THE BIOECONOMIC IMPLICATIONS OF VARIOUS STOCKING STRATEGIES IN THE SEMI-ARID SAVANNA OF NATAL

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Declaration

This thesis is the result of the author's original work except where acknowledged or specifically stated to the contrary in the text. It has not been submitted for any degree or examination at any other university

GRANT HATCH

He waters the mountains from His upper chambers;
the earth is satisfied by the fruit of His work.
He makes grass grow for the cattle,
and plants for man to cultivate -
bringing forth food from the earth.

Psalm 104: 13-14

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ABSTRACT

Climatic and market uncertainty present major challenges to livestock producers in arid and semi-arid environments. Range managers require detailed information on biological and economic components of the system in order to formulate stocking strategies which maximise short-term financial risk and minimise long-term ecological risk. Computer-based simulation models may provide useful tools to assist in this decision process. This thesis outlines the development of a bioeconomic stocking model for the semi-arid savanna of Natal.

Grazing trials were established at two sites (Llanwarne and Dordrecht) on Llanwarne Estates in the Magudu area of the semi-arid savanna or Lowveld of Natal. The Lowveld comprises a herbaceous layer dominated by *Themeda triandra*, *Panicum maximum* and *P. coloratum* and a woody layer characterised by *Acacia* species. The sites differed initially in range composition. Llanwarne was dominated by *Themeda triandra, Panicum maximum* and *P. coloratum*, while Dordrecht with a history of heavy stocking was dominated by *Urochloa mosambicensis, Sporobolus nitens* and *S. iocladus*. Three treatments were stocked with Brahman-cross cattle at each site to initially represent 'light'(0.17 LSU ha^{-1}), 'intermediate' (0.23 LSU ha^{-1}) and 'heavy' (0.30 LSU ha^{-1}) stocking. Data collected at three-week intervals over seven seasons (November 1986 to June 1993 or 120 measuring periods) provided the basis for the development of a bioeconomic stocking model (LOWBEEF - LOWveld BioEconomic Efficiency Forecasting) which comprised two biological sub-models (GRASS and BEEF), based on step-wise multiple linear regression models, and an integrated economic component (ECON). The GRASS model predicted the amount of residual herbage at the end of summer (kg ha^{-1}) and the forage deficit period (days) over which forage supplementation would be required to maintain animal mass. Residual herbage mass at the end of summer (kg ha^{-1}) was significantly related ($P<0.01$) to cumulative summer grazing days (LSU gd ha^{-1}), rainfall (mm) (measured 1 July to 30 June) and range condition (indexed as the sum of the proportions of *T. triandra*, *P. maximum* and *P. coloratum*). The forage deficit period (days) over which herbage mass declined below a grazing cut-off of 1695 kg ha^{-1} was significantly related ($P<0.01$) to

residual herbage mass at the end of summer. The BEEF model predicted the livemass gain over the summer (kg ha^{-1}) which was significantly related ($P<0.01$) to rainfall (mm) and stocking rate (LSU ha^{-1}), but interestingly not to range condition. The economic component (ECON) reflected the difference between gross income (R ha^{-1}) and total costs, which were based on fixed and variable cost structures (using 1993 Rands), including demand-related winter feed costs, to reflect net returns to land and management (R ha^{-1}).

A conceptual model of range dynamics, based on three discrete states, was developed to summarise the effects of rainfall and stocking rate in the semi-arid savanna. State 1, characterised by *S. iocladus* and *S. nitens*, was associated with heavy stocking. Movement towards State 2, characterised by *T. triandra* and *P. maximum*, was associated with periods of above-average rainfall. Drought conditions, which comprised a major system disturbance, led to stability at State 3, dominated by *U. mosambicensis*. Post-drought recovery was influenced by pre-drought composition and stocking levels where tuft numbers, basal cover and seed-bank size were significantly reduced by increased stocking within a sward dominated by species of low stature such as *Aristida congesta* subsp *congesta*, *Urochloa mosambicensis*, *Sporobolus nitens*, *Sporobolus iocladus* and *Tragus racemosa*. It was suggested that extensive soil loss may lead to stabilisation across an irreversible threshold at a forth state characterised by shallow rooted species such as *Tragus racemosa* and *Aristida congesta* subsp. *congesta*.

Sensitivity of optimum economic stocking rate and net return to price and interest rate fluctuations, and wage and feed cost increases were examined for various rainfall and range condition scenarios. Net return and optimum economic stocking rate increased as rainfall and range condition increased through the effect of increased residual herbage mass at the end of summer, decreased forage deficit periods and reduced supplementary feed costs. Net return was highly responsive to changes in beef price where an increase in beef price led to an increase in optimum economic stocking rate and net return. The effect of reduced prices may be compounded by dry seasons, where supply-driven decreases in price may occur.

This suggested that for dry seasons the optimum stocking rate was the lightest within the range of economic stocking rates. Although an increase in interest rates would increase variable costs and lead to reduced returns, the influence of interest rates on enterprises will vary in relation to farm debt loads. Increased labour costs would result in a corresponding decline in net return although optimum economic stocking rate would remain unaffected. Increased supplementary feed cost had little influence on net return relative to the effect of demand-driven increases in feed costs as rainfall decreased.

The distribution of net returns for stocking strategies of 0.20, 0.30 and 0.40 LSU ha^{-1} and climate-dependent stocking (where stocking levels were varied in relation to rainfall and hence forage availability) and range condition scores of 10, 50, 80 and a dynamic range model were examined for a 60 year rainfall sequence (1931-1991). While a range score of 10 would see residual herbage mass decline to below a grazing cut-off of 1695 kg ha^{-1} before the end of summer, a range score of 80 suggested that, irrespective of stocking strategy within the range investigated, herbage would not become limiting. This suggested that irrespective of stocking strategy a range score of 10, established across an irreversible soil loss threshold, would reflect accumulated losses over the 60 year period. In contrast, a range score of 80 would lead to positive accumulated returns. A dynamic range model (where range composition was related to previous seasons rainfall) and a climate-dependent stocking strategy, suggested that herbage would not become limiting by the end of summer and forage deficit periods would be restricted to an average of 88 days per year. Such an approach would yield a higher accumulated cash surplus than fixed stocking strategies.

Incorporation of stochastic rainfall effects allowed the development of cumulative probability distributions based on 800-year simulations to evaluate the risk associated with various stocking strategies. Range condition played a major role in determining the risk of financial loss where decreased range condition was associated with enhanced risk. An increase in stocking rate resulted in increased variability in returns. Although the risk of forage deficits and financial losses may be reduced with lighter stocking, this may be at the cost of reduced returns during wetter seasons. Increased stocking may increase the probability of higher returns during wetter seasons although this may be at the cost of increased risk of forage deficits and highly negative returns during dry seasons. Importantly, ecological risk may increase as stocking is increased. A flexible or climate-dependent strategy, where stock numbers are adjusted according to previous seasons rainfall, may combine the financial benefits of each approach and reduce financial risk. Although errors may carry high ecological costs where, for example, the effect of an above-average rainfall season would be to increase stock numbers into a subsequent dry season, the probability of incurring such error was low.

Current livestock production systems in the semi-arid savanna of Natal based on breeding stock may not be appropriate in a highly variable environment where low rainfall may require extended periods of supplementary feeding or force the sale of breeding stock. A change in emphasis from current systems to a mixed breeding system, where the level of breeding stock would be set at the optimum economic stocking rate for drier seasons, may decrease both financial and ecological risk. Growing stock may either be retained or purchased during wetter seasons to reach the optimum economic stocking rate for such seasons. Although growing stock may display a greater tolerance to restricted forage intake (during drier seasons) than would breeding stock, additional growing stock may be rapidly sold in response to declining rainfall with no influence on the breeding system. Integration of wildlife into current cattle systems may be an important means of reducing financial risk associated with variable rainfall and profitability and ecological risk associated with woody plant encroachment.

CHAPTER 1

INTRODUCTION

Climatic and market uncertainties present complex management challenges to livestock producers in semi-arid savanna systems. Although productivity is determined largely by stochastic rainfall events (O' Connor 1985; Ellis & Swift 1988; Hoffman & Cowling 1990), the major decision affecting the level of livestock production is stocking rate (Riechers *et al.* 1989). The stocking strategy[1] selected by the manager may influence short-term profitability (Danckwerts & King 1984) and in the long-term influence the natural resource-base (Foran & Stafford-Smith 1991). Stocking strategies which maximise short-term return, but which lead to resource degradation, are clearly not sustainable (Hart *et al.* 1988). Stocking decisions consequently determine the ecological and financial risk profile to which the enterprise is subjected. In South Africa range managers tend to increase stocking rate in response to diminishing financial returns (Danckwerts 1989), and while this may be questioned even on the basis of short-term financial returns, in the long-term this may lead to resource degradation.

While the risk associated with rainfall variability is largely beyond the control of the range manager, stocking decisions must be made to address the consequences of variability in the forage environment. This is particularly relevant to the semi-arid savanna of Natal where low and erratic rainfall presents major challenges to livestock producers. Stocking rate therefore comprises the most important management decision affecting both the level of livestock production and financial return in arid and semi-arid systems.

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[1] Stocking strategy may be defined as the combination of strategic management decisions made by the range-user to satisfy ecological, economic and social objectives. These incorporate stocking rate, burning timing and frequency, grazing system, type and level of supplementation, breeding and marketing decisions.

Although research has tended to concentrate on biological components of the system, livestock producers require detailed information on the interaction between biological and economic components in order to formulate viable management approaches. Importantly, this information should be presented to producers in a readily useable form. Computer-based simulation modelling may provide a useful means to integrate the biological and economic components of the system and allow range managers to assess the short-term financial and long-term ecological consequences of various stocking strategies. Simulation modelling may be particularly relevant to highly variable systems where managers with varying attitudes to risk could assess the outcome of different strategies. This thesis outlines the development of a bioeconomic model for commercial beef producers in the semi-arid savanna of Natal using data collected from an extensive grazing trial over a seven year period. It was envisaged that the framework developed in this study for the semi-arid savanna could form the basis for the development of similar models for other beef producing regions of Natal. Importantly, models relating biological and economic components of highly variable systems could then be expanded to include non-commercial objectives, as may occur under communal land tenure, and develop risk management strategies for different socio-economic conditions.

The overall objective of the research programme, which comprises grazing trial sites located in the Southern Tall Grassveld, Natal Sour Sandveld and Lowveld (Acocks 1988), is the development of a grazing capacity model for the drier beef-producing regions of Natal (Turner 1990). The programme was initiated in 1986 by the Department of Grassland Science, University of Natal in collaboration with the Department of Agricultural Development and Roodeplaat Grassland Institute. Turner (1988, 1990) collected the base-line range condition data (1986) and examined the influence of stocking rate, range condition and rainfall on herbage and animal production over the period 1986 to 1989. This thesis reports on an extended time series of the grass and animal production data (1986 to 1993). The specific objectives of this investigation were:

1) to examine the economic implications of various range management decisions;
2) to examine the conceptual relationship between stocking strategy and the costs of range degradation;
3) to develop predictive models relating seasonal patterns in herbage and animal production to stocking strategy, range condition and rainfall;
4) to examine the influence of stocking strategy on range composition dynamics;
5) to examine the influence of stocking strategy on post-drought recovery patterns;
6) to develop a bioeconomic simulation model to examine the influence of biological and economic factors on profitability and risk; and
6) to outline management implications for livestock producers in the semi-arid savanna of Natal.

Thesis format

This thesis comprises a number of separate chapters, each forming the basis of a publication. While a degree of repetition, particularly with regard to referencing and procedure, was unavoidable an underlying philosophy has been developed through successive chapters. Chapters 2 and 3 provide a conceptual framework and background to the study, and attempt to integrate biological and economic factors which influence decision-making in extensive production systems. Chapter 4 places the study in a regional context and outlines the experimental procedure. Chapters 5 and 6 provide the biological basis for the development of a bioeconomic stocking model. Chapters 7 and 8 examine the influence of stocking strategy on range dynamics and address the issue of sustainability and ecological risk. These concepts are incorporated into the bioeconomic model outlined in Chapter 9, evaluated in Chapter 10 and applied in Chapters 11, 12 and 13. Chapter 14 provides a summary of ideas and outlines their relevance to range management in semi-arid savanna systems.

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CHAPTER 2[1]

COSTING STRATEGIC DECISION-MAKING IN RANGE MANAGEMENT

INTRODUCTION

Range managers are faced with a daunting array of biological and economic management decisions. Every decision is associated with benefits and costs (including risk), the outcome of which affects both current profitability and future viability of the enterprise. While strategic decision-making in range management may attempt to minimise risk; environmental and market uncertainty,[2] which are largely beyond the control of the grazier, may strongly influence the outcome of any decision. Environmental uncertainty may arise through climatic variability, while market uncertainty may be associated with fluctuating product prices and increasing input costs. Exogenous factors, such as changes in the demand for meat, which influence product prices, or changes in interest rates, which influence the costs of factors of production, may further contribute to market uncertainty and risk.

Increases in input costs at a greater rate than increases in product prices has led to a cost:price squeeze which has adversely affected the profitability of agriculture

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[1] Hatch GP Costing strategic decision-making in range management. In: Tainton NM (ed.) *Veld and pasture management in South Africa.* 2nd edition. Shuter & Shooter, Pietermaritzburg (in press)

[2] Decision theory has evolved beyond early definitions of risk and uncertainty (e.g. Knight 1921), which distinguished between an ability to attach probability levels to the outcome of any decision (risk) and situations where probabilities could not be given (uncertainty), to a definition based on objective and subjective probabilities which refer to situations where complete information is lacking (Doll & Orazem 1984).

in the short-term.[3] This emphasises the importance of the economic evaluation of alternative management strategies under risky ecological and economic[4] conditions. On what criteria then should range management decisions be based and how should these decisions be made? This chapter provides an outline of how underlying economic and biological principles may be integrated to provide a basis for strategic decision-making in range management.

INTEGRATING THE PRINCIPLES OF RANGE MANAGEMENT AND ECONOMICS

Economics is the science that deals with the allocation of scarce resources among competing uses (Samuelson 1964) and the study of the choices people make with respect to scarce resources (Atkinson 1982). Range economics may be described as the science of simultaneously applying the principles of economics and range management to determine the economic consequences of alternate range management decisions (Workman 1986). This is fundamental to the development of production systems which are both environmentally and economically sustainable.

The resources of concern in range economics may be land (rangeland), labour (labour and management) or produced resources (capital, infrastructure and vehicles) (Workman 1986). Competing uses may include decisions such as whether to stock sheep or cattle, or apply rotational or continuous grazing. These decisions create the problem of choice and the possibility of lost opportunities, i.e. a particular resource-use decision implies that other choices are forgone. **Opportunity cost** is defined as the value of the best alternative foregone (Atkinson 1982).

The principles of **supply** and **demand** are important economic concepts which may influence livestock production from rangelands. The demand for a product, for

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[3] In the long-term farmers may adjust by, for example, substituting relatively cheaper inputs or by becoming more efficient.

[4] While financial costs refer to direct (or accounting) costs, economic costs include all costs i.e. financial, opportunity and externality costs.

example beef, is the quantity willingly bought per unit time at a specific price (Workman 1986). Demand for a product is not independent of the demand for other products, i.e. demand indicates a relative preference for the product.[5] The concept of demand for a product may be enlarged into a **demand function**, which outlines the quantities willingly purchased per unit time at various prices. The schedule is based on the law of demand which states that as the price of a product increases, fewer units are purchased and *vice versa* (Workman 1986).

The concept of supply is analogous to demand, except that supply is based on producers' wishes, and is defined as the quantity of a product willingly offered for sale at a given price (Workman 1986). A **supply schedule** outlines the various quantities offered for sale over a given time at various product prices.

The point at which the quantity demanded is equal to the quantity willingly supplied is the price at which **market equilibrium** occurs. Should the quantity demanded be greater than the quantity willingly supplied at a given price, i.e. there is excess demand, then product price will increase until the market clears. In contrast, if the quantity supplied exceeds the quantity demanded at a given price, i.e. there is an excess supply, then the product price will fall until the market clears (Atkinson 1982) (Figure 1).

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[5] Relative preference is based on a number of factors such as consumer choice, the availability of suitable alternatives, and the relative price of the product.

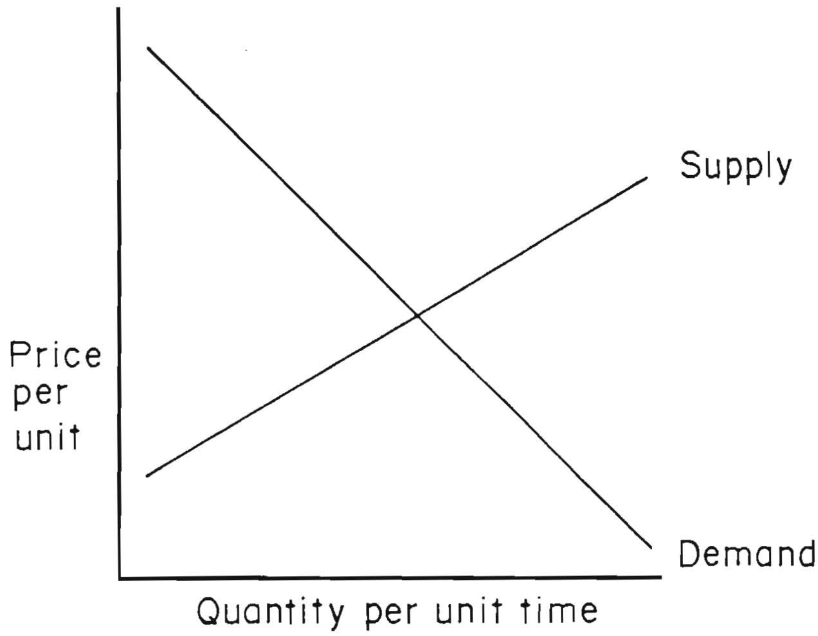

Figure 1 The conceptual relationship between supply and demand. Market equilibrium is indicated at the point of intersection of the supply and demand functions.

Movement along the demand (supply) schedule indicates the consumers' (producers') response to change in price only. Workman (1986) outlined the factors which may lead to shifts in the demand and supply schedules for a product. Shifts in the demand schedule (for a product such as beef) are related to:

1) the price of substitutes (such as lamb or poultry);
2) the price of complements (products which are consumed with the product e.g. beer);
3) consumer preferences (these may include fears regarding cholesterol levels or use of hormones in production);
4) population growth (which influences the number of buyers in the market);
5) disposable income (which may be influenced by the state of the economy and government policies); and

6) consumer expectations regarding product prices and income levels.

Movement along the supply schedule indicates change in the quantity of the product offered in response to a change in price. Shifts in the supply curve (for a product such as beef) may be related to:

1) the price of production inputs (such as labour and fuel);
2) government policy (which includes subsidies, stocking limits and taxation);
3) the prices of alternative products (such as mutton and wool)
4) weather (rainfall variability which influences forage and livestock production); and
5) the number of producers in the market.

The **price elasticity of demand** (E_d) is defined as the percentage change in the quantity of a product demanded (Q) associated with a one percent change in the price of the product (P) (Atkinson 1982).

$$E_d = \frac{\Delta Q}{\Delta P} \cdot \frac{P}{Q}$$

If the absolute value of E_d is greater than one then the price elasticity of demand is considered to be relatively elastic (i.e. highly responsive to changes in price) and if less than one relatively inelastic. Workman *et al.* (1972) indicated that the price elasticity of demand for beef in the United States for the period 1947 to 1967 was -0.67 (i.e. relatively inelastic), which implied that the quantity of beef purchased would decrease by 0.67 percent for each one percent increase in beef price. Conversely, the price flexibility of demand for beef (i.e. response of prices to changes in quantity) may be expected to be greater than one in absolute terms which implies that a one percent decrease in the quantity of beef produced would increase the beef price by more than one percent. This suggests that although the beef industry as a whole would benefit by decreasing production, producers who individually have little influence on the market would benefit by increasing production (Workman 1986). The price elasticity of demand for beef in South Africa was estimated to be slightly less inelastic (-0.77) than in the United States (Hancock 1983), as consumer response to changes in beef price (e.g. supplementing poultry for beef in response to price increases) is greater than for more affluent nations such as the United States.

DETERMINING THE OPTIMUM MIX OF TYPES AND CLASSES OF LIVESTOCK

Single species production systems, based primarily on cattle, may be inefficient users of forage resources, particularly in multi-strata vegetation such as savanna, and may yield lower returns than multi-species systems based on resource partitioning (eg. Aucamp *et al.* 1983; Cumming 1993). The 'cost-price' squeeze has emphasised the importance of multi-species systems which may increase income per unit area of land. Economic theory suggests, however, that there is a limit to the amount (e.g. of animal products) which may be produced in a given time period, i.e. a production possibilities curve which is determined by the quantity and quality of resources available (Atkinson 1982). Increased production of any one product in a multi-species system (e.g. beef) will withdraw resources (both forage and factors of production) from another enterprise (e.g. sheep), giving rise to opportunity costs of increased production of any product. Overall productivity per unit area may be increased by a combination of livestock types and decrease both financial and environmental risk associated with single species systems. For example, although the integration of goats into a beef system would require a reduction of 40 percent of the cattle stocking rate to account for grazing by goats (Stuart-Hill 1987), an integrated system would be three times as profitable as a beef only system (Aucamp *et al.* 1984). In addition, control of bush encroachment through a combination of burning and goats would result in a twofold increase in annual return per Rand invested than would chemical control of bush in a beef system (Trollope *et al.* 1989). Similarly, the return per unit area from an integrated beef/game system is greater than from a beef system alone (Cumming 1993) through more efficient resource use.

The question which now arises is how to determine the optimum combination of classes or types of animals to satisfy economic objectives. Linear programming may provide a useful tool to identify the optimum combination of classes of livestock within a single-species system (Angirasa *et al.* 1981). For example, for calculating the relative proportions of cows, replacement heifers, steers and bulls in a beef system, or for calculating the optimum combination of different types of livestock such as the relative numbers of sheep and cattle in a mixed cattle/sheep system. Linear programming may serve as a useful tool to examine the influence of changes in input costs or product prices on the optimum mixes of classes or types of livestock and indicate the optimum economic stocking levels under environmental variability (Angirasa *et al.* 1981).

Increased focus is likely to be directed towards multi-species systems as pressure to improve output per unit land area, in response to declining real revenue, is increased. Increasingly, livestock may fail to satisfy economic objectives, particularly in arid and semi-arid systems, and alternative forms of land-use, such as recreation and tourism, will become important. For example, game farming may increasingly be integrated into cattle operations as returns from cattle decline. Interestingly, tourism and recreation options are seen as alternatives to sheep farming in the High Country of New Zealand, which is based on non-sustainable inputs of fertilizers and oversowing (South Island High Country Committee 1992). In other words, the sustainability of intensive agriculture, supported by subsidies, is likely to be increasingly questioned by a largely urban-based population. This may encourage changes in land-use and movement out of domestic stock such as sheep and cattle.

DETERMINING THE OPTIMUM LEVEL OF PRODUCTION

While range managers are forced to allocate scarce resources between different and often competing forms of use, the question which now arises is how much to produce? Stocking rate is the management variable which has the greatest influence on levels of productivity (Jones & Sandland 1974; Hart 1978), profitability (Foran & Stafford-Smith 1991; Hatch & Tainton 1993) and the resource-base (Danckwerts & King 1984). Determination of the optimum level of production or stocking rate may be based on the marginality principle (Atkinson 1982), which implies that total output increases at a decreasing rate as stocking rate is increased. As stocking rate is increased, production per hectare will increase at a decreasing rate as a consequence of less forage available per animal, reduced dietary quality, selection of less preferred forage and competition between animals (Jones & Sandland 1974; Wilson 1986). Stocking rate models may be integrated with economic theory to reflect the economic consequences of various stocking rates (e.g. Booysen *et al.* 1975; Danckwerts & King 1984; Workman 1986).

Production per hectare may be given as a quadratic function[6] derived from the product of individual animal performance and stocking rate (Workman 1986). Maximum gain per hectare corresponds to a stocking rate (G_{max}) (Figure 2). The gross value of seasonal animal production may be calculated as the product of seasonal livemass gain (kg ha^{-1}) and product price (R kg^{-1}). Gross income would be maximised at a stocking rate corresponding with G_{max}. Profitability is calculated as gross income less total costs, which comprises the sum of fixed and variable costs. Fixed costs are those that are incurred irrespective of the level of stocking such as depreciation on infrastructure and machinery, and fixed labour costs. Variable costs increase proportionately as stocking rate increases, and include interest costs of livestock purchased (based on market interest rates or the opportunity cost), supplementary feed, veterinary (including dipping and inoculation costs) and marketing costs (including transport and agent's commission). While costs may vary between enterprises due to different economies of size, i.e. the larger the enterprise the greater the ability to reduce costs per unit of production, the relationship between stocking rate and variable costs for a given enterprise is assumed to be linear (Figure 2).

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[6] The rate of decline in individual animal performance with increased stocking rate may, in practice, be curvilinear due to non-uniform temporal and spatial distribution of forage resources, and increased forage harvesting costs (Heitschmidt & Taylor 1991).

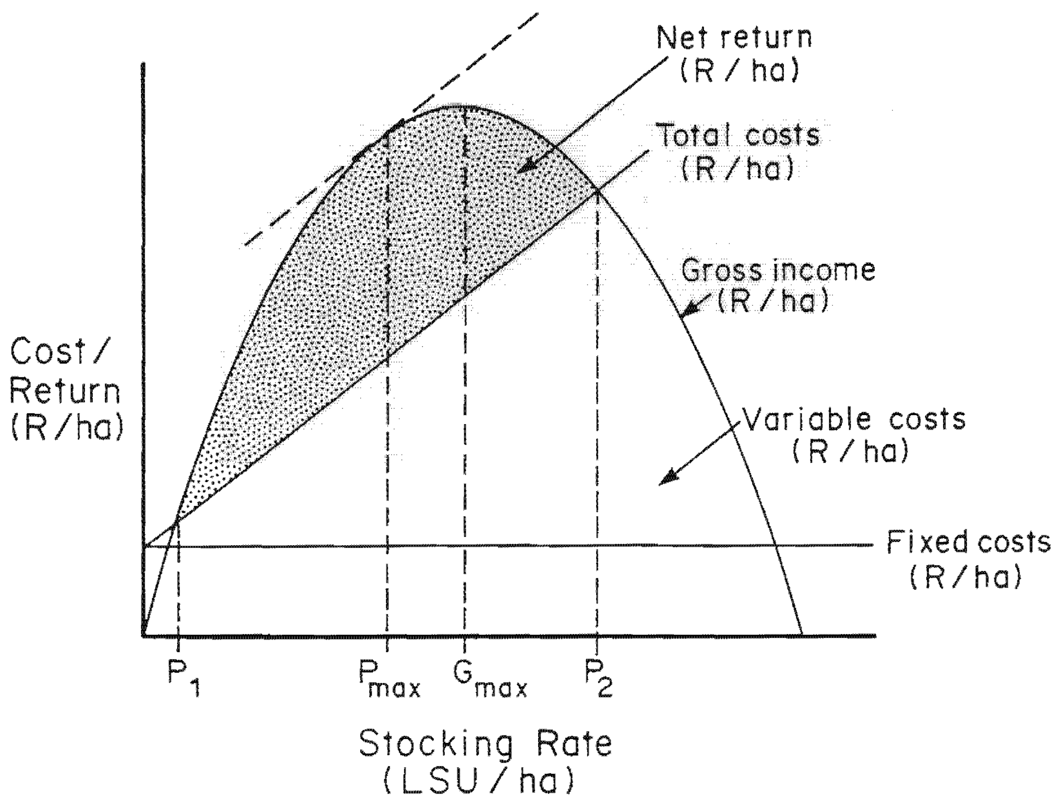

Figure 2 The relationship between stocking rate (LSU ha^{-1}), fixed and variable costs (R ha^{-1}), gross income (R ha^{-1}), and net return (R ha^{-1}) and optimum economic stocking rate (P_{max}) (LSU ha^{-1}) for a given livestock enterprise. LSU denotes large stock units

The range of economic stocking rates (P_1-P_2), i.e. where financial returns are positive, occurs where gross income exceeds total costs. Stocking rates below P_1 and above P_2 fail to cover total costs. The stocking rate at which profitability is maximised (P_{max}) occurs where the distance between the gross income and total cost function is at its greatest. This may be given as the point where a line parallel to the total cost function is tangential to the gross income function (Figure 2). Shifts in the supply:demand relationship for the product, for example beef, may result in an increase in product price, an outward shift of the gross income function and an increase in the economic optimum stocking rate (Figure 3a), i.e an increase in stocking rate from P_{max1} to P_{max2}. Conversely a decrease in product price would lead to a decrease in optimum economic stocking rate. An increase in fixed costs, such as increased labour costs, which result in an outward, parallel shift in total costs, would lead to reduced profits, but at the same economic optimum stocking rate (Figure 3b). An increase in variable costs would see an increase in the slope of the total cost curve and a decrease in optimum economic stocking rate (Figure 3c).

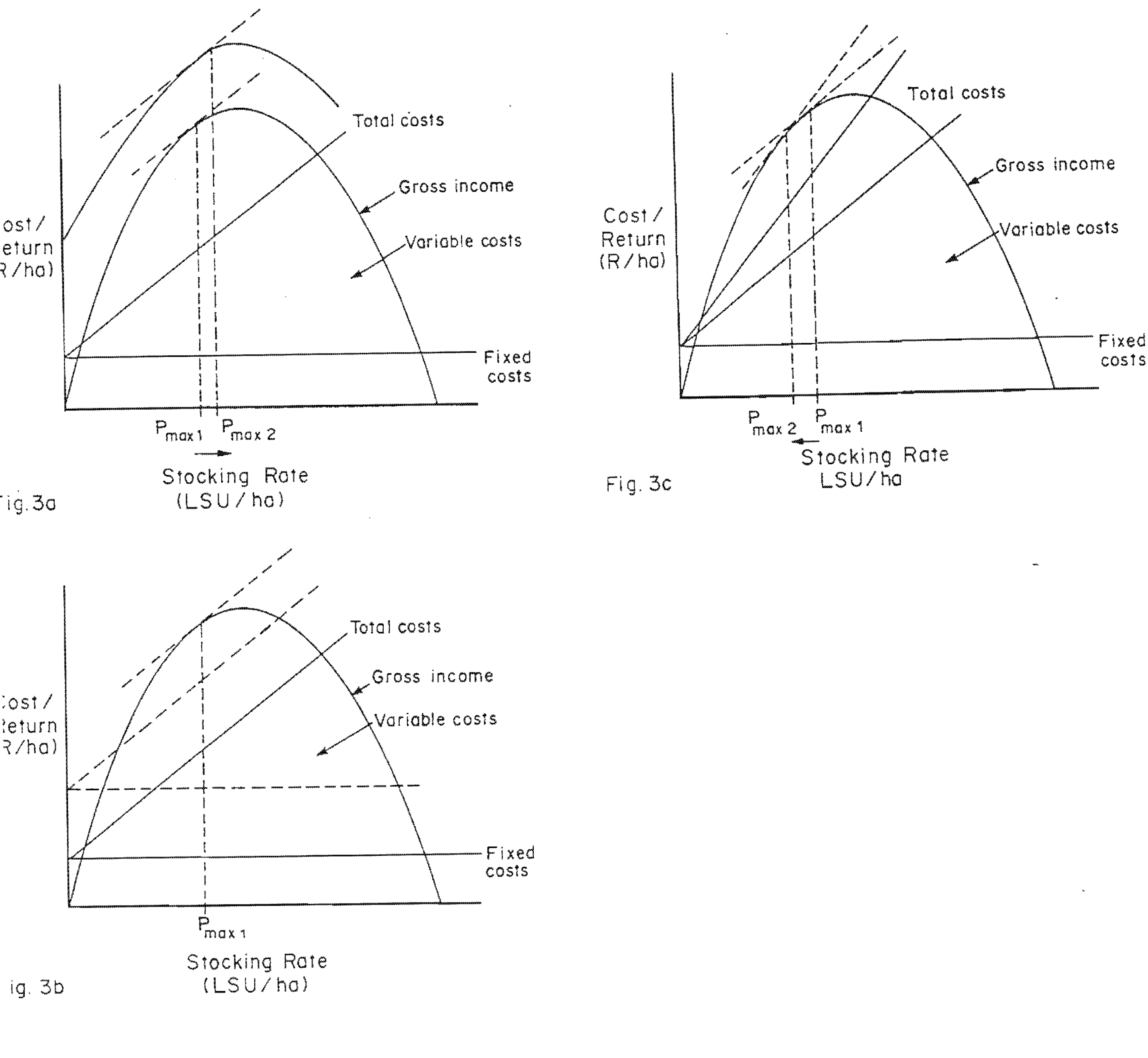

Figure 3 The influence of a) an increase in product price (R kg^{-1}), b) an increase in fixed costs (R ha^{-1}) and c) an increase in variable costs (R animal $unit^{-1}$) on net return and optimum economic stocking rate for a given livestock enterprise

The stocking rate at which maximum profitability is reached will always lie below that at which maximum gain per hectare or gross income is attained (Booysen *et al.* 1975; Danckwerts & King 1984; Workman 1986).[7] This suggests that if the optimum economic stocking rate does not exceed the long-term carrying capacity of the resource then the stocking rate should be both economically and ecologically sustainable (as suggested by Perrings & Barbier 1993).

Despite the apparent simplicity of the economic theory outlining the economic optimum stocking rate, farmers in South Africa tend to increase stocking rates in response to diminishing financial returns. This results in stocking rate exceeding the carrying capacity of the area (Danckwerts 1989), which may lead to range degradation (Danckwerts & King 1984). The economic stocking rate model has limited ability to simulate the outcome of alternative management strategies (Gordon & Hutchings 1993), price structures and user objectives (Seligman *et al.* 1989), and does not cope well with temporal and spacial production variability (Stafford-Smith & Foran 1993) which may lead to fluctuating economic optimum stocking rates. Jameson (1989) argued that economic analyses usually assume that conditions for convergence to an optimum exist while ecologists frequently cite concepts of hysteresis and discontinuity to indicate that convergence to an optimum stocking is not possible. Clearly, the conceptual model clearly cannot be applied to highly variable situations and has obvious limitations, but it does provide a useful basis for integrating the biological and economic principles which determine the optimum level of production.

MANAGING UNDER UNCERTAINTY

Mean annual rainfall is inversely correlated with its coefficient of variation (Tyson 1986). Rainfall variability in arid and semi-arid regions is consequently a major determinant of system dynamics (O' Connor 1985; Cowling 1986; Ellis & Swift 1988; Hoffman & Cowling 1990), which results in spatial and temporal variation

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[7] In theory, if variable costs decreased to zero then the optimum economic stocking rate would equal that at which maximum production per hectare would be attained i.e. G_{max}.

in forage production and hence carrying capacity (Danckwerts & Tainton 1993). Rainfall may limit the potential for the introduction of pastures and so extensive livestock systems in arid and semi-arid regions are based primarily on rangelands, where forage productivity may vary by up to 800 percent (Stafford-Smith & Foran 1988), and carrying capacity by up to 700 percent, between seasons (Tainton & Danckwerts 1989). While system dynamics are strongly climate-driven, stocking rate and rainfall may interact to determine range composition dynamics (Peel *et al.* 1991; Hatch *et al.* 1993). The economic optimum stocking rate is consequently temporally-variable, while the probability of 'droughts'[8] may increase as stocking rate is increased (Noy-Meir & Walker 1986; Heitschmidt & Taylor 1991).

A number if stocking strategies may be developed to deal with variability (e.g. Foran & Stafford-Smith 1991; Danckwerts & Tainton 1993). Strategies may follow:

1) a low stocking rate approach, which may attempt to avoid the consequences of all but the most severe droughts (Foran & Stafford-Smith 1991). The relatively high costs of land and fixed inputs, and opportunity cost of wasted forage, may often preclude this option;

2) a trader or climate-dependent approach, which may attempt to track seasonal patterns in forage availability by marketing livestock at the onset of drought (Foran & Stafford-Smith 1991). Widespread destocking during droughts may lead to reduced livestock prices through market oversupply, while high costs are associated with error, i.e. failure to recognise the difference between a dry spell and a drought may result in poor stock condition and low prices. As droughts are often regional the availability of suitable livestock for restocking may be limited (Danckwerts & Tainton 1993). In addition, unfavourable tax conditions may be associated with the sale and repurchase of livestock, which may influence financial viability of the enterprise during stress periods;

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[8] Although drought may be associated with periods of reduced or below average rainfall, these effects are often confused with forage deficits as a consequence of increased stocking (Gillard & Monypenny 1990).

3) 'moderate' levels of stocking, which may be maintained through droughts by supplementation of conserved or bought feed. The size of the conserved forage reserve should be directly related to rainfall variability (Jones 1983). Purchase of supplementary feed during droughts may be expensive as demand-driven price increases may occur; and

4) a combination of low stocking (approach 1) and the trader approach (2), which may involve levels of breeding stock (set at the forage production potential of dry seasons) complemented with speculative growing stock, which may be readily marketed during dry periods. In addition, growing stock are less likely to be affected by forage deficits than reproductive stock and may recover rapidly following droughts (through compensatory growth). The major advantage of this approach is the ability to capitalise on specific market conditions, such as during the post-drought seasons when stock prices may rise in response to demand created by restocking.

Although these options represent a broad range of strategies which may be adopted, strategic planning is important in dealing with the consequences of a variable forage environment. This may best be achieved through a flexible management approach, such as adaptive management (Walters 1986; Stuart-Hill 1989), although such approaches are rarely applied to situations where they are most appropriate (Stafford-Smith & Foran 1993).

Rainfall variability in humid regions, in contrast, is low and forage production and carrying capacity are largely consistent between seasons. While the amount of forage may not be limiting, quality may become limiting during the dormant season and this may limit livestock production. The high rainfall does provide opportunities for the integration of dryland or irrigated pastures into rangeland systems, which may alleviate the consequences of variability in forage quality. As range composition dynamics are strongly determined by biotic interactions (Cowling 1991), excessive stocking may lead to changes in range composition (van Niekerk *et al.* 1984; Hardy & Hurt 1989). Although the separation of heterogenous vegetation types with fencing and the application of rotational grazing systems is widely recommended in southern Africa to control the frequency and intensity of defoliation of individual grass plants, increase livestock productivity (Roux 1968; Booysen & Tainton 1978), ration forage and allow the accumulation of forage reserves (Danckwerts & Tainton 1993), maintain or improve range condition (Foran *et al.* 1978), and control the extent of area-selective grazing and facilitate herd management (O' Reagain & Turner 1992), there is little empirical evidence to support the benefits of rotational grazing (O' Reagain & Turner 1992). In addition, a major disadvantage of rotational grazing systems is the high cost of fencing (ca R3 000 km^{-1} for 6 strand cattle fencing and up to R30 000 km^{-1} for game fencing), which may be economically justified only if discounted over periods greater than the planning horizons of most graziers (Mentis 1991). Electric fencing, at less than a third of the cost of barbed-wire fencing, may provide a viable alternative and considerably reduce the development costs of fencing. Maintenance costs of electric fencing may, however, be higher than conventional fencing over the long-term. Importantly, rotational grazing may allow considerable management flexibility, for example where single-sire mating is applied.

DEVELOPMENT COSTS & OPTIONS

The development of sustainable livestock production systems implies that the range manager will incur opportunity costs (Figure 4).

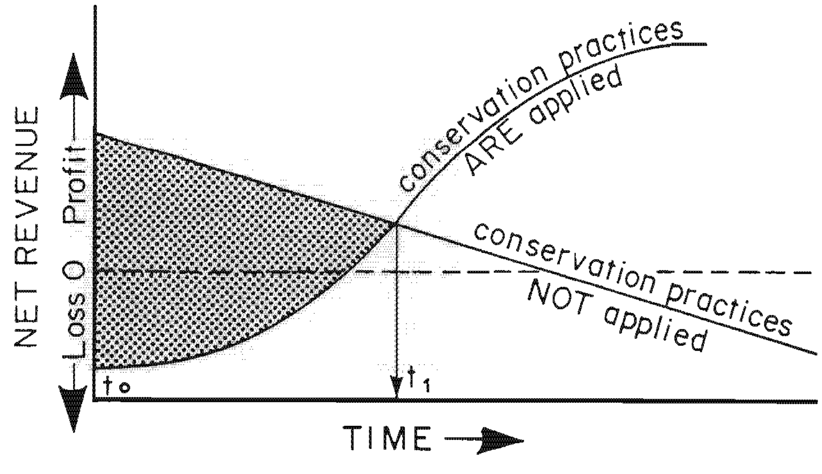

Figure 4 The effect of the adoption of conservation measures on net revenue over time (after Barlowe 1982 cited by Danckwerts & Tainton 1993)

Although the value of a resource may depend on future expected returns (Chapter 3), initial net return must exceed welfare requirements and the time (t_0 - t_1) must not exceed the strategic planning horizon of the range manager for conservation measures to be considered (Danckwerts & Tainton 1993). The question which arises is does investment in development and conservation farming pay?

Strategic planning of developments, such as pasture establishment, fencing or watering points, which may lead to increased future returns, are strongly influenced by the time value of money. One Rand invested today will be worth less in one year through depreciation associated with inflation, risk and opportunity costs (Workman 1986). Discounting provides a mathematical means of calculating the present value of a future sum of money as follows:

$$V_0 = V_n (1+i)^{-n}$$

where V_o is the present sum, V_n is the future value after n years and i is the interest rate (after Workman 1986).

The difference between the discounted investment costs and benefits of any development provides the net present value (NPV) of investment in range development (Stuth *et al.* 1991). If NPV is zero then investment in development is estimated to earn the same rate of return as alternate investments. A NPV greater than zero suggests that the rate of return on development exceeds the return from other opportunities, while a negative NPV suggests that development is not economically justified. The choice of interest rates on which to base discounted present values may vary (Workman 1986) and consequently influence development decisions.

Although sedentary agriculture, with the sub-division of heterogenous landscapes, characterises arid and semi-arid production systems in South Africa, the development costs of fencing and water provision in areas of low and variable productivity may exceed potential returns. The NPV of development in these environments may be highly negative. Present systems may therefore be neither ecologically nor economically sustainable without generous state subsidies. A return to nomadic pastoralism has been suggested to deal with temporal variability in forage productivity (Cowling 1991; Danckwerts & Tainton 1993), although this cannot be addressed in isolation of current land-tenure arrangements in commercial agriculture based on private ownership. In contrast, humid systems may yield greater returns on investment than arid/semi-arid systems and positive NPV of development may prevail.

THE ROLE OF SIMULATION MODELLING APPROACHES

Integration of a large number of environmental and economic factors which determine productivity is clearly a complex, but fundamental part of developing viable management strategies. Computer-based simulation models provide a valuable basis for the examination of alternative management strategies. Simulation modelling provides a basis from which dynamic and stochastic processes, too complex for rigid mathematical models (Anderson 1974), may be addressed in environments where little quantitative data are available (Starfield & Bleloch 1991). A trend toward increased availability of cost-effective personal computing has revolutionalised the global economy, and allowed smaller firms to employ the same levels of logistical support and to access financial models previously limited to large firms (*The Economist* 1993). Similarly, enhanced computing power has accelerated the application of simulation modelling to range management, and includes the development of bioeconomic and decision-support models. Models may also be useful for defining research priorities (Black *et al.* 1993).

