loss (g)}}{1920.6}$$

The trend in soil loss should not be interpreted independently of annual rainfall patterns. The incorporation of these data into the decision-making process regarding managed removals of large herbivores has been covered in Section 8.3.2.2.

8.4.1.2 Monitoring of run-off

Background: As a first priority, run-off should be monitored in those catchments where an integrated erosion reclamation programme has been undertaken with the aim of reinstating hydrological processes. The rationale for such a programme, and the decision-making process regarding fire management and large herbivore population control, have been covered in Section 8.3.2.1. The effect on long term run-off rates of allowing large herbivore populations to reach ecological carrying capacity should be monitored in selected catchments, using a predictive model for run-off. Because the emphasis is on long-term trends, predictive equations derived from the rainfall simulator data are again the most appropriate here, as the standardised conditions of rainfall intensity and amount allow for between-year comparisons. Monitoring should be aimed at separating the relative effects of climate, fire, large herbivores and microherbivores on vegetation, and thus on run-off.

Recommendation: Assessment sites which include a topographical sequence from valley bottom to upland situations should be established in selected catchments following the experimental design of Wills (1985). At each site the following variables should be measured, using Walker's (1976) method: soil capping, litter cover and surface cover.

Soil capping is defined as the percentage of the total surface area of the $1m^2$ quadrat which is capped, and litter cover is the percentage of dead, nonwoody plant material in contact with the soil surface. Surface cover has been defined in Section 8.4.1.1.

Based on these three variables, the mean percentage run-off should be calculated for each treated catchment using the following equation:

run-off (%) = 5.5126 + 0.9777 soil capping + 0.5852 surface cover − 0.4692 litter cover.

The upper 95% confidence limit is calculated as follows:

run-off (%) = 5.5126 + 1.36081 soil capping + 1.07934 surface cover − 0.21575 litter cover.

The lower 95% confidence limit is calculated using the equation below:

run-off (%) = 5.5126 + 0.51452 soil capping + 0.09105 surface cover − 0.72273 litter cover.

The predicted percentage run-off and its 95% confidence limits should be plotted each year to detect a long-term trend in predicted run-off. The relative effects of climate, fire and herbivory on run-off rates should be assessed to determine which factor/s contribute to increased run-off rates, so that the appropriate management action can be taken if necessary (see Section 8.3.2.1).

8.4.2 Future research projects

As a result of this study, the need for two further research projects has been identified.

8.4.2.1 The influence of soil moisture balance on the spatial distribution of grasslands in H&UGR

Background: Tinley (1982) contends that the most important factor determining the spatial distribution of forest, savanna and grassland is soil moisture balance, which is influenced by the presence or absence of a pan horizon (i.e. an impermeable subsurface horizon in the soil) and its depth within the soil profile, and by soil permeability to rain. An excess of soil moisture on a perennial or seasonal basis is the major factor determining the presence of open grassland (Tinley, 1982), yet current management is aimed at maintaining open grassland primarily through burning and, except for Macdonald (1979), no research work has been directed at the influence of soil moisture balance on the spatial distribution of grasslands in H&UGR.

The presence of a clay pan horizon is very clear in vlei grasslands, which have the Mdoni tree *Syzygium cordatum* as an important component (J. Venter - *pers. obs.*). These occur in restricted areas in HGR (Macdonald, 1979). Depression mesic grassland and grassland on ridge crests are also thought to be maintained as grassland through impeded drainage (Macdonald, 1982a). Since grasslands are of such limited extent in H&UGR (Watson & Macdonald, 1983), and because soil moisture balance, rather than fire, may be an important determinant of the spatial distribution of grassland in H&UGR, the research project below is proposed.

Recommendation: A research project should be initiated to examine the influence of soil moisture balance on the spatial distribution and dynamics of grasslands in H&UGR.

The aims of this project should be:

a) to determine the current and past extent of grasslands in H&UGR and in adjacent KwaZulu;

b) to identify the determinants of grassland structure and functioning;

c) to determine the relative importance of soil moisture balance, herbivory and fire in influencing the structure and controlling the functioning of grassland; and

d) to make management recommendations regarding the future management of grasslands in H&UGR to ensure their continued existence and, if possible, their enlargement.

8.4.2.2 The identification of geomorphological processes functioning in H&UGR, the determination of a sediment budget and the implications for reserve management

Background: A recurrent theme in these management recommendations has been a need to separate the relative effects of fire, climate and herbivory on vegetation cover and hence on soil loss. An extensive literature review indicated that in temperate areas herbivore stocking rate affected sediment yield from grazed catchments (see Section 7.3.2), yet in this study such a relationship was not evident. Semi-arid savannas are characterised by large fluctuations in annual rainfall, and data from this study show that it is rainfall, rather than stocking rate, which is the major factor influencing vegetation cover and hence soil loss. Because managers are incapable of managing rainfall, they should not be trapped into manipulating those components of the system which they have the ability to manage, if these components do not contribute significantly to soil loss. The question that needs to be addressed is: "Do the components of the system which can be managed make a significant contribution to the sediment budget of the system?" If they do, there is justification in manipulating them, but if they do not contribute significantly to soil loss then there is no point in manipulating them.

Seen in a broader perspective, there is the need to identify the important geomorphological processes operational in the system. Soil erosion must be seen in its geomorphological context, and there is evidence that in Mkuzi Game Reserve erosion reclamation effort is being expended on arresting the natural geomorphological process of rejuvenation of a landscape (P.S. Goodman - *pers. comm.*). The revised primary objective clearly states that ecological processes should be allowed to operate without interference unless certain conditions are encountered. To implement this objective, geomorphological processes need to be identified and their relevance to reserve management determined.

Recommendation: A research project should be initiated with the following aims:

a) to determine the relative contributions that fire, rainfall, microherbivores and large herbivores make to the sediment budget of H&UGR, primarily through their influence on vegetation cover;

b) to identify those important geomorphological processes operational in H&UGR and to elucidate their relevance to reserve management; and

c) to make management recommendations based on a) and b) above regarding ungulate population control, fire management and erosion reclamation control.

