



## Conservation in the optimal use of rangelands

Charles Perrings<sup>a,\*</sup>, Brian Walker<sup>b</sup>

<sup>a</sup>*Environment Department, University of York, YO1 5DD, York, UK*

<sup>b</sup>*Wildlife and Ecology Division, CSIRO, Canberra, Australia*

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### Abstract

In previous papers we have considered the optimal mix of biodiversity in semi-arid rangelands, focusing on the steady state. This paper addresses the question of conservation in the optimal use of rangelands, where conservation is understood to mean maintenance of the system in a ‘natural’ state. We consider a rangeland that may exist in one of two states. In the ‘natural’ state, its dynamics are regulated by fire. In the ‘managed’ state, its dynamics are regulated by grazing pressure by livestock. We show that the optimal use of rangelands may include its maintenance in both states at different points in time, depending on initial conditions and the set of relative prices.

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### 1. Introduction

It is impossible not to be struck by the sharp divisions between those who argue for the conservation of multi-species ecosystems and those who argue for their sustainable use. Frequently, conservation is assumed to mean preservation and is assumed to be incompatible with any use. Indeed, the ‘conservation value’ of ecosystems is often discussed as if it is completely independent of the value of such systems in any other use. But if conservation is an alternative to exploitation, then conservation is only rational if the conservation value of the system is at least as great

as its value in any other use. Increasingly, there is a perception that biodiversity conservation at a national level is not well served by a strategy that seeks 100% protection of the remaining wildlife refugia, but offers no protection to the rest. It is better served by a strategy that offers the appropriate level of protection to 100% of area over which a nation has sovereignty (Perrings and Gadgil, 2002). This implies that conservation should be an element of use, and that it should be possible to identify the conservation element in any optimal policy.

At one level this is easy to do. ‘Conservation’ typically focuses on the protection of stocks, while ‘use’ focuses on the regulation of flows. Any ecosystem management problem can be cast in state–space terms as an optimal control problem. In a wildlife management problem, for example, the wildlife stocks are the state variables of the problem, and the offtake

\* Corresponding author. Tel.: +44-1904-432-997; fax: +44-1904-432-998.

E-mail address: [cap8@york.ac.uk](mailto:cap8@york.ac.uk) (C. Perrings).

from each stock is a control variable. Any optimal offtake policy automatically implies an optimal stock conservation policy. The optimal level of stock conservation then depends on the value of the resource in situ relative to its value in the market place (corrected for externalities). An optimal stock conservation policy may mean that stocks will be kept at levels below the steady state equilibrium (if it exists) of the unexploited system. But so long as the value of the resource in situ is greater than its value once extracted, stocks will be conserved at positive levels.

It is also possible to identify the conservation phase in an optimal control policy in a very straightforward way. Where the optimal control problem has a certain structure (such that the Hamiltonian of the problem is linear in the control), then the optimal policy involves the most rapid approach to the optimal stock level. If initial wildlife stocks are below the optimal level, the optimal policy will include a conservation phase (no offtake) until stocks have built up to the optimal level.

This paper approaches the problem of conservation in ecosystem use in exactly this way. But it considers the case where the optimal policy reflects the dynamics of species interactions, and where the optimal control problem may not have the sort of structure that makes identification of an initial conservation phase straightforward. It is motivated by the case of semi-arid rangelands, and uses a model of the optimal use of rangelands (Perrings and Walker, 1997) to illustrate the problem. This is used to explore the implications of the hierarchical structure of ecosystems for the dynamics of conservation.