Bioeconomic simulation modelling may be described as the mathematical description of integrated biological and economic components of agricultural production systems (Denham & Spreen 1986). Simulation models have been developed at the scale of 1) component processes, which has led to the development of theoretical models, 2) component parts of systems, 3) whole systems or farms, and 4) regional level based on farm surveys. The major advantage of bioeconomic modelling is the ability to deal with the complexity of inter-related biological and economic relationships, which are often difficult to conceptualise, and which cannot be examined in traditionally designed experiments (Forbes & Oltjen 1986). Bioeconomic models, which may be used to predict the influence of varying levels of input variables on production and profitability, may be either deterministic or stochastic. Deterministic models may predict the outcome of decisions over time with no attached probability distribution. Stochastic models incorporate random variables with the consequence that the output of successive simulations may vary despite the input data and constants remaining the same. A variety of models include BEEF (Loewer & Smith 1986), PASTURE (Tharel *et al.* 1986), GRAZPLAN (CSIRO 1986), STATIC (Taylor & Rudder 1986), GRAZE (Parsch & Loewer 1987), SHEEPO (Whelan *et al.* 1987), RANGEPACK - Herdecon (Stafford-Smith & Foran 1988; Foran *et al.* 1990), DYNAMA (Holmes 1989) and SMART (Hart 1989).

Model development has not proceeded in any logical, hierarchial order, but rather sporadically to satisfy the specific needs of researchers (Forbes & Oltjen 1986), with little progression from the development of sub-system modelling to the development of whole-farm models. Chudleigh & Cezar (1982) argue that the extent of duplication of models suggests that a more co-operative approach to modelling is required to prevent the concept from deriving mere heuristic value. Possible reasons for the slow progress in model development may be that:

1) models are often poorly documented or not published (Chudleigh & Cezar 1982);
2) model developers may have different objectives which leads to the development of discipline-specific models;
3) models developed under specific conditions may have limited value when extrapolated beyond the scope of the original data; and
4) model developers work in diverse environments, where the relative importance of input variables may vary considerably.

The global trend towards the privatization of science has led to the development of decision-support models based on market research and which identify a clear need in a defined market. The Australian CSIRO's RANGEPACK HerdEcon (Stafford-Smith & Foran 1988) is a good example of such a model where range managers, extension personnel, administrators and scientists provided the background for model development. The model has been applied to examine the consequences of various stocking strategies (Stafford-Smith & Foran 1988, 1990, 1992, 1993; Foran *et al.* 1990; Foran & Stafford-Smith 1991). However, despite the availability of detailed decision-support software, application remains limited amongst range managers and complex models remain largely the tools of researchers.

Although the traditional paradigm (e.g. Dyksterhuis 1949) of unidirectional composition change in rangelands in response to disturbance (eg. Foran *et al.* 1978; Tainton 1986; Bosch *et al.* 1989) has been challenged by models based on a multi-directional change between discrete states (Westoby *et al.* 1989), where thresholds may restrict movement between states (Friedel 1991), few attempts have been made to integrate models of ecosystem dynamics into bioeconomic models. A frame-based modelling approach (Starfield *et al.* 1993) may be useful to describe system dynamics within a framework of discrete states and which may be linked into economic models. Integration of conceptual models of range dynamics and bioeconomic models may be an important consideration where the influence of various stocking strategies or combinations of livestock enterprises may influence ecosystem processes, such as rates of soil loss or bush densities. These may provide feedback into the system which may influence future levels of productivity and hence sustainability.

Simple spreadsheet models may provide invaluable management tools to assist in strategic management planning and may be used to address "what if" questions. Easily constructed deterministic bioeconomic models may be used to assess the outcome of various strategic management decisions, determine the optimum combinations of types and classes of livestock and calculate optimum economic stocking rates. By manipulating input or independent variables, such as fixed and variable costs, the effect on output or dependent variables, such as net return per livestock unit or area, may be evaluated.

For example, what would be the impact of an increase in interest rates, the cost of labour or an increase in the beef price on the profitability of an extensive beef long-yearling production system in the semi-arid savanna of Natal? Although these may be calculated by hand, what would be the effect if all three occurred simultaneously? A simple spreadsheet model could address these issues and allow the user to test a number of scenarios and the sensitivity of net return to various cost parameters (Table 1). A number of linked spreadsheet models could be developed to integrate components in multi-species systems.

Table 1 Output from a simple spreadsheet model for a beef long-yearling production system in the semi-arid savanna of Natal relating fixed and variable costs to income and return and reflecting a) the influence of a 15 percent increase in the beef price, b) a 20 percent increase in the cost of labour, c) a one percentage point increase in the interest rate and d) simultaneous increases in the beef price (15 percent), labour cost (20 percent) and interest rate (one percentage point) on net returns to land and management

Assumptions			
animal units		800 LSU	
land area		3200 ha	
stocking rate		0.25 LSU ha^{-1}	
interest rate		15 percent	
beef price		R2.50 kg livemass	
initial animal mass		300 kg	
summer mass gain		180 kg	
Cost structures			
fixed costs ha^{-1}	**R**	**variable costs LSU^{-1}**	**R**
improvements	2.50	supplementary feed	73.00
vehicles	6.90	veterinary	18.00
machinery	3.40	marketing	21.00
labour	20.40	miscellaneous	5.00
electricity	4.40	interest	112.50
Total	37.60		229.50

Output	Total	ha^{-1}	LSU^{-1}
Income	360000.00	112.50	450.00
Costs	303920.00	94.98	379.90
Net return	56080.00	17.53	70.10

a) Influence of a 15 percent increase in the beef price

	Total	ha^{-1}	LSU^{-1}
Income	414000.00	129.38	517.50
Costs	317420.00	99.19	396.78
Net return	96580.00	30.18	120.73

b) Influence of a 20 percent increase in the cost of labour

	Total	ha^{-1}	LSU^{-1}
Income	360000.00	112.58	450.00
Costs	316208.00	98.82	395.26
Net return	43792.00	13.69	54.74

c) Influence of a one percentage point increase in interest rates

	Total	ha^{-1}	LSU^{-1}
Income	360000.00	112.58	450.00
Costs	309280.00	96.65	386.60
Net return	50720.00	15.85	63.40

d) Simultaneous increases in the beef price (15 percent), labour cost (20 percent) and interest rate (one percentage point)

	Total	ha^{-1}	LSU^{-1}
Income	141000.00	129.38	517.50
Costs	331436.00	103.57	414.30
Net return	82563.20	25.80	103.20

CONCLUSIONS

Environmental variability and market uncertainty may strongly influence the outcome of any range management decision. This implies that, even within an adaptive management framework, it may be impossible to collect, assimilate and conceptualise the amount of information on which to base or to predict the long-term outcome of any range management decision. Temporal variability may only be conceptualised at the scale of the human lifespan. This emphasises the importance of simulation modelling approaches to assessing the impact of strategic decisions on long-term profitability and sustainability. Simulation models, ranging from simple spreadsheet models through to detailed decision-support models, may provide invaluable tools for the development of a pro-active management approach based on long-term sustainability. The challenge for range scientists is to provide appropriate data sets and relevant decision-support tools which integrate models of ecosystem processes with the biology and economics of the system.

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CHAPTER 3[1]

COSTING RANGE DEGRADATION: A SOUTHERN AFRICAN PERSPECTIVE

ABSTRACT

The consequences of range degradation through grazing have been well documented, although confusion arises where vegetation change as a consequence of grazing is described as 'overgrazing'. Despite evidence that conservative stocking yields greater financial returns than higher levels of stocking, range-users continue to apply stocking levels which lead to range degradation and reduced financial returns. Overgrazing is, however, a rational response to an uncertain environment with high discount rates and strong market distortions, and where the consequences of degradation are not readily apparent. An integration of ecological, economic and social components of the system may be required to address the challenge of range degradation through overgrazing.

INTRODUCTION

Agricultural challenges in the 1960's centred around the objective of maximising agricultural production to sustain the demand created by a world population explosion (*The Economist* 1991). World population growth has continued unabated and agriculture today faces the daunting task of feeding the world population within the context of widespread environmental degradation. Controversy surrounding global environmental change highlights such issues as ozone-layer depletion and global warming, although reconciling population demands with natural resource sustainability is perhaps a greater and more immediate threat. Although the area of cropland in developing countries is projected to increase by 10% to 850 million hectares by the turn of the century, the area of land per

[1] Hatch GP, Tainton NM & Ortmann GF 1993 Costing range degradation. Paper presented at the 28th Annual Congress of the Grassland Society of Southern Africa, Harare, Zimbabwe.

inhabitant will actually decline from 0.9 to 0.6 hectares during this period (*The Economist* 1991). Greater demands will therefore be placed on existing resources.

Recent changes in the vegetation of South Africa may be indicative of a world-wide trend in ecological degradation. Although Acocks' (1988) view, that vegetation change in South Africa was the consequence of mismanagement, has been challenged by suggestions that rainfall is the primary agent determining vegetation dynamics (e.g. Ellis & Swift 1988; Hoffman & Cowling 1990; Behnke & Scoones 1993), there is considerable evidence that rangelands degrade under extreme stocking pressure (e.g. van Niekerk *et al.* 1984; Arianoutsou-Faraggitaki 1985; Foran & Stafford-Smith 1991). Although it has been argued that overgrazing may lead to the dominance of annual or invader species (Jackson 1985), desertification (Babiker 1985) and increased incidence of famine (Sinclair & Fryxell 1985), climate and grazing clearly interact to determine range dynamics. Despite considerable research effort, little has been achieved in preventing, or at least restricting, the rate of range degradation in Southern Africa (Thomas 1992). Tainton (1987) attributed the failure of collective effort to address the problem of environmental degradation, to the lack of altruistic actions by individuals in a materialistic society which responds to a Darwinian survival instinct (Hardin 1977), regardless of the cost to society and future generations.

Against the background of large-scale environmental degradation is the need for livestock production to satisfy the demands of the world's population. This must be reconciled with resource conservation to ensure sustained return, and emphasises the need for the development of livestock systems which are both ecologically and socially sustainable.

While climatic and market uncertainties present complex management problems for livestock producers, the major decision affecting the level of livestock production is stocking rate (Riechers *et al.* 1989). The stocking strategy[2] selected by the land-manager may have a major impact on short-term profitability (Hatch & Tainton 1993a), while in the long-term may influence the resource-base (Foran & Stafford-Smith 1991). Stocking strategy decisions made by land-users must satisfy both economic and biological objectives to ensure sustained livestock production, while neglecting either implies that the system will ultimately fail. Returns from rangelands may be in the form of livestock products in commercial production systems or internalised in the form of livestock numbers in the subsistence context. Overstocking which leads to changes in range condition and reduced forage production potential, and hence lower levels of livestock production, will result in lower economic returns and reduced human welfare. In South Africa, farmers tend to increase stocking rate in response to diminishing financial returns which results in stocking rate exceeding the carrying capacity of the area (Danckwerts 1989). While such overstocking may be questioned even on the basis of short-term economic return, its long-term effects may result in range degradation (Danckwerts & King 1984). This occurs irrespective of user objectives and the system is locked into a downward spiral of resource degradation. Major system change may occur, with stability reached at a new and lower level of production, and a threshold beyond which the system cannot recover is established (Friedel 1991). This paper examines the bioeconomic consequences of various stocking strategies and provides a conceptual framework from which the issue of range degradation may be examined.

COSTING OVERGRAZING AND RANGE DEGRADATION

While rangeland degradation through overgrazing has been widely documented, confusion arises where vegetation change as a consequence of grazing is described as overgrazing (Wilson & MacLeod 1991). It is important to recognise that

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[2] Stocking strategy may be defined as the combination of strategic management decisions made by the range-user to satisfy ecological, economic and social objectives. These incorporate stocking rate, burning timing and frequency, grazing system, type and level of supplementation, breeding, and marketing decisions.

rangelands evolved under a specific set of conditions which are changed under grazing by domestic herbivores. Change may not be detrimental where livestock production remains unchanged. Range degradation in livestock production systems may be defined as a permanent decline in the rate at which land yields livestock products (Abel & Blaikie 1989) or as a measure of the reduction in the capacity of a landscape to produce forage from rainfall (Walker 1993). Perceptions of overgrazing may also be value-based and dependent on user objectives. Optimum range condition may consequently vary between seasons and management objectives (Stuart-Hill & Aucamp 1993). Overgrazing may therefore be defined as grazing which leads to changes in the resource-base, reduced productivity and human welfare and is therefore non-sustainable.

The influence of stocking strategy on the plant and soil resource may be strongly environment-dependent. Mean annual rainfall is inversely correlated with its coefficient of variation (Tyson 1986). Rainfall variability in arid and semi-arid regions is consequently a major determinant of system dynamics (Ellis & Swift 1988), which results in seasonal fluctuations in forage production and hence carrying capacity (Danckwerts & Tainton 1993). Stocking rate and rainfall interact to determine range composition dynamics (Peel *et al.* 1991), and drastic changes may occur over short time periods (Chapter 8). Range composition changes may be associated with decreased forage production (Danckwerts 1982, Turner 1990, Snyman & Fouche 1993), rather than any change in sward quality (Hatch & Tainton 1993a) and with reduced cover (Tainton 1981), which may increase the potential for soil erosion. Stocking decisions may attempt to avoid the consequences of drought through low stocking, or by marketing animals at the onset of drought (Foran & Stafford-Smith 1991), while overstocking may require extended periods of expensive supplementary feeding. The immediate costs of overgrazing in arid and semi-arid environments may therefore be reflected by increased periods of supplementary feeding, as a consequence of reduced forage production (Chapter 12, 13), while in the long-term soil loss may lead to the dominance of species of low forage production potential (Chapter 8). Either may lead to increased risk and reduced returns.

In contrast, rainfall variability in humid regions is low, and forage production and hence carrying capacity are largely consistent between seasons. Range composition dynamics are strongly determined by stocking strategy, with excessive stocking leading to changes in composition (van Niekerk *et al.* 1984, Hardy & Hurt 1989). Overgrazing may lead to reduced forage production (van Niekerk *et al.* 1984, Turner 1990), while as composition changes animals are forced to graze less preferred species (Hatch & Tainton 1993b). O' Reagain & Mentis (1990) argued that, as no relationship between range condition and dietary quality was evident, the relationship between range condition and animal performance may be questioned. In contrast, Turner (1990) indicated that range condition played a major role in determining animal production from rangeland. The relationship between range condition and animal production in the humid grasslands is therefore a contentious issue. Importantly, changes in range composition may influence the length of the grazing season and a 'souring' of the sward may require extended periods of supplementation. As supplementary feed comprises 40% of the variable cost of livestock production in the humid grasslands of Natal (Crockart & Berry 1993), range users tend to minimise the period of supplementation to reduce costs. Inadequate supplementation may consequently have a detrimental effect on animal production (O' Reagain & Mentis 1988) and low calving percentages may be related to inadequate winter supplementation of beef cows (Meaker 1978). The cost of overgrazing in humid environments may therefore be reflected by increased supplementation costs (Hatch 1992), rather than by direct reductions in animal production. Failure to provide adequate supplementation may result in reduced calving percentages and hence lost income. These may be exacerbated by grazing-induced changes in grass species composition.

The influence of stocking strategy on soil loss and runoff

Although erosion is a natural process which has shaped the southern African landscape, rates of soil erosion under present land-use have been drastically increased (Scotney & McPhee 1992) and could be as high as 12 to 22 times geological rates of loss (Martin 1987).[3] Although estimates of soil loss for southern African rangelands are highly variable (Table 1), increased rates of soil loss have been related to increased stocking (Dunford 1954; Rauzi & Hanson 1966; Sartz & Tolsted 1974; Warren *et al.* 1986; Venter *et al.* 1989; Clarke 1993) and a decline in range condition (Snyman *et al.* 1987). Grazing may increase the rate of soil loss and runoff through a reduction in grass cover (Zobisch 1993), while a change from perennial to pioneer-dominated range may be associated with increased rates of soil loss, runoff and reduced infiltration (Snyman & van Rensburg 1986). This may result in a loss in rangeland productivity (Scotney & McPhee 1992).

Table 1 Soil loss measures for southern African rangelands (t ha^{-1} annum^{-1})

Soil loss (t ha^{-1} annum^{-1})	Source	Description
3.0	Alder (1985)	South Africa
18.4	Anon. (1988)	Lesotho
6.2	Snyman *et al.* (1986)	pioneer range
0.9	Snyman *et al.* (1986)	climax range
1.0 - 11.0	Abel & Stocking (1987)	Botswana
0.3	Ivy (1991)	good cover
4.0	Ivy (1991)	poor cover
2.8	Clarke (1993)	stable soil
9.8	Clarke (1993)	unstable soil

While it should be recognised that other forms of land-use, such as forestry and cropping, result in extensive soil disturbance and may have a far greater impact than grazing on the rate of soil loss per unit area, rangelands comprise 80% of the land area of southern Africa. The accumulated soil loss from rangeland may consequently be considerable. For example, an increase in the rate of soil loss from 0.5 to 1.5 t ha^{-1} annum^{-1} in the Lowveld and Arid Lowveld (Acocks 1988),

[3] Estimated to be 0.13 - 0.16 t ha^{-1} 1000^{-1} years (Braune & Looser 1989).

which occupy an area of 1.2 m ha in Natal (Edwards 1974), would imply an increase in soil loss from 0.6 to 1.8 m t annum^{-1}.[4]

The concept of externalities

While part of the costs of overgrazing are internalised and borne by the range-user in the form of reduced financial returns, other costs are less obvious and are external to the system (Figure 1).

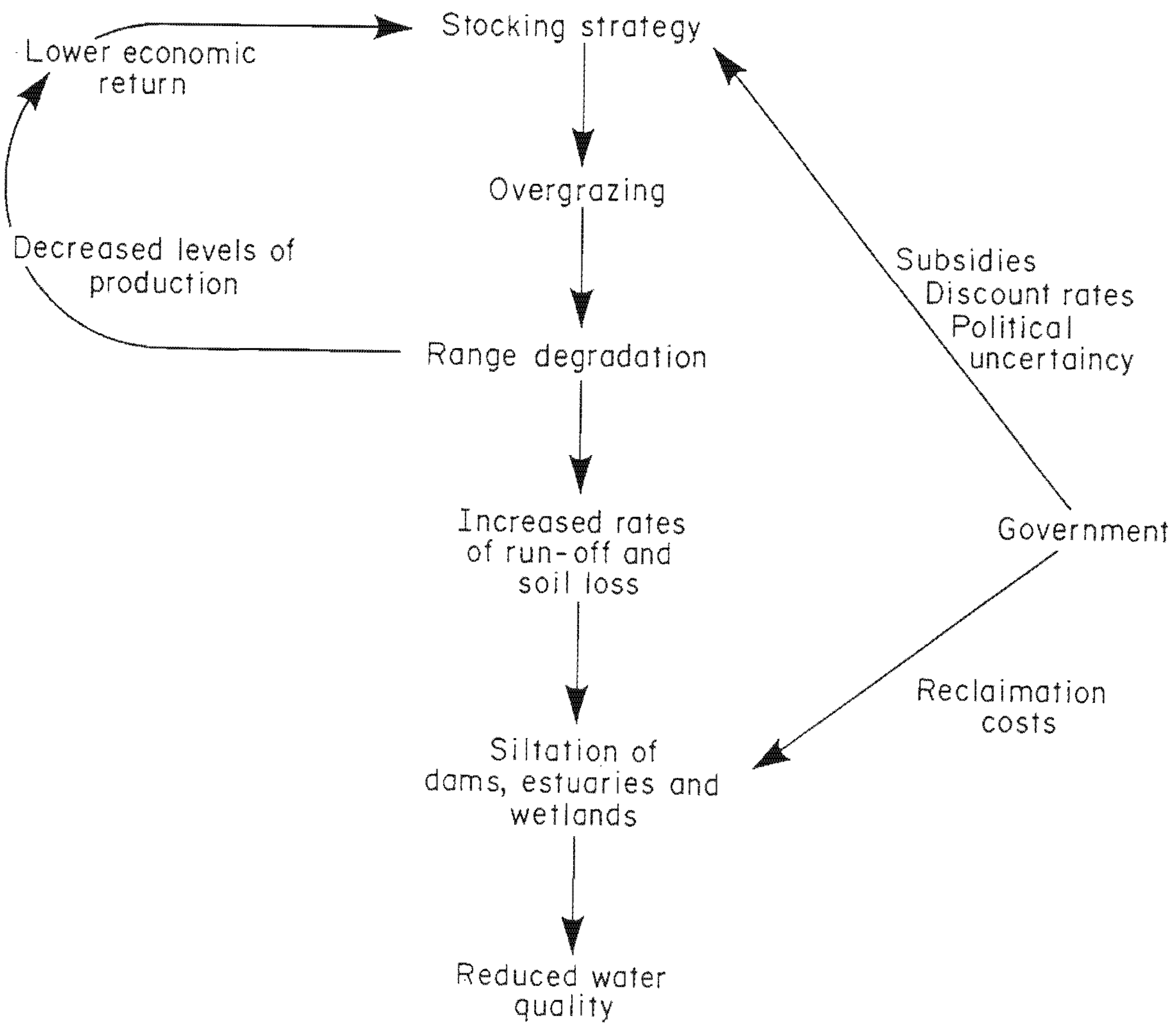

Figure 1 The conceptual relationship between the costs and effects of overgrazing

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[4] Importantly, not all soil is lost to the system and patterns on erosion and deposition may characterise arid and semi-arid landscapes (Pickup 1985, 1991).

Reductions in water quality

While soil loss through overgrazing may lead to decreased range productivity, soil pollution of rivers, estuaries, wetlands and dams through increased sedimentation rates may result in reductions in both water quality and storage capacity. This may present a major limitation to future development in southern Africa. Sedimentation rates, as the consequence of all forms of land-use, for South African rivers range between 0.07 and 5.89 t ha^{-1} yr^{-1} (Rooseboom *et al.* 1992) and may be as high as 20.5 t ha^{-1} yr^{-1} for rivers in Lesotho (Makhoalibe 1984). The costs of reduced water quality, which includes reduced lifespans of dams, dam replacement and dredging of estuaries, are borne by the state and therefore by society. Scotney & McPhee (1990) estimate that 32% of the capacity of the Welbedacht Dam on the Caledon River was lost to siltation during the first three years of its life, while Braune & Looser (1989) estimate that the annual cost of soil loss in South Africa may exceed R50 m.

THE CONCEPT OF DISCOUNTING AND DISCOUNT RATES

While stocking strategies based on low or conservative stocking in arid and semi-arid environments may aim to decrease the risk of forage shortages, and in humid environments reduce the risk of range composition change, the range-user will incur some form of opportunity cost. Sustained return cannot be economically justified if the long-term return is less than the return on immediate and full exploitation (le Maitre 1990). This is the premise on which continued commercial whaling is justified (le Maitre 1990). Mentis (1991) applied this approach to compare single and multi-camp grazing systems and concluded that under high interest rate conditions (i.e. high opportunity costs), multi-camp systems could not be economically justified.

Range managers are therefore faced with the dilemma of stocking conservatively to ensure sustained return, but at some cost of lost production. Mentis (1985) argued that since conservation farming fails to maximise benefits in the short-term, graziers are tempted to profit at the expense of sustained production. The value of a resource is, however, dependent on expected future net returns and therefore investment in conservation measures which increase future expected returns will increase the value of the resource. Cline (1993) argued that there are conceptually two rates which may be used to evaluate resource management decisions: opportunity cost or the social rate of time preference (SRTP). The SRTP is simply the value people place on present consumption over future consumption. The SRTP value incorporates a 'utility-based' discount rate, which takes into account declining marginal utility as income increases. Thus the greater an individual's wealth the lower the 'utility-based' discount rate and the greater the value placed on future consumption or conservation. Conversely, the value may increase with decrease in income and greater emphasis is placed on current consumption. This has major implications for resource management decisions made by range-users under subsistence conditions where basic needs take preference.

If graziers place a premium on current consumption then the rate at which the future is discounted must be high. What factors of the SRTP value may enhance the tendency for current consumption?

THE ROLE OF GOVERNMENT AND MARKETS

Government activity, which influences both monetary policy and hence discount rates, and the level of political uncertainty, influences the SRTP value. Arguably, the SRTP presently applied to rangeland stocking decisions in South Africa may be greater than the opportunity cost due to the high level of political uncertainty. Present land redistribution policies in Zimbabwe, which are aimed at short-term political benefit rather than concerns with equity or efficiency (*The Economist* 1993), are likely to enhance the SRTP value and favour current consumption.

In addition, government may influence resource management decisions through subsidies. The provision of 'drought' subsidies in arid and semi-arid areas may decrease individual farmer's financial risk and encourage increased or optimistic stocking levels.[5] Further, changes in range condition may increase susceptibility to 'drought' and forage shortages may occur during 'normal' rainfall years (Snyman & Fouche 1993). The association of subsidies and agriculture is not unique to South Africa. Governments have suppressed market forces in agriculture with subsidies, tariffs, quotas and monopoly-purchasing boards. Consumers in industrial countries pay US$300 billion a year in taxes and higher prices to support agriculture, while the net welfare loss caused by farm policies is US$100 billion a year (*The Economist* 1992). Efforts to liberalise global trade in agricultural products through the GATT talks were delayed largely through protectionist attitudes of governments who seek to protect agriculture within their own countries.

Range managers cannot be expected to apply management strategies which are financially untenable, nor be expected to act altruistically given the lack of evidence of such behaviour. This formed the central thesis of Hardin's (1968) widely debated argument that communal management of grazing resources ultimately leads to resource degradation. Subsidisation or legislation may be necessary to ensure sustained resource-use where society has an interest in conservation (Danckwerts & Tainton 1993), i.e. to reduce the cost of externalities. Subsidies need, however, to be carefully and fairly applied to sustain agricultural production without creating forms of market distortion which lead to resource degradation.

Market uncertainty, over which individual range-users have little control, is a further source of risk for livestock producers. Workman *et al.* (1972) argue that although the demand for beef in the United States is relatively price-flexible (which implies that a one percent decrease in the quantity of beef produced results in a greater than one percent increase in the price of beef and an increase in total market revenue), increases in cattle numbers by individual range-users are

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[5] Subsidies to farmers in drought-affected areas may increase expected returns and introduce a new source of risk that may raise farmer's discount rates. Both precipitate higher stocking rates.

economically rational decisions as they have no direct control over the market. The control of meat prices through state marketing boards, which may influence the quantity of meat demanded, may create further market distortions and lead to increased stocking rates on rangelands (provided meat prices are higher than market-clearing prices).

DISCUSSION

Despite evidence to suggest that economic return may be optimised under stocking rates considerably lighter than those at which maximum production per unit area is attained (Wilson & MacLeod 1991), graziers continue to adopt stocking strategies which lead to changes in range composition and increased rates of soil loss, and hence lower productivity and profitability. This tendency continues to vex rangeland researchers and extensionists. A number of factors may contribute to this conundrum. Grassland scientists have in the past concentrated on the development of techniques to quantify range condition change (e.g. Foran *et al.* 1978; Tainton 1986; Bosch *et al.* 1987) and attempted to understand system dynamics (e.g. Dyksterhuis 1949; Westoby *et al.* 1989; Laycock 1991). Few attempts have been made to relate these to economic return, or to reductions in return which may be associated with range composition change. Integration of the dynamics of range change and social and ecological sustainability have received even less attention. Where integration has been attempted, the complexity of the management challenges facing producers (e.g. variable rainfall, fluctuating product prices) have often been overlooked.

Despite the dependence on rangelands by people in the developing areas of southern Africa, agricultural needs carry a relatively low priority relative to more basic needs (Thomas 1992). Commercial range-users are likely to ascribe a similar low importance to range management decisions, possibly because the consequences of overgrazing are slow and insidious, often difficult to quantify and not immediately apparent. In addition, the direct costs of overgrazing are not readily quantified. This leads to the short-term receiving preference over long-term considerations.

Why then do range users overgraze? Simply, under conditions of high discount rates and uncertainty, and where the costs of resource degradation are largely externalised and borne by society, overgrazing is a rational decision for any individual. Government subsidies reduce the risk associated with low rainfall and forage deficits, decrease the farmer's discount rate and increase expected future returns which encourages optimistic or over-stocking. Poorly applied subsidies may consequently enhance the rate of range degradation through reduced risk and market distortions.

Range-users are unlikely to respond to impassioned pleas to conserve rangelands. The costs of overgrazing, both within and external to the system, need to be quantified to evaluate the impact of various stocking strategies on profitability. These are fundamental to addressing the process of range degradation through overgrazing, while failure to consider, and correctly apportion, these costs will see range degradation continue unabated. Simulation modelling and the development of decision-support tools may provide a useful basis from which the costs of various management actions may be assessed.

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CHAPTER 4

THE LOWVELD OF NATAL: STUDY AREA, EXPERIMENTAL DESIGN AND MANAGEMENT

INTRODUCTION

The semi-arid savanna or Lowveld and Arid Lowveld (Acocks 1988) occupy an area of ca 12 000 km^2 or 14% of north-eastern Natal (Edwards 1974) on the broad coastal plane between the Indian Ocean and the Inhlangdena mountains, and extending inland along the major river systems (Figure 1). Altitude varies between 150 and 600 m above sea level. The vegetation, which delimits the southern extent of the migration of true tropical species (Werger 1978), comprises both a tree and grass layer. The heavy soils of the Komatipoort System are characterised by an open *Acacia nigrescens-Sclerocarya birrea* woodland with a grass sward dominated by *Themeda triandra, Panicum maximum* and *P. coloratum*. Over-utilization is often associated with increased proportions of *Sporobolus iocladus, S. nitens* and *Urochloa mosambicensis*. The sandy, granite-derived soils are characterised by a *Combretum apiculatum-Terminalia sericea* woodland with *Hyperthelia dissoluta* which degrades to *Digitaria eriantha* with over-utilisation (Acocks 1988).

Rainfall may vary considerably, both temporally and spatially and, while the mean annual rainfall ranges between 500 and 800 mm, seasons of less than 100 mm or over 1000 mm may occur (CCWR 1993). Cropping potential is consequently limited and is confined to the valley systems of the major rivers such as the Umfolozi and Pongola systems. Important sub-tropical products include sugar-cane, cotton, citrus and bananas. Maize and sorghum are important crops in the subsistence areas of kwaZulu.

Historically Zululand was a hunters paradise with a broad spectrum of game species (Delagorgue 1847). Eradication of the tsetse fly in the first half of this century (Steele 1968) saw a rapid expansion of cattle enterprises into the Lowveld, largely at the expense of wildlife. Various game species remain locally abundant. These include kudu (*Tragelaphus strepsiceros*), impala (*Aepyceros melampus*), nyala (*Tragelaphus angasii*), steenbuck (*Raphicerus campestris*), reedbuck (*Redunca arundinum*), warthog (*Phacochoerus aethiopicus*), grey duiker (*Sylvicapra grimmia*), and bushbuck (*Tragelaphus scriptus*). Less common species within conservation areas include waterbuck (*Kobus ellipsiprymnus*), zebra (*Equus burchelli*), giraffe (*Giraffa camelopardalis*), hippo (*Hippopotamus amphibias*), white rhino (*Ceratotherium simum*), black rhino (*Diceros bicornis*) and elephant (*Loxodonta africana*).

Extensive land-use systems range from cattle through to mixed cattle/game and various forms of wildlife enterprises, including sport and trophy hunting, and non-consumptive uses such as tourism. Bush encroachment in cattle systems, and where fires are largely excluded, presents a major management problem, particularly as the costs of bush eradication are high. Wildlife are therefore increasingly seen as an alternative to cattle, as regional tourism initiatives increasingly focus both nationally and internationally on ecotourism potential. Cattle remain, however, regionally important in both subsistence and commercial systems.

Cattle numbers are largely maintained on commercial ranches, despite frequent droughts[1], through supplementation with conserved feeds (important pasture species include *Cenchrus ciliarus* and various *Cynodon* species) and sugar-cane residues. Considerable stock mortalities occur under subsistence pastoralism during droughts. Although goats play an important role in communal grazing systems, due to their resilience to restricted forage availability and ability to both graze and browse, they have been largely ignored by commercial producers despite the availability of local markets.

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[1] Although drought may be associated with periods of reduced or below average rainfall, these effects are often confused with forage deficits as a consequence of increased stocking (Gillard & Monypenny 1990).

STUDY AREA

The study was conducted on two grazing trial sites (Appendix 1) established in 1986 as part of the Northern Natal Range Programme (Figure 1) (Turner 1990; Hatch 1992) which is aimed at the development of a grazing capacity model for the drier regions of Natal (Turner 1990). Criteria for the selection of the sites and experimental design considerations were outlined by Turner (1990). Sites are run collaboratively with farmers who provide the land, cattle and labour assistance.

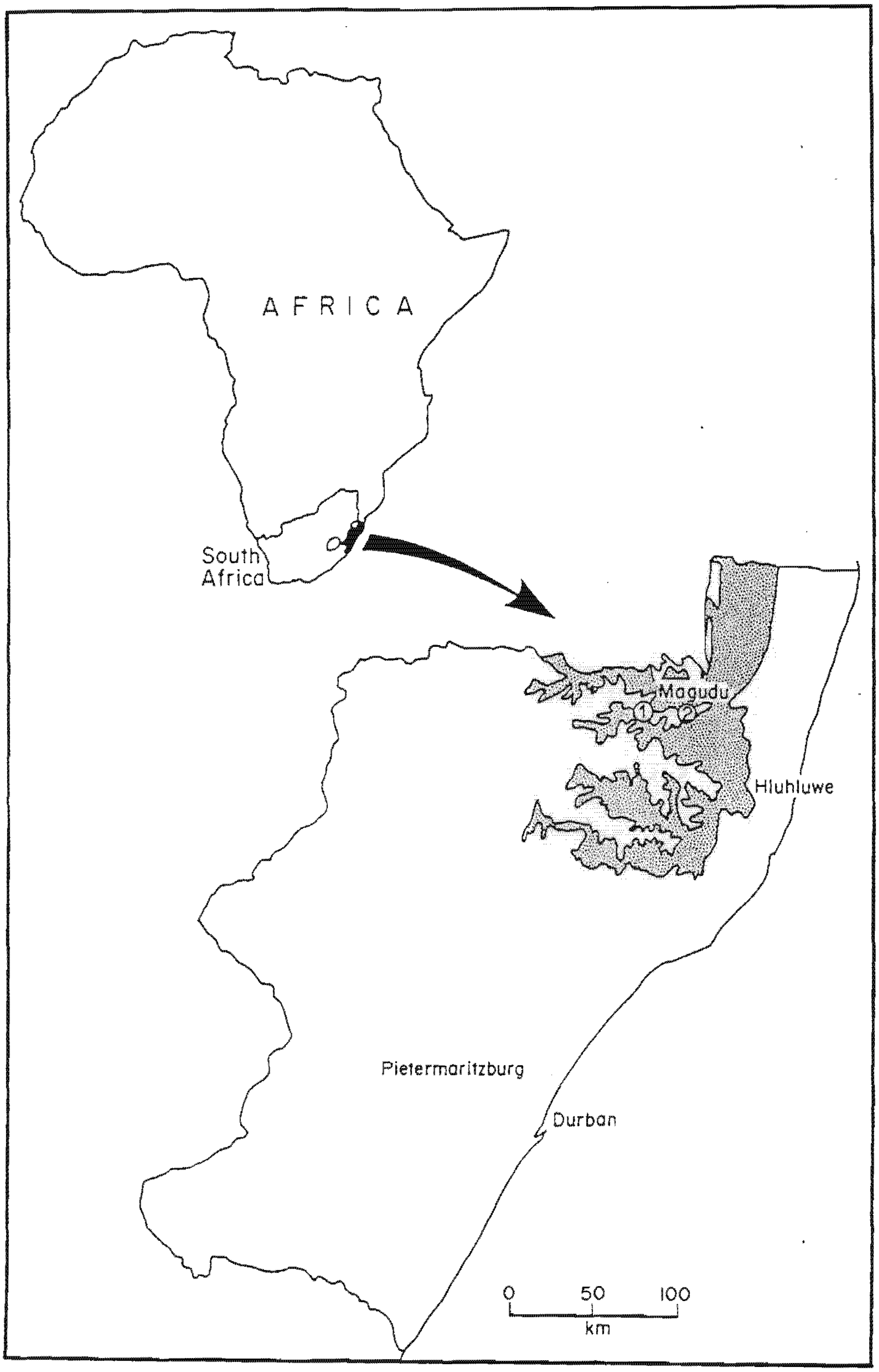

Figure 1 The Lowveld of Natal in regional context indicating the location of the sites at Llanwarne (1) and Dordrecht (2)

The sites differed in range composition (Appendix 2). Llanwarne (27° 35′ S 31° 45′ E), located at an altitude of 320 m above sea level, was dominated by *T. triandra*, *P. maximum* and *P. coloratum*. Dordrecht (27° 36′ S 31° 46′ E), located also at 320 m above sea level, with a history of heavy stocking was dominated by *U. mosambicensis*, *S. nitens* and *S. ioclados*.[2] Soils of the Komatipoort System, predominantly of the Swartland form with Clovelly, Hutton, Mispah, Glenrosa and Bonheim forms, were represented at the sites (Soil Classification Working Group 1991) (Appendix 3).

A two-camp rotational grazing system at each of three stocking rate treatments gave a total of six camps per site. The sites were burnt prior to establishment of the trial (winter 1985). The L1 and M1 camps at Llanwarne (see Appendix 1) were burnt again during the winter of 1990 following an accumulation of rank grass material during previous seasons. Camps received alternate spring and autumn rests, while the period of stay depended on the season and was therefore variable.

The current recommended grazing capacity for the Lowveld is 0.14 LSU ha^{-1} (Anon. 1981), although this recommendation is clearly not being followed by range managers. Currently applied stocking rates may range between 0.5 and 0.17 LSU ha^{-1} on commercial ranches and be as high as 0.7 to 1.0 LSU ha^{-1} under subsistence pastoralism. Fundamental to the development of a bioeconomic model was the examination of stocking rates which would occur within the range of the positive net returns (Chapter 2). Stocking rates which fell outside of the range of economic stocking rates were therefore not considered. The three treatments at each site were stocked at the start of each season with long-yearling Brahman-cross cattle to represent 'light' (0.17 LSU ha^{-1})[3], 'intermediate' (0.23 LSU ha^{-1})

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[2] Tree densities were similar at both sites.

[3] A large stock unit (LSU) was defined as a 450 kg steer gaining 500 g d^{-1} on herbage with a mean digestible energy content of 55% which provided 75 MJ of metabolisable energy per day (Meissner *et al.* 1983).

and 'heavy' (0.30 LSU ha^{-1}) stocking.[4] Six long-yearlings were allocated to the light and intermediate treatments at Dordrecht and seven to the heavy while at Llanwarne eight were allocated to the light and intermediate and nine to the heavy treatments respectively. The area of land was varied to provide the desired range in stocking treatments (Appendix 1). The actual stocking rates were subsequently recalculated for the summer and winter of each season using the animal unit equivalents approach of Aldermann & Barber (1973) which is based on the partitioning of total energy requirements (NE_{mp}) into energy for maintenance (NE_m) and growth (NE_p) as follows:

$$NE_m = 5.67 + 0.061W$$

$$NE_p = \frac{G(6.28 + 0.0188W)}{(1 - 0.3G)}$$

where W = livemass (kg)

G = livemass gain (kg)

The animal unit equivalence of an animal may be calculated by dividing the calculated NE_{mp} by 41.8 (the NE_{mp} of a LSU). Stocking rate (LSU ha^{-1}) may then be derived by dividing the number of LSU equivalents by the area (ha) (Turner & Tainton 1990). This approach to calculating stocking rates implied treatments were not constant over time and a general increase in stocking rate with summer mass gain and decrease with winter mass loss was anticipated.

[4] As it was not possible to exclude game from the sites stocking rates may be greater than those applied. It was not possible to record the occurrence of game on the sites and the assumption was made that game densities, and the impact of game was similar throughout the area.

The experimental cattle were supplemented with sugar-cane tops (20 kg LSU^{-1} d^{-1}) during periods of forage deficit which occurred during the winters of 1992 (for all stocking rates) and 1993 (for the intermediate and heavy treatments at Dordrecht).

DATA COLLECTION

Cattle were weighed at three-weekly intervals to give mean mass change over the previous 21 day period. Cattle were replaced annually in October. Herbage mass was estimated at each weighing date as the mean of 50 readings recorded with a pasture disc-meter (Bransby & Tainton 1977) on a fixed diagonal transect in each camp. Data were collected from November 1986 to June 1993 or for 120 three-week periods.

Range condition was assessed in 1986, 1988, 1990, 1993 and 1994 using the nearest-plant method (Foran *et al.* 1978) to collect 150 points at three meter intervals on each of two fixed transects in each camp, from which the proportional species composition was calculated for each camp following the nomenclature of Gibbs Russell *et al.* (1990). The distance to the nearest plant (cm) was recorded during the 1993 and 1994 surveys to provide an index of plant density and hence cover. Tree composition, density and structure were assessed in 1986, 1990 and 1994 within two fixed 5 m by 30 m transects located at opposite corners of each camp. All tree species in each transect were identified following the nomenclature of Pooley (1993) and placed into one of six heights classes (0-0.5 m, 0.5-1.0 m, 1.0-2.0 m, 2.0-3.0 m, 3.0-4.0 m and >4.0 m). Daily rainfall records were kept for each site. Data collection is ongoing at each site.

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CHAPTER 5

SEASONAL HERBAGE ACCUMULATION PATTERNS IN THE SEMI-ARID SAVANNA OF NATAL[1]

ABSTRACT

Grazing trials were established at two sites in the semi-arid savanna or Lowveld of Natal. The sites differed initially in range composition. Llanwarne was dominated by *Themeda triandra, Panicum maximum* and *P. coloratum*, and Dordrecht by *Urochloa mosambicensis, Sporobolus nitens* and *S. iocladus*. Three treatments were stocked with Brahman-cross cattle at each site to represent 'light' (0.17 LSU ha^{-1}), 'intermediate' (0.23 LSU ha^{-1}) and 'heavy' (0.30 LSU ha^{-1}) stocking. Pasture-disc meter data collected over 120 three-week periods were used to develop a step-wise multiple linear model (GRASS) to predict the amount of residual herbage at the end of summer (kg ha^{-1}) and the period (days) over which forage supplementation would be required to maintain animal mass. Residual herbage mass at the end of summer (kg ha^{-1}) was significantly related ($P<0.01$) to cumulative summer grazing days (LSU gd ha^{-1}), rainfall (mm) and range condition (indexed as the sum of the proportions of *T. triandra*, *P. maximum* and *P. coloratum*). The forage deficit period (days) over which herbage mass declined below a grazing cut-off of 1695 kg ha^{-1} was significantly related ($P<0.01$) to residual herbage mass at the end of summer. The GRASS model may provide a useful tool to assess the influence of various summer stocking strategies on residual herbage mass at the end of summer and the risk of winter forage deficits.

INTRODUCTION

Rainfall and hence forage variability in arid and semi-arid systems presents a major challenge to livestock producers (Chapter 3). Although forage or financial buffers

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[1] Hatch GP & Tainton NM 1995 Seasonal herbage accumulation patterns in the semi-arid savanna of KwaZulu-Natal. *Afr. J. Range & Forage. Sci.* (in press)

may minimise the effects of system variability (Pickup & Stafford-Smith 1993), excessive stocking will increase the probability of forage deficits and may influence the financial viability of the enterprise (Chapter 3). Range managers require detailed information on the influence of various stocking strategies on patterns in forage availability to be able to formulate viable management systems. Important decision times are 1) at the end of the summer, when stock numbers should be adjusted to the amount of residual herbage if winter supplementation and associated costs are to be avoided, and 2) at the start of spring, when decisions regarding summer stocking levels must be made in relation to expected rainfall. The rate of summer herbage accumulation may depend on stocking rates, rainfall and range condition (Noy-Meir 1978, Willms *et al.* 1986, Turner 1990). This paper examines the influence of stocking strategy, rainfall and range condition on seasonal patterns of herbage accumulation, the implications this may have for potential forage deficits and outlines the development of a herbage production model (GRASS) for the semi-arid savanna of Natal.