8.5 CHAPTER SUMMARY

a) As it is impossible to measure progress towards the goal of balancing soil loss with soil genesis in the long-term, and hence establish whether any management action is succeeding in achieving this goal, the recommendation is that the soils policy should be changed to the following: "to minimise the rate of accelerated erosion and thereby its effect on populations of species which can sustain themselves within the reserves' boundaries and which would be threatened with extinction in the absence of the reserves."

b) Another recommendation is that the soil reclamation objective should be slightly modified to follow the format of the primary objective so that the conditions that justify management intervention can be clearly stated. The modified soil reclamation objective is: "soil erosion is an ecological process which should be allowed to operate without interference unless:

i) the rate of erosion has been accelerated by the activities of recent man;

ii) accelerated soil erosion, particularly donga incision, threatens rare habitat types; and

iii) accelerated soil erosion jeopardises the future of the reserves because of its socio-political consequences."

c) To minimise the rate of accelerated erosion, a practical management goal should be: "to identify areas of active erosion resulting from road and track construction and if possible to prevent, or at least minimise, such erosion by the correct alignment and drainage of roads and tracks; and the closure and reclamation of nonessential management tracks and service roads." To help in achieving the above goal, a technical manual detailing the principles involved in road planning, siting, construction and drainage should be drawn up for use in NPB conservation areas.

d) To prevent soil erosion from threatening rare habitat types, erosion reclamation effort should be directed at identifying those rare habitat types threatened by accelerated soil erosion, and particularly donga incision into hydrologically maintained mesic grasslands. Once identified, eroding donga heads should be stabilised and, if possible, attempts should be made to reinstate hydrological processes by reconstructing the original base level of the local landscape and by the removal of invading woody plant species.

e) To prevent the future of the reserves being jeopardised because of different perceptions of reserve management objectives, articles which outline the conservation objectives of H&UGR, the rationale for an ecological approach to management, and the implications of such an approach, should be published in the popular and scientific literature. The specific purpose of these articles should be to elicit critical comments on the ecological approach to management.

f) Since the potential for soil loss is highest when large herbivore populations erupt due to violent oscillations in vegetation and herbivore biomass, these oscillations should be dampened by applying appropriate control measures. The intrinsic rate of increase of certain large herbivore populations should be dampened when they are in the exponential phase of population growth. These control measures should be implemented when animal densities are well below their peak and should be reduced progressively.

g) An integrated management approach should be adopted to prevent the loss of hydrologically maintained grassland. Apart from the management actions listed in d) above, fire management should be aimed at maximising infiltration into the treated catchment by preventing litter cover from falling below the critical level of 10%. An adaptive management approach should be followed where rainfall, herbivory and vegetation are monitored to determine what large herbivore density, if any, conflicts with the goal of maximising infiltration in the treated catchments.

h) Future monitoring of soil loss should be implemented via the annual range condition evaluation and should include the measurement of surface cover and mean grass height, using Walker's (1976) method. By means of predictive models based on these two variables, the mean soil loss and 95% confidence limits can be calculated for reserve sections, and the trend in soil loss monitored over time. This trend should however be interpreted in conjunction with annual rainfall data because of the influence of rainfall on primary production, and hence on surface cover and mean grass height.

i) If the predicted mean soil loss for any reserve section exceeds 2,2 t/ha for two consecutive years, and if it can be demonstrated that large herbivores are responsible for the reduction in vegetation cover, then removals of certain species should be implemented to reduce the rate of soil loss. These removals should be implemented on an experimental basis, with control areas where no population reductions are undertaken and should cease once the predicted soil loss rates drop below 2,2 t/ha.

j) As a first priority, future monitoring of run-off should be implemented in treated catchments, and should include the measurement of soil capping, litter cover and surface cover, using Walker's (1976) method. Based on these three variables, mean percentage run-off and 95% confidence limits can be calculated for selected catchments, using predictive equations. Monitoring should be aimed at separating the relative effects of climate, fire and herbivory on vegetation, and hence on run-off.

k) As grasslands are limited in extent in H&UGR, and since fire management has not been successful to date in maintaining grasslands, a research project should be initiated to examine the influence of soil moisture balance on the spatial distribution and dynamics of grasslands in H&UGR. The aims of this project should include the determination of the current and past extent of grasslands; the identification of determinants of grassland structure and functioning and their relative importance; and the drawing up of management recommendations regarding the future management of grasslands.

1) A second research project should be undertaken to determine which abiotic and biotic components of the system contribute the most to the sediment budget of H&UGR. It should also identify those important geomorphological processes operational in the reserves and elucidate their relevance to reserve management.

REFERENCES

AITKEN, A.P. 1973. Assessing systematic errors in rainfall-runoff models. *J.Hydrol.* 20:121-136

ALLEN, R. 1980. How to save the world: Strategy for world conservation. London: Korgan Page Ltd. 150pp.

ALLEN, R. 1982. Goals of ecosystem conservation. *In*: *Conservation of ecosystems: Theory and practice*. *eds* W.R. Siegfried & B.R. Davies. South African National Scientific Programmes Report no: 61. Pretoria: CSIR, Cooperative Scientific Programmes. 97pp.

ANON. 1985a. Hluhluwe and Umfolozi Game Reserves combined management plan. Part I: Information. Unpubl. Natal Parks Board document. 107pp.

ANON. 1985b. Workshop on management strategies in Hluhluwe and Umfolozi Game Reserves held at Masinda Lodge, U.G.R. Unpubl. Natal Parks Board Report. 21pp.

ANON. 1985c. Hluhluwe and Umfolozi Game Reserves combined management plan. Part II: Objectives, Zonation, Management Programme. Unpubl. Natal Parks Board document. 182pp.

ANON. 1985d. Mfolozi catchment plan: Pilot project area resource evaluation. Report of the Mfolozi catchment planning committee. Pietermaritzburg: Natal Town and Regional Planning Commission. 32pp.

ANON. 1987. Wildtuin se riviere - "'n Bedreigde spesie." *S.A. Waterbulletin* 13(3):16-19.

BARLOW, T. 1979. The unfinished soil conservation task. Conservation Viewpoint Editorial. *J. Soil & Water Cons.* 254-255.