The starting point here is provided by Holling's observations about the interaction between the spatial scale of ecological systems and their dynamics. Adapting Simon's (1974) analysis of 'hierarchical' systems, Holling's early work on boreal forests had shown how the dynamics of the system reflect interactions between 'transformational cycles' range from the leaf over a period of days to the forest over a period of years. It established the importance of variation in the speed of the dynamics of systems at different spatial and temporal scales. Hierarchical systems are nested systems existing at different spatial and temporal scales, each with its own dynamics. Small fast-moving systems are embedded in large slow-moving systems. Generally, the small fast-mov-

ing systems are constrained by the large slow-moving systems, but there also occur junctures at which smaller systems are able to disrupt larger systems (Ludwig et al., 1978; Holling, 1992). In ecology, this prompted development of analyses at the landscape scale that focused on interactions between biotic and abiotic processes at different scales (Allen and Starr, 1982; O'Neill et al., 1986; Levin, 1992).

This work has influenced research on the economics of ecosystem management by changing our perception of the interdependence of spatial and temporal structure. Levin et al. (1998), Holling et al. (2002) and Holling and Gunderson (2002) have argued that the insights into the behaviour of hierarchical ecological systems can and should be applied to the economics of renewable resources/ecosystems. Ecological-economic systems are hierarchical, in that they consist of a structure of subsystems, each operating at distinct spatial and temporal scales both in interaction with each other, and with the systems of the natural environment. Holling et al. (2001) refer to this as a 'panarchy'. They argue that it is possible to evaluate the evolution of such systems within the framework of interacting 'adaptive cycles'. Cycles are characterised by three things: the 'inherent potential' or 'wealth' of the system; its 'connectedness' which determines its flexibility or rigidity; and its resilience or adaptive capacity.

The importance of spatial structure is obvious. A landscape may contain a number of populations whose interactions determine the dynamics of the general system, and its potential for its exploitation. Those interactions, and hence the dynamics of the system, are physically structured by topography, hydrology, vegetation cover and so on. In marine systems, for example, Brown and Roughgarden (1997) analysed a model barnacle system to show the implications of physical structure for spatial dynamics, and hence for the optimal exploitation of the resource. In ecological-economic systems human activities structure the environment within which other species exist, and this constrains the dynamics of those species. Sanchirico and Wilen (1999) consider the optimal exploitation of a multi-location fishery in which the level of fishing effort in each 'patch' affects the dynamics of fish stocks in that patch.

The temporal structure of the system is also increasingly recognised to be important. Implicitly,

models of renewable natural resource extraction assume that the dynamics of the social system ‘contain’ the dynamics of the exploited population. That is, the decision-maker is assumed to operate at a temporal scale (over a horizon) that extends beyond the renewal period of the exploited population. If this is not the case, the resource is assumed to be exhaustible, and its dynamics of little consequence. In fact, neither position is consistent with the theory of hierarchical systems. For one thing, the dynamics of the large slow-moving systems that are taken to be exogenous to the economic problem may be sensitive to changes in the small fast-moving systems. An illustration from the folklore of complex systems is the butterfly effect. It implies that localised short-term decisions affecting the dynamics of small fast-moving systems may have consequences for the time behaviour of large slow-moving systems.

An obvious example is the effect of the fast dynamics of many pests and pathogens on human populations. Epidemics involve the explosive growth of infectious agents within a host population, often affecting the dynamics of that population. HIV in Africa is a current example, but there are numerous other examples of human societies that have been transformed by such epidemics. Typically, epidemics are treated as stochastic events, but an understanding of the temporal interactions between pathogen and host might make them at least partially predictable. Indeed, the development of what might be described as economic epidemiology—an offspring of ecological economics—is stimulated by exactly this insight (Ehrlich, 2000; Holling et al., 2002; Delfino and Simmons, 2000).

A practical application of an understanding of the interconnectedness of systems over space and time concerns the level at which to manage environmental problems. Failure to understand how human behaviour affects and is affected by the dynamics of other species means failure to take this into account in decision-making. This turns out to be closely related to the policy question: what is the right level at which to address a problem? The European principle of subsidiarity holds that the right level at which to manage a problem is the lowest level that contains all of the relevant effects. To establish the right level we need to understand the spatial and temporal reach of the problem. For example, the control of a potentially invasive pathogen in one location may be a local problem if

there are no trade/travel links between that location and the rest of the world. It becomes a global problem if there are such links.