PROCEDURE

Grazing trials were established at two adjacent sites in the semi-arid savanna of Natal or Lowveld and data were collected over a seven year period (1986-1993). The sites differed in range composition. Llanwarne (27° 35′ S 31° 45′ E), located at a altitude of 320 m above sea level, was dominated by *Themeda triandra*, *Panicum maximum* and *P. coloratum*. Dordrecht (27° 36′ S 31° 46′ E), located also at 320 m above sea level, was initially dominated by *Urochloa mosambicensis*, *Sporobolus nitens* and *S. ioclados*. Soils of the Komatipoort System, predominantly of the Swartland form with Clovelly, Hutton, Mispah, Glenrosa and Bonheim forms, were represented at the sites (Soil Classification Working Group 1991).

Three treatments at each site were stocked at the start of each season with long-yearling Brahman-cross cattle to represent 'light' (0.17 LSU ha^{-1})[2], 'intermediate'

[2] LSU denotes large stock unit as defined by Meissner *et al.* (1983).

(0.23 LSU ha^{-1}) and 'heavy' (0.30 LSU ha^{-1}) stocking. Game, primarily impala (*Aepyceros melampus*), nyala (*Tragelaphus angasii*), kudu (*Tragelaphus strepsiceros*) and warthog (*Phacochoerus aethiopicus*), occurred at both sites. The actual stocking rates applied in each season were calculated for the summer and winter using the animal unit equivalents approach of Aldermann & Barber (1973). A two-camp rotational grazing system for each stocking rate treatment gave a total of six camps per site. Camps received alternate spring and autumn rests while the period of stay depended on the season and was therefore variable. Herbage mass was estimated at three-week intervals (from November 1986 to June 1993) as the mean of 50 readings recorded with a pasture disc-meter (Bransby & Tainton 1977), on a fixed diagonal transect in each camp. This gave a total of 120 recording dates for each camp. Mean disc-meter heights for each camp were converted to an estimate of herbage mass (kg ha^{-1}) using the generalised calibration equation developed for the two sites by Turner (1990) where herbage mass (kg ha^{-1}) = 882 + 271 * (mean disc height in cm). Patterns in herbage mass consequently followed mean disc-meter heights.

Range condition was assessed in 1986, 1988, 1990, 1993 and 1994 using the nearest-plant method (Foran *et al.* 1978) to collect 150 points at three meter intervals on each of two fixed transects, from which a proportional species composition was calculated for each camp following the nomenclature of Gibbs Russell *et al.* (1990) (Appendix 2). Daily rainfall (mm) records were kept for each site and total seasonal rainfall was calculated from 1 July to 30 June.

A step-wise multiple linear regression approach (Steel & Torrie 1981) was used to reflect the influence of cumulative summer grazing days (indexed as LSU grazing days ha^{-1}), range condition (indexed as the sum of the proportions of *T. triandra*, *P. maximum* and *P. coloratum*)[3] and rainfall (mm) (recorded from 1 July to 30 June) on residual herbage (kg ha^{-1}) at the end of summer and winter.

A forage deficit model was developed to reflect the relationship between the residual herbage mass at the end of summer (kg ha^{-1}), i.e. at the last date on which >15 mm of rain was recorded, and the expected number of winter grazing days (LSU gd ha^{-1}) to the actual length of the winter period (days), i.e. the period until the first spring rains of >15 mm was recorded.[4] The expected number of winter grazing days assumed a dry matter intake (including wastage) of 15 kg LSU^{-1} day^{-1} of herbage above 4 cm (1695 kg ha^{-1}) i.e. the grazing cut-off (Walker 1980; Turner 1990).

RESULTS AND DISCUSSION

Rainfall

Total seasonal rainfall patterns reflected considerable spacial and temporal variability (Figure 1), where the mean for the seven year period was 569 ± 62.7 mm and 612 ± 73.7 mm at Llanwarne and Dordrecht respectively. Rainfall was slightly below the long-term mean of 518 mm at each site during the 1986/87 season, consistently higher for the 1987/88 to 1990/91 seasons and considerably below the mean during the 1991/92 season (Figure 1).

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[3] The sum of proportions of the *T. triandra, P. maximum* and *P. coloratum* was strongly correlated with a forage score for each camp based on all species (r = 0.98 P<0.01 DF = 47) (after Trollope 1990) (subjectively allocated forage scores are indicated in Appendix 2a). A key species approach based on the sum of proportion of these species was therefore considered to be a reliable indicator of forage production potential for the semi-arid savanna (Turner 1988). Importantly, these species could easily be applied by range managers with little training (Tainton 1988) and would provide a relevant practical approach to range assessment.

[4] The period over which sward growth continued after the last date in summer on which >15 mm of rain was recorded and the delay before growth commenced after the date on which >15 mm was recorded in spring were assumed to be equal.

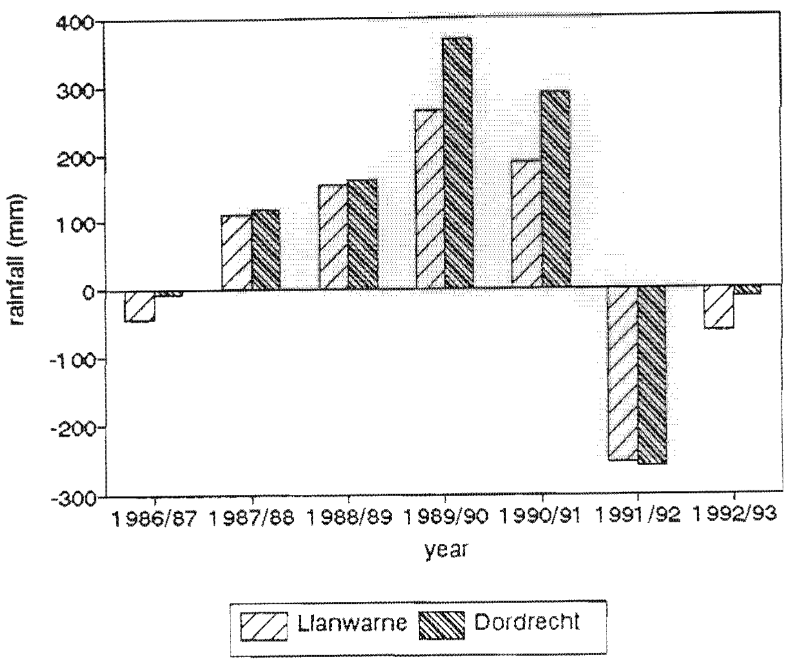

Figure 1 Variation in total seasonal rainfall (mm) about the long-term mean of 518 mm at Llanwarne and Dordrecht

Seasonal herbage accumulation patterns

Residual herbage mass closely followed rainfall patterns, with increased estimates associated with periods of rainfall (Figure 2). Interestingly, major rainfall events (such as occurred during period 52) did not result in correspondingly high rates of herbage accumulation. This suggested that runoff during extreme events, and increased infiltration below the rooting depth of the grass layer, may limit the effectiveness of the rainfall per mm. Importantly, soil loss during such events, particularly where they coincide with periods of reduced herbage availability and hence cover, may be extremely high.

Herbage mass tended to be lower at Dordrecht than at Llanwarne, despite often higher rainfall at Dordrecht (Figure 1). These differences were most pronounced during drier seasons (Figure 2). A general decline in herbage mass at both sites from period 80 was associated with the 1991/92 drought. The effect was less pronounced for the lighter stocked treatments (Figure 2) and clearly illustrated the benefits of conservative stocking strategies (as outlined in Chapter 2) on residual herbage.

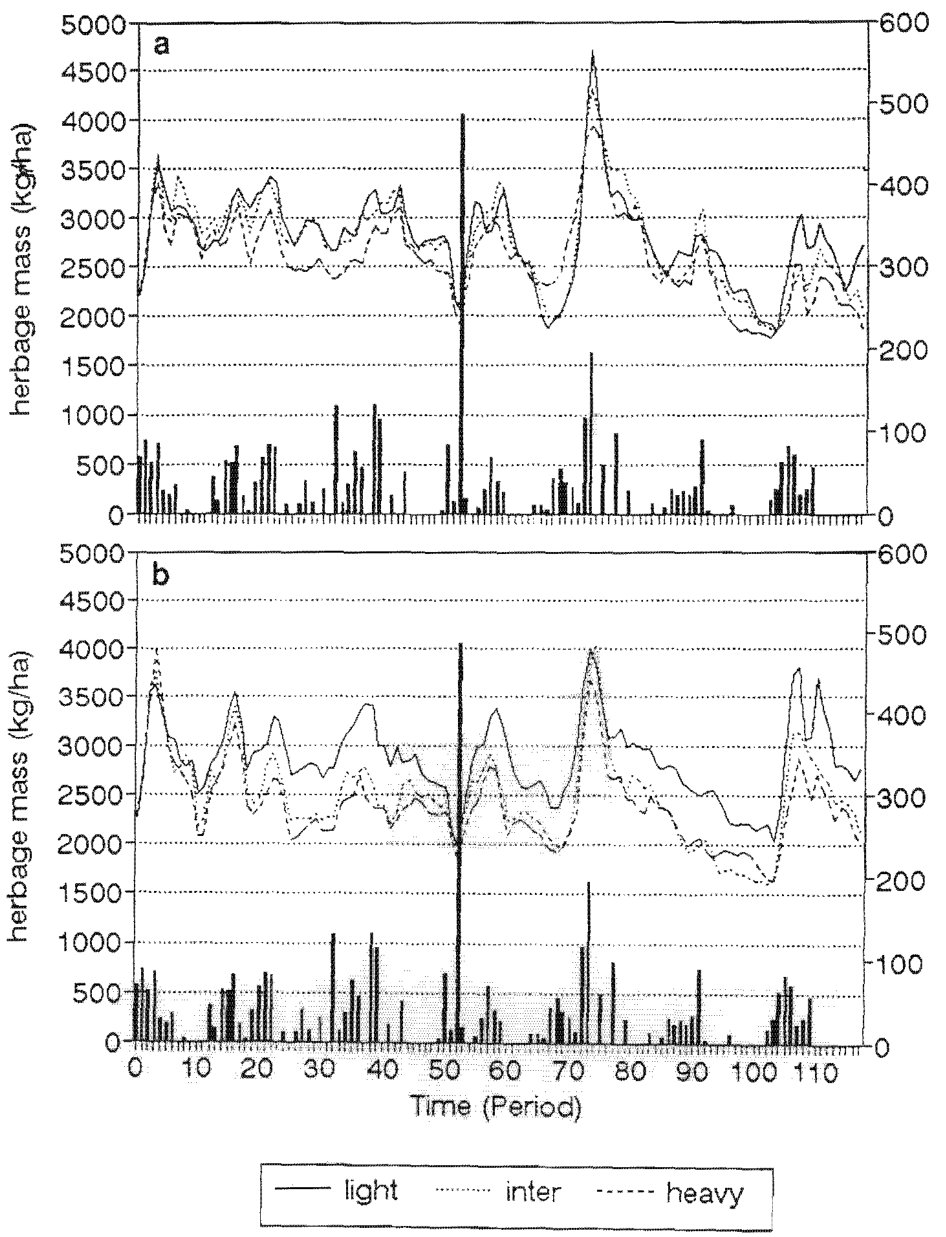

Figure 2a Patterns in herbage mass (kg ha^{-1}) in relation to rainfall (mm) during three-week periods from November 1986 (period 1) to June 1993 (period 120) for the a) group 1 and b) group 2 camps at Llanwarne

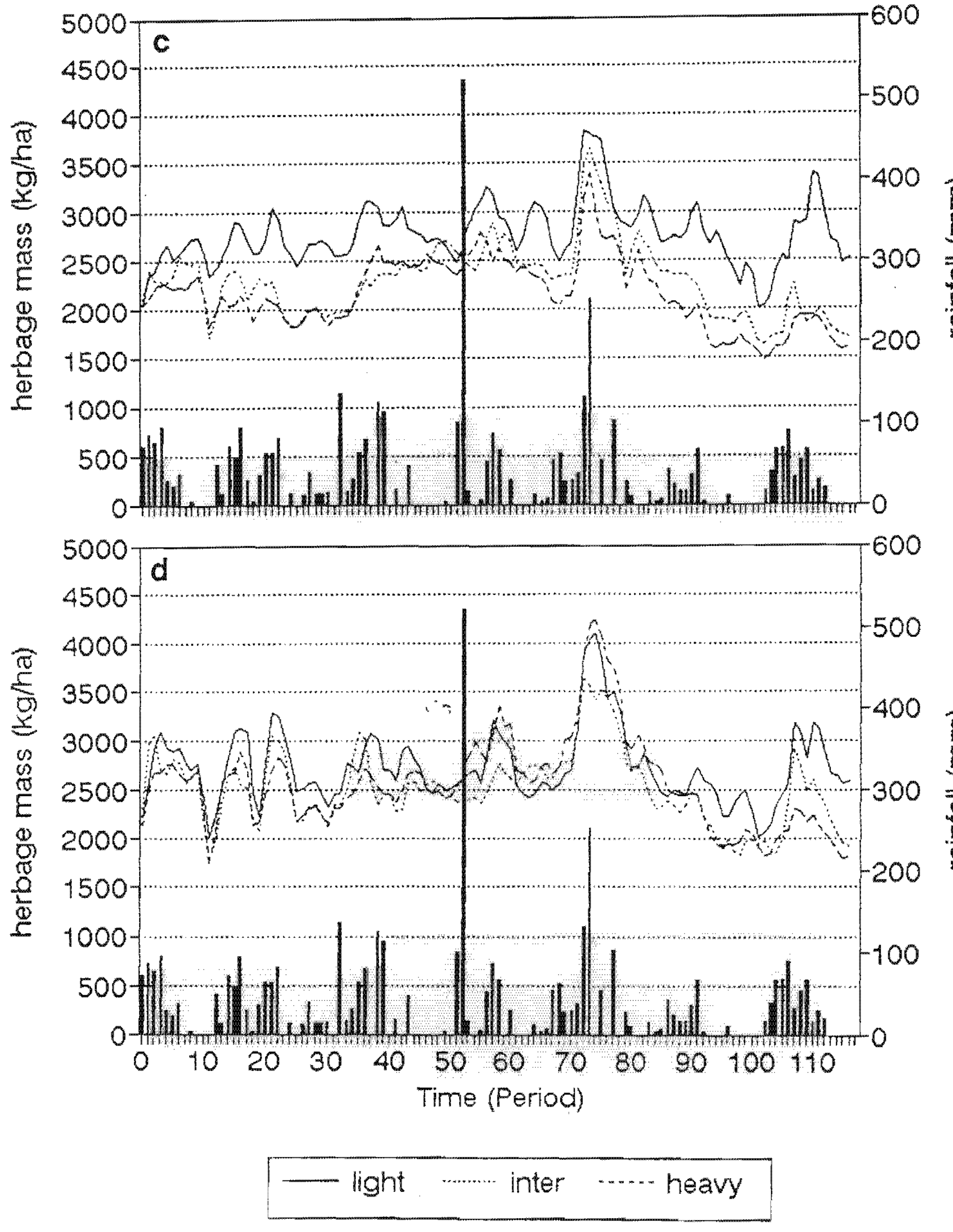

Figure 2b Patterns in herbage mass (kg ha^{-1}) in relation to rainfall (mm) during three-week periods from November 1986 (period 1) to June 1993 (period 120) for the c) group 1 and d) group 2 camps at Dordrecht

Patterns in herbage mass suggested a negative influence of stocking rate with consistently higher estimates for the light relative to the intermediate and heavy treatments at each site (Figure 2). Differences is herbage mass estimates between camps within stocking treatments reflected the consequences of various periods of stay.

The relationship between total seasonal rainfall and season length

The period (days) over which rain was recorded, i.e. from the first date in spring to the last date in summer on which >15 mm was recorded, was significantly related ($P<0.01$) to rainfall (mm) (measured 1 July to 30 June) (Table 1).

Table 1 **A linear regression model relating season length (the length of the period (days) over which rain was recorded, i.e. from the first date in spring to the last date in summer on which >15 mm was recorded) to rainfall (mm) (n = 42)**

Variables in model	Coefficient	SE	t-value	R^2	SE_p	Model signif.
Constant	179.36300	17.52	10.24 **	0.50	34.77	**
Rainfall	0.17959	0.03	6.09 **			

SE denotes standard error of coefficient, SE_p standard error of dependent variable based on input independent variables

** $P<0.01$

This implied that seasons of above-average rainfall tended to be longer than lower rainfall seasons (Table 2). This may influence the amount of herbage produced and hence influence patterns of animal production.

Table 2 **The length of the summer and winter periods (days) in relation to total seasonal rainfall at Llanwarne and Dordrecht (calculated from the first date in spring to the last day in summer on which >15 mm was recorded)**

	Llanwarne			Dordrecht		
Season	Rainfall	Summer	Winter	Rainfall	Summer	Winter
1986/87	511	147	127	475	147	127
1987/88	636	210	104	627	210	105
1988/89	680	253	251	672	252	252
1989/90	879	169	168	638	187	149
1990/91	799	189	189	677	201	189
1991/92	243	105	231	264	105	231
1992/93	437	170	-	532	210	-

Prediction of residual herbage mass at the end of summer

Rainfall and range condition were both positively related ($P<0.01$) to the estimate of residual herbage (kg ha^{-1}) at the end of summer, while cumulative summer grazing days had a significant ($P<0.01$) but negative effect on residual herbage mass (Table 3).

Table 3 **A step-wise multiple linear regression model relating cumulative summer grazing days (LSU gd ha^{-1}), rainfall (mm) and range condition (indexed as the sum of proportions of *T. triandra, P. maximum* and *P. coloratum*) to the amount of residual herbage at the end of summer (kg ha^{-1}) (defined as the last date in summer on which >15mm of rain was recorded) (n = 84)**

Variables in model	Coefficient	SE	t-value	R^2	SE_p	Model signif.
Constant	1775.249493	205.50	8.84 **	0.54	290.46	**
Cumulative summer grazing days	-14.102845	2.51	-5.61 **			
Rainfall	1.371655	0.18	7.81 **			
Range condition	12.4506	3.69	3.37 **			

SE denotes standard error of coefficient, SE_p standard error of dependent variable based on input independent variables
** $P<0.01$

The amount of residual herbage at the end of summer may vary not only in relation to rainfall and cumulative summer grazing days but, importantly, may increase as the proportional abundance of *T. triandra, P. maximum* and *P. coloratum* increases. Similar increased herbage production with increased dominance of a few grass species, i.e. decreased species diversity, has been reported for temperate North American rangeland (McNaughton 1968) and the Lowveld (Turner 1990). Increased range condition would considerably increase the residual herbage mass at the end of summer (Figure 3) and may decrease the probability of forage deficit and hence the cost of winter supplementation for a given stocking strategy.

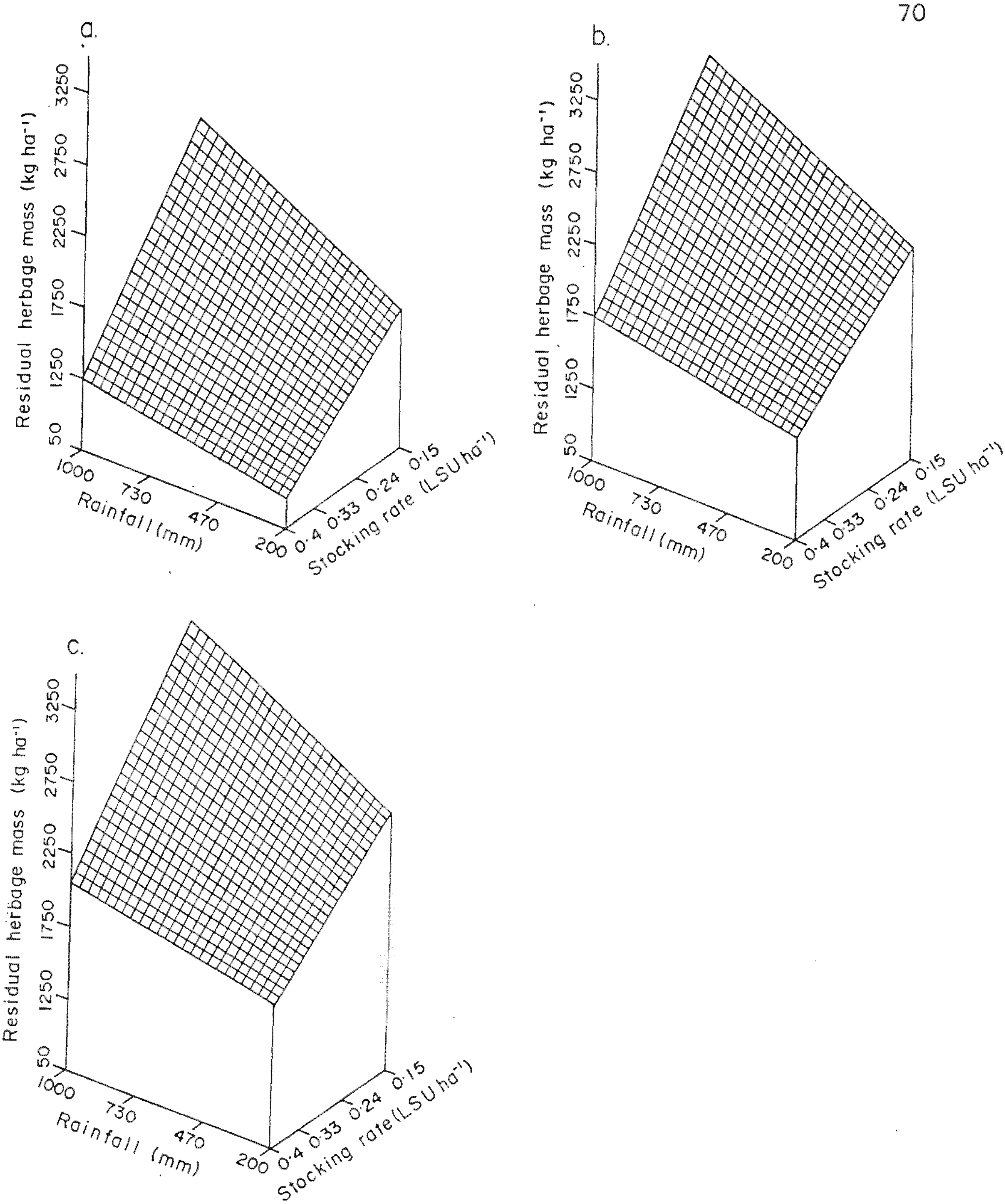

Figure 3 Residual herbage at the end of summer in relation to summer stocking rate (LSU ha^{-1}) and rainfall (mm) for range conditions (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) of a) 10, b) 50 and c) 80

Prediction of residual herbage mass at the end of winter

Although residual herbage mass at the end of winter in the semi-arid savanna may be expected to be related to residual summer herbage mass and winter cumulative grazing days (Turner 1990), residual herbage mass was significantly related only to the amount of residual herbage at the end of summer ($r = 0.50$ $P<0.01$ DF = 70) in this study. This was counter-intuitive, but may be the consequence of 1) rainfall during winter, which may result in some plant growth,[5] and 2) supplementary feeding during periods of forage deficit (where forage declined below 1695 kg ha^{-1}) during the winters of 1992 (for all stocking treatments) and 1993 (for the intermediate and heavy treatments at Dordrecht). Cattle were supplemented during these periods with sugar-cane tops (20 kg LSU d^{-1}).

The relationship between residual herbage mass at the end of summer and the period of forage deficit

The length of the winter period (days) varied between seasons in relation to seasonal rainfall patterns (Table 2). The length of the forage deficit period (days) when herbage mass declined below 1695 kg ha^{-1} was significantly related ($P<0.01$) to the residual herbage mass at the end of summer (Table 4).

[5] Although winter rainfall may result in sward growth, the response appeared to be a "greening" of the sward rather than herbage mass accumulation.

Table 4 **A linear regression model relating forage deficit period (days) when herbage mass declined below 1695 kg ha^{-1} to residual herbage at end of summer (kg ha^{-1}) (n = 72)**

Variables in model	Coefficient	SE	t-value	R^2	SE_p	Model signif.
Constant	261.778967	33.52	7.81 **	0.45	37.99	**
Residual herbage mass at the end of summer	-0.081917	0.01	-6.98 **			

SE denotes standard error of coefficient, SE_p standard error of dependent variable based on input independent variables
** P<0.01

THE DEVELOPMENT OF A HERBAGE MASS MODEL (GRASS) FOR THE SEMI-ARID SAVANNA OF NATAL

Although the coefficients of determination in this study accounted for c. 50% of the variability in the data, this was not be unexpected given the temporal and spatial scale at which the trial was conducted (Stielau K. Department of Statistics and Biometry, University of Natal, P.O. Box X01, Scottsville 3209, South Africa), where natural variability could lead to coefficients of determination of the magnitude recorded in this study. This did, however, imply that the value of the dependent variables, residual herbage mass and the forage deficit period, would vary within a range of values. Importantly, the relationships indicate those factors which significantly influenced residual herbage mass and the length of the forage deficit period and therefore provide a useful basis for the development of a herbage mass model for the semi-arid savanna.

Residual herbage mass at the end of summer, which was related to cumulative summer grazing days, rainfall and range condition (Table 3), influenced the period over which winter supplementation would be required (as outlined in Figure 4). The stocking strategy selected by the range manager may therefore influence the financial viability of the enterprise, even in the short-term, where decisions regarding summer stocking levels need to consider the risk of winter forage deficits. In addition, although range composition dynamics in semi-arid systems may be related to rainfall patterns (Chapter 3), overgrazing which leads to range degradation (as outlined in Chapter 3) may also reduce the amount of residual herbage at the end of summer (Figure 3) and contribute to increased risk of winter forage deficits. This may be an important direct cost associated with overgrazing.

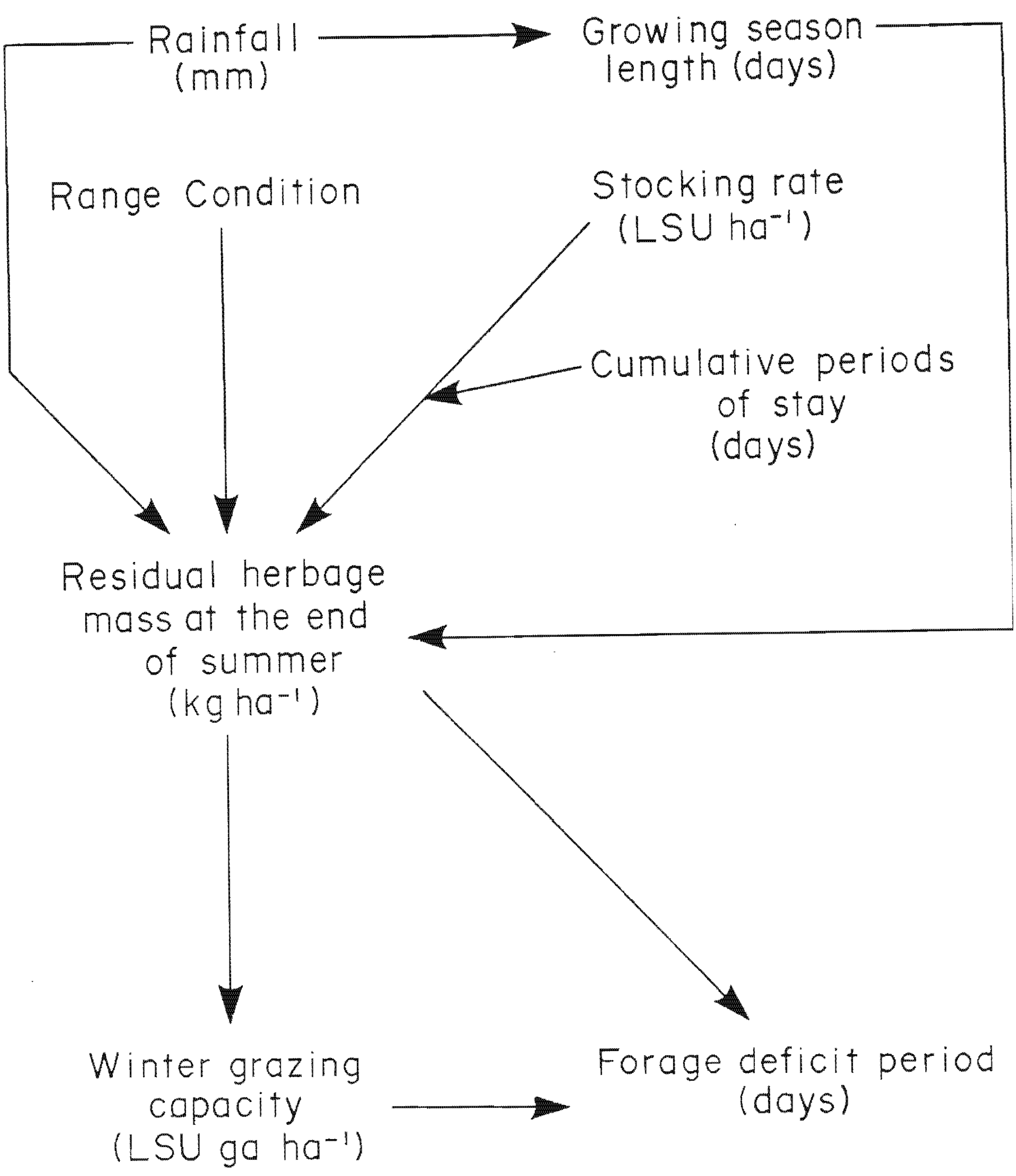

Figure 4 The **GRASS** model outlining the relationship between cumulative summer grazing days (LSU gd ha^{-1}), range condition (indexed as the sum of proportions of *T. triandra*, *P. maximum* and *P. coloratum*) and rainfall (mm), and residual summer herbage mass (kg ha^{-1}) and the period (days) over which supplementary feeding may be required to maintain animal condition

Soil heterogeneity (e.g. depth, nutrient status, erosion patterns) may also influence herbage production (Kelly & Walker 1976; Rutherford 1980; Scholes 1990) and lead to spatial variability at the camp scale (Archer & Smiens 1991; Pickup & Stafford-Smith 1993). Although this may limit the application of the model developed in this study to other sites, refinement of the model to incorporate a broad range in soil, management and rainfall conditions would be an important future consideration and may considerably improve the predictive ability of the model. The conceptual framework within which the GRASS model was developed may provide a valuable planning tool to assess the influence of various stocking strategies, range and rainfall conditions on residual herbage at the end of summer and winter forage deficits in the semi-arid savanna of Natal. Incorporation of stochastic elements, such as rainfall and season length, would provide an indication of the risk associated with various stocking strategies.

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CHAPTER 6

SEASONAL BEEF PRODUCTION PATTERNS IN THE SEMI-ARID SAVANNA OF NATAL

ABSTRACT

Grazing trials were established at two sites in the semi-arid savanna or Lowveld of Natal. The sites differed initially in range composition. Llanwarne was dominated by *Themeda triandra, Panicum maximum* and *P. coloratum*, and Dordrecht by *Urochloa mosambicensis, Sporobolus nitens* and *S. iocladus*. Three treatments were stocked with Brahman-cross cattle at each site to initially represent 'light' (0.17 LSU ha^{-1}), 'intermediate' (0.23 LSU ha^{-1}) and 'heavy' (0.30 LSU ha^{-1}) stocking. Cattle mass data collected over 120 three-week periods were used to develop a step-wise multiple linear regression model (BEEF) where summer mass gain (kg ha^{-1}) was significantly related ($P<0.01$) to total seasonal rainfall (mm) (measured 1 July to 30 June) and stocking rate (LSU ha^{-1}). Winter mass loss (kg ha^{-1}) was related to residual herbage mass at the end of summer (kg ha^{-1}) and the length of winter (days). Although range condition did not significantly influence summer mass gain, patterns in winter mass losses were influenced by residual herbage at the end of summer, which suggested that grass species in the Lowveld may differ in production potential rather than in quality. The BEEF model may provide a useful tool to assess the influence of various stocking strategies and rainfall conditions on summer mass gain.

INTRODUCTION

Although stocking rate may influence individual animal performance (Mott 1960; Riewe 1961, Jones & Sandland 1974), livestock production in semi-arid systems may be determined largely by temporal rainfall variability which influences patterns in herbage availability (Chapter 5). This paper examines the relationship between rainfall, range condition, stocking rate and livestock production patterns in the semi-arid savanna with the aim of developing a model to predict seasonal patterns in livemass change.

PROCEDURE

Data collection procedures follow those outlined in Chapter 5.

A step-wise multiple linear regression approach (Steel & Torrie 1981) was used to reflect the influence of stocking treatment (LSU ha^{-1}), range condition (indexed as the sum of the proportions of three key forage species, *T. triandra*, *P. maximum* and *P. coloratum*)[1] and total seasonal rainfall (mm) (recorded from 1 July to 30 June) on summer mass gain (kg ha^{-1}).[2] A winter loss model examined the influence of residual summer herbage mass (kg ha^{-1}) and the length of the winter period (days) (defined as the last date in summer to the first date in the subsequent spring when >15 mm of rain was recorded) on winter mass loss (kg ha^{-1}).[3]

RESULTS AND DISCUSSION

Livemass gain per individual animal (kg LSU^{-1}) tended to decrease and gain per area (kg ha^{-1}) increase as stocking rates increased across treatments. Livemass gain was influenced by rainfall patterns (Chapter 5) and decreased summer gains were

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1 The sum of proportions of the *T. triandra, P. maximum* and *P. coloratum* was strongly correlated with a forage score for each camp based on all species ($r = 0.98$ $P<0.01$ $DF = 47$) (after Trollope 1990) (subjectively allocated forage scores are indicated in Appendix 2a). A key species approach based on the sum of proportion of these species was therefore considered to be a reliable indicator of forage production potential for the semi-arid savanna (Turner 1988). Importantly, these species could easily be applied by range managers with little training (Tainton 1988) and would provide a relevant practical approach to range assessment.

2 Livemass gain was expressed per unit land area to link into an economic model based on cost per unit land area (Chapter 9).

3 The period over which herbage accumulation continued, after the last date in summer on which >15 mm of rain was recorded, and the delay before herbage accumulation commenced after the date on which >15 mm on rain was recorded in spring, were assumed to be equal.

evident for drier seasons[4] (1986/87, 1991/92) (Table 1). Patterns in winter mass loss (kg LSU^{-1}) reflect little difference between stocking treatments and sites, with the exception of the 1991/92 season where losses were substantial. Losses invariably increased, however, as stocking rates were increased at both sites. Supplementary feeding (sugar-cane tops at 20 kg LSU^{-1} d^{-1}) for all stocking treatments during the winter of 1992 and for the intermediate and heavy at Dordrecht during the winter of 1993 restricted mass loss at these times.

[4] Seasons extended over two calender years where the summer period of livemass gain commenced in the first year (e.g. 1986 in the 1986/87 season) and ended when mass loss began in the winter of the following year (e.g. 1987 in the 1986/87 season). The length of the summer and winter periods were related to patterns in mass gain and loss and consequently did not equal 365 days i.e a calender year.

Table 1 **Summer livemass gain and winter mass loss per individual animal (kg LSU^{-1}) and per hectare (kg ha^{-1}) for 'light' (L) (0.17 LSU ha^{-1}), 'intermediate' (M) (0.23 LSU ha^{-1}) and 'heavy' (H) (0.30 LSU ha^{-1}) stocking at Llanwarne and Dordrecht**

		Llanwarne				Dordrecht			
		Summer		Winter		Summer		Winter	
Season[1]	Treat.	/LSU	/ha	/LSU	/ha	/LSU	/ha	/LSU	/ha
1986/87	L	174	29.6	11	1.9	173	26.0	3	0.5
	M	161	40.3	15	3.8	158	30.0	4	0.8
	H	158	47.4	17	5.1	148	38.5	21	5.5
1987/88	L	231	34.7	11	1.7	225	42.8	13	2.5
	M	233	53.6	21	2.5	247	64.2	19	4.9
	H	246	78.8	33	10.6	229	71.0	21	6.5
1988/89	L	224	33.6	0	0	201	28.1	1	0.1
	M	207	43.5	0	0	198	33.7	1	0.2
	H	199	55.7	0	0	193	42.5	1	0.2
1989/90	L	182	27.3	10	1.5	197	33.5	4	0.7
	M	176	38.7	14	3.1	188	39.5	18	3.8
	H	153	39.8	21	5.5	177	47.8	18	4.9
1990/91	L	226	33.9	4	0.6	213	32.0	1	0.2
	M	221	48.6	1	0.2	215	40.9	4	0.8
	H	220	61.6	3	0.8	212	53.0	3	0.8
1991/92	L	154	23.1	38	5.7	200	36.0	72	13.0
	M	136	28.6	41	8.6	167	36.7	51	11.2
	H	147	42.6	36	10.4	150	40.5	62	16.7
1992/93	L	215	30.1	0	0.0	224	38.1	0	0.0
	M	224	49.3	8	1.8	188	37.6	52	12.0
	H	202	54.5	29	8.7	192	50.5	45	13.5

The length of the period (days) over which cattle gained mass tended to be longer at Llanwarne (302 ± 65) than at Dordrecht (269 ± 33) (Table 2). Similarly, the length of the period of mass loss was shorter at Llanwarne (51 ± 44) than at Dordrecht (86 ± 32). Average daily gains (ADG) were consequently lower at Llanwarne than at Dordrecht given similar mass gain for the summer, but over a longer period at Llanwarne. Importantly, the longer the period of winter mass loss the greater the cost of winter supplementation to maintain animal condition.

Table 2 **The length of the period (days) of summer mass gain and winter loss at Llanwarne and Dordrecht**

Site	Llanwarne		Dordrecht	
Season	Gain	Loss (days)	Gain	Loss (days)
1986/87	189	85	253	63
1987/88	357	21	252	105
1988/89	357	0	273	84
1989/90	294	63	252	121
1990/91	348	42	327	42
1991/92	231	126	231	126
1992/93	336	126	294	126

Development of a summer mass gain model

Summer livemass gain per hectare (kg ha^{-1}) was significantly related ($P<0.01$) to rainfall (mm) and stocking rate (LSU ha^{-1}) (Table 3).

Table 3 A step-wise multiple linear regression model relating rainfall (mm) and stocking rate (LSU ha^{-1}) to livemass gain per hectare (kg ha^{-1})

Variables in model	Coefficient	SE	t-value	R^2	SE_p	Model signif..
Constant	-25.317706	8.17	-3.10 **	0.75	5.80	**
Rainfall	0.095159	0.03	3.42 **			
Rainfall^2	-0.000073	0.00	-2.88 **			
Stocking rate	180.861889	17.25	10.48 **			

SE denotes standard error of coefficient, SE_p standard error of dependent variable based on input independent variables

** $P<0.01$

Summer mass gain was not significantly related to range composition. In contrast, the amount of residual summer herbage (kg ha^{-1}) was influenced by range composition (Chapter 5). This suggested that livemass gain over the summer would not be significantly influenced by changes in range composition, provided herbage did not become limiting during the summer. Grass species in the semi-arid savanna may therefore differ in terms of production potential rather than quality. Changes in range composition may consequently influence patterns in winter mass loss rather than summer mass gain provided summer stocking rates are not excessive.

A significant quadratic relationship between livemass gain per hectare and rainfall implied that livemass gain would increase at a decreasing rate as rainfall increased, attain a maximum at 700 mm and then decline. This may be the consequence of increased growth and hence increased steminess of *T. triandra* and *P. maximum* during higher rainfall seasons. This would have acted to reduce intake and hence performance. In practise, range managers may increase stocking pressure to compensate for increased production during wetter seasons by reducing the grazed area. Accumulated herbage in ungrazed camps may then provide a drought reserve or be burnt to restrict bush thickening.

Despite considerable evidence to suggest that the relationship between stocking rate and livemass gain per unit area is quadratic (Mott 1960, Cowlishaw 1969, Conway 1974, Jones & Sandland 1974, Chapter 2), at least until maximum gain is attained (Heitschmidt & Taylor 1991), this tendency was not evident in this study. This may be related to the restriction of stocking treatments to a range of likely economic stocking rates in this study (Chapter 3). These rates did not result in summer forage limitations. The stocking rate at which maximum gain per hectare would be reached was therefore calculated from the slope (a) and intercept (b) coefficients of the linear relationship between stocking rate and individual animal performance (Edwards 1981) (Table 4).

Table 4 **Calculation of the stocking rate (LSU ha^{-1}) at which maximum gain per hectare (G_{max}) would be attained at Llanwarne and Dordrecht (based on the slope (a) and intercept (b) coefficients of significant ($P<0.01$) linear relationships between stocking rate and average daily gain)**

Season	Llanwarne				Dordrecht			
	a	b	R^2	G_{max}	a	b	R^2	G_{max}
1987	0.84	1.13	0.99	0.37	0.77	0.32	0.77	0.50
1988	1.94	3.29	0.84	0.29	0.05	0.32	0.06	-
1989	1.18	1.64	0.96	0.36	2.25	2.66	0.99	0.39
1990	0.80	1.02	0.82	0.39	1.18	1.30	0.99	0.45
1991	4.94	7.40	0.91	0.33	1.17	2.02	0.85	0.29
1992	0.59	0.59	0.12	-	0.50	0.37	0.89	0.68
1993	2.21	2.35	0.55	0.47	0.58	0.41	0.47	0.71
Mean				0.37				0.50

In theory, the stocking rate at which maximum summer gain per hectare would be attained (G_{max}) would be considerably higher at Dordrecht than at Llanwarne (Table 4). This may be attributed to the dominance of *S. iocladus* and *S. nitens* at the former, where the intake on less productive species may be higher during the summer than for more productive (and hence more stemmy) species such as *T. triandra* and *P. maximum*. Greater summer individual animal performance during drier seasons (1991/92) at Dordrecht than at Llanwarne (Table 1) may be the consequence of greater accessibility to higher quality forage at Dordrecht while cattle at Llanwarne were forced to select lower quality stem material. Importantly, despite higher summer production, even during drier seasons, little forage is likely to remain at the end of summer on pioneer dominated range (Chapter 5). This may account for the longer periods of winter mass loss at Dordrecht which may influence the period over which supplementary feeding may be required.

Development of a winter mass loss model

Winter mass loss (kg ha^{-1}) was negatively related ($P<0.01$) to residual summer herbage mass (kg ha^{-1}) and positively related ($P<0.01$) to winter length (days) (Table 5). Cumulative winter grazing days (LSU gd ha^{-1}) did not influence the extent of mass loss. This may be the consequence of rainfall and hence some growth during winter[5], supplementation during periods of forage deficit and the ability of the cattle to browse during adverse conditions. Residual herbage mass and the length of the winter would be unaffected by these factors.

[5] Importantly, although winter rainfall did not appear to influence patterns in herbage mass (Chapter 4), the extent of cattle mass loss may be reduced by increased sward quality as grass growth or "greening" occurred after rain fell.

Table 5 **A step-wise multiple linear regression model relating residual summer herbage mass (kg ha^{-1}) and the length of the winter (days) (from the last date in summer to the first date in the subsequent spring on which >15 mm of rain was recorded) to winter mass loss (kg ha^{-1})**

Variables in model	Coefficient	SE	t-value	R^2	SE_p	Model signif.
Constant	8.209615	3.59	2.29 **	0.51	1.97	**
Winter length	0.037206	0.01	4.03 **			
Residual summer herbage mass	-0.002806	0.00	-2.43 **			

SE denotes standard error of coefficient, SE_p standard error of dependent variable based on input independent variables

** P<0.01

As residual herbage mass was a function of cumulative summer grazing days (LSU gd ha^{-1}), rainfall (mm) and range condition (indexed as the sum of the proportions of *T. triandra*, *P. maximum* and *P. coloratum*) (Chapter 5), it may be inferred that winter mass loss would be related to each of these factors. Although range composition was not significantly related to summer mass gain, the extent of winter mass loss may be strongly influenced by range condition through its effect on residual summer herbage mass (Figure 1).