BARNES, D.L. & Franklin, M.J. 1970. Runoff and soil loss on a sandveld in Rhodesia. *Proc. Grassld. Soc. sth. Afr.* 5:140-144.

BARNES, D.L., Odendaal, J.J. & Beukes, B.H. 1982. Use of the dry-weight-rank method of botanical analysis in the eastern Transvaal highveld. *Proc. Grassld. Soc. sth. Afr.* 17:79-82.

BEARDALL, G.M., Joubert, S.C.J. & Retief, P.F. 1984. An evaluation of the use of correspondence analysis for the analysis of herbivore-habitat selection. *S. Afr. J. Wildl. Res.* 14(3):79-88.

BELL, R.H.V. 1981. An outline of a management plan for Kasungu National Park, Malawi. *In: Problems in management of locally abundant wild mammals. eds* P.A. Jewell & S. Holt. New York: Academic Press. 361pp.

BLACKBURN, W.H., Knight, R.W. & Wood, M.K. 1982. *Grazing on watersheds: a state of knowledge.* Texas Agricultural Experiment Station report no: MP 1496. Texas: Texas A&M University. 32pp.

BLAIR-RAINS, A.R. 1981. Erosion risk assessment and land classification for conservation design. *In: Soil Conservation : problems and prospects. ed.* R.P.C. Morgan. Chichester: John Wiley. 576pp.

BLANDFORD, D.C. 1981. Rangelands and soil erosion research : a question of scale. *In: Soil Conservation : problems and prospects. ed.* R.P.C. Morgan. Chichester : John Wiley. 576pp.

BOTKIN, D.B. 1982. Ecological characteristics of ecosystems. *In: Conservation of ecosystems: Theory and practice. eds* W.R. Siegfried & B.R. Davies. South African National Scientific Programmes Report no: 61. Pretoria: CSIR, Cooperative Scientific Programmes. 97pp.

BOTKIN, D.B. 1984. The garden of the unicorn: The ecosystem context for the management of endangered species. *In: Conservation of threatened natural habitats. ed.* A.V.Hall. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

BOURQUIN, O., Vincent, J. & Hitchins, P.M. 1971. The vertebrates of Hluhluwe Game Reserve - Corridor (State land) - Umfolozi Game Reserve Complex. *Lammergeyer* 29:1-56.

BOWLAND, A.E. 1985. The effect of fire, overgrazing and drought on the abundance and diversity of small mammals in the Central Complex , and the indirect impact on predators. M.Sc. thesis. Pietermaritzburg: University of Natal, Dept. of Zoology. 141pp.

BRANSON, F.A., Gifford, G.F., Renard, K.G. & Hadley, R.F. 1981. Rangeland Hydrology. Toronto: Kendall/Hunt Publishing Co. 339pp.

BROOKS, P.M. 1983. Should the non-cull area in Umfolozi Game Reserve be retained to provide management information: A perspective. Unpubl. Natal Parks Board Report. 6pp.

BROOKS, P.M., Macdonald, I.A.W. & Whateley, A. 1980. Assessment of the state of the gravel roads in Hluhluwe Game Reserve and northern corridor in April 1980, with suggestions for an improved approach to roads planning throughout Natal's reserves. Unpubl. Natal Parks Board Report. 15pp.

BROOKS, P.M. & Macdonald, I.A.W. 1982. An ecological case history of the Hluhluwe-Corridor-Umfolozi Game Reserve Complex, Natal, South Africa. First draft. Unpubl. Natal Parks Board Report. 74pp.

BROOKS, P.M. & Macdonald, I.A.W. 1983. The Hluhluwe-Umfolozi Reserve : an ecological case history. *In*: *Management of large mammals in African conservation areas. ed.* R.N. Owen-Smith. Pretoria : Haum Educational. 297pp.

BROWN, L.R. 1981. World population growth, soil erosion and food security. *Science* 214(4524):995-1001.

CARTER, L.J. 1977. Soil erosion : the problem persists despite the billions spent on it. *Science* 196(4288):409-411.

CASS, A., Savage, M.J. & Wallis, F.M. 1984. The effect of fire on soil and microclimate. *In*: *Ecological effects of fire in South African Ecosystems. eds* P. de V. Booysen & N.M. Tainton. Berlin: Springer-Verlag. 426pp.

CAUGHLEY, G. 1976. Wildlife management and the dynamics of ungulate populations. *In*: *Applied Biology*. Volume I. *ed.* T.H. Coaker. New York: Academic Press. 358pp.

CAUGHLEY, G. 1981. Overpopulation. *In*: *Problems in management of locally abundant wild mammals*. *eds* P.A. Jewell & S. Holt. New York: Academic Press. 361pp.

CAUGHLEY, G. & Walker, B.H. 1983. Working with ecological ideas: Ecological concepts. *In*: *Guidelines for the management of large mammals in African conservation areas*. *ed.* A.A. Ferrar. South African National Scientific Programmes. Report no: 69. Pretoria: CSIR, Cooperative Scientific Programmes. 95pp.

COE, M.J., Cumming, D.H. & Phillipson, J. 1976. Biomass and production of large African herbivores in relation to rainfall and primary production. *Oecologia* 22:341-354.

COSTIN, A.B. & Gilmour, D.A. 1970. Portable rainfall simulator and plot unit for use in field studies of infiltration, runoff and erosion. *J. Appl. Ecol.* 7:193-200.

CROUSE, R.P., Corbett, E.S. & Seegrist, D.W. 1966. Methods of measuring and analysing rainfall interception by grass. *Int.Ass.Sci.Hydrol.Bull.* 11:110-120.

CURRIE, P.O. 1975. Grazing management of ponderosa pine-bunchgrass ranges of the central Rocky Mountains: The status of our knowledge. U.S. Forest Service. Research paper no: RM-159. 24pp.

DADKHAN, M. & Gifford, G.F. 1980. Influence of vegetation, rock cover and trampling on infiltration rates and sediment production. *Water Resources Bulletin* 16(6):979-986.

DENSHAM, W.D. & Wills, A.J. 1984. Minutes of animal population control meeting and a workshop on an expert system to aid in the formulation of animal removal strategies. Unpubl. Natal Parks Board Report. 28pp.