To identify the implications of cross-scale species interactions for biodiversity conservation in rangelands the paper first considers the links between ecosystem structure and dynamics—between topology and persistence of states on nature. It then discusses the characteristics of semi-arid rangelands and constructs a model with which to explore these linkages. Finally, it offers a discussion of the implications for conservation.

## 2. Resilience and the dynamics of conservation

Ecology works with a rather different set of stability measures than economics. These include measures of ‘resistance’, ‘persistence’ and ‘resilience’ as well as stability. Resistance is a measure of the capacity to resist change. It is therefore a measure of local stability. Persistence is a measure of the capacity of the system in some state to endure. It is related to the global stability of the equilibrium corresponding to that state. Resilience is interpreted in two different ways, one corresponding to the local stability of an equilibrium, the other corresponding to its global stability. We wish to focus on the latter.

The Holling (1973) measure of resilience is a measure of the size of a disturbance needed to dislodge a system from its stability domain. This makes it a measure of the size of the stability domain corresponding to some attractor. Resilience is measured by the size of the perturbation that will cause the system to flip into some other stability domain. More generally, it is the conditional probability that it will flip into another stability domain given (a) its current state and (b) the disturbance regime.

If a system in some state is not at equilibrium, and is subject to disturbances, its sustainability depends on whether it can withstand those disturbances. In general, if an ecological economic system can exist in multiple stable states, and if it may at any point in time be far from equilibrium, then we should be as interested in its behaviour in the neighbourhood of the unstable equilibria (the unstable manifolds between

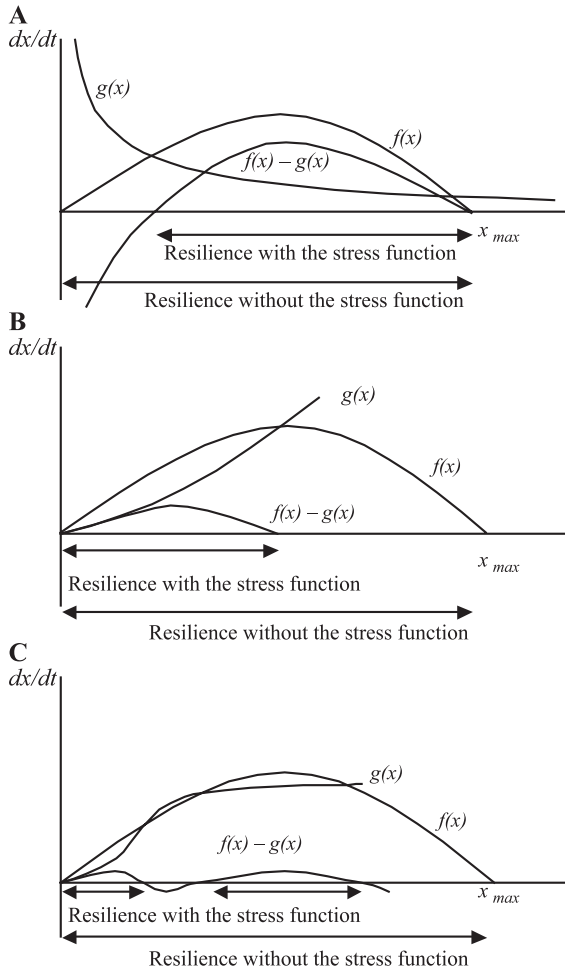


Fig. 1. Resilience with and without stress.

states) as we are in the neighbourhood of the stable equilibria.<sup>1</sup>

A second and related property of dynamical ecological systems is that their susceptibility to shocks depends on their position in the renewal cycle. Holling describes ecological systems as passing through four phases. A first phase involves the rapid accumulation of both biomass and structure (complexity). A second phase involves high and relatively stable

<sup>1</sup> In agroecosystems generally the impact of price shifts on crop choices, pesticide and fertilizer regimes all have the potential to induce a change of state, and to involve hysteresis. Hysteresis implies that the choice or control variables that induced the flip in the first place need to be returned beyond those levels if they are to induce a return flip.

biomass and structure, and corresponds to the climax state in traditional ecology. A third phase involves the rapid dissolution of structure and loss of biomass, and a final phase involves the reconfiguration or rebirth of the system. It is particularly vulnerable to shocks in the second phase. Indeed the dissolution and reconstruction phases are frequently triggered by relatively minor shocks.