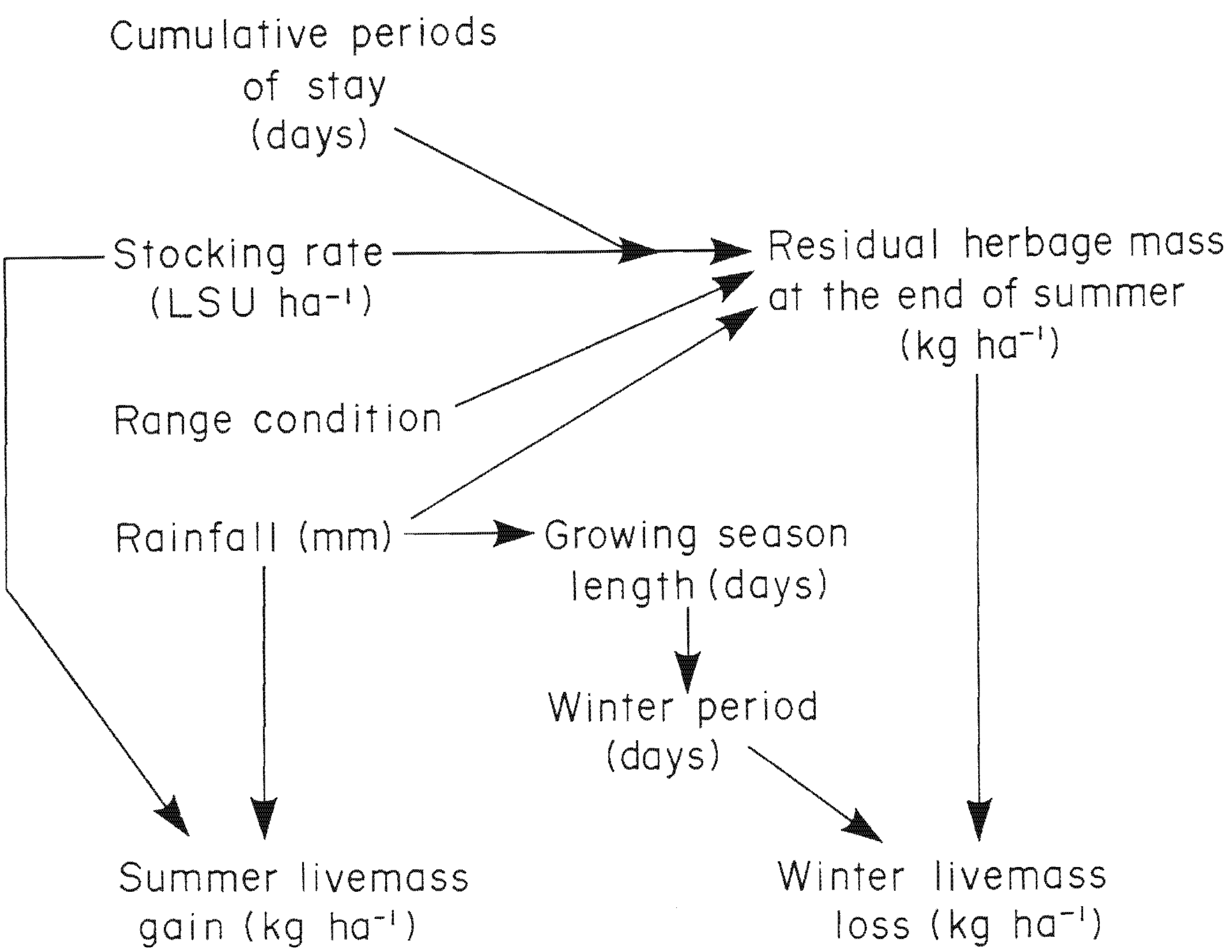

Figure 1 The relationship between the rainfall (mm), stocking rate (LSU ha^{-1}), range condition (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) and summer mass gain and winter mass loss (kg ha^{-1}) in the semi-arid savanna of Natal

CONCLUSIONS

Although winter mass loss was related to residual summer herbage and winter length, the relationship was confounded by winter rainfall, supplementation during drier seasons and the ability of the cattle to browse during adverse conditions. Although the winter model may provide an indication of the factors which influence winter mass loss, it may be limited in its application. Addition of the amount of available browse during periods of forage deficit and estimates of browse intake in relation to stocking rate may consequently be important additions to allow the refinement of a winter mass loss model.

Although the coefficients of determination observed in this study accounted for c. 50% of the variability, this was not be unexpected given the temporal and spatial scale at which the trial was conducted (Stielau K. Department of Statistics and Biometry, University of Natal, P.O. Box X01, Scottsville 3209, South Africa), where natural variability could lead to coefficients of determination of the magnitude recorded in this study. This did, however, imply that the dependent values would vary within a wide range. Importantly, the significant relationship between rainfall, stocking rate and summer mass gain provided a useful basis for the development of a summer mass gain model (BEEF) for the semi-arid savanna. The model may provide a useful tool to assess the influence of various stocking strategies on production risk through the incorporation of stochastic rainfall effects. Incorporation of the BEEF model into an economic model may provide an indication of the influence of stocking strategy on the probability of obtaining a given level of income.

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CHAPTER 7

THE INFLUENCE OF RAINFALL AND GRAZING ON RANGE COMPOSITION DYNAMICS IN THE SEMI-ARID SAVANNA OF NATAL

ABSTRACT

Range composition dynamics were examined over an eight year period (1986-1994) at two sites in the semi-arid savanna or Lowveld of Natal. The sites differed initially in range composition. Llanwarne was dominated by *Themeda triandra*, *Panicum maximum* and *P. coloratum*, and Dordrecht by *Urochloa mosambicensis*, *Sporobolus nitens* and *S. iocladus*. Three treatments were stocked with Brahman-cross cattle at each site to initially represent 'light' (0.17 LSU ha^{-1}), 'intermediate' (0.23 LSU ha^{-1}) and 'heavy' (0.30 LSU ha^{-1}) stocking. Detrended Correspondence Analysis revealed distinct patterns in grass species dynamics, which appeared to be determined by rainfall although modified by stocking rate. A state-and-transition model of range dynamics, based on three discrete states, was developed to summarise the effects of rainfall and stocking rate. State 1, characterised by *S. iocladus* and *S. nitens*, was associated with increased stocking. Movement towards State 2, characterised by *T. triandra* and *P. maximum*, was associated with periods of increased rainfall. Drought conditions, which comprised major system disturbance, led to stability at State 3 dominated by *U. mosambicensis*. Post-drought recovery was determined by pre-drought composition and stocking levels. Soil loss during successive rainfall cycles may lead to stabilisation across an irreversible threshold at a forth state characterised by shallow rooted species such as *Tragus racemosa* and *Aristida congesta* subsp. *congesta*.

INTRODUCTION

The conceptual model of range dynamics based on successional theory (Clements 1916, Dyksterhuis 1949), which suggested that grazing altered species composition in a predictable manner and change could be measured as the relative difference between the current and climax vegetation (Joyce 1993), has been challenged by models based on non-equilibrium community dynamics, alternative steady states and transition thresholds (Ellis & Swift 1988, Westoby *et al* 1989, Archer & Smiens 1991, Friedel 1991, Laycock 1991). Although range dynamics and productivity in arid and semi-arid environments are largely determined by rainfall (McDonald 1982; Ellis & Swift 1988; O' Connor 1985, 1991; Peel *et al.* 1991), the major decision affecting the level of livestock production is stocking rate (Riechers *et al.* 1989). Range dynamics may determine levels of herbage and livestock productivity (Chapter 5, 6) and hence profitability, although optimum range condition may change from season to season and with management objectives (Stuart-Hill & Aucamp 1993). This paper examines the relationship between stocking strategy, rainfall and grass species dynamics, and outlines a conceptual model for the semi-arid savanna of Natal.

PROCEDURE

Data collection procedures followed those outlined in Chapter 5.

Proportional species composition data from each camp, site and survey were pooled and ordinated using Detrended correspondence analysis (DCA), in the form of DECORANA (Hill 1977), to elucidate patterns in the distribution of grass species between and within sites and surveys. The distance to the nearest plant (rounded up to the nearest centimetre) was recorded at 100 points per camp during the 1993 and 1994 surveys to provide an index of minimum bare area. Distance measures play a major role in determining estimates of basal cover in tufted rangeland and are negatively related to estimates of cover (Hardy & Tainton 1993). Estimates of minimum bare area within sites and surveys (ln transformed to reduce the influence of outlier values) were examined using a Kruskal-Wallis test of group medians (Siegel 1956; Steel & Torrie 1981). Distributions of ln transformed distance measures were compared for camps across surveys using the Wilcoxon two-sample signed-ranks test (Siegel 1956). Daily rainfall records (mm) were kept for each site and total seasonal rainfall was calculated from 1 July to 30 June each year.

RESULTS AND DISCUSSION

Seasonal rainfall patterns reflected considerable spacial and temporal variability (Figure 1), where the mean for the seven year period was 569 ± 62.7 mm and 612 ± 73.7 mm at Llanwarne and Dordrecht respectively. Rainfall was slightly below the long-term mean of 518 mm at each site during the 1986/87 and 1992/93 seasons, consistently higher for the 1987/88 to 1990/91 seasons and considerably below the mean during the 1991/92 season (Figure 1).

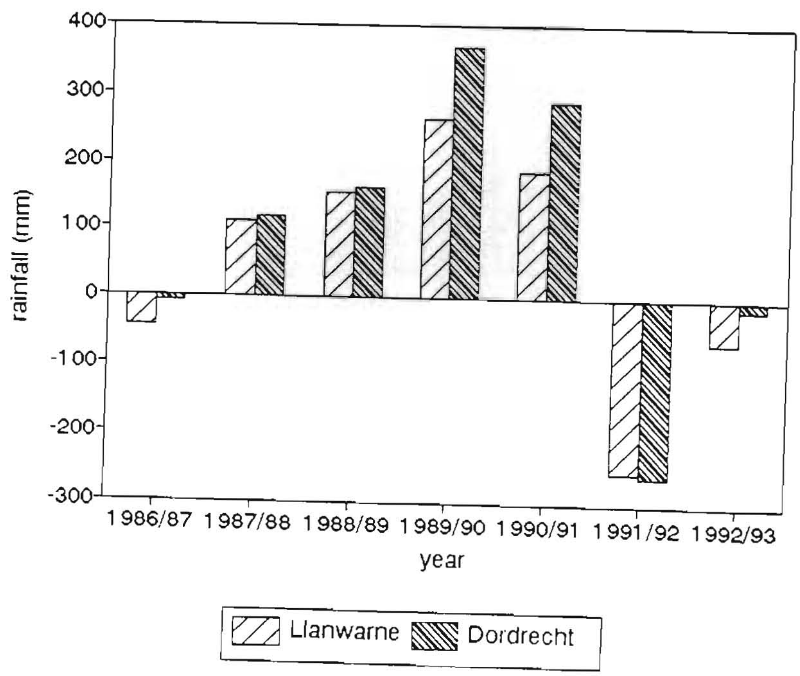

Figure 1 Variation in seasonal rainfall (mm) about the long-term mean of 518 mm at Llanwarne and Dordrecht

An ordination of the species composition data over all sites, treatments and surveys revealed a broad separation of species on axes 1 and 2 with eigen scores 0.23 and 0.10 respectively (Figure 2). Axis 2 appeared to comprise a gradient from perennial species characteristic of undisturbed sites, such as *T. triandra*, though to pioneer species such as *U. mosambicensis* and *Tragus racemosa* (Figure 2).

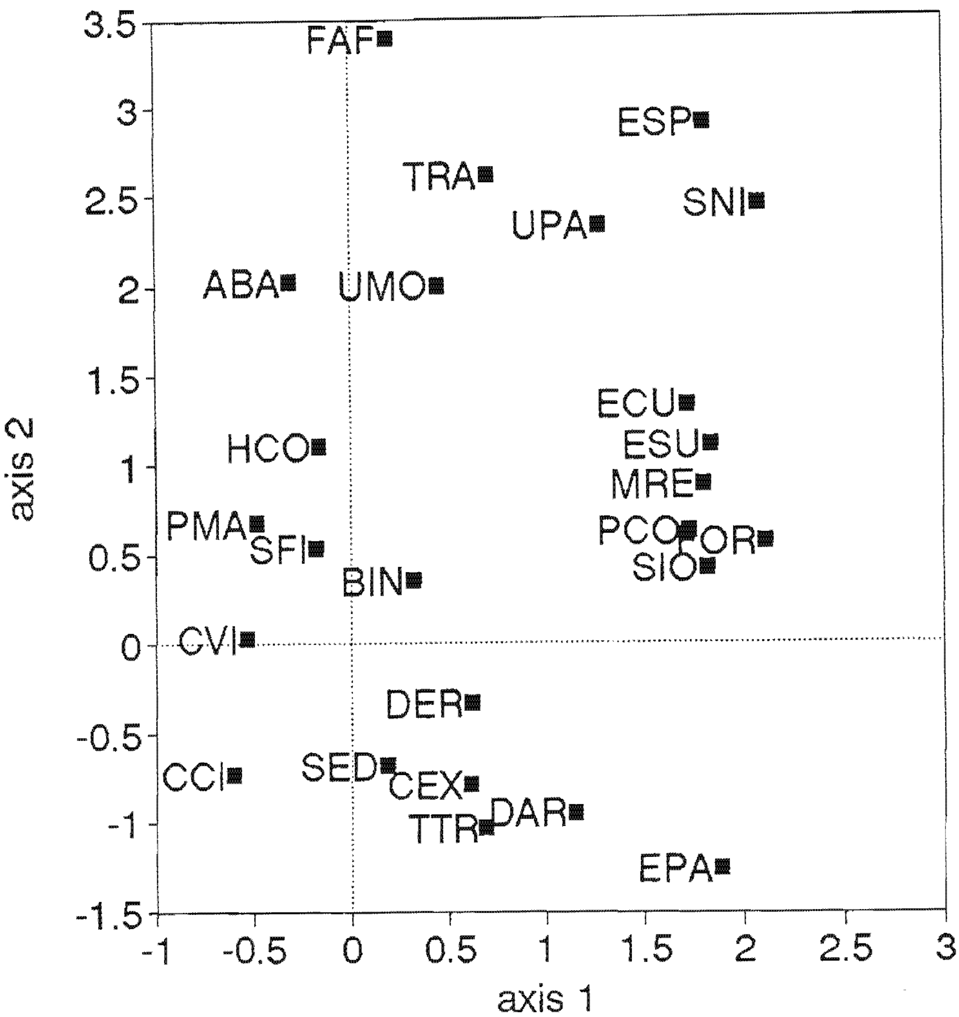

Figure 2 Ordination of grass species data across sites, treatments and surveys on DCA axis 1 and 2. Species acronyms are ACO - *Aristida congesta* subsp *congesta*, BIN - *Bothriochloa insculpta*, CCI-*Cenchrus ciliarus*, CEX - *Cymbopogon excavatus*, CVI - *Chloris virgata*, DAR - *Digitaria argyrograpta*, DER - *D. eriantha*, ECU - *Eragrostis curvula*, ESU - *E. superba*, EPA - *Eustachys phaspaloides*, ESP - *Eragrostis* species, FAF - *Fingeruthria africana*, FOR - forb species, HCO - *Heteropogon contortus*, MRE - *Melanis repens*, PMA - *Panicum maximum*, PCO - *P. coloratum*, SED - sedge species, SFI - *Sporobolus frimbriatus*, SIO - *S. iocladus*, SNI - *S. nitens*, TRA - *Tragus racemosa*, TTR - *Themeda triandra*, UMO - *Urochloa mosambicensis*, UNA - unallocated bare ground and UPA - *U. panicoides*

An examination of trends in proportional abundance of the six most abundant species revealed distinct temporal patterns in composition (Figure 3). *Panicum coloratum* displayed no clear trend in abundance over the eight year period and it appeared to be unaffected by either grazing or rainfall. In contrast, *P. maximum* followed distinct temporal trends and increased in proportional abundance, particularly at Llanwarne. The increase was most pronounced for the 'light' treatments and a decline in abundance with decreased rainfall was apparent for the 'heavy' treatments at Dordrecht. Although *T. triandra* was not abundant at Llanwarne in 1986, it tended to increase in proportional abundance over the 1988 to 1990 period and then decline to 1994. This pattern coincided with periods of above (1988 to 1990) and below-average rainfall (1990 to 1993) (Figure 1). Similarly, although *T. triandra* occurred with low abundances at Dordrecht in 1986, it tended to increase until 1990 and then decline to 1994. Importantly, *T. triandra* was represented at low levels under 'light' stocking at Dordrecht through to 1994. Although *S. nitens* was well represented in the H1 camp at Dordrecht in 1986 its proportional abundance tended to decline during the below-average rainfall period (1990-1993). *Sporobolus iocladus* was present throughout the survey period at relatively consistent levels at both sites. In contrast, *U. mosambicensis* tended to increase in proportional abundance in response to decreased rainfall to the extent that it comprised over 40% of the sward in the H1 camp at Dordrecht in 1994 (Figure 3b).

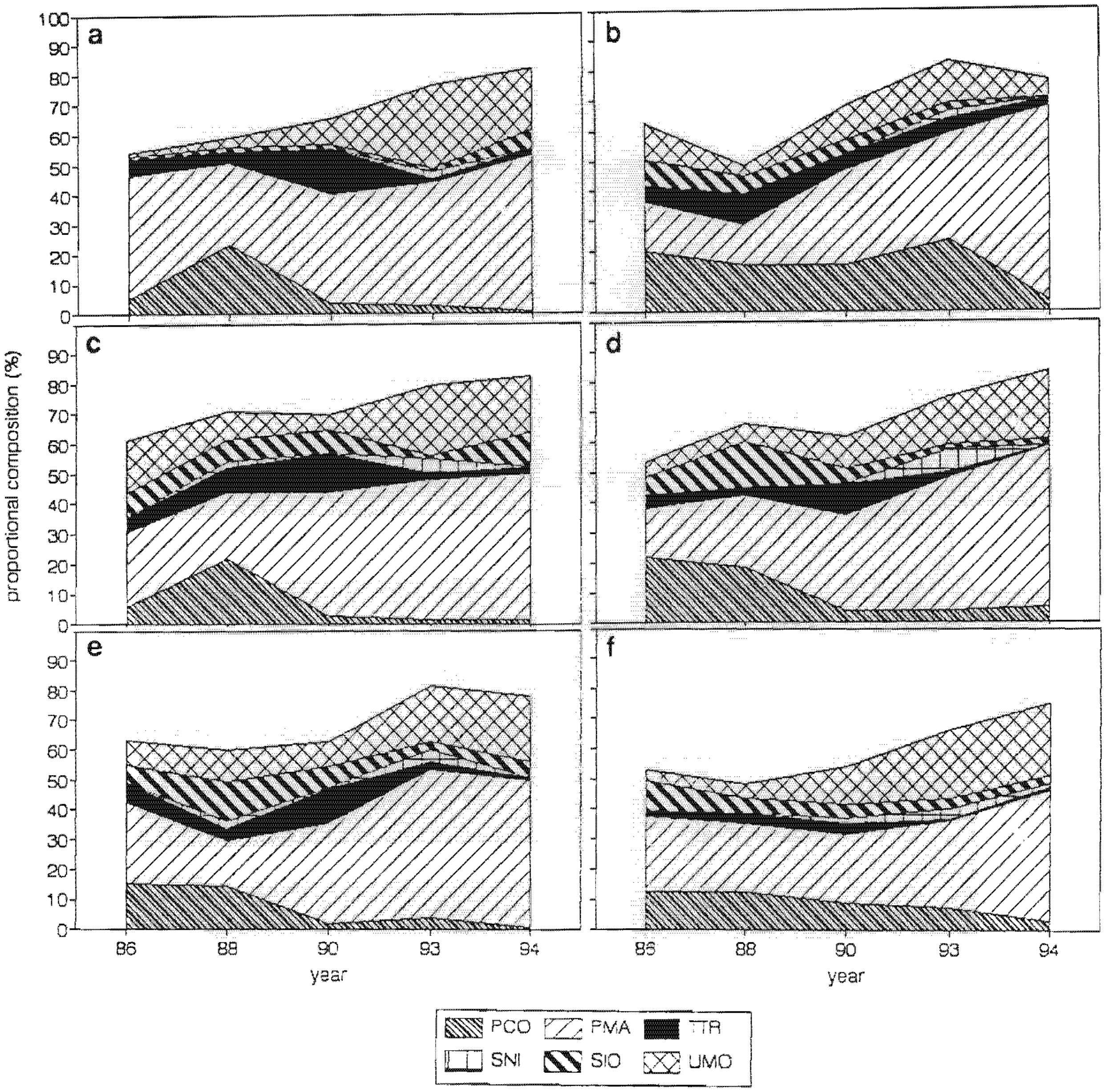

Figure 3a Proportional abundances of six grass species over a eight year for the 'light' [a) L1, b) L2]; 'intermediate' [c) M1, d) M2] and 'heavy' [e) H1, f) H2] treatments at Llanwarne. Species acronyms are PCO - *Panicum coloratum*, PMA - *P. maximum*, TTR - *Themeda triandra*, SNI - *Sporobolus nitens*, SIO - *S. iocladus* and UMO - *Urochloa mosambicensis*

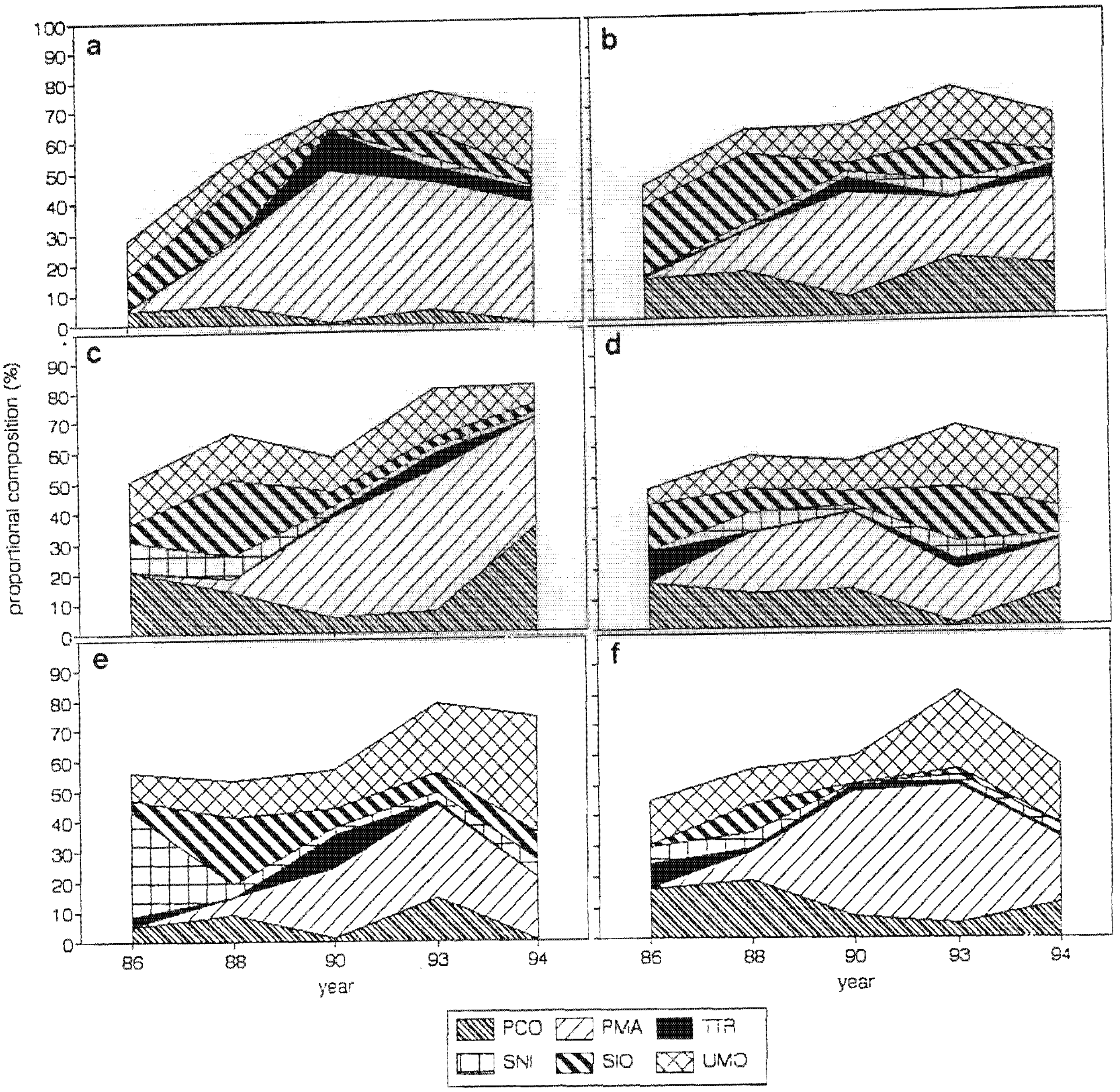

Figure 3b Proportional abundances of six grass species over a eight year period for the 'light' [a) L1, b) L2]; 'intermediate' [c) M1, d) M2] and 'heavy' [e) H1, f) H2] treatments at Dordrecht. Species acronyms are PCO - *Panicum coloratum*, PMA - *P. maximum*, TTR - *Themeda triandra*, SNI - *Sporobolus nitens*, SIO - *S. iocladus* and UMO - *Urochloa mosambicensis*

An ordination of samples revealed distinct temporal patterns which appeared to be associated with rainfall (Figure 4). Sites at Llanwarne tended to remain within the boundary of their initial composition during the above-average rainfall period (1988-1990). The 1993 survey recorded movement into a new domain characterised by *U. mosambicensis* following the 1991/1992 drought. Trajectories during the 1994 survey reflected movement towards the original domain (occupied during the 1990 survey), although the extent of movement appeared to be reduced with increased stocking (Figure 4).

The sites at Dordrecht, which were initially characterised by *S. iocladus, S. nitens* and *U. mosambicensis*, tended to move towards the *P. maximum/T. triandra* domain during the above-average rainfall period (1988 to 1990) (Figure 4), largely in response to an increase in the abundance of *P. maximum* (Figure 3). Although the pattern in species movement for 'light' treatments in response to the 1991/92 drought was similar to that outlined for Llanwarne, patterns for the 'intermediate' and 'heavy' treatments were less clear (Figure 4).

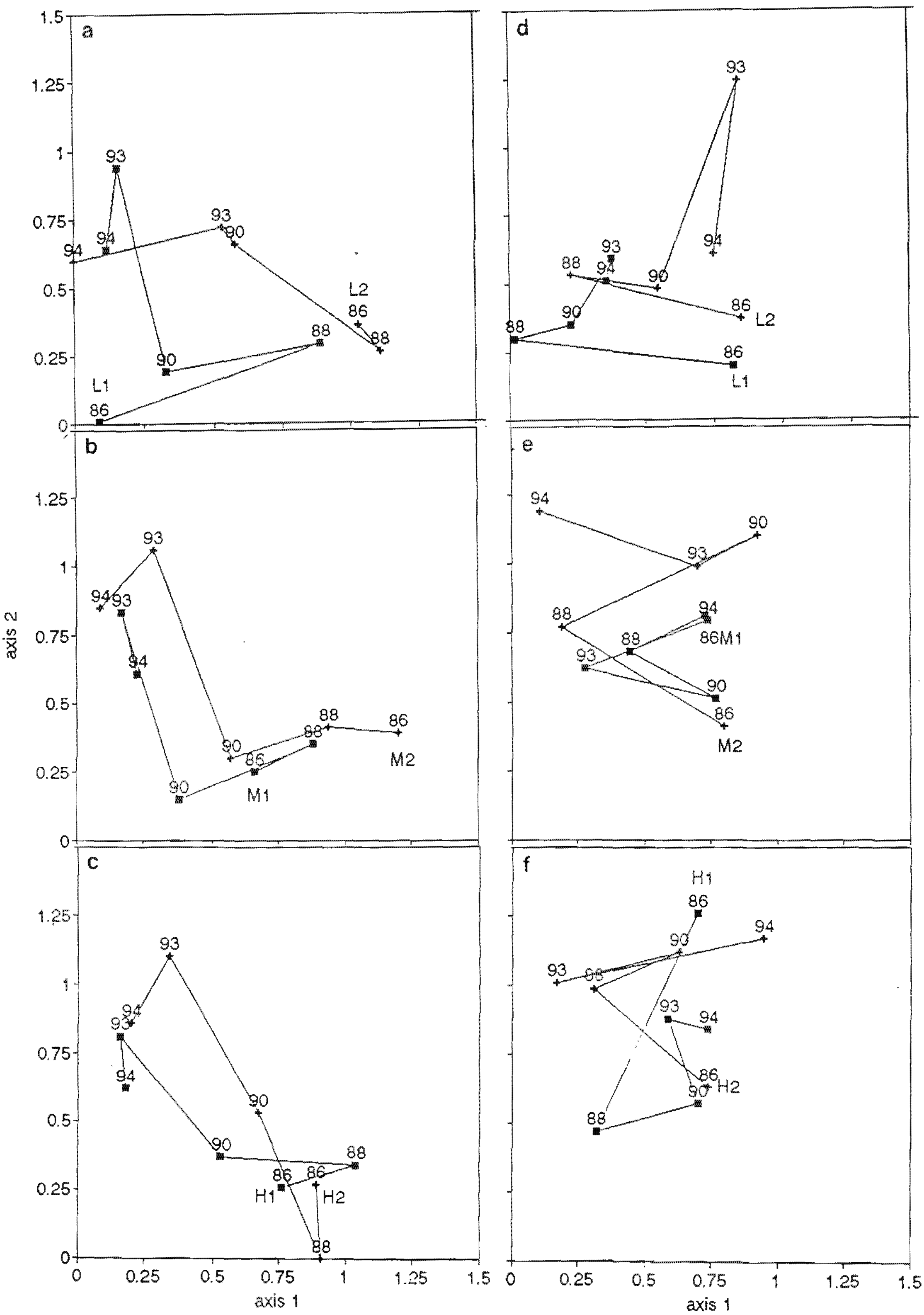

Figure 4 Ordination of sample sites for the a) 'light' b) 'intermediate' and c) 'heavy' treatments at Llanwarne and the d) 'light', e) 'intermediate' and f) 'heavy' treatments at Dordrecht respectively

Although basal cover (indicated by decreased mean distance) generally increased in the post-drought recovery period (Figure 5), the increase was significant at Llanwarne only ($P<0.05$). Stocking treatment had no significant influence on basal cover ($P>0.05$). Interestingly, basal cover measures appeared to be higher at Dordrecht than at Llanwarne, particularly in the 'heavy' treatment. This was counter-intuitive but may be related to a dominance of *U. mosambicensis*, a prostrate growing species, at the former.

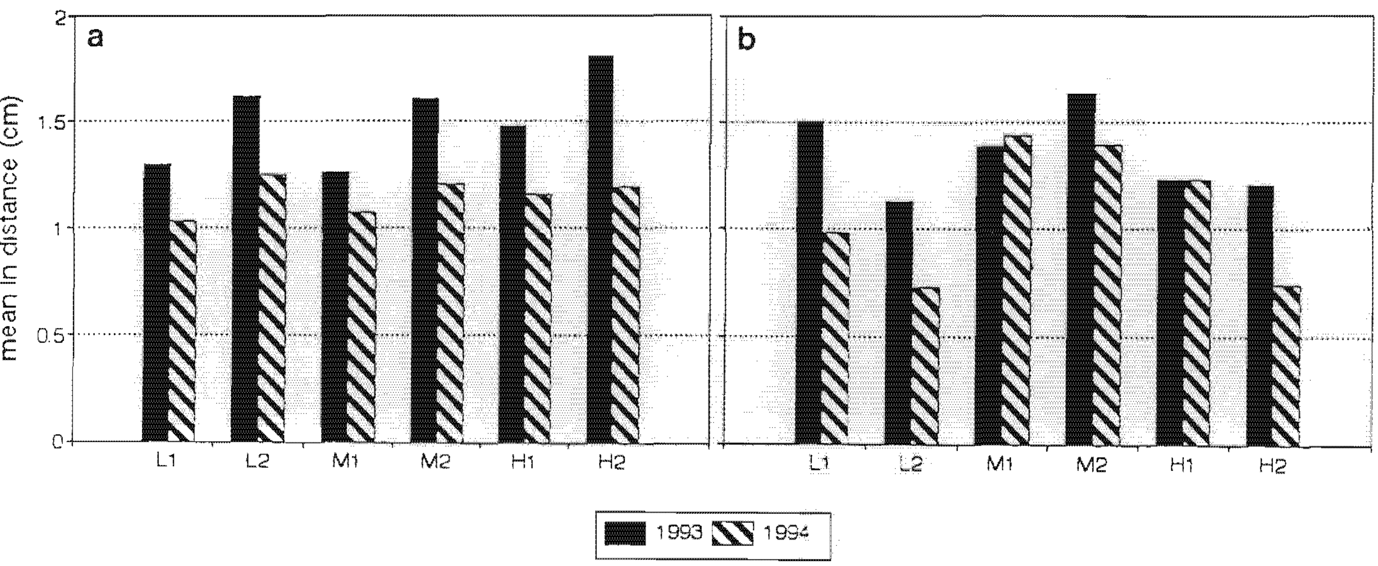

Figure 5 Mean ln transformed distances (cm) for the stocking treatments at a) Llanwarne and b) Dordrecht. L, M and H denote 'light', 'intermediate' and 'heavy' stocking treatments respectively

A STATE-AND-TRANSITION MODEL OF GRASS SPECIES DYNAMICS IN THE SEMI-ARID SAVANNA OF NATAL

Rainfall appeared to play on over-riding role in determining range dynamics in this study, although stocking acted to influence or modify the effect of rainfall particularly during the post-drought recovery phase (e.g. Chapter 8). McDonald (1982) and Emslie (1985) reported a similar pattern in the semi-arid savanna of Natal where the frequency of perennial grasses increased with increased rainfall while the frequency of pioneer species increased following dry seasons.

A state-and-transition model, based on three discrete states each characterised by grass species which responded to rainfall and grazing, was developed to summarise grass species dynamics in response to rainfall and grazing in this study (Figure 6).

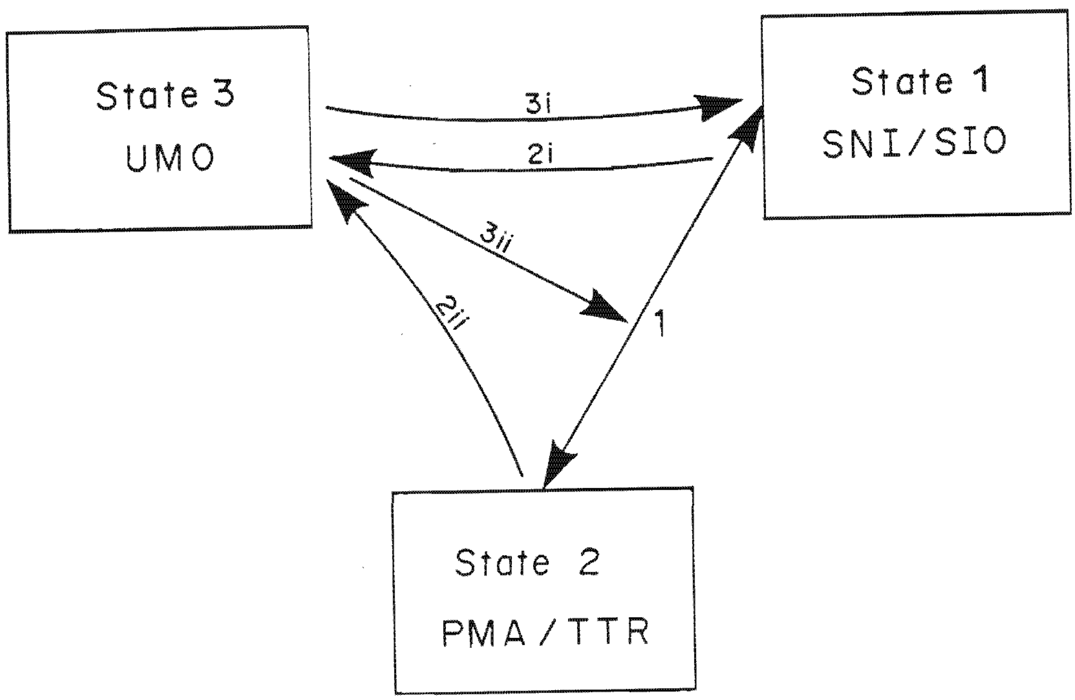

Figure 6 A state-and-transition model of the response of grass component of a semi-arid savanna to rainfall and grazing where SNI denotes *Sporobolus nitens*, SIO - *S. iocladus*, PMA - *Panicum maximum*, TTR - *Themeda triandra* and UMO - *Urochloa mosambicensis*

It is suggested that the position of sites along transition 1 may be related to both long-term grazing history and seasonal rainfall (Figure 6). State 1, characterised by *S. iocladus* and *S. nitens*, may be associated with increased stocking and perhaps increased soil loss (evident as sheet erosion at Dordrecht) and increased tree density. Movement towards state 2, following increased rainfall (Figure 4) and reduced grazing pressure, may be associated with increased proportions of *T. triandra* and *P. maximum* (Figure 3). A drought (i.e. rainfall <250 mm) would comprise a major system disturbance and, irrespective of stocking strategy or position of sites on transition 1, would lead to stability at state 3 (Figure 4) dominated by *U. mosambicensis* (Figure 3) following movement along transitions

2i or 2ii. O' Connor & Pickett (1992) observed that although the seed bank size of *U. mosambicensis* increased with increased stocking it was well represented across a range of stocking levels. This species may play a vital role in stabilising the post-drought soil surface and account for the increased cover with increased stocking at Dordrecht (Figure 5). The extent of post-drought recovery along transition 3 would depend on grazing pressure, with increased movement associated with lower levels of stocking. A general movement of sites towards state 1 in the post-drought phase (outlined as 3i and 3ii) is suggested, although the position sites would occupy would depend on pre-drought position on gradient 1 and hence long-term grazing history.

Soils and topography may influence spatial plant distributions (Archer & Smiens 1991) and hence influence grass species dynamics. For example, Emslie (1985) reported marked differences in basal cover and seedling establishment on Swartland and Shortland soils in the Umfolozi area of the Lowveld. Validation of the conceptual model proposed in this study for a range of soil conditions would consequently be an important consideration.

The question of relevance to range mangers is: are range dynamics primarily rainfall-driven and consequently beyond the control of the manager? It may be difficult to address this question within the relatively short time frame in this study. An assessment of sites with similar soils under communal grazing (stocked at ca 0.3 to 0.7 LSU ha^{-1}) at the same time as the 1994 survey in this study, revealed a dominance of *Tragus racemosa* and *Aristida congesta* subsp. *congesta* (Hatch in prep.) i.e. a similar post-drought recovery recorded in this study did not take place. These species are adapted to shallow rooting conditions (Tainton *et al.* 1983) consistent with eroded sites and are of poor grazing value. This suggested that a history of excessive stocking may lead to the development of an alternate or forth state following extensive soil loss across a threshold which would restrict recovery (Friedel 1991). The development of a forth state may follow a number of rainfall cycles. The pattern of movement may be stepwise rather than linear (Archer & Smiens 1991) and may be associated with extremes of rainfall or grazing impacts.

Soil loss at each stage may eventually lead to a permanent shift to a new domain dominated by shallow rooted species of low forage value as irreversible thresholds are encountered. Walker (1980) and Walker *et al.* (1981) argued that management aimed at sustained production in semi-arid systems may reduce system resilience and increase susceptibility to perturbation. This would be at a cost of reduced production as the system became dominated by pioneer species. Importantly, a change in range composition across irreversible thresholds may be associated with reduced herbage and livestock productivity (as outlined in Chapter 5, 6) and hence reduced human welfare. The challenge for range management is to develop stocking strategies which consider irreversible thresholds and prevent them being reached. Dynamic system modelling approaches may play an important role in developing such understanding.

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CHAPTER 8

POST-DROUGHT RECOVERY PATTERNS IN THE SEMI-ARID SAVANNA OF NATAL

ABSTRACT

The influence of two stocking treatments (0.17 and 0.30 LSU ha^{-1}) on post-drought sward recovery was investigated at two sites in the semi-arid savanna of Natal. The sites differed initially in range composition. Llanwarne was dominated by *Themeda triandra*, *Panicum maximum* and *P. coloratum*, and Dordrecht by *Urochloa mosambicensis, Sporobolus nitens* and *S. iocladus*. Five months of post-drought resting had no consistent effect on tuft numbers, seedling recruitment, species composition or basal cover. Seed bank size decreased as stocking increased. Although increased stocking, particularly at Dordrecht, did not appear to influence seedling recruitment, tuft numbers and basal cover were reduced within a sward dominated by species of low stature such as *Aristida congesta* subsp *congesta*, *Urochloa mosambicensis*, *Sporobolus nitens*, *Sporobolus iocladus* and *Tragus racemosa*. Management recommendations based on post-drought resting are unlikely to be adopted by range managers given high opportunity costs after considerable supplementary feeding costs may have been incurred, and where benefits, such as improved range condition and hence increase herbage production, are not immediately apparent. In addition to the increased cost of forage supplementation during droughts, excessive stocking may be associated with reduced herbage and animal production, and hence reduced profitability, in the post-drought period. Reduced basal cover as a consequence of increased stocking may lead to increased rates of soil loss and contribute to the cost of externalities.

INTRODUCTION

Although range composition dynamics in semi-arid systems appeared to be influenced by rainfall (Chapter 7), stocking may influence post-drought sward recovery (Danckwerts & Stuart-Hill 1988; Chapter 7). While some destocking (largely non-breeding stock) may occur on commercial ranches during droughts, stock numbers are largely maintained during and after droughts through supplementary feeding. Reduced livestock prices, as the result of increased supply to markets during droughts, and elevated post-drought prices may act as further disincentives to destock during droughts. In contrast, stock mortalities under communal pastoralism are often higher during droughts than on commercial ranches, as access to markets is restricted by poor infrastructure and conserved feeds are not readily available (Hatch in press). This may allow rapid sward recovery following droughts (Stuart-Hill 1992), and may emulate the grazing pattern under wild ungulates and nomadic pastoralism prior to sedentary agriculture. The 1991/92 drought, where rainfall was 40% of the long-term mean of 518 mm, provided an opportunity to examine the effect of pre- and post-drought stocking on sward recovery in the semi-arid savanna of Natal.

PROCEDURE

Experimental layout and management follow that detailed in Chapter 5.

One of each of the 'light' (L1) and 'heavy' (H1) stocked treatments at each site was selected as the study area (Chapter 4). Ten 1 m by 1 m quadrats were randomly located and permanently marked in each of the camps in August 1992. Five soil cores (8 cm diameter by 15 cm depth) were collected from each quadrat site and germinated in seedling trays (at a depth of 2 cm), laid out in a randomised block design, under overhead mist-sprayers in a controlled environment. The number of seedlings which emerged weekly were recorded until germination ceased (a five-week period). Although the seedling emergence technique may provide an incomplete assessment of seed bank size and composition (Thompson & Grime 1989), the approach was considered suitable to provide an index of seed bank size for each treatment. An analysis of variance (randomised block design) of seedling numbers (square-root transformed), followed by a least significant difference (LSD) test, examined differences in mean seed bank size within and between treatments and sites (Steel & Torrie 1981).