DENT, M.C. & Schulze, R.E. 1987. The hydrological data manager and digitization in 1985 : points to ponder in the development of a new digitizing system. *Water SA* 13(1):49-52.

DIAMOND, J.M. 1982. Implications of island biogeography for ecosystem conservation. *In*: *Conservation of ecosystems: Theory and practice. eds* W.R. Siegfried & B.R. Davies. South African National Scientific Programmes Report no: 61. Pretoria: CSIR, Cooperative Scientific Programmes. 97pp.

DIAMOND, J.M. 1984. Management for maintenance of species diversity. *In*: *Conservation of threatened natural habitats. ed.* A.V.Hall. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

DISSMEYER, G.E. 1982. Developing a USLE cover-management (c) factor procedure for forest conditions. *In*: *Proceedings of the workshop on estimating erosion and sediment yield on rangelands*. U.S. Dept. of Agriculture, Agricultural Research Service report no: ARM-W-26. 228pp.

DOWNING, B.H. 1972. A plant ecological survey of the Umfolozi Game Reserve, Zululand. Ph.D. thesis. Pietermaritzburg: Univ. of Natal, Botany Dept. 179pp.

DOWNING, B.H. 1980. Relationship between rock substrate, landform and soil in Umfolozi Game Reserve, as explained for conservation purposes. *Lammergeyer* 30:32-48.

DUNNE, T. 1979. Sediment yield and land use in tropical catchments. *J.Hydrol.* 42:281-300.

DUNNE, T., Dietrich, W.E. & Brunengo, N.J. 1978. Recent and past erosion rates in semi-arid Kenya. *Z. Geomorph. N.F.* 29:130-140.

DUNNE, T., Dietrich, W.E. & Brunengo, N.J. 1980. Simple, portable equipment for erosion experiments under artificial rainfall. *J. Agric. Eng. Res.* 25:161-168.

DU PLESSIS, M.C.F. & Mostert, J.W.C. 1965. Afloop en grondverliese by die landbounavorsingsinstituut Glen. *S. Afr. Tydskr. Landbouwet.* 8:1051-1060.

DURNFORD, E.G. 1954. Surface runoff and erosion from Pine grasslands on the Colorado front range. *J. For.* 52:923-927

EDWARDS, P.J. 1984. The use of fire as a management tool. *In*: *Ecological effects of fire in South African Ecosystems.* *eds* P. de V. Booysen & N.M. Tainton. Berlin: Springer-Verlag. 426pp.

EL-SWAIFY, S.A. 1981. Controlling erosion problems. Propelling our way out of the merry-go-round. *In*: *Soil Conservation : problems and prospects. ed.* R.P.C. Morgan. Chichester: John Wiley. 576pp.

ELTRINGHAM, S.K. 1979. The ecology and conservation of large African mammals. London: Macmillan Press Ltd. 286pp.

EMSLIE, R.H. 1985. A critical assessment of past wildlife management and a rationale for the proposed ecological management approach. *In*: *Background documents for workshop on ecologically based management in HGR-UGR.* Unpubl. Natal Parks Board Report. 38pp.

FEELY, J.M. 1980. Did Iron-Age man have a role in the history of Zululand's wilderness landscapes? *S.A. J. Sci.* 76:150-152.

FERRAR, A.A. *ed.* 1983. Guidelines for the management of large mammals in African conservation areas. South African National Scientific Programmes. Report no: 69. Pretoria: CSIR, Cooperative Scientific Programmes. 95pp.

FREDRICKSEN, R.L. 1970. Erosion and sedimentation following road construction and timber harvest on unstable soils in three small western Oregon watersheds. U.S.Dept. of Agriculture, Forest Service Res.Pap. PNW-104. Portland: Pacific Northwest Forest and Range Exp.Stn. 15pp.

FREUND, R.J. & Minton, P.D. 1979. Regression methods : A tool for data analysis. New York: Dekker. 261pp.

FROST, P., Medina, E., Menaut, J-C., Solbrig, O., Swift, M. & Walker, B.H. 1986. Responses of savannas to stress and disturbance: A proposal for a collaborative programme of research. Biology International Special Issue-10: 1-82.

FULLER, M.F. & Lury, D.A. 1977. Statistics workbook for social science students. Oxford: Philip Allan Publ. Ltd. 250pp.

GAUCH, H.G. 1982. Multivariate analysis in community ecology. Cambridge: Cambridge University Press. 298pp.

GEBHARDT, K.A. 1982. Use of erosion models on western rangelands. *In*: *Proceedings of the workshop on estimating erosion and sediment yield on rangelands.* U.S. Dept. of Agriculture, Agricultural Research Service report no: ARM-W-26. 228pp.

GIFFORD, G.F. & Hawkins, R.H. 1978. Hydrological impact of grazing on infiltration : a critical review. *Water Resourc.Res.* 14(2):305-313.

GIFFORD, G.F. & Hawkins, R.H. 1979. Deterministic hydrologic modelling of grazing system impacts on infiltration rates. *Water Resources Bulletin* 15(4):924-934.

GIFFORD, G.F. & Springer, E.P. 1980. A selected annotated bibliography on grazing hydrology. Research report no: 50. Logan: Utah Agricultural Experiment Station. 33pp.

GIFFORD, G.F. & Whitehead, J.M. 1982. Soil erosion effects on productivity in rangeland environments: where is the research? *J. Range Manage.* 35(6):801-802.

GLYMPH, L.M. 1975. Evolving emphasis in sediment-yield predictions in present and prospective technology for predicting sediment yields and sources. U.S.Dept.of Agriculture, Agricultural Research Service report no: ARS-S-40:1-9.

GOODMAN, P.S. 1984. Soil erosion in ecological perspective. *In*: *Notes on a training course on soil erosion and reclamation*. Unpubl. Natal Parks Board report. 125pp.

GOODMAN, P.S. 1987. *Pers. comm.* Mkuzi Game Reserve, P.O. Mkuze, 3965, South Africa.

GOTFRYD, A. & Hansell, R.I.C. 1985. The impact of observer bias on multivariate analyses of vegetation structure. *Oikos* 45:223-234.

GREEN, R.H. 1979. Sampling design and statistical methods for environmental biologists. New York: Wiley. 257pp.