If we think about resilience in the sense of Holling as a measure of the size of the stability domain, and use a compensatory growth function to illustrate the implications of stresses on the system, it is easy to see how it affects standard analysis of the extraction of renewable resources. Consider a simple renewable resource problem in which growth of some species may be described by a compensatory (say logistic) function. Suppose that  $f(x)$  defines the stress-free growth of the stock  $x$ , and that  $g(x)$  is a stress function describing the impact of some economic activity on the growth of the species. The net growth function of the species is then  $f(x) - g(x)$ . Panel A in Fig. 1 illustrates how such a stress function might reduce the resilience of the system more at the growth phase than at the climax phase. Panel B illustrates the opposite case. Panel C indicates the case where the stability domain of the affected ecosystem is fragmented by the source of stress.

Now suppose that the source of stress is stochastic process with mean,  $g^M(x)$ , and that the upper and lower realisations of  $f(x) - g(x)$  can be described by the curves  $f(x) - g^U(x)$  and  $f(x) - g^L(x)$ . While the states corresponding to the mean and upper realisations of  $g(x)$  are ‘absorbing’—i.e., belonging to two distinct basins of attraction—the states corresponding the lower realisation of  $g(x)$  are merely persistent. That is, they do not belong to distinct basins of attraction. For some realisations of  $g(x)$ , there is a route between the two

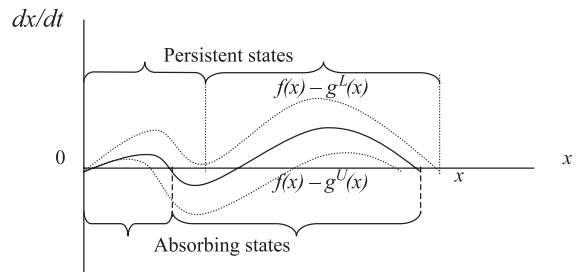


Fig. 2. Persistent states of nature in a stochastic system.

states. If the system were originally in the lower steady state, it might well stay there for some considerable time, but eventually it would switch from the lower to the upper state (Fig. 2).

### 3. The rangeland model

The idealised system modelled here is a semi-arid savanna typical of Central and Southern Africa. A stylised description of this system follows. It has a mean annual rainfall of around 450 mm, but rainfall is highly variable. The coefficient of variation is around 40%. Soils are variable, ranging from sands to heavy clays, as is vegetation. Vegetation is dominated by *Colophospermum mopane* in the low veld, but *Acacia* spp. tend to become dominant on heavier soils. Grasses comprise both perennial and annual species. The relative importance of perennials increases with rainfall and decreases with grazing pressure, but in general perennials dominate grass biomass. Grass production is generally very sensitive to rainfall, but perennials are much less variable than annuals (Taylor and Walker, 1978; Kelly and Walker, 1976).

The balance between grass and woody vegetation depends on soils, the rainfall regime and—most importantly for this paper—the fire regime (Scholes and Walker, 1993). Since woody vegetation dominates grasses in competition for light, nutrients and water, sandy soils are largely associated with woodland or shrubland, and grasses are sparse. By contrast, grasses are more competitive on heavier soils because a higher proportion of rainfall is retained in the upper layers of the soil where grass has most of its roots. However, it is common for such soil types to support multiple vegetation ‘states’ (Westoby et al., 1989). This depends on the role of fire. Fire keeps the vegetation in a relatively open state. If fire is excluded, cohorts of woody plants become established during good rainy seasons and develop into thickets. The thicket then excludes grass from developing even if grazing is excluded, and hence may dominate until re-structuring of the woody vegetation through wood–wood competition and the consequent death of trees allows grass to come back into the system.