Five of the quadrats in each camp were protected from grazing with metal-framed wire cages (1.8 by 1.8 by 0.9 m) for six months (September 1992 - April 1993), while the remaining five were grazed. The number and diameter of 'dead' tufts was recorded for each quadrat during September 1992 (date 1). The number of tufts which recovered during the subsequent spring, the number of seedlings, and their heights (cm) were recorded in December 1992 (date 2) and in January 1993 (date 3). Tuft and seedling numbers, tuft and seedling heights, and basal cover (calculated as percent area covered by plant bases) were compared between paired exclosures and grazed areas within treatments with the Wilcoxon two-sample signed-ranks test (Siegel 1956). The Mann-Whitney test (Siegel 1956) was used to compare variables between stocking rates within sites.

The absolute abundance of grass species, based on presence/absence in 25 sub-quadrats (0.2 by 0.2 m), was determined for each quadrat in April 1993 following the notation of Gibbs Russell *et al.* (1990). Detrended correspondence analysis (DCA), in the form of DECORANA (Hill 1979), was used to elucidate patterns in the distribution of species between and within sites. Differences in ranking of samples on DCA axis 1 between grazed and exclosure plots were compared using the Wilcoxon signed-ranks test (Siegel 1956), and between grazed quadrats across stocking treatments and sites using the Kruskal-Wallis test (Siegel 1956).

RESULTS

While seed bank size was lower at Dordrecht than at Llanwarne, 'heavy' stocking at Dordrecht significantly reduced ($P < 0.01$) mean seed bank size relative to 'light' stocking at Llanwarne (Table 1). There was no consistent effect of stocking treatment across sites.

Table 1 **Mean seed bank size (square-root transformed) for the 'light' (L1) and 'heavy' (H1) treatments at Llanwarne and Dordrecht. Distributions with different superscripts were significantly different ($P \leq 0.05$).**

Site	Treatment	Mean
Llanwarne	L1	2.73^{a}
	H1	2.43^{ab}
Dordrecht	L1	2.18^{ab}
	H1	1.44^{b}

LSD: 1.04 (5%)
1.42 (1%)

Although grazing significantly reduced tuft and seedling heights relative to exclosure quadrats within treatments at each site (Table 2), mean tuft and seedling numbers did not appear to be affected by a six month post-drought rest (Figure 1). Although basal cover was significantly lower in grazed than in protected quadrats at Llanwarne, these differences were already evident at date 1 (Figure 1). No significant differences in basal cover were apparent at Dordrecht. It appeared therefore that a six month post-drought rest had little immediate effect on the grass layer.

Table 2 **The influence of post-drought resting on the recovery of the grass layer at date 2 in the 'light' (L1) and 'heavy' (H1) treatments at Llanwarne and Dordrecht**

Site	Llanwarne		Dordrecht	
Treatment	L1	H1	L1	H1
Wilcoxon signed-rank value	T	T	T	T
Tuft number	1.70 NS	1.95 NS	1.16 NS	2.85 **
Seedling number	0.07 NS	1.63 NS	2.45 *	1.33 NS
Cover (%)	2.30 *	2.81 **	0.78 NS	1.79 NS
Tuft height (cm)	3.26 **	3.34 **	3.10 **	2.16 *
Seedling height (cm)	2.61 **	2.34 *	2.46 *	2.43 *

NS = non-significant
* = $P<0.05$
** = $P<0.01$

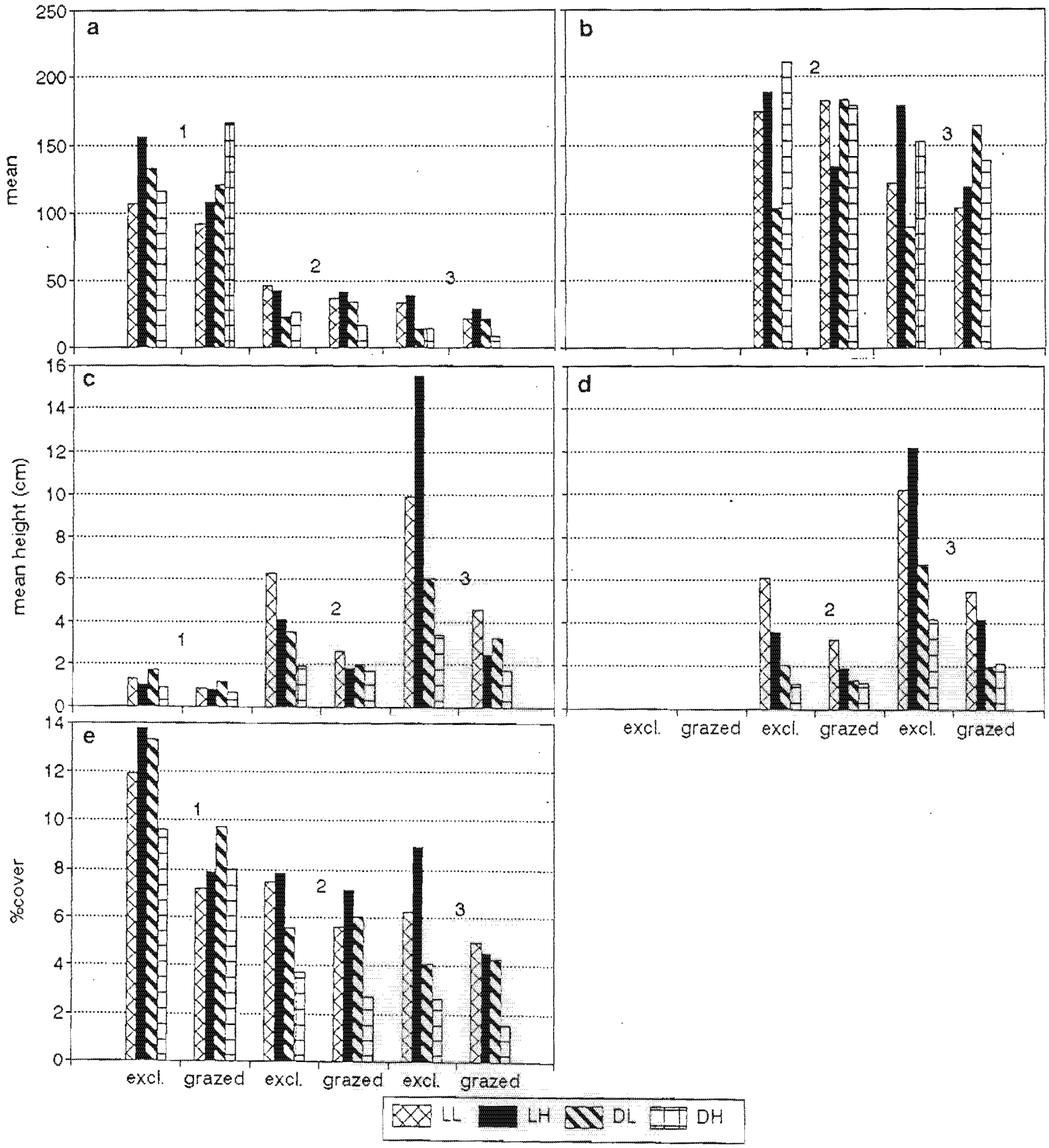

Figure 1 The influence of post-drought resting on a) mean tuft number, b) mean seedling number, c) tuft height (cm), d) seedling height (cm), and e) basal cover at measuring dates 1) September 1992, 2) December 1992 and 3) January 1993 where LL and LH denotes Llanwarne 'light' and 'heavy' and DL and DH Dordrecht 'light' and 'heavy' stocking treatments respectively

A comparison of variables between stocking treatments revealed that tuft numbers did not differ significantly between stocking treatments within sites (Table 3). Tuft mortalities (tuft numbers at date 2 expressed as a percentage at date 1) were, however, considerably higher at Dordrecht (77% and 83% for the 'light' and 'heavy' treatments respectively) than at Llanwarne (53% and 67% for the 'light' and 'heavy' treatments respectively).

Table 3 Influence of 'light' (0.17 LSU ha^{-1}) and 'heavy'(0.30 LSU ha^{-1}) stocking on sward recovery in December 1992 (date 2) and January 1993 (date 3) at Llanwarne and Dordrecht

Site Sampling date Mann Whitney U Statistic	Llanwarne date 2 U	 date 3 U	Dordrecht date 2 U	 date 3 U
Tuft number	0.27 NS	0.76 NS	1.55 NS	1.48 NS
Seedling number	0.49 NS	0.30 NS	0.68 NS	0.38 NS
Cover (%)	0.49 NS	0.57 NS	2.31 *	2.83 **
Tuft height (cm)	1.36 NS	0.87 NS	1.78 NS	2.76 **
Seedling height (cm)	2.34 *	1.32 NS	2.43 *	2.23 *

NS = non-significant
* = P<0.05
** = P<0.01

Seedling numbers did not differ significantly between stocking treatments at either site (Table 3), despite the relatively smaller seed bank size at Dordrecht (Table 1). Although basal cover was not significantly influenced by stocking at Llanwarne, 'heavy' stocking significantly reduced basal cover at Dordrecht (Table 3). Grazing tended to decrease tuft and seedling height at both sites (Figure 1).

The effect of increased stocking at Dordrecht in the post-drought period was reflected further in a comparison of the sites in an ordination (Figure 2). Samples for the 'heavy' treatment at Dordrecht, characterised by *Aristida congesta* subsp *congesta*, *U. mosambicensis*, *Sporobolus nitens*, *S. iocladus* and *Tragus racemosa*, tended to be located towards the left of the ordination diagram. In contrast, samples from Llanwarne tended to be located on the right of DCA axis 1 and were characterised by *P. maximum*, *P. coloratum* and *T. triandra*. No significant differences ($P > 0.05$) in DCA axis 1 scores were evident between grazed and graze-excluded plots over all treatments and sites.

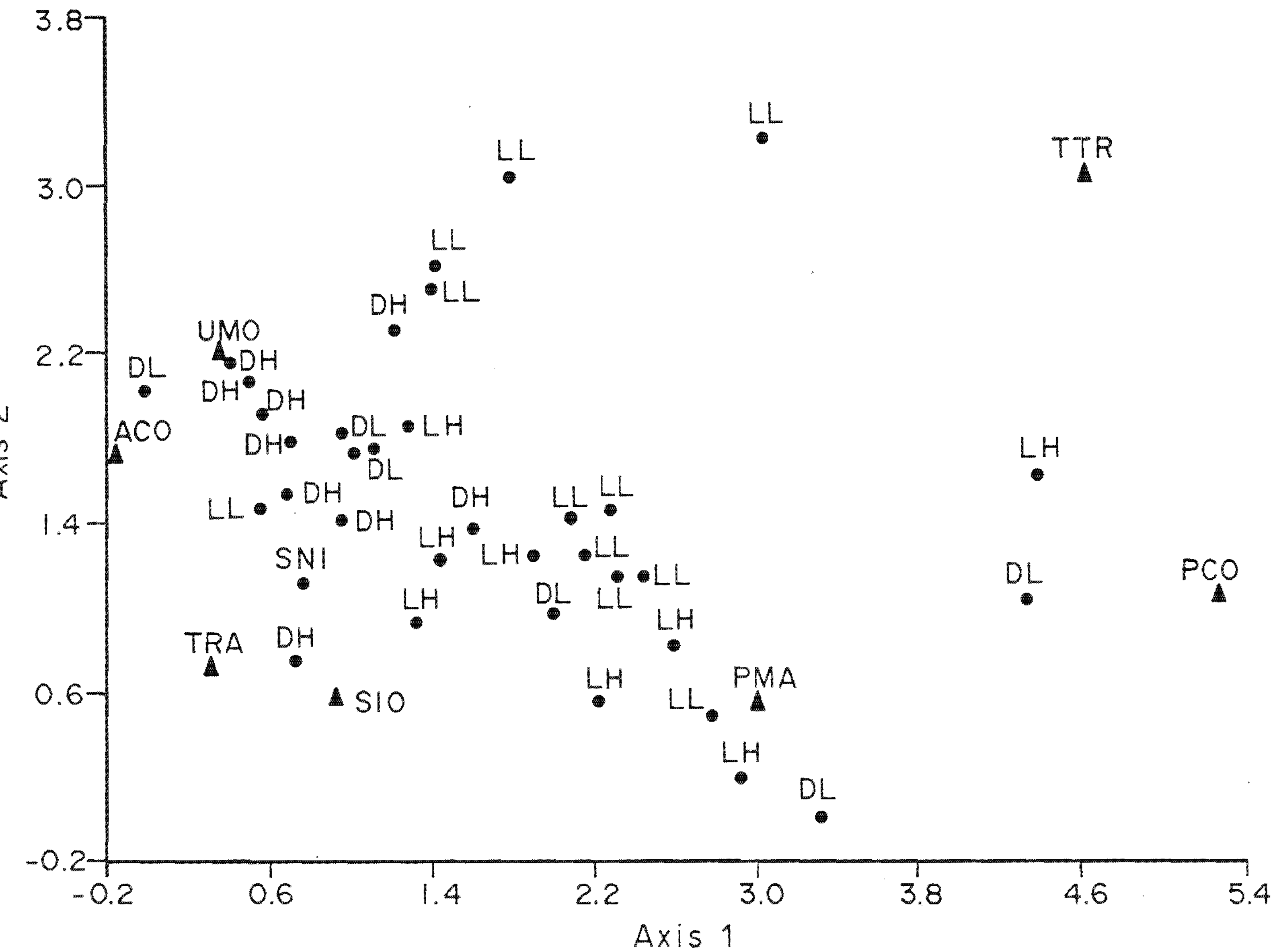

Figure 2 Ordination of samples (●) and species (▲) on DCA axes 1 and 2. LL and LH denote Llanwarne 'light' and 'heavy' and DL and DH Dordrecht 'light' and 'heavy' treatments respectively. Species acronyms are ACO - *Aristida congesta* subsp *congesta*, PCO - *Panicum coloratum*, PMA - *Panicum maximum*, SIO - *Sporobolus iocladus*, SNI - *Sporobolus nitens*, TRA - *Tragus racemosa*, TTR - *Themeda triandra* and UMO - *Urochloa mosambicensis*

It is suggested that DCA axis 1, with an eigen value of 0.66, represented a gradient from pioneer through to perennial-dominated sites, i.e. from those characterised by *S. nitens, T. racemosa* and *U. mosambicensis* to those characterised by *T. triandra, P. maximum* and *P. coloratum*. A comparison of the ranking of sites along ordination axis 1 revealed significant differences ($P<0.01$) between treatments (Figure 3).

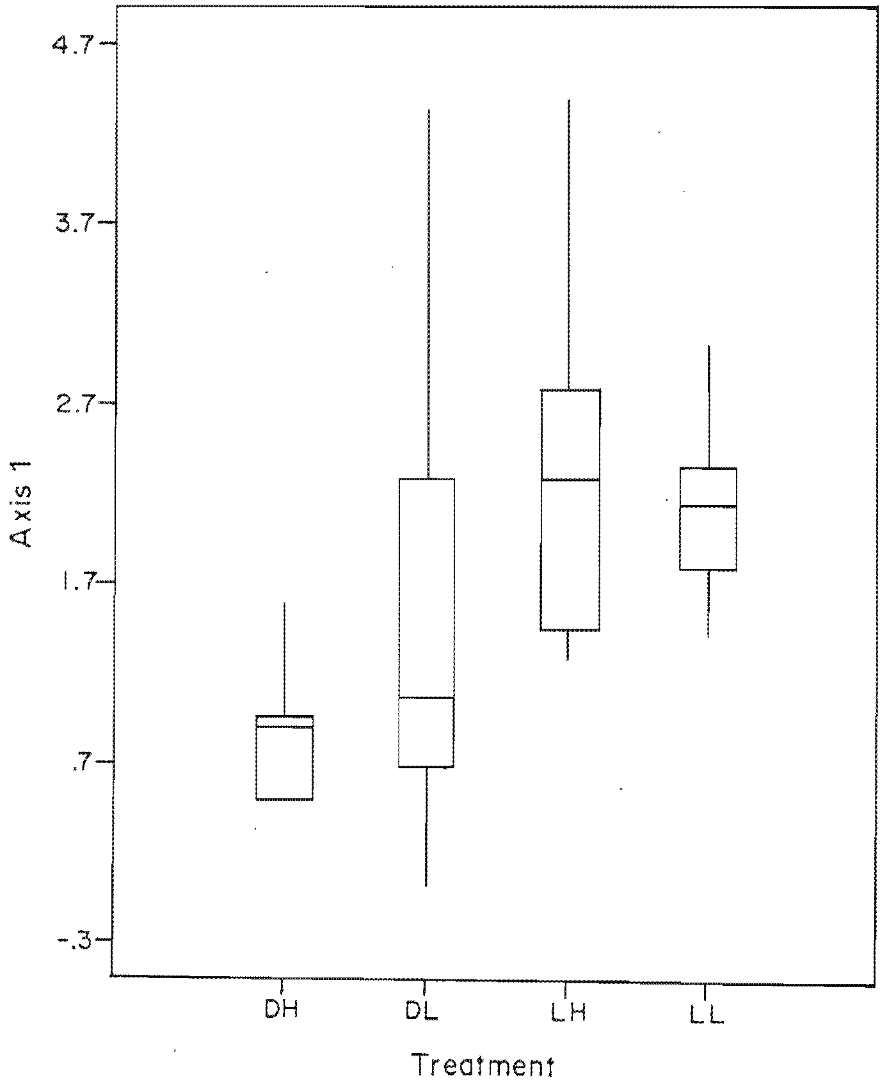

Figure 3 A Box and Whisker plot of DCA axis 1 sample scores for the light (L) and heavy (H) treatments at Llanwarne (L) and Dordrecht (D) respectively. The data range is indicated by a vertical line, upper and lower quartiles are separated by the median within each rectangle

DCA axis 1 score was significantly correlated ($r = 0.62$ $P<0.01$) with mean tuft and seedling heights of grazed samples. This suggested that a change from perennial to pioneer grass species may be associated with a reduction in plant height which may be associated with reduced herbage production (as outlined in Chapter 5).

DISCUSSION

Increased stocking, particularly with reduced range condition, reduced grass seed bank size. Although seedling numbers were not affected by increased stocking, the sward tended to be dominated by pioneer species such as *A. congesta* subsp *congesta*, *U. mosambicensis*, *S. nitens*, *S. iocladus* and *T. racemosa*. O' Connor & Pickett (1992) indicated that while seed production of pioneer species may be maintained under intense grazing pressure, seed production, and hence contribution to the seed bank, of perennial species such as *T. triandra*, may be reduced. Although seed bank composition was not measured in this study it is likely that a post-drought dominance of pioneer species with increased stocking may be the consequence of a seed bank dominated by these species. Seed bank size and composition may consequently be an important indicator of past management and sward recovery potential in semi-arid environments.

Six months of post-drought resting had no consistent effect on tuft numbers, seedling recruitment, species composition or basal cover, although plant height was significantly reduced by grazing. Interestingly, Danckwerts & Stuart-Hill (1988) reported significant increases in tuft mortalities as a consequence of drought in semi-arid rangeland in the eastern Cape.

Increased stocking was associated with increased tuft mortalities and reduced basal cover at Dordrecht, although not at Llanwarne. Similar associations between increased stocking and decreased cover have been reported for semi-arid rangelands in Australia (Scattini 1973) and North America (Branson 1985). Importantly, decreased basal cover may be associated with increased runoff and soil loss from rangelands (McPhee & Smithen 1984; Snyman & Fouche 1991; Emmerich & Cox 1992) and hence increased costs of externalities (Chapter 3).

Although Walker (1980) argued that excessive stocking may enhance system resilience, post-drought dominance of pioneer species may have important economic implications. In addition to increased risk and hence cost of supplementation during droughts, increased stocking may lead to reduced herbage and animal production, and profitability, in the post-drought period.

Management recommendations based on post-drought resting (e.g. Stuart-Hill 1992) are unlikely to be adopted by range managers given high opportunity costs, after substantial supplementary feeding costs may have been incurred, and where benefits, such as improved range condition (Chapter 7) and increase herbage production (Chapter 5), are not immediately apparent. As the detrimental effects of drought may persist for a number of seasons (Danckwerts & Stuart-Hill 1988), the monitoring of the parameters in this study over a number of subsequent seasons is an important consideration.

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CHAPTER 9

TOWARDS THE DEVELOPMENT OF A BIOECONOMIC STOCKING RATE MODEL FOR THE SEMI-ARID SAVANNA OF NATAL

ABSTRACT

Rainfall variability is a major determinant of both system dynamics and profitability of livestock enterprises in semi-arid environments. Range managers require detailed information on the financial and ecological implications of various strategies in order to formulate viable management systems. Data collected over seven seasons (1986-1993), from a series of extensive grazing trials in the semi-arid savanna of Natal with cattle stocked at three rates, were used to develop a spreadsheet-based bioeconomic stocking model. The model (LOWBEEF) comprised two biological sub-models based on step-wise multiple regression models and an integrated economic component. The BEEF sub-model related seasonal livemass gain (kg ha^{-1}) to stocking rate (LSU ha^{-1}) and rainfall (mm). The GRASS sub-model related residual herbage at the end of summer (kg ha^{-1}) to summer stocking intensity (LSU gd ha^{-1}), range condition (indexed as the sum of proportions of *T. triandra, P. maximum* and *P. coloratum*) and rainfall (mm). The period (days) over which supplementary feeding would be required to maintain cattle mass was related to residual summer herbage mass. The biological sub-models were linked to an economic component model (ECON) to reflect the influence of various environmental and economic parameters on profitability.

INTRODUCTION

Climatic and market uncertainties present complex management problems for livestock producers in arid and semi-arid savanna systems. Although productivity is determined largely by stochastic rainfall events (Hoffman & Cowling 1990; Chapter 5), the major decision affecting the level of livestock production is stocking rate (Riechers *et al.* 1989). The stocking rate strategy selected by the manager may have a major impact on short-term profitability while in the long-term may influence the natural resource-base (Foran & Stafford-Smith 1991). Stocking strategies which degrade the resource-base are consequently non-sustainable (Hart *et al.* 1988) and stocking decisions determine both the financial and ecological risk to which the enterprise is subjected. In South Africa range managers tend to increase stocking rate in response to diminishing financial returns (Danckwerts 1989), which may enhance the potential for resource degradation through soil loss and bush encroachment. While high discount rates, price uncertainty and market distortions may favour short-term considerations (Chapter 3), there is the need to provide ecological information in an economic context (Stafford-Smith & Foran 1990) to encourage the development of sustainable livestock production systems. This paper outlines the philosophy underlying the development of a bioeconomic stocking model (LOWBEEF) which may act as a tool to assist decision-making for extensive beef enterprises in a semi-arid savanna.

KEY DESIGN ELEMENTS

Grazing trials established at two adjacent sites in the semi-arid savanna or Lowveld of Natal provided a seven year data set (November 1986 - June 1993) from which component biological sub-models were developed (Chapter 5, 6). The biological sub-models were then integrated into an economic sub-model to provide a spreadsheet-based bioeconomic model - LOWBEEF (LOWveld Bioeconomic Efficiency Forecasting) (Figure 1), which provides the basis for the development of a decision-support model (Hatch & Goosen in prep.).

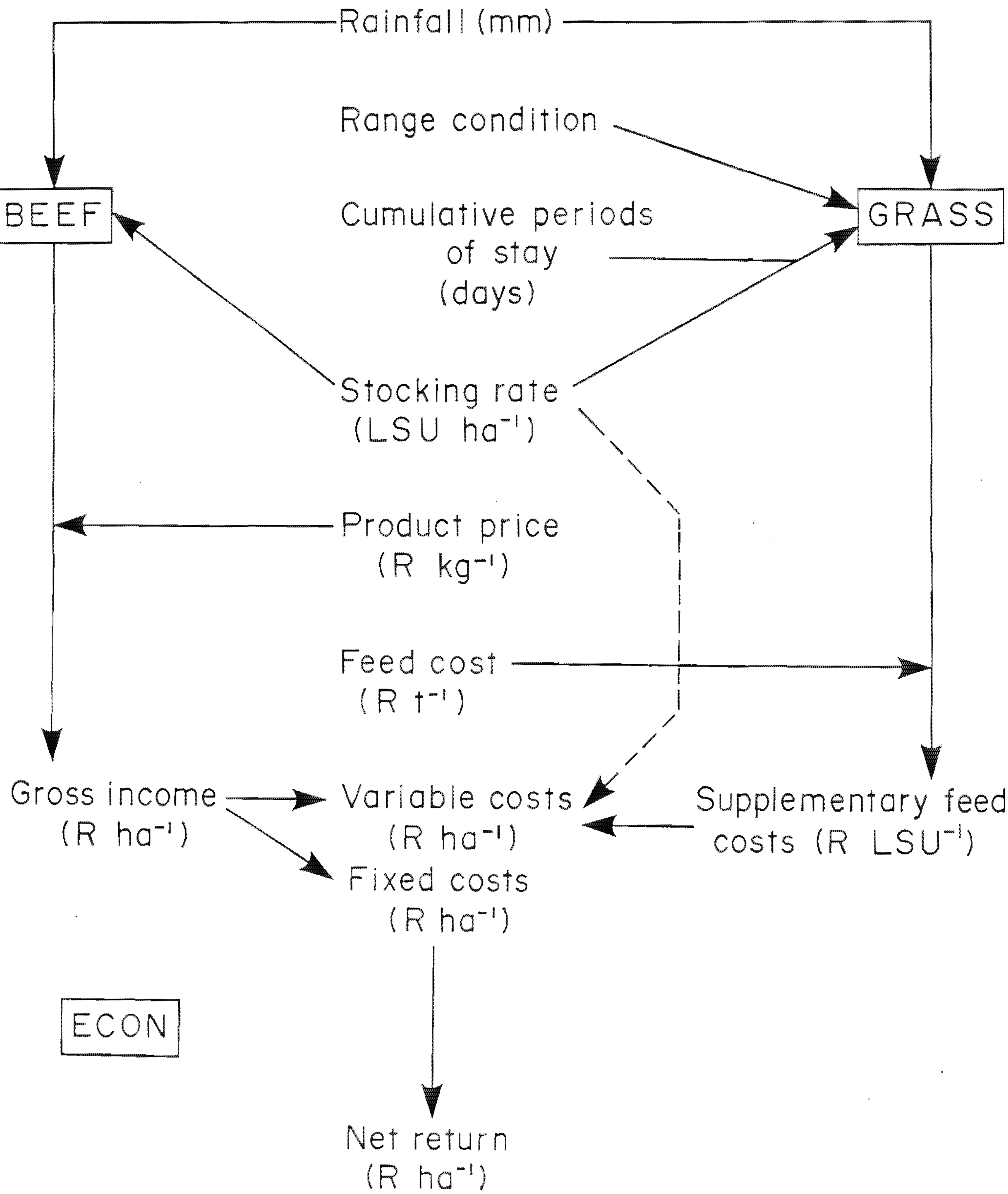

Figure 1 The relationship between the biological (BEEF and GRASS) and economic (ECON) component sub-models of LOWBEEF (LOWveld Bioeconomic Efficiency Forecasting)

Data format

The biological inputs were designed to be readily available or in a format which could be measured by range managers at the camp scale (outlined in bold). These include:

i) **total seasonal rainfall (mm)** (measured from 1 July to 30 June), which is recorded routinely on most farms or available from regional climatic databases (such as the Computing Centre for Water Research, University of Natal or Agrometerology Section, Department of Agriculture);

ii) **range composition**, which is based on the sum of proportion of three abundant species i.e. *Themeda triandra*, *Panicum maximum* and *P. coloratum*. These species may be identified with little training, particularly during the spring/summer period when they flower. In addition, data collected on fixed transects over a number of years may prove useful for monitoring;

iii) stocking rate, which may be calculated using the animal unit equivalents approach of Aldermann & Barber (1973) (Chapter 4) based on **animal mass (kg)** data collected at regular intervals from farm scales; and

iv) cumulative summer grazing days, which may be calculated for each camp based on stocking rates and **cumulative periods of stay (days)**.

Economic data may be obtained from farm records, economic survey data (such as the Mail-In Record Scheme maintained by the Directorate of Agricultural Economics) or COMBUD cost projections published by the Directorate of Agricultural Economics.

Summer animal sub-model (BEEF)

The BEEF sub-model (Chapter 6) predicted summer beef production (kg ha^{-1}) in relation to total seasonal rainfall (mm) (recorded from 1 July to 30 June), and stocking rate (LSU ha^{-1})[1] as follows:

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[1] LSU denotes large stock unit as defined by Meissner *et al.* (1983).

$$G = -25.317706 + 180.861889(SR) + 0.095159(RAIN) - 0.000073(RAIN)^2$$

$$n = 42, R^2 = 0.75$$

where G = summer livemass gain (kg ha^{-1})

SR = stocking rate (LSU ha^{-1})

RAIN = rainfall (mm)

Summer herbage mass sub-model (GRASS)

The GRASS sub-model (Chapter 5) reflected the influence of range condition score, rainfall (mm) and cumulative summer grazing days (indexed as LSU grazing days ha^{-1}) on residual herbage mass at the end of summer (kg ha^{-1}) (Chapter 5) as follows:

$$HMS = 1775.249493 - 14.102845(GD) + 12.4506(RANGE) + 1.371655(RAIN)$$

$$n = 84, R^2 = 0.54$$

where HMS = residual herbage mass at end of summer (kg ha^{-1})

GD = cumulative summer grazing days (LSU gd ha^{-1})

RANGE = sum of proportions of *T. triandra*, *P. maximum* and *P. coloratum*

RAIN = rainfall (mm)

Winter deficit period

A forage deficit component of GRASS (Chapter 5) reflected the number of days over which supplementary feeding would be required to maintain animal mass.[2] This was defined as the difference between the number of grazing days provided by residual summer herbage mass (assuming a dry matter intake, including wastage, of 15 kg LSU^{-1} d^{-1}) and the actual length of the winter period (from the

[2] Range managers may in practise accept a certain level of winter mass loss, particularly for growing stock where compensatory growth in the following spring may restore lost mass.

last date in summer to the first date in the following spring on which >15 mm of rain was recorded) as follows:

$$DEFICIT = 261.778967 - 0.081917(HMS)$$
$$n = 72, R^2 = 0.45$$

where DEFICIT = forage deficit period (days)
HMS = residual herbage at end of summer (kg ha^{-1})

Economic sub-model (ECON)

The ECON sub-model reflected gross income per hectare (R ha^{-1}) as the product of summer livemass gain (kg ha^{-1}) and the beef price (R kg^{-1}) as follows:

$$I_g = P_{beef} * G$$

where I_g = gross income (R ha^{-1})
P_{beef} = beef (sale) price (R kg^{-1} livemass)
G = summer mass gain (kg ha^{-1})

Net return to land and management (R ha^{-1}) was calculated as gross income less total costs (R ha^{-1}), which comprised the sum of fixed (R ha^{-1}) and variable (R LSU^{-1}) costs as follows:

$$I_n = I_g - [C_f + (C_v * SR)]$$

where I_n = net return to land and management (R ha^{-1})
I_g = gross income (R ha^{-1})
C_f = fixed costs (R ha^{-1})
C_v = variable cost (R LSU^{-1})
SR = stocking rate (LSU ha^{-1})

Fixed costs comprise those that are incurred irrespective of the level of stocking, such as depreciation on infrastructure and machinery, and fixed labour costs. Variable costs increase proportionately as stocking rate increases, and include interest costs of livestock purchased (based on market interest rates or the opportunity cost), supplementary feed (based on the winter forage deficit component of GRASS), veterinary (including dipping and inoculation costs) and marketing costs (including transport and agent's commission) (Chapter 2) (Table 1).

Table 1 Assumed cost structures in 1993 Rands for an extensive beef-production system in the semi-arid savanna of Natal (after Whitehead 1993)

fixed costs ha^{-1}	R	variable costs LSU^{-1}	R
improvements	2.50	veterinary	18.00
vehicles	6.90	marketing	21.00
machinery	3.40	miscellaneous	5.00
labour	20.40	interest	112.50[a]
electricity	2.20	feed[b]	
miscellaneous	2.20		

[a] Assuming an initial animal mass of 300 kg, interest rate of 15 percent and beef (purchase) price of R2.50 kg^{-1}

[b] season dependent (Table 2)

Interest costs on the purchase of cattle were calculated as follows:

$$I_c = \frac{IRATE}{100} * \frac{PERIOD}{365} * (M * P_{beef})$$

where I_c = interest cost (R LSU^{-1})

IRATE = interest rate (percent)

Period = purchase to sale date (days)[3]

M = initial animal mass (kg)

P_{beef} = beef (purchase) price (R kg^{-1} livemass)[4]

The cost of supplementation required to maintain mass over the forage deficit period predicted by GRASS was calculated as follows:

$$C_{feed} = DEFICIT * INTAKE * P_{feed}$$

where C_{feed} = winter feed cost (R LSU^{-1})

DEFICIT = forage deficit period (days)

INTAKE = daily intake rate (kg dry matter d^{-1})[5]

P_{feed} = feed cost (R kg^{-1})

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[3] Taken to be one year (365 days) in this study.

[4] Although beef purchase and sale prices may differ in practice, they were assumed to be equal in this study.

[5] 15 kg LSU^{-1} d^{-1} (including wastage) was assumed.

The cost of supplementary feed[6] may depend on rainfall which would influence the aggregate demand for the feed in a region and hence the cost (Table 2).[7]

Table 2 **Assumed feed cost structures (R t^{-1}) for three rainfall (mm) conditions in the semi-arid savanna of Natal**

Rainfall (mm)	Feed cost (R t^{-1})
<400	300.00[a]
400 - 600	200.00[b]
>600	100.00[b]

[a] Prevailing cost during the 1992/93 season
[b] Estimated

DISCUSSION

The LOWBEEF model may provide a useful decision-support tool to examine the relative importance of various biological and economic determinants of profitability and optimum economic stocking rate. A deterministic model may, however, be of limited use in a variable environment where rainfall may influence system dynamics (Chapter 7). Although the outcome of alternate management strategies may be evaluated using historical rainfall sequences, such long-term data sets are rare. Stochastic procedures have consequently been used to generate weather data sets comparable to actual records for a given location (Wright & Hanson 1991). The incorporation of stochastic elements, such as rainfall, based on simple probabilities may greatly enhance model utility and allow the user to investigate the influence of management on financial and ecological risk.

Although several models have been developed to model grass growth (e.g. Noy-Meir 1978), animal production (e.g. McKeon *et al.* 1982; Turner 1990) and

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[6] For example, hay or sugar-cane tops.

[7] Although it may be possible to linearly relate feed cost to rainfall, in practise the demand may be stepwise as reduced rainfall would lead to forage shortages and sudden increases in feed demand.

financial return (e.g. Stafford-Smith & Foran 1990), few bioeconomic models have considered or integrated system dynamics, despite the availability of system models (e.g. Starfield *et al.* 1993). Incorporation of models of system dynamics (such as outlined in Chapter 7) may vastly improve the predictive ability of the bioeconomic model. This is fundamental to addressing the issue of long-term sustainability and in evaluating the economic implications and ecological risk of various stocking strategies.

Although the data on which the model was developed were based on three levels of fixed stocking, range managers may in practise adopt a range of strategies (as outlined in Chapter 2). These may include conservative or low stocking to avoid the consequences of drought and associated forage deficits through to 'heavy' stocking with increased probabilities of forage deficits. Variable stocking strategies may take advantage of increased forage availability during wetter seasons and avoid the consequences of drier seasons through stock reductions. In addition, the type of system in the semi-arid savanna of Natal may vary from beef to beef/game and game systems (Chapter 4). Within beef enterprises these may vary from weaner through mixed cow and steer systems to speculative systems based on the purchase and sale of various classes of stock. The BEEF sub-model in this study was based on growing stock which may show a greater tolerance to forage deficits than would breeding stock. Foran & Stafford-Smith (1991) argue that gain in saleable mass may ignore reproductive stock and the production of replacement stock. Importantly, the forage deficit concept incorporated into the LOWBEEF model considers the effect of increased summer stocking on winter forage deficits and hence feed costs. This would be highly relevant to breeding stock where the loss of condition over the winter period, as a consequence of poor nutrition, may lead to decreased spring calving rates (Meaker 1978).

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CHAPTER 10

MODEL VALIDATION AND TESTING

The result of a mathematical development should be continuously checked against one's own intuition about what constitutes reasonable biological behaviour. When such a check reveals disagreement then the following possibilities must be considered:

a. a mistake has been made in the formal mathematical development;
b. the starting assumptions are incorrect and/or constitute a too drastic oversimplification;
c. one's own intuition about the biological field is inadequately developed; or
d. a penetrating new principle has been discovered.

HJ Gold *Mathematical modeling of biological systems* (in Gleick J 1987 *Chaos: making a new science.* Sphere, London)

INTRODUCTION

Quantitative ecological modelling provides a useful tool for the study of complex system dynamics where little quantitative data are available (Starfield & Bleloch 1991). The modeller is, however, faced with a dilemma by the conflicting needs to summarise the system into manageable quantitative elements (brevity) and to capture the essence of the system (detail). Identification of the primary determinants of system dynamics may overlook small but important elements (often described as 'noise'), which acting in isolation or in concert may drive the system. This is the premise on which chaotic behaviour in biological systems is based (May 1976).

The question which arises is how well does the LOWBEEF model represent reality? Model evaluation criteria may be based on issues such as how well the model serves the purpose for which it was designed and secondly on what underlying assumptions were made (Starfield AM, Department of Ecology, Evolution and Behaviour, University of Minnesota, Minneapolis). Model validation consequently forms an important component of the modelling process (van der Molen & Pintér 1993) (Figure 1)

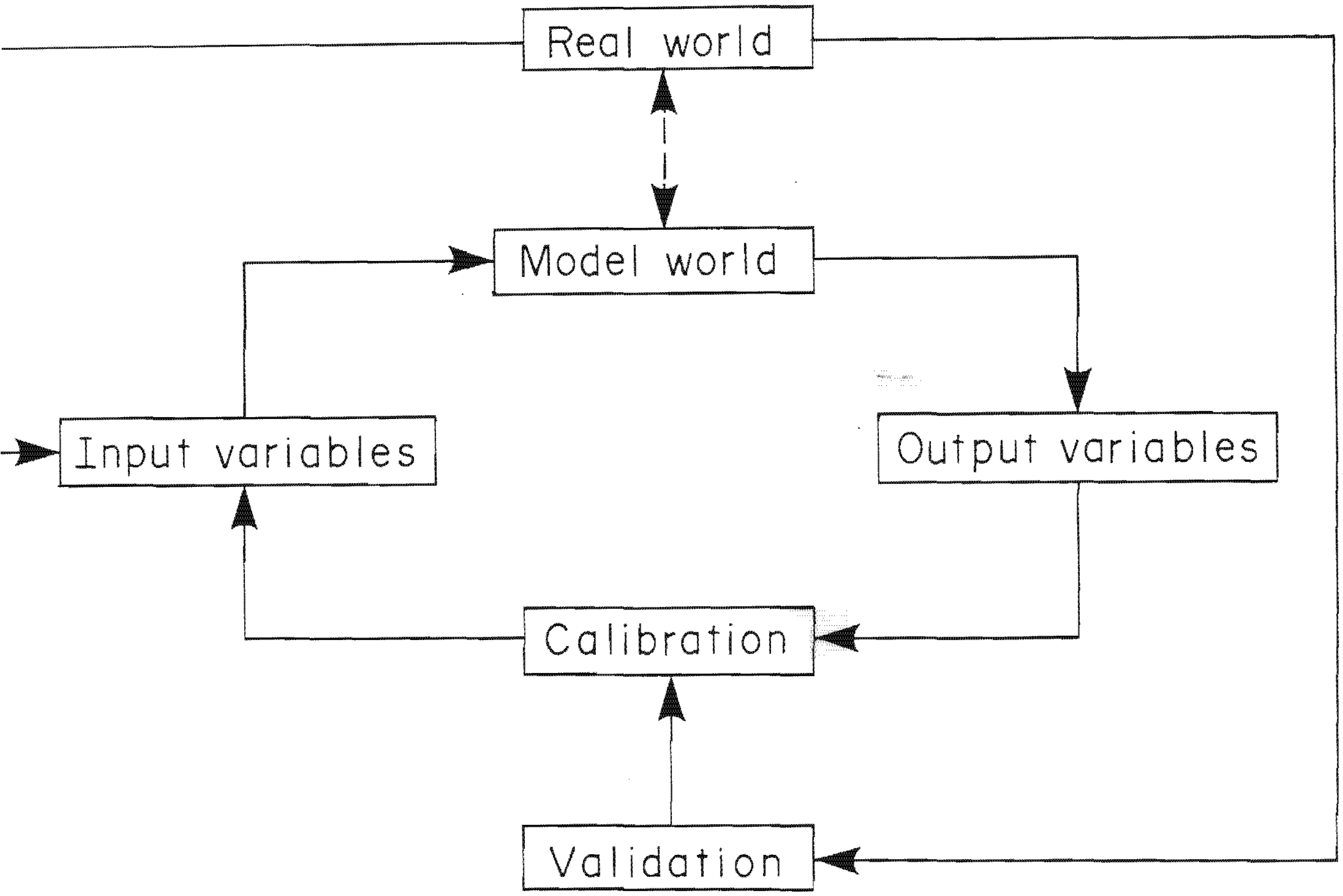

Figure 1 The conceptual relationship between the components of model development

Although model validation may provide a comparison of predicted values with the real world (observed values) (McKinion & Baker 1982), validation cannot guarantee the accuracy of predictions (Power 1993) nor unquestionably establish the 'truth' of a model (Konikow & Bredehoef 1992; Power 1993). Oreskes *et al.* (1994) subsequently argued that because the validation or verification[1] of models of natural systems (which are never closed) is impossible, models remain of largely heuristic value. Model validation can therefore never prove a model to be valid i.e. to represent reality, but may prove a model to be invalid (Harrison 1990), i.e. to poorly represent reality.

MODEL VALIDATION PROCEDURE

Data from a number of sources may be considered for model validation (Dent & Blackie 1979). These may include:

1) a component of the original data set, where the data are partitioned into model development and model testing sets. This approach may lead to a loss of data for development of the primary model and consequently weaken the model. The test data set may subsequently be included if the model is accepted;
2) an independent data set collected from different sites; or
3) data collected from the same sites, after the model development data set was collected.

Although approach 2 would provide for unbiased validation with independent data, such data were not available for the semi-arid savanna. Data collected during the 1993/94 season from the two sites at which LOWBEEF was developed, which may be considered to comprise an independent data set (Starfield AM, Department of Ecology, Evolution and Behaviour, University of Minnesota, Minneapolis), were consequently used to validate the GRASS and BEEF components of the LOWBEEF model.

Within a wide range of validation techniques, regression (or correlation) of observed versus predicted data may provide a useful validation procedure (Mayer & Butler 1993). Paired predicted and observed data for the 1993/94 season were

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[1] While verification implies an establishment of truth, validation does not imply an establishment of truth (although truth is not excluded). Rather it refers to the establishment of legitimacy (Oreskes *et al.* 1994).

compared using regression analysis (Dent & Blackie 1979; Mayer & Butler 1993). The slope and intercept coefficients of the resultant model were then compared with the slope and intercept coefficients (i.e. one and zero respectively) of the perfect-fit model (y = x), using the t-test (Steel & Torrie 1981).

RESULTS

Validation of the GRASS sub-model

The GRASS component of the LOWBEEF model was validated by a comparison of predicted and observed residual herbage masses at the end of summer (kg ha^{-1}) using data for the summer of 1993/94 where rainfall of 572 and 658 mm was recorded at Llanwarne and Dordrecht respectively.

The predicted and observed residual herbage masses at the end of summer (kg ha^{-1}) were significantly related ($r = 0.75$ $P<0.01$ DF = 10) (Figure 2), and the resultant model did not differ significantly ($P<0.05$) from the expected y = x model.