GREENACRE, M.J. 1984. Theory and applications of correspondence analysis. London: Academic press. 364pp.

GROBLER, J.H. 1983. Management plan format and conservation policy. Unpubl. Natal Parks Board report. 25pp.

GROBLER, J.H. 1984. Management planning : the Natal Parks Board (R.S.A.) approach. *Parks* 9(1):13-15.

GROSSMAN, D. 1982. Primary production of rangeland : practical and interpretive problems. *Proc. Grassld. Soc. sth. Afr.* 17:76-78.

HALL, A.V. *ed.* 1984. Conservation of threatened natural habitats. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

HALL, M. 1979. The influence of prehistoric man on the vegetation of the Umfolozi-Corridor-Hluhluwe Reserve Complex. Paper presented at the workshop on vegetation dynamics of the Central Complex, Hluhluwe Game Reserve, 10-12 August 1979. Unpubl. Natal Parks Board Report. 5pp.

HANKS, J., Densham, W.D., Smuts, G.L., Jooste, J.F., Joubert, S.C.J., le Roux, P. & Milstein, P. le S. 1981. Management of locally abundant mammals - The South African experience. *In*: *Problems in management of locally abundant wild mammals*. *eds* P.A. Jewell & S. Holt. New York: Academic Press. 361pp.

HAYLETT, D.G. 1960. Run-off and soil erosion studies at Pretoria. *S.A.J.Agric.Sci*. 3(3):379-394.

HOPE, A.S. 1980. Estimation of catchment moisture status for the SCS stormflow model. M.Sc. Eng. thesis. Pietermaritzburg: Univ. of Natal, Dept. of Agric. Eng. 149pp.

HUDSON, N.W. 1981. Soil conservation. London: Batsford. 324pp.

IUCN. 1980. The World Conservation Strategy. Gland: IUCN. 47pp.

JEFFERS, J.N.R. 1982. Modelling. London: Chapman & Hall. 80pp

JEWELL, P.A. & Holt, S. *eds* 1981. Problems in management of locally abundant wild mammals. New York: Academic Press. 361pp.

JOHNSON, C.W., Schumaker, G.A. & Smith, J.P. 1980. Effects of grazing and sagebrush control on potential erosion. *J. Range Manage.* 33(6):451-454.

JOHNSON, D.N. & Gerber, J. 1986. Birds of Umfolozi Game Reserve : additional and confirming records. *Lammergeyer* 36:51-52.

JOHNSON, R.A. & Wichern, D.W. 1982. Applied multivariate statistical analysis. New Jersey: Prentice-Hall. 594pp.

KELLY, R.D. & Walker, B.H. 1976. The effects of different forms of land use on the ecology of a semi-arid region in south-eastern Rhodesia. *J.Ecol.* 64:553-576.

KNOTT, A.P. 1985. Herbivore population censusing in the Hluhluwe and Umfolozi Game Reserves. Unpubl. Natal Parks Board monitoring document. 14pp. + appendices.

KNOTT, A.P. & Brooks, P.M. 1986. An evaluation of helicopter and fixed-wing censusing in Umfolozi Game Reserve. Unpubl. Natal Parks Board report. 15pp.

KNOTT, J.M. 1973. Effects of urbanization on sedimentation and flood flows in Colma Creek Basin, California. U.S. Geol. Surv. Open-file Rep. California: Menlo Park. 54pp.

LANG, R.D. 1979. The effect of ground cover on surface runoff from experimental plots. *J. Soil Cons. N.S.W.* 35(2):108-114.

LARSON, W.E. 1981. Protecting the soil resource base. *J. Soil and Water Cons.* 36(1):13-16.

LARSON, W.E., Pierce, F.J. & Dowdy, R.H. 1983. The threat of soil erosion to long-term crop production. *Science* 219:458-465.

LUSBY, G.C., Reid, V.H. & Knipe, O.D. 1971. The effects of grazing on the hydrology and biology of the Badger Wash Basin in western Colorado, 1953-66. U.S. Geological Service. Water-Supply paper no: 1532-B. 73pp.

MACDONALD, I.A.W. 1979. The effects of gulley erosion on the vegetation of Hluhluwe Game Reserve. Paper presented at the workshop on vegetation dynamics of the Central Complex, Hluhluwe Game Reserve, 10-12 August 1979. Unpubl. Natal Parks Board Report. 5pp.

MACDONALD, I.A.W. 1982a. A first proposal for the definition and delimitation of sites of biological sensitivity in the Hluhluwe-Umfolozi Complex. Unpubl. Natal Parks Board Report. 11pp.

MACDONALD, I.A.W. 1982b. The influence of short term climatic fluctuations on the distribution of savanna organisms in southern Africa. M.Sc. thesis. Durban: University of Natal, Dept. of Biological Sciences. 290pp.

MACDONALD, I.A.W. 1984. An analysis of the role of the Hluhluwe-Umfolozi Game Reserves Complex in the conservation of the avifauna of Natal. *Proc. V. Pan-Afr. Orn. Congr.* 601-637.

MACDONALD, I.A.W. & Birkenstock, P.J. 1980. Birds of the Hluhluwe-Umfolozi Game Reserve Complex. *Lammergeyer* 29:1-56.

MANNERING, J.V. 1981. The use of soil loss tolerances as a strategy for soil conservation. *In*: *Soil conservation : problems and prospects*. *ed.* R.P.C.Morgan. Chichester: John Wiley. 576pp.

MANNETJE, L.'t. & Haydock, K.P. 1963. The dry-weight rank method for the botanical analysis of pasture. *J. Br. Grassl. Soc.* 18:268-275.

MARSHALL, C.J. 1982. There's a track winding back. *J.Soil Cons.* 38(2):81-84.

MCCALLA, G.R., Blackburn, W.H. & Merrill, L.B. 1984. Effects of livestock grazing on sediment production, Edwards Plateau of Texas. *J. Range Manage.* 37(4):291-294.

MCGINTY, W.A., Smeins, F.E. & Merrill, L.B. 1978. Influence of soil, vegetation and grazing management on infiltration rate and sediment production of Edwards Plateau rangeland. *J.Range Manage.* 32:33-37.