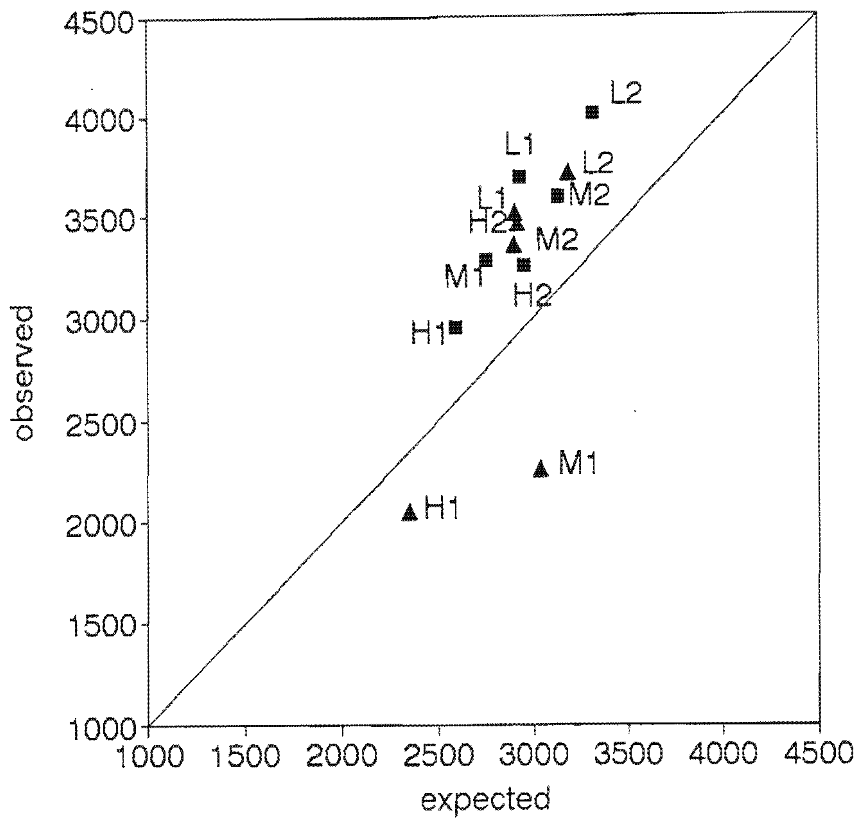

Figure 2 The relationship between expected and observed residual herbage masses at the end of summer (kg ha^{-1}) reflecting deviation about the model y = x where **L**, **M** and **H** denotes 'light', 'intermediate' and 'heavy' stocking for the group 1 and 2 camps at Llanwarne (■) and Dordrecht (▲) respectively

Interestingly, the M1 and H1 camps at Dordrecht reflected considerably lower observed (2258.7 and 2047.0 kg ha^{-1}) than predicted (3035.6 and 2349.3 kg ha^{-1}) residual herbage mass estimates (Figure 2). Although exclusion of these data would improve the correlation coefficient (to 0.89), these outliers may serve to reflect an important consequence of increased stocking.

An increase in post-drought stocking pressure led to a sward dominated by pioneer species of low forage production potential (Chapter 8). The discrepancy between the predicted and observed values for the M1 and H1 camps at Dordrecht (which were grazed in the post-drought recovery period) suggests that the cost of increased post-drought stocking may be a subsequent reduction in summer grass growth, reflected as a reduction in residual herbage mass at the end of summer of 776.9 and 302.3 kg ha^{-1}, which may lead to a reduction in winter grazing capacity of 52 and 20 LSU gd ha^{-1} respectively.

Validation of the BEEF sub-model

The predicted and observed summer mass gains (kg ha^{-1}) for the 1993/94 season were significantly correlated ($r = 0.99$ $P<0.01$ DF $= 4$) (Figure 3), and the resultant model did not differ significantly ($P>0.01$) from the expected $y = x$ model.

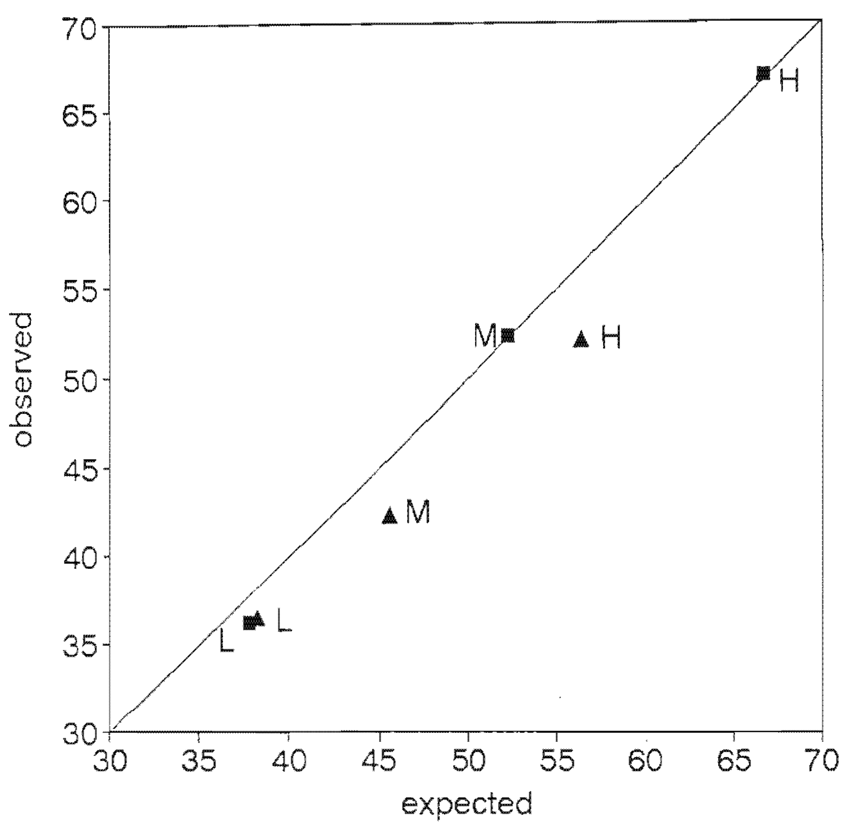

Figure 3 The relationship between expected and observed summer livemass gains (kg ha^{-1}) reflecting deviation about the model $y = x$ where L, M and H denotes 'light', 'intermediate' and 'heavy' stocking at Llanwarne (■) and Dordrecht (▲) respectively

DISCUSSION

The sample size with which the GRASS (n = 12) and BEEF (n = 6) models were validated was small. Although collection of additional data would increase the observed sample size, model validation can never prove a model to be valid (Harrison 1990), but may provide the basis on which to reject a model. The validation in this study procedure revealed that the GRASS and BEEF models may be able to reliably predict (at least for the sites at which the data were collected) the amount of residual herbage at the end of summer (kg ha^{-1}) and summer livemass gains (kg ha^{-1}). The LOWBEEF model may consequently provide a useful basis from which to examine the bioeconomic consequences of various stocking strategies for the semi-arid savanna of Natal. Incorporation of additional data, collected during the 1993/94 and subsequent seasons, into the existing LOWBEEF model may considerably improve the predictive ability of the model.

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CHAPTER 11

DETERMINING OPTIMUM ECONOMIC STOCKING RATE AND PROFITABILITY IN AN UNCERTAIN ENVIRONMENT

ABSTRACT

A spreadsheet-based bioeconomic stocking model (LOWBEEF), based on two biological sub-models (GRASS and BEEF) and an integrated financial component (ECON), was used to determine optimum economic stocking rate (LSU ha^{-1}) and net return (R ha^{-1}), and sensitivity to price and interest rate fluctuations and wage and feed cost increases, for various rainfall and range condition scenarios in the semi-arid savanna of Natal. Net return and optimum economic stocking rate increased as rainfall and range condition increased through the effect of increased residual herbage mass at the end of summer (kg ha^{-1}) and consequently decreased forage deficit periods and reduced supplementary feed costs. Net return was highly responsive to changes in beef price where an increase in beef price led to an increase in optimum economic stocking rate and net return. The effect of reduced prices may be compounded by dry seasons, where supply-driven decreases in price may occur. This suggested that for dry seasons the optimum stocking rate was the lightest within the range of economic stocking rates. Although an increase in interest rates would increase variable costs and lead to reduced returns, the influence of interest rates on farm profitability may vary in relation to individual farm debt loads. Increased labour costs (which comprise a fixed cost) would result in a corresponding decline in net return although optimum economic stocking rate would remain unaffected. Increased supplementary feed cost had little influence on net return relative to the effect of demand-driven increases in feed cost as rainfall decreased.

INTRODUCTION

Range managers in arid and semi-arid environments are faced with the challenge of rainfall variability and market uncertainty which influence short-term financial viability and ecological risk. Although application of the concept of a carrying capacity to arid and semi-arid environments has been questioned (Ellis & Swift 1988; Behnke & Scoones 1993), range managers require detailed information of the bioeconomic consequences of various stocking strategies. The development of financially and ecologically sustainable strategies, which maximise profitability while minimising ecological risk, is consequently a challenge. Decision-support models may provide useful tools to examine the sensitivity of profitability and optimum economic stocking rate to various biological and financial inputs and assist in the development of viable management systems.

PROCEDURE

A spreadsheet-based bioeconomic simulation model (LOWBEEF), developed from data collected at two sites (November 1986 - June 1993) (Chapter 9), was used to determine the optimum economic stocking rate (OESR) (LSU[1] ha^{-1}), and net return to land and management (NR)[2] (R ha^{-1}) at the optimum economic stocking rate, within a range of stocking rates from 0.10[3] to 0.50 LSU ha^{-1} for various rainfall (mm) and range condition (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) scenarios. Cost structures for an extensive beef system in the semi-arid savanna of Natal (Table 1) were based on available data in 1993. The model was used to examine the sensitivity of optimum economic stocking rate and net return to price and interest rate fluctuations, and wage and feed cost increases.

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1 LSU denotes large stock unit as defined by Meissner *et al.* (1983).

2 Net return was defined as the difference between gross income (R ha^{-1}) and total costs (R ha^{-1}) which comprised the sum of fixed and variable costs.

3 Although the optimum economic stocking rate may decline to zero during dry seasons (where the loss would then equal fixed costs), complete destocking is unlikely to be considered by range managers where unfavourable tax conditions would be attracted (by the sale of all stock), cash flows and cattle breeding programmes would be severely disrupted and repurchase of stock during the post-drought period (when demand-driven price increases may occur) would severely influence the financial viability of the enterprise.

Table 1 **Assumed cost structures in 1993 Rands for an extensive beef-production system in the semi-arid savanna of Natal (after Whitehead 1993)**

fixed costs ha^{-1}	R	variable costs LSU^{-1}	R
improvements	2.50	veterinary	18.00
vehicles	6.90	marketing	21.00
machinery	3.40	miscellaneous	5.00
labour	20.40	interest	112.50[a]
electricity	2.20	feed[b]	
miscellaneous	2.20		

[a] Assuming an initial mass of 300 kg, interest rate of 15 percent and a beef price of R2.50 kg^{-1}

[b] season dependent (Chapter 2)

DETERMINING OPTIMUM ECONOMIC STOCKING RATE

Although the distribution of net returns or profitability may vary in relation to rainfall, stocking rate and range condition (Figure 1), an optimum economic stocking rate may be calculated as the stocking rate at which net return is maximised (Table 2).

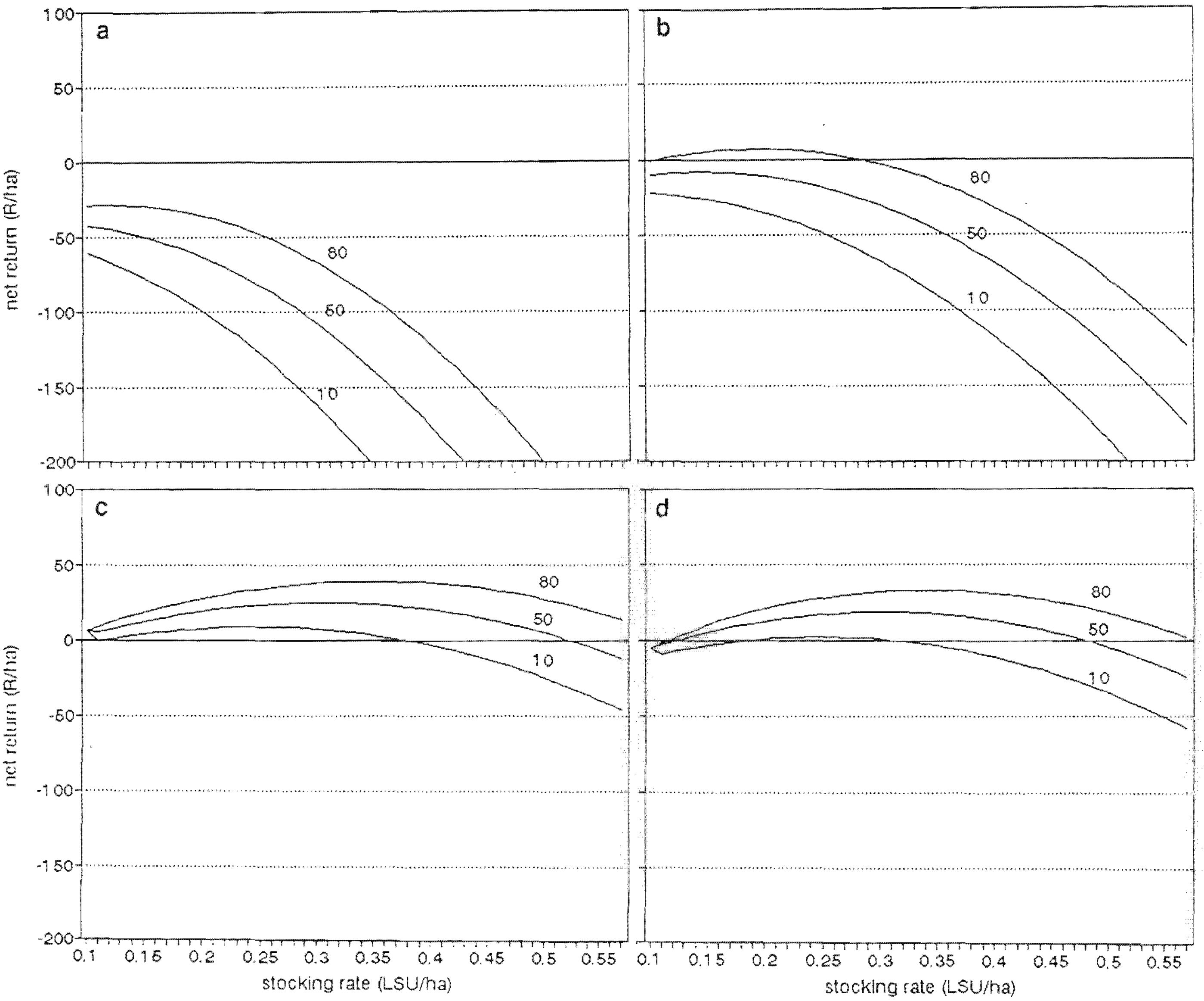

Figure 1 Distribution of net returns (R ha^{-1}) for range condition scores (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) of 10, 50 and 80 for rainfall of a) 300 mm, b) 500 mm, c) 700 mm and d) 900 mm

***maximum* and *P. coloratum*) on optimum economic stocking rate (SR) (LSU ha^{-1}) and net return (NR) (R ha^{-1}) for an extensive beef enterprise in the semi-arid savanna of Natal**

	Range score															
	10		20		30		40		50		60		70		80	
Rain	SR	NR	SR	NR	SR	NR	SR	NR	SR	NR	SR	NR	SR	NR	SR	NR
200	0.10	-79.66	0.10	-75.09	0.10	-70.52	0.10	-65.96	0.10	-61.39	0.10	-56.82	0.10	-52.46	0.11	-47.69
250	0.10	-69.82	0.10	-65.25	0.10	-60.68	0.10	-56.12	0.10	-51.55	0.10	-46.98	0.10	-42.42	0.11	-37.63
300	0.10	-60.89	0.10	-56.32	0.10	-51.76	0.10	-47.19	0.10	-42.62	0.10	-38.00	0.10	-33.49	0.12	-28.41
350	0.10	-52.87	0.10	-48.32	0.10	-43.74	0.10	-39.17	0.10	-34.61	0.10	-30.04	0.11	-25.40	0.13	-20.07
400	0.10	-32.30	0.10	-29.26	0.10	-26.22	0.10	-22.90	0.14	-19.05	0.15	14.65	0.17	-9.72	0.19	-4.26
450	0.10	-26.80	0.10	-23.25	0.10	-20.69	0.12	-17.19	0.14	-13.17	0.16	-8.66	0.17	-3.62	0.19	1.95
500	0.10	-22.21	0.10	-19.16	0.10	-15.99	0.13	-12.34	0.14	-8.21	0.16	-3.54	0.18	1.60	0.19	7.24
550	0.10	-18.52	0.10	-15.48	0.10	-12.17	0.13	-8.37	0.15	-4.08	0.16	0.60	0.18	5.93	0.20	11.65
600	0.24	4.71	0.25	8.47	0.27	12.49	0.29	16.72	0.30	21.21	0.32	25.19	0.33	30.86	0.35	36.04
650	0.24	6.32	0.26	10.09	0.27	14.11	0.29	18.34	0.30	22.82	0.32	27.50	0.33	32.43	0.34	37.56
700	0.24	7.01	0.26	10.81	0.27	14.82	0.29	19.05	0.30	23.51	0.31	28.18	0.33	33.08	0.34	38.19
750	0.24	6.80	0.26	10.60	0.27	14.62	0.28	18.84	0.30	23.30	0.31	27.95	0.33	32.82	0.34	37.92
800	0.24	5.67	0.26	9.49	0.27	13.51	0.28	17.73	0.30	22.17	0.31	26.82	0.33	31.66	0.34	36.73
850	0.24	3.63	0.26	7.46	0.27	11.48	0.28	15.70	0.30	20.13	0.31	24.77	0.33	29.58	0.34	34.63
900	0.24	0.68	0.26	4.52	0.27	8.55	0.28	12.76	0.30	17.18	0.31	21.81	0.33	26.59	0.34	31.62
950	0.24	-3.18	0.26	0.66	0.27	4.69	0.28	8.91	0.30	13.32	0.31	17.93	0.32	22.73	0.34	27.69
1000	0.24	-7.97	0.26	-4.10	0.27	-0.07	0.27	4.14	0.30	8.54	0.31	13.40	0.32	17.93	0.33	22.89

Note: area bounded by dashed line indicates rainfall conditions below which optimum economic stocking rate would decline to zero and net return would be a loss of R37.60 ha^{-1} i.e. fixed costs

Optimum economic stocking rate and net return increased as rainfall increased as the consequence of increased residual herbage mass at the end of summer (kg ha^{-1}) which acted to reduce the forage deficit period (days) (as outlined in Chapter 5) and hence the cost of supplementation. As the relationship between summer livemass gain (kg ha^{-1}) and rainfall was quadratic (Chapter 6), net return reached a maximum with rainfall of 700 mm and then declined (Figure 1). The substantial increase in net returns with an increase in rainfall from 550 to 600 mm would be related to the decreased supplementary feed costs, i.e. R200.00 to R100.00 t^{-1} (as outlined in Chapter 9).

Although optimum economic stocking rate reached 0.35 LSU ha^{-1}, the ability of the sward to sustain this stocking pressure in the long-term may be questioned, i.e. there may be ecological costs associated with such a high stocking rate (e.g. Chapter 7, 8). Importantly, although range managers may elect to stock below the optimum economic stocking rate at the cost of short-term returns, the accumulated herbage may act as a fodder reserve for dry periods or provide fuel for burning to control bush encroachment. This may reduce both the risk of forage deficits and bush encroachment which may increase long-term returns.

While net returns, even at the optimum economic stocking rate, were generally low (the highest was R38.19 ha^{-1}) and often negative (losses of up to R79.76 ha^{-1}), the effect of reduced rainfall may be compounded by reduced range condition (as outlined in Chapter 7). Jansen *et al.* (1992) reported similar low returns for extensive beef systems in the semi-arid savanna of Zimbabwe where 40 percent of ranches surveyed reflected negative net revenues. This suggested that extensive livestock production systems should plan for both low returns and frequent losses associated with low rainfall.

SENSITIVITY OF OPTIMUM ECONOMIC STOCKING RATE AND NET RETURN TO BEEF PRICE

Individual livestock producers have little influence on the total market for beef (Chapter 2) and are 'price-takers' rather than 'price-setters'. Fluctuating product prices are consequently a major source of risk for livestock producers in arid and semi-arid systems, particularly where prices and supplementary feed costs may be related to rainfall patterns (Chapter 9). The LOWBEEF model was used to examine the sensitivity of net return to changes in beef price (R kg^{-1}) from a base of R2.50 for three rainfall (300, 500 and 700 mm) and range condition (10, 50 and 80) scenarios (Table 3).

Table 3 The influence of beef price (R kg^{-1}) on optimum economic stocking rate (OESR) (LSU ha^{-1}) and net return (NR) (R ha^{-1}) for rainfall of 300 mm, 500 mm and 700 mm, and range conditions (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) of 10, 50 and 80 (base = R2.50 kg^{-1})

Rainfall			300			
Range	10		50		80	
Price	OESR	NR	OESR	NR	OESR	NR
base	0.10	-60.89	0.10	-42.62	0.12	-28.41
1.50	0.10	-71.14	0.10	-52.87	0.10	-39.17
2.00	0.10	-66.01	0.10	-47.75	0.10	-34.04
3.00	0.10	-55.77	0.10	-37.50	0.15	-20.94
3.50	0.10	-50.64	0.12	-31.89	0.18	-11.56
Rainfall			500			
Range	10		50		80	
Price	OESR	NR	OESR	NR	OESR	NR
base	0.10	-22.21	0.14	-8.21	0.19	7.24
1.50	0.10	-39.80	0.10	-27.63	0.12	-18.10
2.00	0.10	-31.00	0.13	-19.22	0.16	-0.67
3.00	0.10	-13.20	0.18	4.87	0.23	23.66
3.50	0.15	-2.09	0.22	20.42	0.27	42.55
Rainfall			700			
Range	10		50		80	
Price	OESR	NR	OESR	NR	OESR	NR
base	0.24	7.01	0.30	23.51	0.34	38.19
1.50	0.10	-22.70	0.17	-13.98	0.22	-5.20
2.00	0.10	-13.15	0.24	2.56	0.28	14.31
3.00	0.31	28.40	0.37	48.33	0.41	66.47
3.50	0.38	54.15	0.43	78.55	0.48	99.13

Net return was highly responsive to changes in beef price (Table 3). For example, an increase in beef price from R2.50 to R3.00 kg^{-1} at a range score of 10 and rainfall of 700 mm would result in an increase in net return from R7.01 to R28.40, i.e. a 305 percent increase in response to a 20 percent increase in beef price. Similarly, optimum economic stocking rate increased as beef price increased. Net returns during drier seasons were strongly negative as the consequence of extended periods of forage supplementation. Reductions in the beef price had the effect of increasing the loss. The effect of dry seasons may consequently be compounded by depressed livestock prices where, as producers sell excess stock in response to reduced rainfall, prices may decrease due to increased market supply. This suggested that for drier seasons the most economic rate would be the lightest possible within the economic range. Stocking strategies which cannot be sustained during dry seasons and where the manager may be forced to sell stock may be subject to increased financial risk, while failure to reduced stocking levels may increase ecological risk (as outlined in Chapter 8).

SENSITIVITY OF OPTIMUM ECONOMIC STOCKING RATE AND NET RETURN TO INTEREST RATE FLUCTUATIONS

Although poor monetary management by government has contributed to inflation in South Africa, recent attempts by the Reserve Bank to reduce inflation through control of the money supply has led to the maintenance of positive real interest rates and hence relatively high nominal interest rates. Interest rates may have a major influence on the profitability of extensive beef enterprises, particularly those with relatively high debt levels. The sensitivity of optimum economic stocking rate and net return to changes in nominal interest rates from 15 percent (Chapter 9) was examined for three rainfall (300, 500 and 700 mm) and range condition (10, 50 and 80) scenarios (Table 4).

Table 4 The influence of change in nominal interest rates on optimum economic stocking rate (OESR) (LSU ha^{-1}) and net return (NR) (R ha^{-1}) for rainfall of 300 mm, 500 mm and 700 mm, and range conditions (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) of 10, 50 and 80 (base = 15 percent)

Rainfall			300			
Range	10		50		80	
Interest rate	OESR	NR	OESR	NR	OESR	NR
base	0.10	-60.89	0.10	-42.62	0.12	-28.41
10	0.10	-57.14	0.10	-38.87	0.12	-23.62
16	0.10	-61.64	0.10	-43.37	0.10	-29.37
18	0.10	-63.14	0.10	-44.87	0.10	-32.02
20	0.10	-64.64	0.10	-46.37	0.10	-32.67
25	0.10	-68.39	0.10	-50.12	0.10	-36.42
Rainfall			500			
Range	10		50		80	
Interest rate	OESR	NR	OESR	NR	OESR	NR
base	0.10	-22.21	0.14	-8.21	0.19	7.24
10	0.10	-18.46	0.16	-2.41	0.21	14.88
16	0.10	-22.97	0.14	-9.26	0.19	5.82
18	0.10	-24.46	0.13	-11.31	0.18	3.03
20	0.10	-25.96	0.12	-13.25	0.17	0.36
25	0.10	-29.71	0.10	-17.53	0.15	-5.77
Rainfall			700			
Range	10		50		80	
Interest rate	OESR	NR	OESR	NR	OESR	NR
base	0.24	7.01	0.30	23.51	0.34	38.19
10	0.28	16.76	0.34	35.42	0.38	51.74
16	0.24	5.22	0.29	21.29	0.34	35.64
18	0.10	1.81	0.28	17.01	0.32	30.72
20	0.10	-1.40	0.26	12.93	0.31	25.99
25	0.10	-8.46	0.23	3.70	0.27	15.31

An increase in interest rates, which would increase variable costs (as outlined in Chapter 2), resulted in a decrease in net return and optimum economic stocking rate (Table 4). This would increase the extent of losses for drier seasons or decreased range scores. Conversely, a decrease in interest rates would lead to an increase in net return and the optimum economic stocking rate. Although net return was relatively sensitive to interest rate fluctuations, the response became less elastic as range condition increased. For example, an interest rate increase from 15 to 25 percent for rainfall of 700 mm resulted in an 84 percent and 60 percent decrease in net return for range scores of 50 and 80 respectively (Table 4). Importantly, the influence of interest rates on profitability may vary between farms in relation to farm debt levels.

SENSITIVITY OF OPTIMUM ECONOMIC STOCKING RATE AND NET RETURN TO CHANGES IN LABOUR COST

The effect of changes in labour legislation

New labour legislation in South Africa, namely the Labour Relations Act and the Basic Conditions of Employment Act, has placed increasing emphasis on the role of labour in agriculture (Goedecke & Ortmann 1993). Farm labour in South Africa currently falls outside the control of major trade unions which have made significant wage demands to employers in other industries. Demands have tended to increase as political change in South Africa has provided trade unions with greater latitude and power. The unionisation of farm labour or the introduction of minimum wage legislation, which may follow labour legislation in other sectors of the economy, may lead to increased wage costs to commercial beef producers. This would negatively affect the profitability of extensive production systems where labour contributes 54 percent of fixed costs (Chapter 9). The question of importance to livestock producers in an uncertain environment is: what would be the effect of a 10, 15 or even a 100 percent increase in the cost of labour and how would net return be affected?

Although an increase in the cost of labour (i.e. a fixed cost) would result in a corresponding decrease in profit, or under reduced rainfall and range condition lead to increased losses (Table 5), optimum economic stocking rate would remain unchanged (as outlined in Chapter 2).

Table 5 **The influence of increased labour cost on net return (NR) (R ha^{-1}) at optimum economic stocking rate for rainfall of 300 mm, 500 mm and 700 mm, and range conditions (indexed as the sum of proportions of *T. triandra, P. maximum* and *P. coloratum*) of 10, 50 and 80**

Rainfall			300	
Range		10	50	80
Percent increase	Cost	NR	NR	NR
base	20.4	-60.89	-42.62	-28.41
10	22.44	-62.93	-44.66	-30.45
15	23.46	-63.95	-45.68	-31.47
25	25.5	-65.99	-47.72	-33.51
50	30.60	-71.09	-52.82	-38.61
100	40.80	-81.29	-63.02	-48.81
Rainfall			500	
Range		10	50	80
Percent increase	Cost	NR	NR	NR
base	20.40	-22.21	-8.21	7.24
10	22.44	-24.25	-10.25	5.20
15	23.46	-25.27	-11.27	4.18
25	25.5	-27.31	-13.31	2.14
50	30.60	-32.41	-18.41	-2.96
100	40.80	-42.61	-28.61	-13.16
Rainfall			700	
Range		10	50	80
Percent increase	Cost	NR	NR	NR
base	20.40	7.01	23.51	38.19
10	22.44	4.97	21.47	36.19
15	23.46	3.95	20.45	35.13
25	25.5	1.91	18.41	33.09
50	30.60	-3.19	13.31	27.99
100	40.80	-13.39	3.11	17.79

Net return became increasingly less responsive to increases in wage cost, or increasingly inelastic, as range condition improved. For example, for a range condition score of 80 and rainfall of 700 mm a 10 percent increase in labour cost would see net return decrease by 5.2 percent. In contrast, at a range condition of 10 and at the same rainfall, net return would be highly responsive to changes in wage cost, i.e. for every 10 percent increase in cost net return would decrease by 29 percent.

Increased labour costs may not strongly influence the financial viability of beef enterprises, even in the short-term, and may be supported under above-average rainfall conditions. The reduction in net return would merely reflect the nominal increase in labour cost. In contrast, under low rainfall conditions, particularly with poor range condition, increased wage costs would lead to increased losses. This may negatively influence the financial viability of the enterprise given the high probability of below-average rainfall seasons in semi-arid savanna regions. In the long-term, livestock producers may respond to increased labour costs by decreasing the size of their labour force in order to reduce overall costs i.e. by becoming more efficient. A decrease in the number of farm workers in arid and semi-arid regions of Australia was linked to legislation-enforced wage increases in the early 1970's (DM Stafford-Smith, CSIRO, P.O. Box 2111, Alice Springs, NT 0871, Australia). Increased rural unemployment in South Africa may exacerbate the effects of rural poverty and lead to increased rates of urbanisation. Importantly, beef farmers have alternatives, e.g. game farming, and may shift out of beef if game farming becomes relatively more profitable. In other words, financial viability of a given enterprise is assessed in relative, rather than absolute terms.

SENSITIVITY OF OPTIMUM ECONOMIC STOCKING RATE AND NET RETURN TO INCREASED FEED COSTS

The cost of supplementary feeding in the semi-arid savanna may be strongly influenced by rainfall, as suggested in Figure 2. In addition, demand-linked increases of the cost of supplementary feed may influence profitability and optimum economic stocking rates. Swanepoel *et al.* (1994) reported that, in the semi-arid savanna of the north-west Transvaal, the real cost of purchased feed increased by 201 percent over the period 1979 to 1991 which coincided with a major drought.

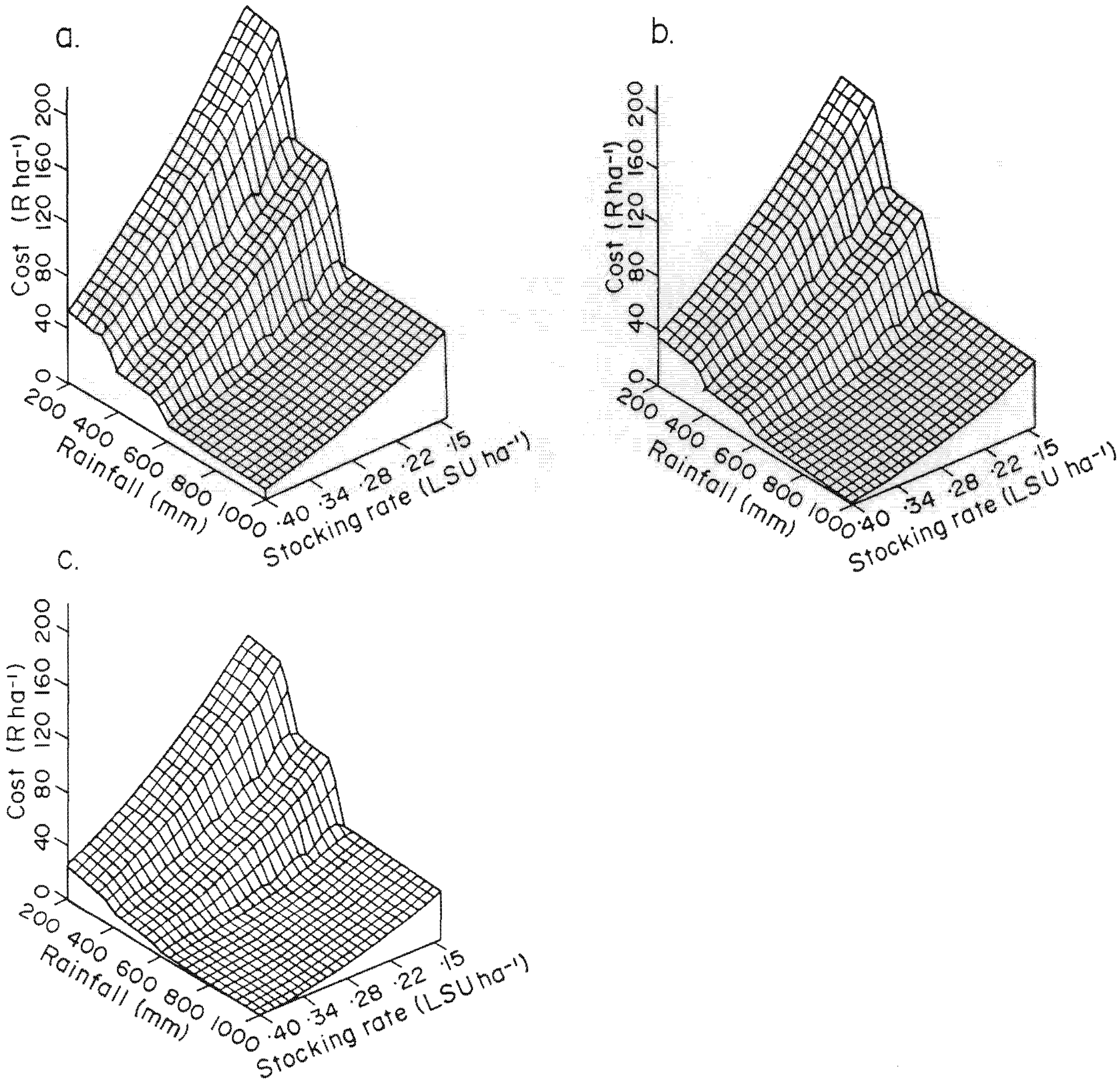

Figure 2 The influence of stocking strategy (LSU ha^{-1}) and rainfall (mm) on winter feed costs (R ha^{-1}) for range condition scores (indexed as the sum of proportions of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of a) 10, b) 50 and c) 80

Feed costs of R300.00, R200.00 and R100.00 t^{-1} for rainfall seasons of less than 400 mm, 400 to 600 and above 600 mm respectively were assumed. The effect of increases in the cost of feed on optimum economic stocking rate and net return were examined for rainfall of 300, 500 and 700 mm and range condition scores of 10, 50 and 80 (Table 6).

Table 6 The influence of increases in winter feed costs on optimum economic stocking rate (OESR) (LSU ha^{-1}) and net return (NR) (R ha^{-1}) for rainfall of 300 mm, 500 mm and 700 mm, and range conditions (indexed as the sum of proportions of ***Themeda triandra, Panicum maximum*** and ***P. coloratum***) of 10, 50 and 80

Rainfall			300			
Range	10		50		80	
Percent increase	OESR	NR	OESR	NR	OESR	NR
base[a]	0.10	-60.89	0.10	-42.62	0.12	-28.41
10	0.10	-65.34	0.10	-45.25	0.10	-30.06
15	0.10	-67.56	0.10	-46.56	0.10	-30.80
20	0.10	-69.79	0.10	-48.39	0.10	-31.43
30	0.10	-74.24	0.10	-50.49	0.10	-32.68
Rainfall			500			
Range	10		50		80	
Percent increase	OESR	NR	OESR	NR	OESR	NR
base[b]	0.10	-22.21	0.14	-8.21	0.19	7.24
10	0.10	-24.63	0.13	-10.31	0.19	5.11
15	0.10	-25.84	0.12	-11.24	0.17	4.34
20	0.10	-27.05	0.12	-12.08	0.17	3.54
30	0.10	-29.47	0.11	-13.57	0.16	2.14
Rainfall			700			
Range	10		50		80	
Percent increase	OESR	NR	OESR	NR	OESR	NR
base[c]	0.24	7.01	0.30	23.51	0.34	38.19
10	0.22	3.30	0.27	19.75	0.32	34.63
15	0.21	1.71	0.26	18.52	0.31	33.08
20	0.19	0.24	0.25	16.62	0.30	31.68
30	0.18	-2.34	0.23	13.97	0.28	29.22

[a]R300.00 t^{-1}, [b]R200.00 t^{-1}, [c]R100.00 t^{-1}

The influence of feed cost increases would depend on rainfall (and feed cost structure) which influences the length of the forage deficit period and hence the cost of supplementary feeding. Net return consequently became increasingly less responsive to increased supplementary feed costs as rainfall and range condition increased. For example, a 30 percent increase in feed costs for rainfall of 500 mm would lead to a 265 and 70 percent decrease in net return for range scores of 50 and 80 respectively. In contrast, for rainfall of 700 mm the decline in net return in response to a 30 percent increase in feed costs would be 41 and 23 percent for range scores of 50 and 80 respectively. This suggested that demand-driven increases in feed cost may have a greater influence on profitability than nominal inflationary increases.

DISCUSSION

Optimum economic stocking and net return were sensitive to rainfall variability, range condition and the level of input variables. It is clearly not possible to provide an economic optimum stocking rate for the semi-arid savanna, although it may be possible to determine a range of economic optimum stocking rates for various cost and price structures, and rainfall and range conditions.

Although the analysis served to demonstrate the effects of varied levels of inputs on output, determination of optimum economic stocking rates and net return may not be as simple in practice where the levels of a number of input variables may vary simultaneously. For example, an increase in inflation may lead to increased nominal interest rates which act to increase costs and eventually prices. This may act to increase variability in returns and hence the risk attached to various stocking strategies. Simulation modelling may consequently provide a useful tool to determine optimum economic stocking rates and net return in relation to variability in both economic and environmental inputs and address simultaneous changes in input variables.

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CHAPTER 12

ASSESSING THE BIOECONOMIC IMPLICATIONS OF VARIOUS STOCKING STRATEGIES IN THE SEMI-ARID SAVANNA OF NATAL USING LONG-TERM RAINFALL RECORDS

ABSTRACT

A spreadsheet-based bioeconomic model (LOWBEEF), developed from data collected at two sites (1986 - 1993) in the semi-arid savanna of Natal, was used to examine the influence of various fixed (0.2, 0.3 and 0.4 LSU ha^{-1}) and variable (where stock numbers were adjusted in relation to previous seasons rainfall) stocking strategies on patterns in residual herbage mass at the end of summer (kg ha^{-1}), the forage deficit period (days) and net return (R ha^{-1}) for various range condition scenarios using a historical rainfall sequence (1931-1991). While a range score of 10 (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*) would see residual herbage mass decline to below a grazing cut-off of 1695 kg ha^{-1} before the end of summer, a range score of 80 suggested that, irrespective of stocking strategy within the range considered, herbage would not become limiting during the summer. The forage deficit period over which residual herbage declined below the grazing cut-off increased as stocking rate increased. This suggested that irrespective of stocking strategy a range score of 10, if established across an irreversible soil loss threshold, would reflect accumulated financial losses over the 60 year period. In contrast, a range score of 80 would lead to positive accumulated net returns. A dynamic range model (where range composition was related to previous seasons rainfall) based on a climate-dependent stocking strategy, suggested that herbage would not become limiting by the end of summer and forage deficit periods would be restricted to an average of 88 days per year. Such an approach would yield a higher accumulated cash surplus than fixed stocking strategies.

INTRODUCTION

Rainfall variability, which is a major determinant of system dynamics (Ellis & Swift 1988; Hoffman & Cowling 1990; Chapter 7), presents major challenges to range managers in arid and semi-arid environments. Range managers require detailed information on the outcome of various stocking strategies in relation to rainfall variability to be able to formulate viable management systems. This paper examines the influence of a range of selected fixed and variable stocking strategies on residual herbage mass at the end of summer (kg ha^{-1}), the forage deficit period (days) and net return (R ha^{-1}) for various range condition scenarios using a historical rainfall sequence in the semi-arid savanna of Natal.

PROCEDURE

A spreadsheet-based bioeconomic model (LOWBEEF) developed from data collected over a seven year period (November 1986 - June 1993) at two sites in the semi-arid Lowveld of Natal (Chapter 9) was used to examine the influence of various cattle stocking strategies and range condition scores on residual herbage mass at the end of summer (kg ha^{-1}), the forage deficit period (days) (when herbage mass declined below a grazing cut-off of 1695 kg ha^{-1} and supplementation would be required to maintain animal mass) and net return[1] (R ha^{-1}) using historical rainfall data. Insufficient rainfall data were available from the sites from which the bioeconomic model was developed and data from the farm Zilverhout (27° 35′ S

[1] Net return was calculated as the difference between gross income (R/ha) and total costs (R/ha) which comprised fixed and variable costs.

31° 44′ E)[2], which spanned a 60 year period (CCWR 1993), were consequently selected (Figure 1).

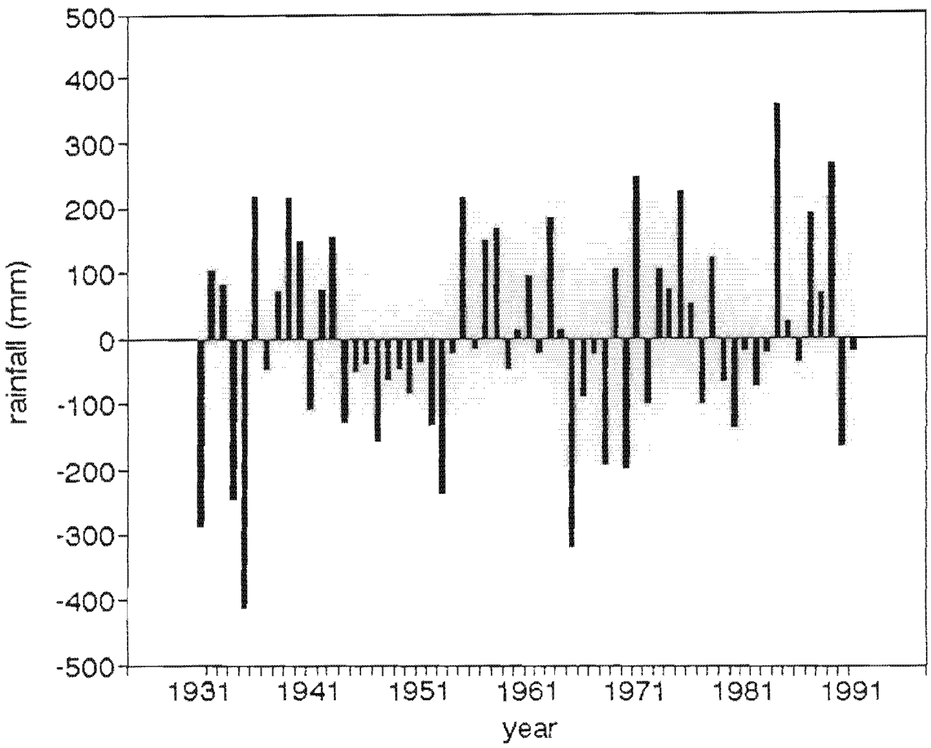

Figure 1 **Seasonal rainfall variability (mm) about the 60 year mean (567 mm) at Zilverhout in the semi-arid savanna of Natal**

Stocking strategies comprised three levels of fixed stocking (0.20, 0.30 and 0.4 LSU ha^{-1})[3], and a climate-dependent strategy, where stocking rate was adjusted according to rainfall during the previous season (Table 1).[4]

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2 Situated ca 20 km from the sites at which the LOWBEEF model was developed.

3 LSU denotes large stock unit as defined by Meissner *et al.* (1983).

4 Although ideally stocking rate would be adjusted within seasons to account for rainfall and hence herbage availability, it would be both impossible to predict the rainfall and impractical to adjust stocking rates accordingly.

Table 1 **A climate-dependent stocking strategy where stocking rate was adjusted according to rainfall in the previous seasons for the semi-arid savanna of Natal**

Rainfall (mm)	Stocking rate (LSU ha^{-1})
< 250	0.15
251 - 400	0.20
401 - 600	0.25
601 -800	0.30
> 800	0.35

Range condition was indexed as the sum of proportions of three key forage species i.e. *Themeda triandra, Panicum maximum* and *P. coloratum*. Although range scores of 10, 50, 80 were considered, range composition in the semi-arid savanna may be determined by seasonal rainfall patterns (Chapter 7). A dynamic range score, where rainfall in the previous season influenced range composition in the subsequent season (Table 2), was therefore included.

Table 2 **Interactions between total seasonal rainfall (mm) in the previous season and range condition (indexed as the sum of proportion of *Themeda triandra, Panicum maximum P. coloratum*) in the semi-arid savanna of Natal**

Rainfall (mm)	Range score
< 250	10
251 - 400	30
401 - 600	50
601 -800	70
>800	80

RESULTS

Residual herbage mass at the end of summer

Patterns in residual herbage mass at the end of summer (kg ha^{-1}) were strongly related to rainfall patterns and range condition (Chapter 5). The static range model revealed a similar pattern where residual herbage mass at the end of summer increased as range score increased (Figure 2 a, b & c). Residual herbage mass decreased with increased stocking and a range score of 10 would see residual herbage mass decline below a grazing cut-off of 1695 kg ha^{-1} before the end of summer for stocking in excess of 0.20 LSU ha^{-1}. In contrast, at a range score of 80, herbage would not become limiting by the end of summer, even at a stocking rate of 0.40 LSU ha^{-1}. Incorporation of a dynamic range score introduced considerable variability and revealed that forage deficits would occur during low rainfall seasons for the 0.30 and 0.40 LSU ha^{-1} strategies, while a climate-dependent strategy would see residual herbage mass at the end of summer consistently above the grazing cut-off (Figure 3).

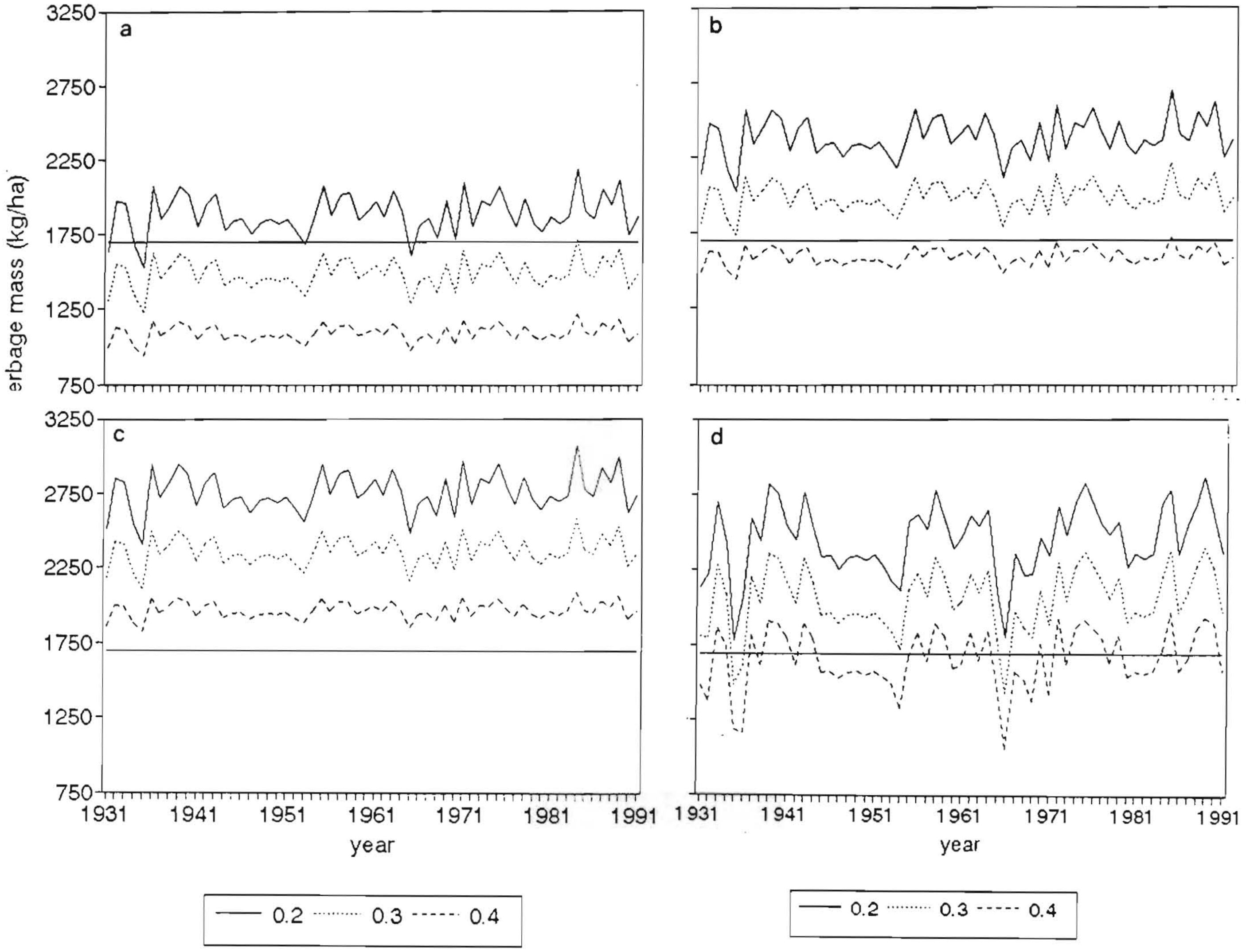

Figure 2 Distribution of residual herbage mass estimates at the end of summer (kg ha^{-1}) (1931 - 1991) for stocking strategies of 0.20, 0.30 and 0.40 LSU ha^{-1} and range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of a) 10, b) 50 c) 80 and d) a dynamic range model

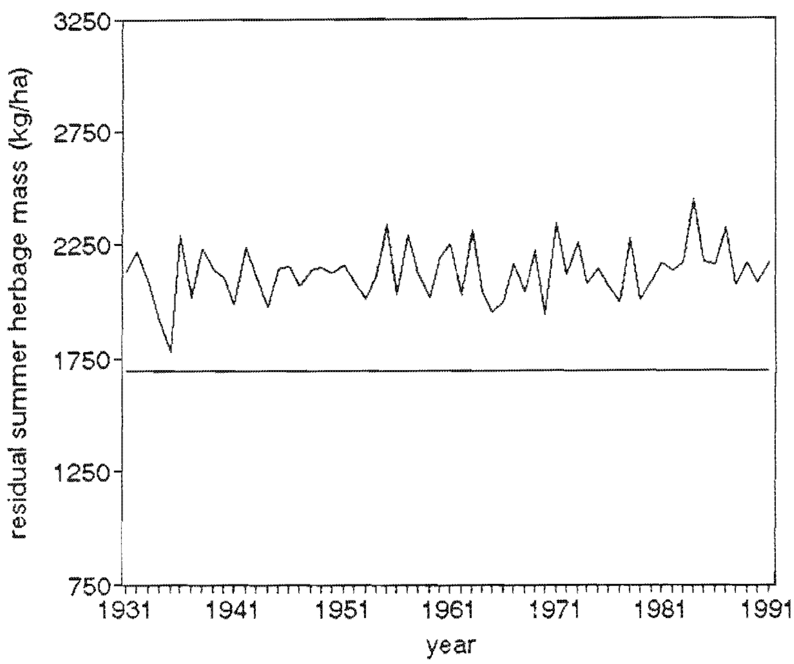

Figure 3 Distribution of residual herbage mass estimates at the end of summer (kg ha^{-1}) (1931 - 1991) for climate-dependent stocking with a dynamic range model

Forage deficit period

Patterns in residual herbage mass at the end of summer influenced the length of the forage deficit period (Chapter 5). The deficit period consequently decreased as range score increased and summer stocking levels decreased (Figure 4). Incorporation of a dynamic range model introduced similar variability (in magnitude but opposite in direction) to that outlined in Figure 2, where the extent of deficits increased as stocking rate increased (Figure 4). A climate-dependent strategy based on a dynamic range model considerably reduced variability in deficit periods which were limited to an average of 88 days per winter over the 60 year period (Figure 5). This contrasted markedly with the fixed strategies (Figure 4) where stock numbers were not reduced in response to decreased rainfall.

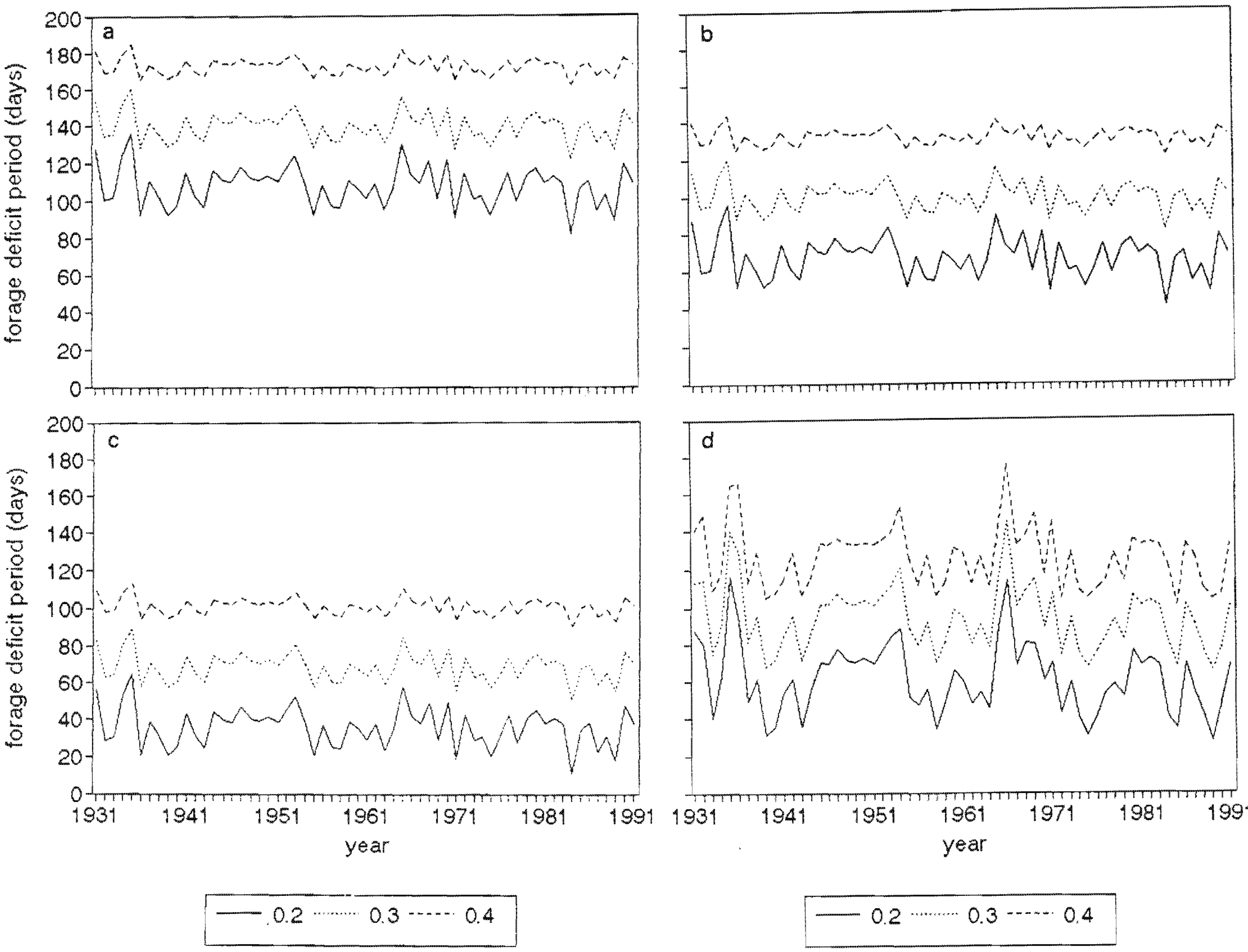

Figure 4 Patterns in forage deficit period (days) (1931 - 1991) for stocking strategies of 0.20, 0.3 and 0.4 LSU ha^{-1} and range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of a) 10, b) 50 c) 80 and d) a dynamic range model

stocking levels, a flexible strategy would yield the highest return over the 60 year period (Figure 7) (Table 1).

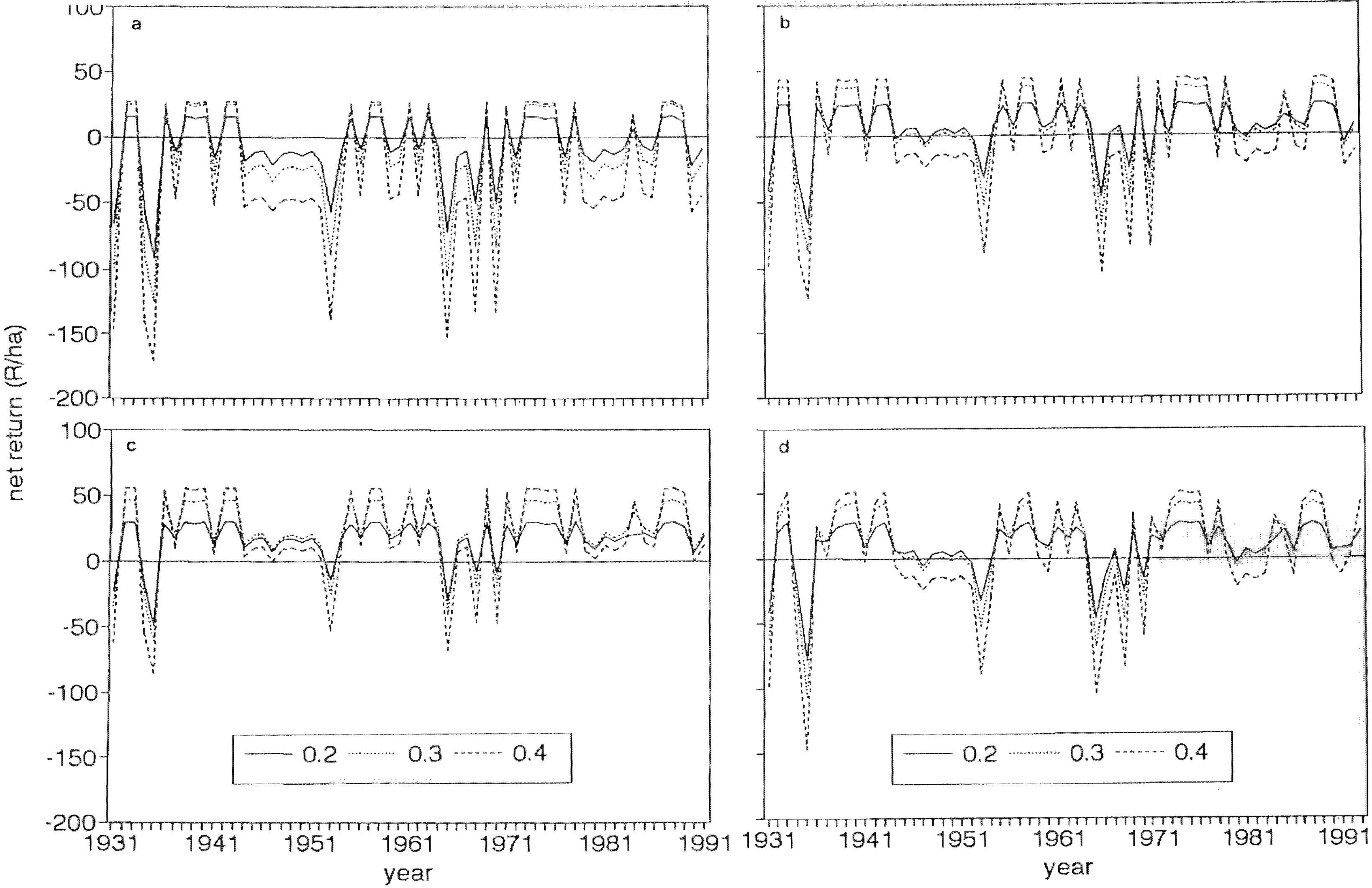

Figure 6 Distribution of net returns (R ha^{-1}) (1931 - 1991) for stocking strategies of 0.20, 0.3 and 0.4 LSU ha^{-1} and range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of a) 10, b) 50 c) 80 and d) a dynamic range model

Table 1 Accumulated cash surplus (in 1993 Rands) (1931- 1991) in relation to stocking strategy and range score (indexed as the sum of proportions of *Themeda triandra, Panicum maximum* and *P. coloratum*)

Stocking strategy (LSU ha^{-1})	Range score 10	50	80	dynamic
0.20	-491.74	364.99	1007.53	483.30
0.30	-844.18	440.91	1404.73	618.38
0.40	-1855.83	-142.37	1147.73	94.25
climate-dependent	-802.91	402.63	1306.78	639.25

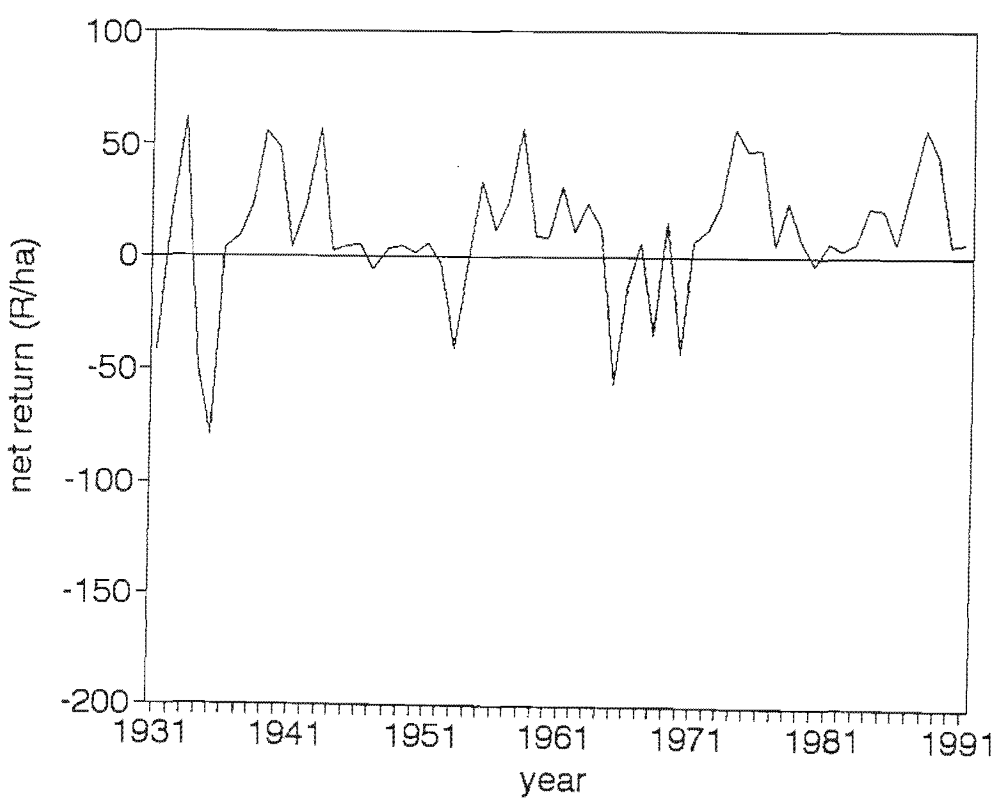

Figure 7 Distribution of net returns (R ha^{-1}) (1931 - 1991) for climate-dependent stocking with a dynamic range model

DISCUSSION

Range dynamics in semi-arid savanna systems may be influenced by rainfall (Chapter 7). A dynamic range model may consequently provide a more appropriate basis for examining the influence of various stocking strategies on profitability than would a static range model. Importantly, a 'permanent' range score of 10 (or less) may be associated with a soil loss threshold beyond which recovery cannot occur (Chapter 7). Such systems, dominated by species of low forage production potential such as *Tragus racemosa* and *Aristida congesta* subsp. *congesta* (Chapter 7), may support livestock production over the summer growth period only. Forage would then become limiting and financial losses would occur irrespective of stocking strategy.

Although conservative stocking (e.g. 0.20 LSU ha^{-1}) reduced losses during drier seasons this was at the cost of higher profits during higher rainfall seasons. A climate-dependent strategy, in contrast, minimised losses during drier seasons and increased returns during wetter seasons. Although a flexible strategy may allow stocking rates to be increased considerably during wetter seasons, to yield greater returns than would fixed stocking, errors may carry high ecological costs (as outlined in Chapter 8). However, the patterns in runs of wet and dry years, which characterise semi-arid systems (McDonald 1982; Foran & Stafford-Smith 1991; Figure 1), suggest a low probability of making such an error.

A range score of 10 (or less), suggesting a sward dominated by *Urochloa mosambicensis* which may occur during the post-drought recovery phase (as outlined in Chapter 7, 8), would result in forage deficits and financial losses irrespective of stocking strategy adopted. Although this may negatively influence the financial viability in the short-term subsequent sward recovery would restore positive net returns. The length of the period of sward recovery (or negative net returns), which may strongly influence the long-term financial viability of the enterprise, may be related to a history of excessive stocking.

Although historical rainfall sequences may provide a useful indication of the possible outcome of various stocking strategies, insufficient data are available to examine the distribution of returns or risk attached to alternate strategies. Incorporation of stochastic rainfall may play an important role in determining financial risk, while integration of models of system dynamics which could provide environmental feedback would address the issue of ecological risk.

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CHAPTER 13[1]

ASSESSING THE RISK OF VARIOUS STOCKING STRATEGIES IN THE SEMI-ARID SAVANNA OF NATAL

ABSTRACT

Rainfall in arid and semi-arid environments is a major determinant of system dynamics which results in seasonal fluctuations in forage production and hence carrying capacity. Stocking strategies may attempt to avoid the consequences of drought through conservative stocking or by marketing stock at the onset of drought, while overstocking may imply extended periods of supplementation and increased ecological risk. The influence of fixed and variable stocking strategies on the distributions of residual summer herbage mass (kg ha^{-1}), forage deficit period (days) and net return (R ha^{-1}) for various range condition scenarios were examined using a spreadsheet-based bioeconomic model (LOWBEEF). The development of cumulative probability distributions may allow range users to assess the level of risk attached to different strategies. Incorporation of the ecological effects of stocking strategies as feedback into the model would be important to evaluate ecological risk.

INTRODUCTION

Range managers in arid and semi-arid systems operate in a complex and uncertain environment where rainfall variability is a major determinant of system dynamics (Ellis & Swift 1988; Hoffman & Cowling 1990). This results in seasonal fluctuations in forage production and hence carrying capacity (Danckwerts & Tainton 1993). Range composition change may be associated with decreased forage production (Danckwerts 1982, Turner 1990, Snyman & Fouche 1993) and reduced cover (Tainton 1981), which may increase the potential for soil erosion.

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[1] Hatch GP, Tainton NM & Ortmann GF 1994 Costing the effects of various stocking strategies in an uncertain environment. *Proc. VII Aust. Range. Congr.* (in press)

Stocking strategies may attempt to avoid the consequences of drought through 'light' stocking, or by marketing animals at the onset of drought (Foran & Stafford-Smith 1991), while overstocking may require extended periods of supplementary feeding. The cost of overgrazing in arid and semi-arid environments may consequently be reflected by increased periods of supplementation, rather than by reduced livestock production, and by increased ecological risk associated with soil loss and woody plant encroachment.

Evaluation of the risk associated with various stocking decisions is fundamental to the development of strategies which meet short-term economic objectives and minimise long-term ecological risk. This paper examines the risk and hence cost associated with stocking strategies for a beef enterprise in the semi-arid savanna of Natal, South Africa.

STUDY AREA

The semi-arid savanna or Lowveld of Natal occupies the broad coastal plain on the eastern seaboard of South Africa and comprises a herbaceous layer dominated by *Themeda triandra*, *Panicum maximum* and *P. coloratum* and a woody layer characterised by *Acacia* species. Rainfall may vary considerably, both temporally and spatially and, while the mean annual rainfall ranges between 500 and 800 mm, seasons of less than 100 mm or over 1000 mm may occur (CCWR 1993). Cropping potential is consequently limited and extensive farming systems comprise beef, beef/game and game enterprises.

PROCEDURE

A spreadsheet-based bioeconomic model (LOWBEEF) developed from data collected over a seven year period at two sites in the semi-arid Lowveld of Natal (Chapter 9) was used to examine the influence of various cattle stocking strategies[2] and

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[2] Stocking strategy may be defined as the combination of strategic management decisions made by the range-user to satisfy ecological, economic and social objectives. These incorporate stocking rate, burning timing and frequency, grazing system, type and level of supplementation,

range condition scores on residual herbage mass at the end of summer (kg ha^{-1}), the forage deficit period (days) (when herbage mass declined below a grazing cut-off of 1695 kg ha^{-1} and supplementation would be required to maintain animal mass) and net return[3] (R ha^{-1}) under stochastic seasonal rainfall (measured 1 July to 30 June) (Table 1). An 800-year simulation was considered adequate (Grieg-Smith 1983) to assess the risk attached to each strategy and cumulative probability distributions (CPD) were developed for each strategy.

Table 1 Rainfall probability intervals for the semi-arid savanna (after Dent *et al.* 1989)

Rainfall (mm)	Probability
< 375	0.2
375 - 606	0.3
606 - 744	0.3
> 744	0.2

Stocking strategies comprised five levels of set stocking (0.20, 0.25, 0.30, 0.35 and 0.40 LSU[4] ha^{-1}) and a climate-dependent strategy, where stocking rate was adjusted according to rainfall during the previous season (Table 2).

breeding and marketing decisions.

3 Net return was calculated as the difference between gross income (R/ha) and total costs (R/ha) which comprised fixed and variable costs.

4 LSU denotes large stock unit as defined by Meissner *et al.* (1983).

Table 2 **A climate-dependent stocking strategy for the semi-arid savanna**

Rainfall (mm)	Stocking rate ($LSU\ ha^{-1}$)
< 250	0.15
251 - 400	0.20
401 - 600	0.25
601 -800	0.30
> 800	0.35

Range condition was indexed as the sum of proportions of three key forage species i.e. *Themeda triandra, Panicum maximum* and *P. coloratum*. Range scores of 10, 50, 80 and a dynamic score, based on a conceptual model where rainfall in the previous season influenced range composition in the subsequent season (Table 3), were examined.

Table 3 **Interactions between seasonal rainfall (mm) in the previous season and range condition in the subsequent season (indexed as the sum of proportion of *Themeda triandra, Panicum maximum P. coloratum*) in the semi-arid savanna**

Rainfall (mm)	Range score
< 250	10
251 - 400	30
401 - 600	50
601 -800	70
>800	80

RESULTS

Residual herbage mass at the end of summer

Residual herbage mass at the end of summer (kg ha^{-1}) was strongly influenced by range condition and stocking strategy, with increased herbage mass estimates associated with reduced stocking and increased range condition (Figure 1).

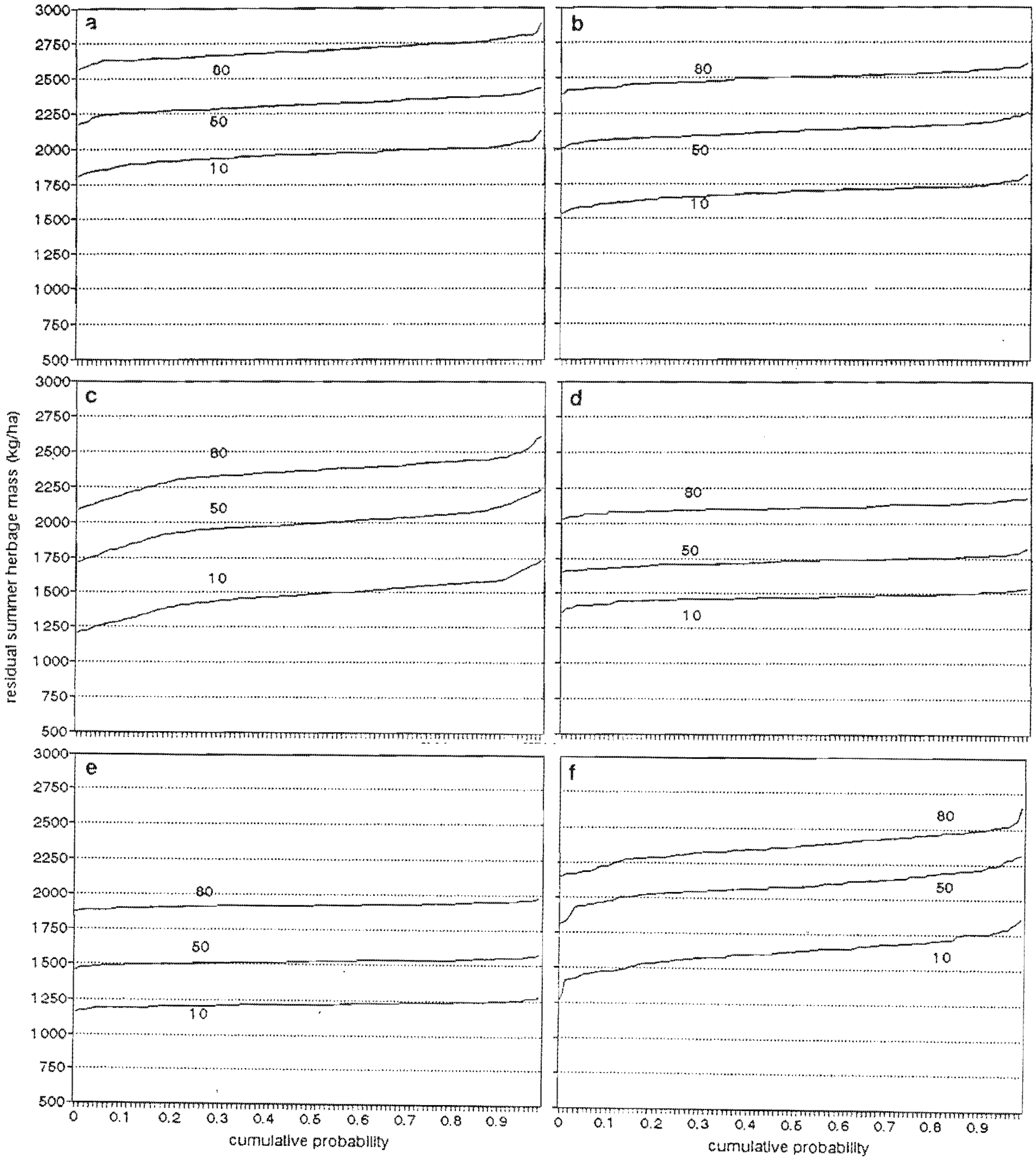

Figure 1 The distribution of residual herbage mass estimates at the end of summer (kg ha^{-1}) for range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of 10, 50 and 80 and stocking strategies of a) 0.20 LSU ha^{-1}, b) 0.25 LSU ha^{-1}, c) 0.30 LSU ha^{-1}, d) 0.35 LSU ha^{-1}, e) 0.40 LSU ha^{-1} and f) climate-dependent stocking

Although the dynamic range model confirmed that increased stocking reduced the amount of residual herbage at the end of summer, there appeared to be little difference between the climate-dependent and 0.30 LSU ha^{-1} strategy (Figure 2).

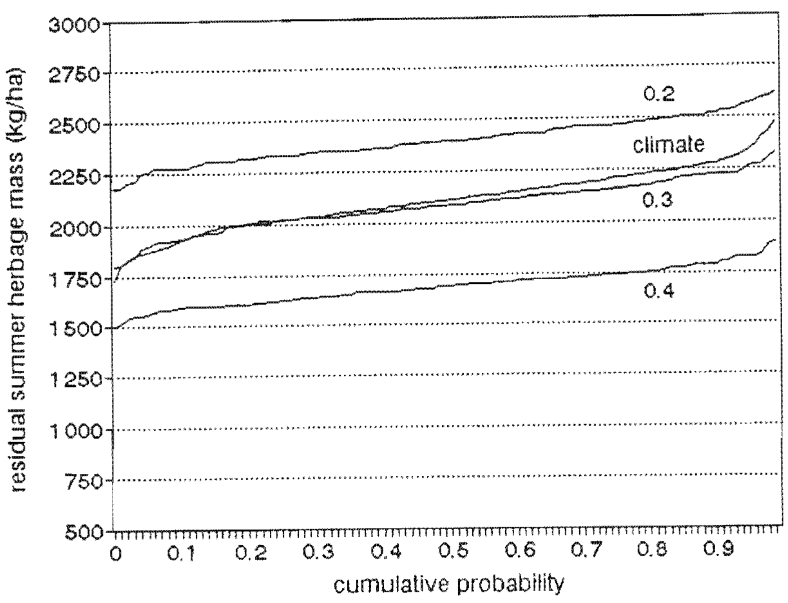

Figure 2 The influence of stocking strategy (0.20, 0.30, 0.40 LSU ha^{-1} and climate-dependent) on the distribution of residual herbage mass estimates at the end of summer (kg ha^{-1}) with a dynamic range model

Forage deficit period

Importantly, residual herbage mass at the end of summer reflected the amount of available herbage for winter. This would influence the forage deficit period over which supplementation may be required. Reduced residual herbage mass at the end of summer, associated with decreased range condition and increased stocking, consequently increased both the probability of forage deficit and length of the deficit period (Figure 3). While this tendency was apparent for the dynamic range model (Figure 4), climate-dependent strategy was able to reduce the period of forage deficit, relative to higher levels of set stocking, by reducing stock numbers during drier seasons.

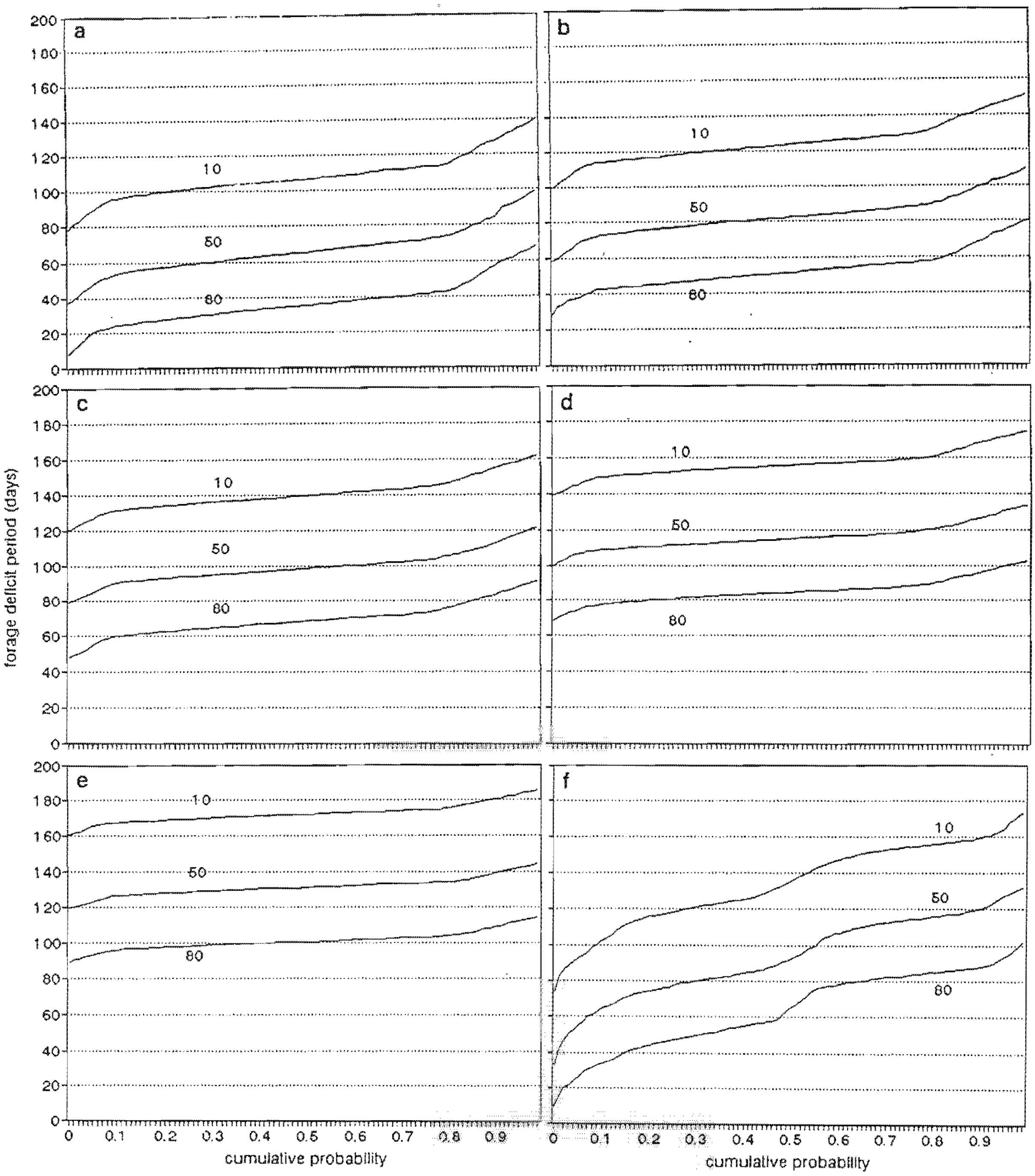

Figure 3 The risk of forage deficit (days) for range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of 10, 50 and 80 and stocking strategies of a) 0.20 LSU ha^{-1}, b) 0.25 LSU ha^{-1}, c) 0.30 LSU ha^{-1}, d) 0.35 LSU ha^{-1}, e) 0.40 LSU ha^{-1} and f) climate-dependent stocking

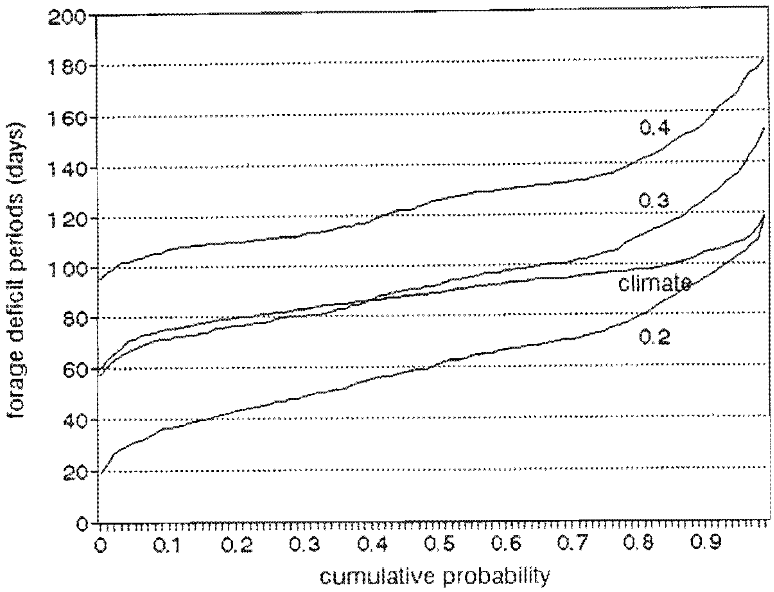

Figure 4 The influence of stocking strategy (0.20, 0.30, 0.40 LSU ha^{-1} and climate-dependent) on the risk of forage deficit (days) with a dynamic range model

Net return

Range condition played an important role in determining the risk of negative returns or losses (R ha^{-1}) (Figure 5). For example, a range score of 10 implied a greater than 50 percent probability of incurring a loss irrespective of stocking strategy while, even for 'heavy' stocking (0.40 LSU ha^{-1}), a range score of 80 had a less than 30 percent chance of showing a loss. Increased stocking acted to influence the slope of the CPD and to increase the variability in returns. This enhanced the probability of both increased profits and losses for the static range (Figure 5) and dynamic model (Figure 6). Climate-dependent stocking reduced the risk of losses during drier seasons while increasing the potential for increased returns during wetter seasons compared to fixed stocking strategies (Figure 6).