MCNAUGHTON, S.J. 1979. Grazing as an optimization process : grass-ungulate relationships in Serengeti. *Am.Nat.*

MCPHEE, P.J., Smithen, A.A., Venter, C.J., Hartmann, M.O. & Crosby, C.T. 1983. The South African rainfall simulator programme for assessing soil loss and run-off. Paper presented at the S.A. National Hydrological Symposium, September, 1983. Pretoria.

MEGAHAN, W.F. 1977. Reducing erosional impacts of roads. FAO conservation guide 1:237-260.

MEGAHAN, W.F. & Kidd, W.J. 1972. Effect of logging roads on sediment production rates in the Idaho Batholith. U.S.Dept. of Agriculture, Forest Service Res.Pap. INT-123. Utah: Ogden, Intermtn.For. and Range Exp.Stn. 14pp.

MELTON, D.A. 1978. Ecology of waterbuck *Kobus ellipsiprymnus* in the Umfolozi Game Reserve. D.Sc. thesis. Pretoria: Mammal Research Institute, Dept. of Zoology. 372pp.

MENTIS, M.T. 1985. Conserving rangeland resources. *J. Grassl. Soc. Sth. Afr.* 2(3):27-31.

MILLER, K. & Child, G. 1983. Goals for protected areas: Management objectives for nature conservation. *In*: *Guidelines for the management of large mammals in African conservation areas. ed.* A.A.Ferrar. South African National Scientific Programmes Report no: 69. Pretoria: CSIR, Cooperative Scientific Programmes. 95pp.

MOLDENHAUER, W.C. 1982. Soil erosion tolerance values. *In*: *Proceedings of the workshop on estimating erosion and sediment yield on rangelands.* U.S. Dept. of Agriculture, Agricultural Research Service report no: ARM-W-26. 228pp.

MOORE, E., Janes, E., Kinsinger, F., Pitney, K. & Sainsbury, J. 1979a. Livestock grazing management and water quality protection. U.S. Environmental Protection Agency. Report no: EPA 910/9-79-67. 148pp.

MOORE, T.R., Thomas, D.B. & Barber, R.G. 1979b. The influence of grass cover on runoff and soil erosion from soils in the Machakos area, Kenya. *Trop. Agric. (Trinidad)* 56(4):339-344.

MORGAN, R.P.C. 1979. Soil erosion. New York: Longman. 113pp.

MYERS, N. 1984. Problems and opportunities in habitat conservation. *In: Conservation of threatened natural habitats. ed.* A.V.Hall. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

NYAMAPFENE, K. 1982. Some perspectives on soil erosion and conservation. *Zimbabwe Sci. News* 16(2):286-288.

O'CONNOR, T.G. 1985. A synthesis of field experiments concerning the grass layer in the savanna regions of southern Africa. South African National Scientific Programmes Report no: 114. Pretoria: CSIR, Foundation for Research Development. 126pp.

ODUM, E.P. 1971. Fundamentals of ecology. Philadelphia: W.B. Saunders Co. 574pp.

OWEN-SMITH, R.N. 1980. A quantitative assessment of the avifauna of the Umfolozi thorn savanna. *Lammergeyer* 30:49-60.

OWEN-SMITH, R.N. *ed.* 1983. Management of large mammals in African conservation areas. Pretoria: Haum Educational. 297pp.

OWENS, L.B. 1974. Rates of weathering and soil formation on granite in Rhodesia. Ph.D. thesis. Indiana: Lafayette, Univ. of Purdue. 144pp.

PACKER, P.E. 1963. Soil stability requirements for the Gallatin elk winter range. *J. Wildl. Manage.* 27:401-410.

PARKER, R.E. 1976. Introductory statistics for biology. London: Edward Arnold. 122pp.

PLATFORD, G.G. 1982. The determination of some soil erodibility factors using a rainfall simulator. *Proc. S.A. Sugar Technologists Assoc.* June 1982:1-4.

PORTER, R.N. 1972. Report on soil erosion associated with roads in the Hluhluwe Game Reserve. Unpubl. Natal Parks Board Report. 7pp.

PORTER, R.N. 1977. Control burning in the Central Complex. Unpubl. Natal Parks Board Report. 8pp.

PRESCOTT-ALLEN, R. 1984. Threatened habitats: The challenge for humanity. *In*: *Conservation of threatened natural habitats*. *ed*. A.V.Hall. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

RICH, J.L. 1911. Recent stream trenching in the semi-arid portion of south western New Mexico, a result of removal of vegetation cover. *Amer. J. Sci.* 32:237-245.

RICKLEFS, R.E., Naveh, Z. & Turner, R.E. 1984. Conservation of ecological processes. *The Environmentalist* 4(8):1-16.

RIQUIER, J. 1982. A world assessment of soil degradation. *Nature and Resources* 18(2):18-21.

ROSEWELL, C.J. 1986. Trends in soil conservation. *J. Soil Cons. N.S.W.* 42(2):131-135.

SCHERTZ, D.L. 1983. The basis for soil loss tolerances. *J.Soil & Water Conservation* 38(1):10-14.

SCHULZE, R.E. 1975. Catchment evapotranspiration in the Natal Drakensberg. Ph.D. thesis. Pietermaritzburg: Univ. of Natal, Geography Dept. 244pp.

SCHULZE, R.E. & Arnold, H. 1980. Digitizing and routine analyses of hydrological data. Pretoria: Water Research Commission report. 137pp.

SCOTT, J.D. 1981. Soil erosion, its causes and its prevention. *In*: *Veld and pasture management in South Africa*. *ed*. N.M. Tainton. Pietermaritzburg: Shuter and Shooter. 481pp.

SIEGEL, S. 1956. Nonparametric statistics for the behavioral sciences. New York: McGraw-Hill. 312pp.

SIEGFRIED, W.R. & Davies, B.R. *eds* 1982. Conservation of ecosystems: Theory and practice. South African National Scientific Programmes Report no: 61. Pretoria: CSIR, Cooperative Scientific Programmes. 97pp.

SINCLAIR, A.R.E. 1974. The social organization of the East African Buffalo (*Syncerus caffer* Sparrman). *IUCN Publ.N.S.* 24:676-689.