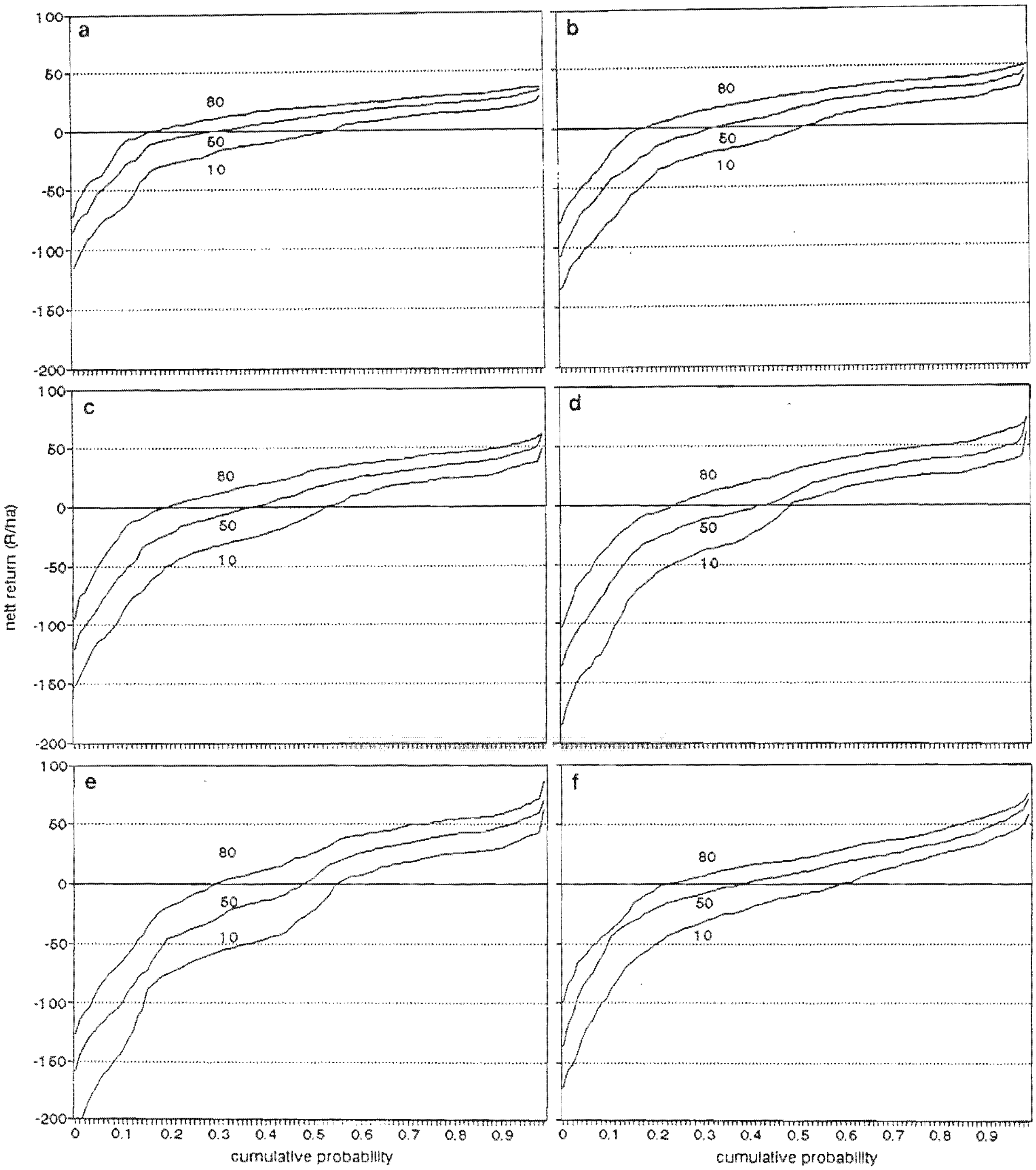

Figure 5 The distribution of net returns (R ha^{-1}) for range condition scores (indexed as the sum of proportion of *Themeda triandra*, *Panicum maximum* and *P. coloratum*) of 10, 50 and 80 and stocking strategies of a) 0.20 LSU ha^{-1}, b) 0.25 LSU ha^{-1}, c) 0.30 LSU ha^{-1}, d) 0.35 LSU ha^{-1}, e) 0.40 LSU ha^{-1} and f) climate-dependent stocking

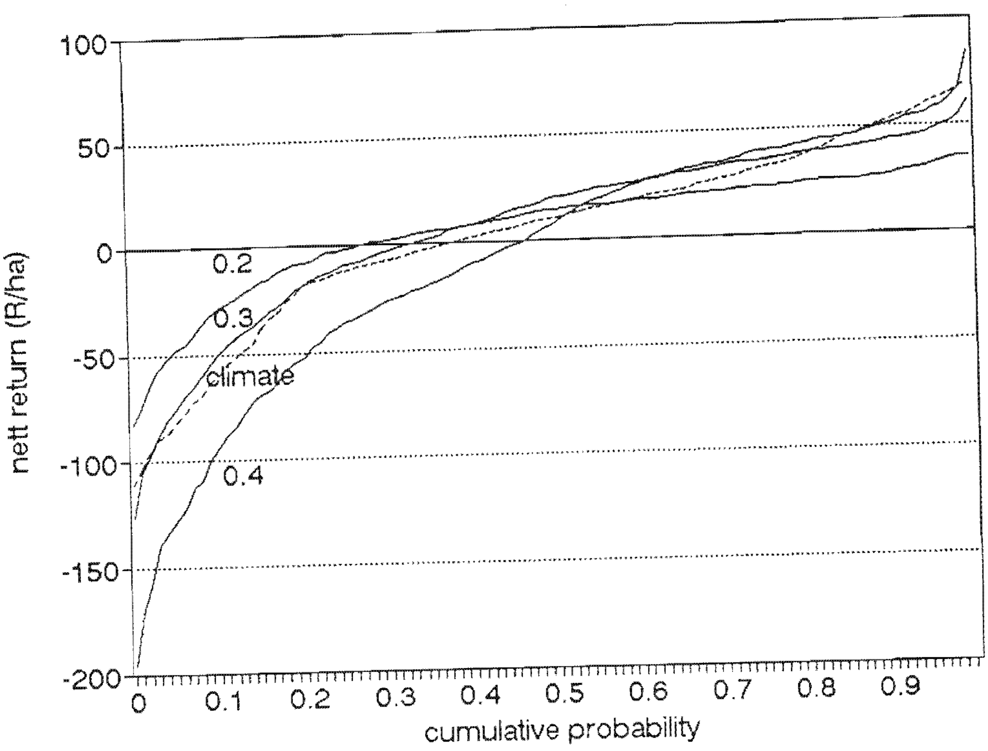

Figure 6 The influence of stocking strategy (0.20, 0.30, 0.40 LSU ha^{-1} and a climate-dependent) on the distribution of net returns (R ha^{-1}) with a dynamic range model

DISCUSSION

Although the risk of forage deficits and financial losses were reduced with lighter stocking, this was at the opportunity cost of higher returns during wetter seasons. In contrast, while increased stocking increased the probability of increased returns during wetter seasons this was at the cost of increased risk of forage deficits and highly negative returns during drier seasons. Importantly, ecological risk may increase as stocking is increased. A flexible or climate-dependent strategy, where stock numbers are adjusted according to previous seasons rainfall, may combine the financial benefits of each approach and reduce financial risk. Errors may carry high ecological costs, where for example the effect of above-average rainfall would be to increase stock numbers into a subsequent dry season with increased ecological risk. Seasonal rainfall in the semi-arid savanna tends to be leptokurtic, suggesting a non-uniform distribution in rainfall i.e. volatility is clustered and runs of 'good' and 'poor' seasons may be expected. An examination of long-term rainfall records for the semi-arid savanna of Natal revealed a less than three percent probability of making such an error.

The development of cumulative probability distributions may allow the range user to assess the level of financial risk attached to different stocking strategies. Although this may address short-term financial objectives, implicit in the development of sustainable strategies is the consideration of ecological risk. Strategies based on increased stocking are not sustainable if they lead to resource degradation. Clearly, stocking in excess of 0.40 LSU ha^{-1} is unlikely to be maintained indefinitely on range with a score of 80, irrespective of the influence of rainfall on range dynamics.

Incorporation of conceptual models of range dynamics would provide feedback into the system and indicate the ecological risk and economic costs associated with various strategies. Although the conceptual model applied in this study reflected the influence of rainfall on range dynamics, stocking strategy may interact with rainfall to determine composition (Chapter 7). Integration of dynamic range models which reflect interactions between rainfall and stocking rate and incorporate the concept of system thresholds (Friedel 1991) may be an important means of assessing the risk of various strategies. The use of cumulative probability distributions derived from bioeconomic models may provide an important extension tool to demonstrate the financial and ecological effects of different strategies.

ACKNOWLEDGEMENTS

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CHAPTER 14

MANAGEMENT IMPLICATIONS FOR LIVESTOCK PRODUCTION IN THE SEMI-ARID SAVANNA OF NATAL

THE INFLUENCE OF RAINFALL AND GRAZING ON RANGE COMPOSITION DYNAMICS

Range composition in this study appeared to be determined by rainfall patterns which led to marked changes over the seven year experimental period (Chapter 7). An increase in the proportion of *Themeda triandra* and *Panicum maximum* was associated with above-average rainfall while a sward dominated by *Urochloa mosambicensis* would be the consequence of a drought. These results were not unexpected and corroborate much of the current literature (e.g. O' Connor 1985; Ellis & Swift 1988; Peel *et al.* 1991) which suggests that range dynamics, and hence productivity, in arid and semi-arid systems are driven primarily by rainfall. These studies have provided the basis for the development of a new paradigm (Mentis *et al.* 1989), based on non-linear (e.g. Westoby et al. 1989) or non-equilibrial system dynamics (e.g. Ellis & Swift 1988), which has strongly countered the traditional range paradigm (e.g. Foran *et al.* 1978) based on unidirectional composition change in response to grazing (Joyce 1993). This has led to suggestions that excessive stocking, as may be experienced under open-access communal land-tenure, has little permanent or irreversible impact on rangelands (e.g Shackleton 1993; Tapson 1993), while application of the concept of a carrying capacity for arid and semi-arid systems has been questioned (e.g. Behnke & Scoones 1993). The danger arises that in replacing the traditional with a new paradigm the impression is created that grazing has no influence on range dynamics in arid and semi-arid systems. Importantly, stocking strategy may interact with rainfall patterns to influence system dynamics, particularly during critical stress periods such as the post-drought recovery phase (Danckwerts & Stuart-Hill 1988; Chapter 8), and lead to the development of alternate states across irreversible thresholds from which recovery cannot occur (Friedel 1991).

It was therefore suggested that soil loss in the semi-arid savanna may lead to stabilisation at a state dominated by *Tragus racemosa* and *Aristida congesta* subsp. *congesta*.

While Mentis *et al* (1989) argued that animal production from range was not related to sward composition, Turner (1990) indicated that range condition may strongly influence potential stocking rates. Although range condition did not influence cattle livemass gain over the summer (within the range of stocking rates applied in this study), range condition strongly influenced the amount of residual herbage at the end of summer and the number of days over which supplementary feeding would be required to maintain animal mass. The cost of excessive stocking in the semi-arid savanna may consequently be related to the increased costs of winter supplementation, rather than to decreased animal production over the summer (as suggested in Chapter 3). Interestingly, Walker (1980) argued that a change in sward composition, as a consequence of excessive stocking, would increase the resilience of a system to disturbances such as drought. Importantly, however, the herbage and animal production potential of a *Themeda triandra-Panicum maximum* dominated sward, during even dry seasons, may considerably exceed that of a degraded system dominated by pioneer species such as *Tragus racemosa* and *Aristida congesta* subsp. *congesta* in even good years.

This suggests that although range dynamics and productivity in semi-arid systems may vary in relation to rainfall, stocking rate may act to modify the effect of rainfall, while exceeding system thresholds (or the carrying capacity) may lead to irreversible change as soil loss occurs. This may lead to stability at a state dominated by species of poor forage production potential. A resilient, degraded system may therefore be the consequence of excessive stocking in the semi-arid savanna, but would certainly not represent the most productive system.

TOWARDS THE DEVELOPMENT OF SUSTAINABLE PRODUCTION SYSTEMS FOR THE SEMI-ARID SAVANNA

The development of sustainable production systems for the semi-arid savanna implies that both financial and ecological risk must be addressed. Strategies based on short-term financial return may ignore long-term economic and ecological costs (as outlined in Chapter 3). Although there is clearly no long-term optimum economic stocking rate for the semi-arid savanna[1], given the variability in biological and economic inputs, it may be possible to identify a range of stocking rates between which the optimum may fluctuate. Importantly, the concept of system boundaries or thresholds must be accommodated to address the issue of ecological risk. From a legislative or regional planning perspective the range in upper stocking limit (for various range and rainfall conditions) would be of interest, i.e the maximum economic stocking rate which could be sustained without resource degradation.

While excessive stocking may increase the risk of forage deficits and reduced financial returns during drier seasons, conservative stocking may reduce the risk of such losses but at the opportunity cost of higher returns during wetter seasons. The optimum economic stocking rate may consequently fluctuate in response to rainfall. It may decrease during drier seasons to the lowest rate within the economic range (as outlined in Chapter 2)[2] and increase considerably during better seasons due to increased herbage availability. This suggests that the most profitable system would be one based on a flexible stocking strategy such as a climate-based stocking approach outlined in this study. Current livestock production systems in the semi-arid savanna, based primarily on breeding stock, may not be appropriate in a highly variable environment as they are unable to adjust stock numbers in response to forage deficits (despite the ability to sell

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[1] It would, however, be possible to retrospectively calculate the optimum economic stocking rate for each season given complete information on biological and financial inputs (as outlined in Chapter 11).

[2] Or decline to zero during dry seasons to reduce the extent of the loss to fixed costs (as outlined in Chapter 11).

excess non-breeding stock) and tend to maintain 'high' stocking rates with the associated feed costs. Although stock numbers may be maintained during wetter seasons, where the economic optimum may reach 0.35 LSU ha^{-1}, any short-term advantage gained by increased stocking may be lost during one dry season where supplementary feeding or drastic stock reductions (of breeding stock) may be required (particularly where beef prices may decrease due to an increase in market supply). In addition, significant ecological costs may be incurred as a consequence of soil loss or woody plant encroachment. The implication is that the production emphasis in a highly variable environment should be based on growing rather than breeding stock. In contrast, in humid rainfall environments where seasonal rainfall variability is low, the production emphasis may be in favour of breeding stock.

While a flexible stocking strategy in the semi-arid savanna would allow range managers to minimise losses during drier seasons and take advantage of increased herbage production during wetter seasons, such a system would need to be incorporated within a livestock production system. Although a purely speculative system, with complete destocking during drier seasons, may reduce supplementation costs and ecological risk, such a system is unlikely to be accepted by range managers due to high financial risk.[3] A mixed breeding system, based on both breeding and growing stock, may be an acceptable compromise[4] where an upper limit of breeding stock would be set at the optimum economic stocking rate for seasons with rainfall of <550 mm (as outlined in Chapter 9). For example, a range score of 30 would imply a stocking rate of 0.10 LSU ha^{-1}, which for a 5000 ha system would equate to 333 cows (assuming that one cow would equal 1.5 LSU). During wetter seasons stock numbers may be increased to the economic optimum by either retaining weaners or by buying in additional stock or

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[3] Associated with fluctuating beef prices (i.e. sale of stock during dry seasons when supply-driven price decreases may occur and repurchase when demand-driven price increases may occur), unfavourable tax conditions (created by the sale of all stock) and erratic cash flows (when suitable stock may often not be available during post-drought recovery periods).

[4] A system based on an integration of breeding and growing stock may also reduce risk relative to breeding or production systems alone.

both, while during drier seasons non-breeding stock would be sold to reduce stocking rates to match available forage. Although the risk remains of reduced profitability associated with supply-driven decreases in beef prices during dry seasons, this may be overcome by adjusting stock numbers at the end of summer in relation to the amount of residual herbage and selling excess stock before prices begin to decline. In addition, growing stock may display greater tolerance to forage deficits (and would regain mass in the subsequent season) than would breeding stock (where reduced conception rates in the subsequent spring may be associated with winter forage deficits).

As the profitability of beef enterprises in the semi-arid savanna may depend largely on rainfall, profits are consequently highly variable and often negative. Integration of browsers into the system may increase profitability and reduce ecological risk and hence costs associated with woody plant encroachment. Despite the popularity of goats in other savanna regions of southern Africa, disease and theft problems have limited the acceptance of goats in the semi-arid savanna of Natal. Although wildlife has tended to be ignored as a potential source of income in the past in favour of cattle, this is rapidly changing as the economic advantages of mixed beef/game systems become apparent. Integration of wildlife enterprises into existing beef systems may consequently enhance profitability (Jansen *et al.* 1992; Cummings 1993) and reduce the financial risk associated with a variable rainfall environment.

While the need for supplementary feeding may be the consequence of excessive stocking during drier seasons, range managers may be tempted to reduce the cost of supplementation by delaying the onset of feeding until animal mass or condition loss occurs. This may lead to both increased opportunity costs (in terms of lost animal production or reduced calving rates in the subsequent season) and increased ecological costs. Cattle may switch to browse under adverse conditions which may limit the extent of mass loss and delay the onset of supplementary feeding. Diet additives, such as polyethylene glycol-based products, may have a similar effect by allowing cattle to overcome the effects of tannins and increase browse intake. A delay in the commencement of supplementation, or alternately the use of browse-intake stimulants, may lead to resource competition between cattle and browsers, such as kudu and nyala. This may considerably increase mortalities of obligate browsers.

There may consequently be major indirect costs associated with delayed supplementation or the use of browse-intake stimulants during forage deficit periods. In the long-term it is suggested that increased utilisation of the browse layer by cattle may lead to a change in species composition, as preferred species such as *Ziziphus mucronata* and *Grewia occidentalis* are replaced with avoided species such as *Dichrostachys cinerea* and *Euclea divinorum*. Although the problem of resource competition may be overcome by pen-feeding stock, this may not be feasible for large livestock enterprises or where stock may not adapt to confined conditions. The forage deficit concept outlined in this study (Chapter 5) may consequently provide an important means of determining when to commence forage supplementation (rather than using actual periods of cattle mass loss which were poorly related to residual herbage mass at the end of summer) (as outlined in Chapter 6). In addition, the forage deficit concept which predicted increasing deficits and hence increased supplementation costs as stocking was increased may provide an important means of assessing the financial and ecological costs of excessive stocking.

As residual herbage mass at the end of summer decreased as stocking rate was increased, fire may be largely eliminated as a management option with excessive stocking. This may increase the risk of bush encroachment in the long-term which may have important implications for long-term sustained return from cattle enterprises.[5] Interestingly, although the optimum economic stocking rate may increase considerably during above-average rainfall seasons (as outlined in Chapter

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5 Although the risk of woody plant encroachment may be reduced by an integration of game into the system, browsers may not occur in sufficiently high densities to have a significant impact in absence of fire, which would act to reduce browse height.

11), it may perhaps pay to stock below the optimum rate and accumulate sufficient herbage to provide high intensity fires to restrict bush encroachment. The long-term economic benefits of such an approach (in terms of reduced bush clearing costs) may outweigh the short-term opportunity costs (of lost cattle production), i.e. the net present value of stocking below the optimum and burning may be greater than that based on stocking at the economic optimum. Importantly, integration of a rotational resting programme into a variable stocking system would allow further flexibility by providing a drought fodder reserve, i.e it would reduce the risk of forage deficits (e.g. Tainton & Danckwerts 1989), and would allow the accumulation of fuel loads to support intense fires to increase top-kill of invasive woody species.

FUTURE RESEARCH CONSIDERATIONS

The coefficients of determination on which the GRASS and BEEF models in this study were based accounted for c. 50% of the data variability suggesting that the dependent variables may vary within a wide range of values. This was not unexpected given the scale at which the trial was conducted and where much unexplained variability would be expected (Clarke P., Department of Statistics and Biometry, University of Natal, P.O. Box X01, Scottsville 3209, South Africa). However, validation of the models using data collected during the 1993/94 season suggested that the simulation modelling approach outlined in this study may provide a useful means of assessing the financial risk of alternate stocking strategies and provide range managers and extension personnel with an important decision-support tool. Caution should, however, be exercised in applying the model beyond the scope of the data from which it was developed. Incorporation of data collected during the 1993/94 and subsequent seasons, which may span a wider range of experimental conditions, could improve the reliability of the GRASS and BEEF sub-models by increasing the amount of variability accounted for the models. This could improve the predictive ability of the LOWBEEF model. It should, however, be noted that despite incorporation of additional data into the models, the proportion of variability in the data set accounted for by the model could remain low due to natural variability in the system.

Interestingly, additional variability could be added to the model by allowing the coefficients, on which the GRASS and BEEF models are based, to vary randomly within the range of values outlined by the 95% confidence intervals (i.e. ± 1.96 * SE). This may imply that the optimum number of simulations (800 years in Chapters 12 and 13) to provide an indication of the risk associated with various strategies would have to be increased considerably to account for the increased variability.

Although this study examined the risk and outcome of various fixed and a climate-dependent strategy (e.g. Chapter 12, 13), a range of strategies may be developed in relation to individual objectives and attitudes to risk. These may be accommodated within an integrated model (Hatch & Goosen in prep.), where the influence of user-defined stocking strategies and range dynamics models on profitability may be examined for historical and stochastic rainfall sequences. An important future consideration would be the integration of varying livestock purchase and sale prices into the model. This would allow the user to assess the influence of the market for beef on stocking strategies and risk. In addition, livestock producers may elect to market stock at the end of summer and avoid the cost of winter supplementation. The model developed in this study considered production at a one year time-step and could not therefore accommodate intra-seasonal fluctuations in stocking rate. Modification of the current model to consider changes in stocking rate within seasons could allow producers to assess the financial viability of various marketing approaches in relation to meat price fluctuations. For example, would it pay to stock heavily during the summer and market stock before winter to avoid winter supplementation costs (i.e. forage deficits)? Importantly, the model would need to incorporate the ecological consequences of these approaches.

Determination of ecological risk, which is fundamental to the development of sustainable stocking strategies, may not be simple given that the consequences of various strategies may only be reflected after a number of seasons (e.g. woody plant encroachment) or may be difficult to evaluate (e.g. soil erosion). Although monitoring the influence of stocking treatment on tree composition and structure will be continued at Llanwarne and Dordrecht, it is possible that the pattern of temporal change may exceed the lifespan of the trial programme. Integration of conceptual models of system change which incorporate environmental feedback into the system, such as the simple dynamic range model applied in this study, may consequently provide a useful basis from which to address ecological risk. Incorporation of the influence of stocking strategy on the risk of soil loss and bush encroachment in relation to rainfall variability would be important considerations.

With the current trend towards multi-species systems (Chapter 2), Integration of mixed species systems comprising beef with game in various combinations would be a useful addition to the current model and may allow users to assess optimum combinations of enterprises. In addition, consideration of the influence of factors such as soil heterogeneity and tree density on herbage production may provide the model with a spatial component.

While the current model addresses production variability and risk for commercial livestock production, range-use in the semi-arid savanna is based on a variety of tenure arrangements and user objectives. Land tenure may vary from private ownership through to open-access communal tenure. These give rise to a range in management objectives and hence attitudes to risk. For example, subsistence conditions may emphasise risk-avoidance rather than profit maximisation strategies while criteria for valuing output may be assessed in terms of draught power, milk, meat and utility-value rather than financial measures as applied in the current model. Expansion of the current modelling philosophy to include non-commercial objectives is an important future consideration.

While there is scope for improvement and expansion of the LOWBEEF model, development of additional models, based on the current framework, for the humid rangelands of Natal using data collected at sites in the Southern Tall Grassveld and Natal Sour Sandveld (Acocks 1988) would be an important future consideration. While increased stocking in the semi-arid savanna may increase the risk of forage deficits and provide a means of assessing the risk of alternate stocking strategies, a similar model for humid rangeland would examine the effect of stocking strategy on the cost of urea-supplementation which comprises a major component of variable costs for livestock producers (Hatch 1992). Importantly, integration of economic parameters into livestock production models currently being developed, using the paradigm outlined in this study, could make a major contribution to the development of sustainable livestock production systems in southern Africa.

CONCLUSIONS

Although simulation modelling and decision-support models may provide management tools to assist in the decision-making process, livestock production in a highly variable environment such as the Lowveld will remain a challenge. There is the need for a constant assessment of the consequences of variables, such as rainfall and product prices, which are beyond the control of the manager and others, such as stocking strategy, which are directly controlled by the manager. The development of a pro-active management strategy based on flexible stocking in relation to environmental and market variability remains the only way in which long-term ecological risk will be minimised and financial viability assured. The art of range management, based on intuition and understanding rather than hard science alone, is likely to remain fundamental to the discipline of range management.

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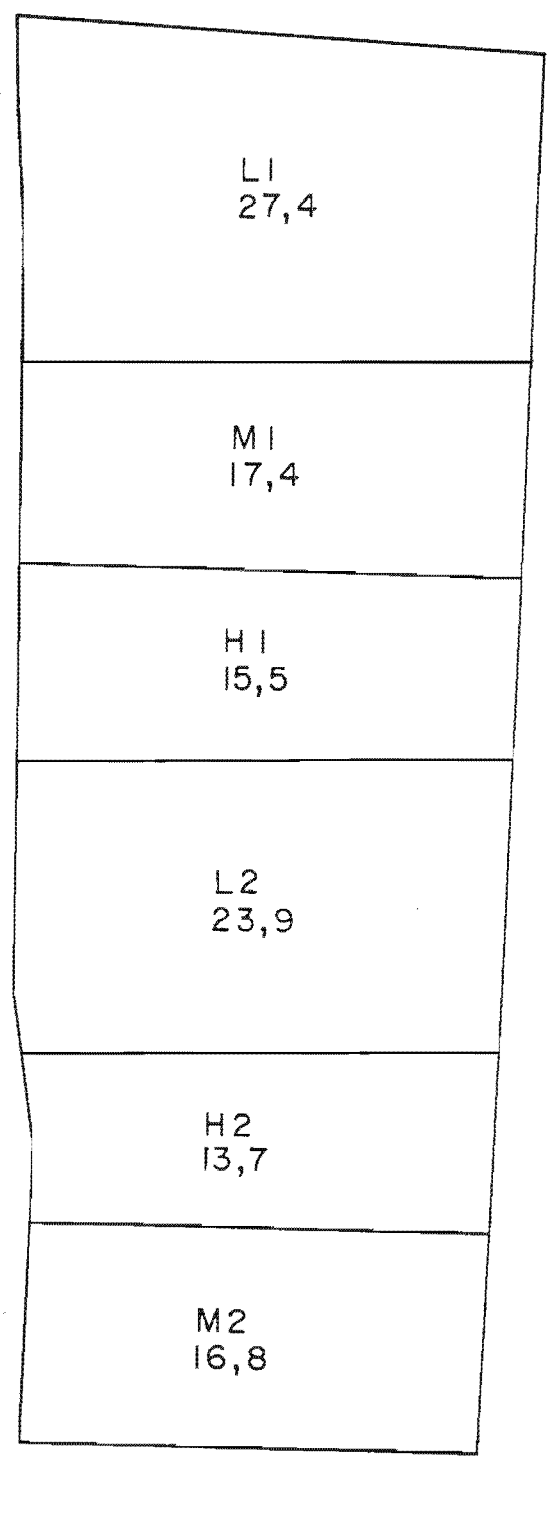
APPENDICES

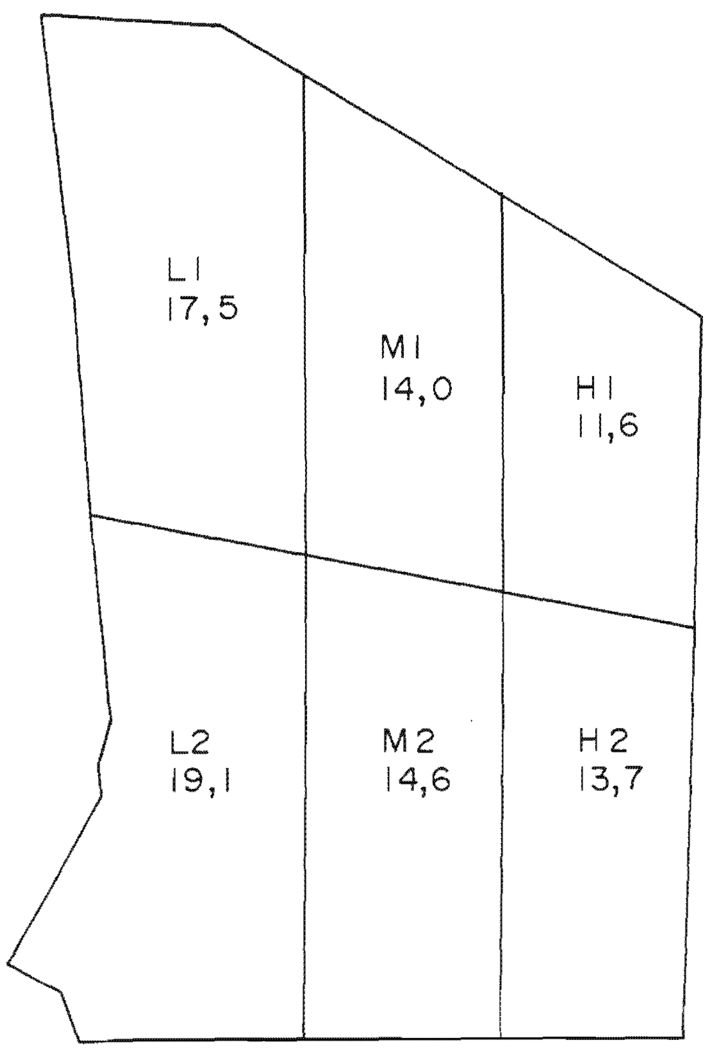
Appendix 1a Experimental site layout at site 1: Llanwarne

scale 1:10 000



Appendix 1b Experimental site layout at site 2: Dordrecht

scale 1:10 000



Appendix 2a **Lowveld grass species acronym key and subjective forage scores**

Acronym	Species	Forage score
ACO	*Aristida congesta subsp. congesta*	1
BIN	*Bothriochloa insculpta*	2
CEX	*Cymbopogon excavatus*	3
CCI	*Cenchrus ciliaris*	4
CVI	*Chloris virgata*	1
DER	*Digitaria eriantha*	5
DAR	*Digitaria argyrograpta*	1
ECU	*Eragrostis curvula*	1
ESU	*Eragrostis superba*	3
ESP	*Eragrostis species*	1
EPA	*Eustachys paspaloides*	1
FAF	*Fingerhuthia sesleriiformis*	1
HCO	*Heteropogon contortus*	3
PCO	*Panicum coloratum*	7
PSP	*Pennisetum species*	1
PMA	*Panicum maximum*	10
MRE	*Melinis repens*	1
SFI	*Sporobolus fimbriatus*	2
SNI	*Sporobolus nitens*	2
SIO	*Sporobolus iocladus*	2
TTR	*Themeda triandra*	7
TRA	*Tragus racemosa*	1
UMO	*Urochloa mosambicensis*	3
UPA	*Urochloa panicoides*	1
FOR	Forb species	
SED	Sedge species	
UNA	Unallocated bare ground	

Appendix 2b Proportional range composition (percentage) at Llanwarne and Dordrecht during the 1986, 1988, 1990, 1993 and 1994 surveys where L, M and H denotes 'light', 'intermediate' and 'heavy' stocking treatments respectively

1986	Llanwarne						Dordrecht					
Species	L1	M1	H1	L2	M2	H2	L1	M1	H1	L2	M2	H2
ACO	0.3	0.3	0.0	0.3	1.0	0.3	0.3	0.0	1.7	0.0	0.0	0.0
BIN	0.3	0.3	2.7	4.0	0.3	6.7	1.0	0.3	0.3	0.7	0.3	1.0
CEX	1.7	2.3	0.3	0.3	0.7	0.0	0.0	0.3	0.0	0.0	0.0	0.0
CCI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
CVI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
DER	5.0	2.7	0.0	0.0	0.0	0.3	0.0	0.0	1.3	0.3	0.3	2.7
DAR	24.7	25.3	23.0	17.7	15.3	22.3	24.0	7.3	10.3	12.0	2.0	2.7
ECU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ESU	1.0	1.7	1.7	0.3	0.7	1.3	0.0	0.0	2.0	6.7	15.3	9.7
ESP	1.0	0.0	0.3	1.0	0.0	0.7	4.7	1.0	4.3	0.3	0.3	0.7
EPA	2.3	1.0	1.0	1.7	2.0	3.3	3.3	0.0	0.3	1.0	1.7	1.3
FAF	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCO	1.0	0.7	1.0	0.3	0.3	0.3	0.0	0.0	0.3	0.0	0.0	0.0
PCO	5.0	5.7	15.3	19.7	22.0	12.7	4.7	20.7	5.3	13.0	15.0	16.0
PSP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PMA	41.3	25.0	27.3	16.7	16.0	25.0	30.0	7.7	6.3	18.7	11.0	22.7
MRE	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
SFI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SNI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.7	35.0	0.0	0.0	5.7
SIO	1.0	8.3	6.0	9.3	6.0	10.3	11.0	6.0	4.0	22.7	15.3	0.7
TTR	6.0	5.0	6.7	5.3	4.7	2.0	0.3	0.7	3.3	1.7	11.0	8.3
TRA	0.7	1.0	1.0	0.0	2.0	2.3	2.3	1.0	2.3	1.7	0.0	2.3
UMO	1.3	17.7	8.3	12.0	5.3	3.7	12.3	14.3	9.0	7.3	5.0	14.7
UPA	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FOR	7.3	3.0	5.0	10.0	17.7	8.7	5.7	25.7	10.0	13.0	20.7	11.7
SED	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
UNA	0.0	0.0	0.0	1.3	6.0	0.0	0.3	5.0	3.7	1.0	2.0	0.0

1988	Llanwarne						Dordrecht					
Species	L1	M1	H1	L2	M2	H2	L1	M1	H1	L2	M2	H2
ACO	0.3	0.3	1.0	1.3	1.7	0.3	0.7	0.0	2.0	0.0	2.0	0.7
BIN	0.7	2.3	2.3	8.3	2.3	2.3	0.3	1.3	0.0	0.3	0.0	0.7
CEX	1.0	2.3	0.0	0.0	0.3	1.0	0.0	0.0	0.3	0.0	0.0	0.0
CCI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CVI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
DER	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.3	5.3
DAR	22.0	15.3	25.0	13.0	17.7	34.0	29.0	13.3	18.7	18.7	7.3	3.0
ECU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	1.7
ESU	2.7	2.0	3.3	6.0	1.3	1.7	2.0	0.0	0.7	3.3	8.3	13.0
ESP	1.0	0.0	1.3	0.7	0.0	0.3	4.0	1.0	3.7	1.0	0.3	8.7
EPA	2.3	0.7	0.3	3.7	0.7	4.7	4.0	0.3	7.7	0.0	0.7	1.3
FAF	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCO	1.0	0.3	0.7	1.3	0.0	0.0	0.3	0.0	0.7	0.0	0.3	0.3
PCO	23.3	22.0	14.3	15.0	18.3	12.7	6.7	13.7	9.0	15.0	12.0	18.7
PMA	27.3	21.7	15.7	13.3	24.0	22.7	20.0	4.3	6.0	13.3	19.3	9.3
PSP	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MRE	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.3	0.3
SFI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SNI	0.7	2.0	2.7	0.0	0.3	0.0	1.0	7.7	4.3	2.0	6.3	5.3
SIO	1.7	7.3	13.3	6.0	15.7	5.0	17.0	25.7	22.3	23.7	8.3	9.3
TTR	3.7	9.0	4.3	11.0	2.7	3.7	0.0	0.3	0.0	0.7	0.7	1.3
TRA	0.0	0.0	0.0	0.0	1.7	0.0	1.0	0.7	1.0	4.7	0.0	1.7
UMO	3.3	9.7	10.3	3.7	6.0	4.7	9.0	15.3	12.3	8.0	11.0	11.7
UPA	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FOR	7.3	5.0	5.0	12.7	7.0	7.0	4.0	9.3	5.3	8.0	15.7	7.7
SED	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
UNA	0.3	0.0	0.3	4.0	0.3	0.0	1.0	7.0	1.0	1.3	7.0	0.0

1990	Llanwarne						Dordrecht					
Species	L1	M1	H1	L2	M2	H2	L1	M1	H1	L2	M2	H2
ACO	1.3	0.3	3.0	5.3	2.0	1.0	0.3	0.3	1.3	0.7	2.3	2.7
BIN	1.3	1.7	2.0	3.0	4.0	11.3	1.0	1.3	6.7	1.7	2.3	1.7
CEX	4.3	3.7	0.7	0.0	1.3	0.3	1.0	0.0	1.3	0.3	0.0	0.0
CCI	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.3	0.3	0.0
CVI	2.0	1.0	2.0	2.0	0.0	6.0	0.0	0.0	0.0	0.0	0.0	0.0
DER	4.0	0.7	2.0	0.7	2.0	0.3	6.3	0.0	0.0	3.3	0.0	0.7
DAR	13.0	17.3	13.0	6.0	15.3	10.7	3.0	17.7	12.3	14.0	5.0	1.3
ECU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.3	0.0
ESU	1.3	0.7	0.7	0.3	1.7	0.0	4.3	1.7	5.0	1.7	1.7	6.7
ESP	1.3	1.7	3.3	1.3	0.0	2.0	0.0	1.7	7.0	1.0	15.3	17.3
EPA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FAF	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCO	0.3	0.0	0.0	0.7	0.3	1.6	0.7	0.0	1.7	2.0	0.7	0.0
PCO	3.7	2.7	2.0	15.0	4.0	9.0	0.7	5.3	1.3	6.3	12.7	7.0
PMA	37.0	41.0	33.7	32.0	31.7	22.7	49.7	33.3	23.0	34.3	25.0	40.7
PSP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MRE	0.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.3
SFI	0.0	0.0	1.3	0.7	0.3	0.0	4.0	0.3	0.7	2.0	2.7	0.3
SNI	0.0	0.0	0.3	0.0	0.0	1.3	0.0	2.0	2.7	2.0	2.0	0.3
SIO	1.7	8.0	7.3	5.0	5.3	5.0	0.7	5.3	6.3	2.7	4.7	0.0
TTR	15.0	13.3	11.7	5.3	10.3	4.3	13.3	1.3	11.3	5.3	0.7	2.7
TRA	0.0	0.0	0.0	1.0	0.3	0.3	0.0	0.0	0.0	0.0	1.3	0.0
UMO	8.7	5.3	8.3	11.3	10.7	12.3	5.3	11.3	13.0	12.7	10.0	9.3
UPA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FOR	3.3	2.0	5.3	8.3	7.7	6.3	8.7	8.7	3.0	8.3	7.7	6.3
SED	1.0	0.3	0.3	0.7	0.3	0.7	0.0	0.7	1.0	0.0	0.0	0.3

1993	Llanwarne						Dordrecht					
Species	L1	M1	H1	L2	M2	H2	L1	M1	H1	L2	M2	H2
ACO	2.4	1.7	3.0	3.0	4.0	8.0	1.7	2.3	2.0	6.3	6.0	4.3
BIN	2.7	2.4	3.7	3.0	1.7	6.4	2.0	2.7	0.0	0.0	3.0	1.3
CEX	1.7	2.7	0.7	0.4	0.7	1.0	0.0	0.0	0.0	0.0	0.0	0.0
CCI	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.3
CVI	0.0	0.7	1.4	0.0	0.7	0.7	0.0	0.0	0.0	0.0	0.7	0.3
DER	0.4	0.0	1.4	0.0	0.7	0.4	0.7	0.7	0.0	0.0	0.7	0.3
DAR	5.0	6.0	4.4	6.0	3.4	4.0	9.7	8.3	10.0	2.0	8.0	4.3
ECU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ESU	0.7	1.7	1.4	0.7	4.0	0.7	1.3	3.3	0.7	0.3	2.3	0.7
ESP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.3	0.7
EPA	0.0	0.0	0.0	0.4	0.4	0.0	1.7	0.3	0.3	0.0	0.3	0.0
FAF	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	1.0	0.3	0.3
HCO	1.4	2.0	0.4	1.7	1.4	1.0	4.0	0.3	0.0	0.3	2.0	0.7
PCO	2.7	1.7	4.0	23.4	3.7	7.3	4.7	7.0	14.7	19.3	0.7	4.7
PMA	41.4	46.4	49.4	35.4	44.3	29.0	41.7	46.7	30.7	19.0	17.7	45.0
PSP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MRE	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SFI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SNI	2.3	5.0	4.0	2.4	7.7	3.7	3.0	1.3	2.7	5.0	5.7	2.0
SIO	1.7	1.4	2.7	2.7	1.7	3.4	9.0	3.7	6.7	13.3	17.7	2.3
TTR	1.3	2.3	3.0	5.4	2.0	0.7	5.0	6.0	1.3	1.0	3.7	1.0
TRA	4.3	2.0	1.7	1.7	9.4	8.7	1.3	0.7	7.0	10.7	10.7	5.6
UMO	27.7	23.4	18.7	13.7	16.0	22.8	13.7	16.3	23.0	17.7	20.3	26.0
UPA	3.3	0.0	0.4	0.4	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0
FOR	1.0	1.0	0.4	0.4	0.0	0.0	0.3	0.3	0.7	0.3	0.0	0.0
SED	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
UNA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

1994	Llanwarne						Dordrecht					
Species	L1	M1	H1	L2	M2	H2	L1	M1	H1	L2	M2	H2
ACO	3.3	1.6	2.3	3.7	5.3	7.3	1.6	4.0	0.3	0.3	1.3	3.8
BIN	1.0	0.7	1.0	4.6	2.3	2.3	2.6	0.6	0.0	2.6	1.0	2.1
CEX	1.0	3.0	0.7	0.3	1.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
CCI	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	1.6	1.6	0.0
CVI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
DER	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7
DAR	9.6	8.0	14.0	7.0	2.0	7.6	14.6	5.0	14.6	10.3	1.0	0.0
ECU	0.0	0.0	0.0	0.0	0.0	0.0	1.3	0.0	1.0	0.0	0.0	0.0
ESU	1.0	1.0	1.3	3.0	0.7	3.3	4.3	2.0	0.3	6.0	0.3	6.9
ESP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	3.0	16.0	11.1
EPA	0.7	0.3	0.3	1.3	1.0	0.3	2.6	0.0	0.0	0.6	0.6	0.7
FAF	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0
HCO	0.3	0.3	0.3	1.0	0.7	0.0	0.6	0.0	0.0	0.3	0.0	0.7
PCO	0.6	1.7	0.6	3.6	5.3	3.3	0.3	35.0	0.3	16.6	12.3	11.4
PMA	52.6	48.3	49.3	64.3	53.3	43.3	39.0	35.6	20.6	28.3	15.6	21.1
PSP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MRE	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SFI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SNI	0.0	0.3	0.3	0.7	0.0	1.3	0.6	1.6	6.0	1.0	1.3	3.5
SIO	6.3	11.7	5.3	0.0	2.3	2.7	4.0	3.3	9.3	3.6	9.3	0.3
TTR	2.3	2.3	1.0	2.3	0.0	1.3	5.0	0.0	0.0	4.6	0.6	0.7
TRA	0.0	0.3	0.3	0.0	0.0	1.3	0.3	0.6	1.3	0.3	2.6	0.0
UMO	20.7	18.7	22.3	6.0	23.0	24.0	21.3	6.6	38.0	12.3	18.0	19.4
UPA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
FOR	1.0	1.0	1.0	1.7	3.6	1.0	0.6	4.0	8.0	5.3	16.0	15.9
SED	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
UNA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix 3a Soil map of Llanwarne

LEGEND

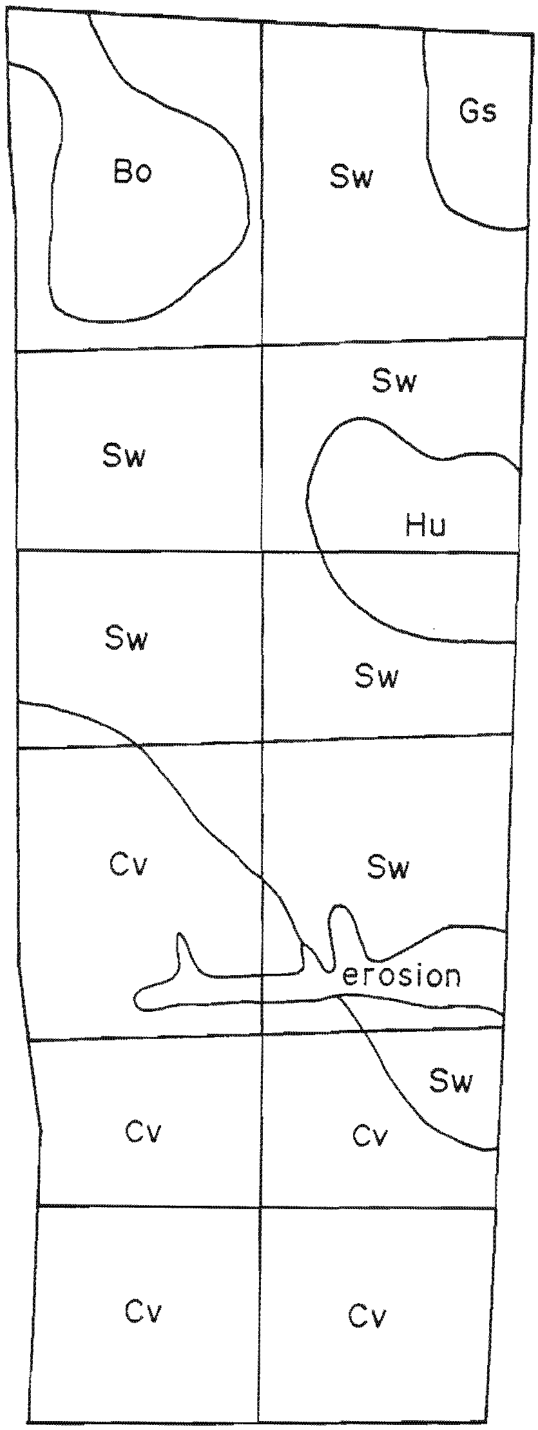
Sw - Swartlands

Bo - Bonheim

Cv - Clovelly

Hu - Hutton

Scale 1 : 10 000



Appendix 3b Soil map of Dordrecht