SINCLAIR, A.R.E. 1975. The resource limitation of trophic levels in Tropical grassland ecosystems. *J.Anim.Ecol.* 44:497-520.

SMITH, D.D. & Whitt, D.M. 1948. Evaluating soil losses from field areas. *Agr.Eng.* 22:173-175.

SMITH, D.R. 1967. Effects of cattle grazing on a Ponderosa Pine-Bunchgrass range in Colorado. U.S. Dept. of Agriculture, Technical Bulletin no: 1371. 60pp.

SMUTS, G.L. 1980. Report on the ecological status of certain reserves in Zululand. Unpubl. Natal Parks Board report. 38pp.

SNEE, R.D. 1977. Validation of regression models : methods and examples. *Technometrics* 19(4):415-428.

SNYMAN, H.A., van Rensburg, W.L.J. & Opperman, D.P.J. 1985. Soil loss and run-off measurements from natural veld with a rainfall simulator. *J.Grassl.Soc.South.Afr.* 2(4):35-40.

SNYMAN, H.A. & van Rensburg, W.L.J. 1986. Effect of slope and plant cover on run-off, soil loss and water use efficiency of natural veld. *J.Grassl.Soc.South.Afr.* 3(4):153-158.

SOKAL, R.R. & Rohlf, F.J. 1981. Biometry. The principles and practice of statistics in biological research. San Francisco: W.H. Freeman. 859pp.

SOULE, M.E. 1984. Conservation in the real world: Real-conserve or conservation-as-usual? *In*: *Conservation of threatened natural habitats. ed.* A.V.Hall. South African National Scientific Programmes Report no: 92. Pretoria: CSIR, Foundation for Research Development. 185pp.

SOULE, M.E. 1985. What is conservation biology? *BioScience* 35(11):727-734.

SOULE, M.E. & Wilcox, B.A. 1980. Conservation Biology: An evolutionary-ecological perspective. Sunderland: Sinauer Associates Inc. 395pp.

SOUSA, W.P. 1984. The role of disturbance in natural communities. *Ann.Rev.Ecol.Syst.* 15:353-391.

SPEAR, P.T. 1980. Umfolozi Game Reserve soil survey data. Unpubl. 1:50 000 soil map.

STEWART, P.G. 1965. Notes on the prevention and control of soil erosion in the Zululand game reserves. Guideline Bulletin no: 1. Unpubl. Natal Parks Board report. 16pp.

STOCKING, M.A. 1973. Towards a model of soil erosion - an example from Rhodesia. *S.A. Geographer* 4(3): 253-257.

STOCKING, M.A. 1981. Conservation strategies for less developed countries. *In*: *Soil conservation : problems and prospects*. *ed.* R.P.C.Morgan. Chichester: John Wiley. 576pp.

SWANSON, N.P. 1965. Rotating-boom rainfall simulator. *Trans. Am. Soc. Agric. Eng.* 8:71-72.

TINLEY, K.L. 1982. The influence of soil moisture balance on ecosystem patterns in southern Africa. *In*: *Ecology of Tropical Savannas*. *eds* B.J.Huntley & B.H.Walker. Berlin: Springer-Verlag. 669pp.

TRIESTE, D.J. & Gifford, G.F. 1980. Application of the Universal Soil Loss Equation to rangelands on a per-storm basis. *J. Range Manage.* 33(1):66-70.

TROLLOPE, W.S.W. 1984. Fire behaviour. *In*: *Ecological effects of fire in South African Ecosystems*. *eds* P. de V. Booysen & N.M. Tainton. Berlin: Springer-Verlag. 426pp.

WALKER, B.H. 1976. An approach to the monitoring of changes in the composition and utilization of woodland and savanna vegetation. *S.A. J. Wildl. Res.* 6(1):1-32.

WALKER, B.H. 1979. Game ranching in Africa. *In*: *Management of semi-arid ecosystems*. *ed.* B.H. Walker. Hamburg: Elsevier Scientific. 398pp.

WALKER, B.H. & Goodman, P.S. 1983. Some implications of ecosystem properties for wildlife management. *In*: *Management of large mammals in African conservation areas*. *ed.* R.N. Owen-Smith. Pretoria: Haum Educational. 297pp.

WALKER, B.H., Emslie, R.H., Owen-Smith, R.N. & Scholes, R.J. 1987. To cull or not to cull: lessons from a southern African drought. *J.Appl.Ecol.* 24:381-401.

WARD, C.J. 1962. Report on scrub control in the Hluhluwe Game Reserve. *Lammergeyer* 2(1):57-62.

WATSON, H.K. 1982. Slope map of Umfolozi Game Reserve. Unpubl. 1:50 000 slope map.

WATSON, H.K. & Macdonald, I.A.W. 1983. Vegetation changes in the Umfolozi-Hluhluwe Game Reserve from 1937 to 1975. *Bothalia* 14(4):265-269.

WHATELEY, A.M. 1986. *Pers. comm.* "Chelsea Gardens", Back Lane, Mickleton, Gloucestershire, England.

WHATELEY, A. & Porter, R.N. 1983. The woody vegetation communities of the Hluhluwe-Corridor-Umfolozi Game Reserve Complex. *Bothalia* 14(4):745-758.

WHATELEY, A. & Brooks, P.M. 1985. The carnivores of the Hluhluwe and Umfolozi Game Reserves : 1972 - 1983. *Lammergeyer* 35:1-27.

WHITE, B. 1978. Road erosion. Scan 212. Victoria: Soil Conservation Authority.

WHITTAKER, R.H. 1965. Dominance and diversity in land plant communities. *Science* 147:250-260.

WHITTAKER, R.H. 1977. Evolution of species diversity in land communities *Evolutionary Biology* 10:1-67.

WIGHT, J.R. & Lovely, C.J. 1982. Application of the soil loss tolerance concept to rangelands. *In: Proceedings of the workshop on estimating erosion and sediment yield on rangelands*. U.S. Dept. of Agriculture, Agricultural Research Service report no: ARM-W-26. 228pp.

WILCOX, B.A. 1980. Insular ecology and conservation. *In: Conservation Biology: An evolutionary-ecological perspective*. *eds* M.E. Soule & B.A. Wilcox. Sunderland: Sinauer Associates Inc. 395pp.

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WILLIAMS, J.R. 1982. Planning research to evaluate the effect of soil erosion on soil productivity. *In*: *Proceedings of the workshop on estimating erosion and sediment yield on rangelands*. U.S. Dept. of Agriculture, Agricultural Research Service report no: ARM-W-26. 228pp.

WILLS, A.J. 1984. Monitoring the extent and effects of fire on vegetation in the Hluhluwe and Umfolozi Game Reserves. Unpubl. Natal Parks Board monitoring document. 9pp. + appendices.

WILLS, A.J. 1985. Guidelines for the annual veld assessment survey in the Hluhluwe and Umfolozi Game Reserves. Unpubl. Natal Parks Board report. 12pp.

WILLS, A.J. 1987. *Pers. comm.* Hluhluwe Game Reserve, P.O. Box 25, Mtubatuba, 3935, South Africa.

WILLS, A.J. & Knott, A.P. 1985. Proposed long-term vegetation and herbivore management goals and strategies in the Hluhluwe and Umfolozi Game Reserves. *In*: *Background documents for workshop on ecologically based management in HGR-UGR, 2-4 December 1985.* Unpubl. Natal Parks Board report. 39pp.

WILSON, A.D. & Tupper, G.J. 1982. Concepts and factors applicable to the measurement of range condition. *J. Range Manage*. 35(6):684-689.

WISCHMEIER, W.H. 1955. Punch cards record run-off and soil loss data. *Agric. Eng*. 36:664-666.

WISCHMEIER, W.H. 1975. Estimating the soil loss equation's cover and management factor for undisturbed areas. *In*: *Proc. Sediment-yield workshop Oxford, Miss*. U.S. Dept. Agriculture, Agricultural Research Service report no: S40:118-124.

WISCHMEIER, W.H. 1976. Use and misuse of the Universal Soil Loss Equation. *J.Soil & Water Cons*. 31:5-9.

WISCHMEIER, W.H., Smith, D.D. & Uhland, R.E. 1958. Evaluation of factors in the soil-loss equation. *Agric. Eng.* 39(8):458.

WISCHMEIER, W.H. & Smith, D.D. 1978. Predicting rainfall erosion losses - a guide to conservation planning. U.S. Dept. of Agriculture, Agr. Handbk. no: 537. 58pp.

WOOD, M.K. & Blackburn, W.H. 1981. Sediment production as influenced by livestock grazing in the Texas rolling plains. *J.Range Manage.* 34(3):228-231.

YOUNG, A. 1976. Tropical soils and soil survey. Cambridge: Cambridge University Press. 468pp.

ACKNOWLEDGEMENTS

Many people have assisted in this study since it began in March 1982. My sincere thanks go to my supervisor, Prof. N. Tainton, for his help and guidance throughout the study and to my cosupervisor, Prof. R. Schulze for his substantial input and advice. Mrs M. Pattenden helped considerably with data input, mapping and figures, as did Messrs A. Whateley, Z. Ngema, A. Mkwanazi and Ms J. Fotheringham in many aspects of the field work. My thanks also go to Mr P. Goodman for his advice and direction since the start of this study; to Prof. P. Clarke, Dr D. Morrey and Mr S. Piper for their statistical advice and Mrs G. Hendry for her assistance with correspondence analysis. Mr A. Smithen provided advice and logistical support for the rainfall simulator trials, and Mrs C. O'Mahoney and Ms K. Wiercks digitised the autographic raingauge charts.

My thanks also go to Dr M. Brooks for his advice, support and for critically reading some chapters and to Drs M. Mentis and M. Fey and Mr B. Russell for help in the planning stages. The rainfall simulator trials were possible with the help of Messrs F. Moolman, M. Taylor, B. Leslie, B. Qwabe, F. Msomi, I. Mngomezulu, B. Mquni, I. Mdaka, M. Nene and M. Mkwanazi. Messrs E. Cox, G. Young and G. Taylor are thanked for the use of the Natal Parks Board water tankers, as are the various Roads Maintenance Unit drivers for delivering the water, and Messrs S. Pillinger, H. Bentley, B. Harris and G. Smith for ensuring a constant supply of water. Messrs A. Knott and T. Sandwith supplied ungulate stocking rate data and Mr A. Wills vegetation monitoring data. Messrs M. Johnston and A. Smith-Bailey described the soil profile pits.

Mr G. Garland is thanked for his help in designing the sediment collection apparatus and Messrs F. du Toit, P. Clark and B. Robbins for the installation of the natural run-off plots and cabling of exclosures. Messrs B. Liggitt, S. Biyela and A.

Mkwanazi negotiated with the local chief and indunas regarding the installation of equipment in KwaZulu and the Cengeni gate guards kept a watchful eye on this equipment which was supplied by the Institute of Natural Resources. Messrs D. Hartley and B. Howells are thanked for the competent manner in which they burnt the firebreaks around the plots and Mr G. Smith for ensuring that no fires from KwaZulu would reach the plots during a particular burning season. Mr R. Emslie helped with the analysis and interpretation of data and commented on some chapters, and Mrs M. Smith provided statistical advice.

Mrs M. Pattenden and Mrs M. Venter typed various drafts of the thesis and Mrs S. Klarer is thanked for critically reading the entire thesis. Mrs R. Harvey typed the abstract and corrected certain figures. Dr M. Brooks assisted in obtaining CSIR funding for this project and Mr B. Dinkleman administered the CSIR trust fund. Messrs A. Wills and T. Sandwith lent support by taking over my normal duties and workload while I was on study leave and had considerable input into the management recommendations chapter. The financial support of the Natal Parks Board and the Foundation for Research Development of the CSIR is gratefully acknowledged. The Natal Region of the Dept. of Agriculture and Water Supply assisted in supplying a rainfall simulator and computer facilities at Cedara. To all the people and institutions mentioned above I would like to offer my sincere thanks, as the completion of this study would have been impossible without the generous help and support received.

Lastly, I would like to thank my mother for her logistical assistance during the write-up and my father for his encouragement and continued interest in this study.