On more finely textured soils, grass is seldom excluded. In dry years, woody vegetation dies back to the amount permitted by the available soil water. In wet years, woody vegetation regenerates but not enough to

make use of plant-available soil moisture, enabling grass to take up the unused water. Once established, grass competes with woody vegetation by reducing the amount of water available. Indeed, the greater the variability of rainfall, and the lower the mean annual rainfall, the less woody vegetation can be supported.

Following Perrings and Walker (1995, 1997), this stylised description is reflected in a simplified model that groups grasses, woody vegetation and wild herbivores in three state variables. We focus on the implications of a control sequence associated with a given set of market and environmental conditions when the system is not initially at equilibrium. Aside from environmental conditions, production of grass and wood depends on competition between plants, grazing pressure by wildlife, and the effects of fire. Grazing pressure is assumed to affect grass and woody biomass in different ways. Specifically, herbivores are assumed to consume grass more than woody biomass. Fire is not used strategically, but occurs if the fuel load is sufficient. The only direct control is offtake from wild herbivores, implying that the simplified model captures elements of both game ranches and hunting concessions. We are concerned with the implications of a control sequence for the conservation of the biodiversity in the system.

Consider, first, the optimal conservation effort in the simplest case. The social decision-maker is assumed to choose a level of offtake,  $k(t)$ , to maximise the net benefits from the use of the ecosystem, where this is the difference between the revenues from harvest,  $p(t)k(t)$ , and the costs of maintaining the system,  $c(x(t), y(t), z(t))$ :

$$\max_{u(t)} \int_{t=0}^{\infty} e^{-\delta t} [p(t)k(t) - c(x(t), y(t), z(t))] dt$$

subject to the equations of motion for the state variables wildlife,  $x(t)$ , grass,  $y(t)$ , and woody biomass,  $z(t)$ .

$$\dot{x} = f(x(t), y(t), z(t)) - k(t)$$

$$\dot{y} = g(x(t), y(t), z(t))$$

$$\dot{z} = h(x(t), y(t), z(t))$$

$$x(0) = x_0, y(0) = y_0, z(0) = z_0$$

$$k(t) \geq 0.$$

The current value Hamiltonian for this problem is:

$$\begin{aligned}\tilde{H} = & [p(t)k(t) - c(x(t), y(t), z(t))] \\ & + \lambda[f(x(t), y(t), z(t)) - k(t)] \\ & + \mu[g(x(t), y(t), z(t))] + \zeta[h(x(t), y(t), z(t))]\end{aligned}$$

The maximum principle requires that  $p(t) = \lambda(t)$ . Given that the Hamiltonian is linear in the control, the approach to the optimum is ‘most rapid’, implying that if the initial level of the state variables is less than the optimum, then there will optimally be a conservation phase during which  $k(t) = 0$ . That is, the optimal control obeys the law:

$$k(t) = \begin{cases} 0 & \text{if } p(t) < \lambda(t) \\ k^* & \text{if } p(t) = \lambda(t) \end{cases}$$

So long as the market price of the harvested resource is less than its social opportunity cost—its value to society—the stock of the resource should be allowed to build up naturally. The important element here is the rate of growth in the value of the natural asset relative to that of produced capital. In Holling’s terms, this is the relative ‘speed’ of the natural variables of the system. The Hotelling rule for renewable resources implies that conservation is efficient so long as the growth in value of a natural resource inclusive of its regeneration rate is greater than or equal to the growth in value of reproducible capital. It follows that if the price of a renewable resource is constant over time, and its in situ rate of growth (‘speed’) is greater than the rate of interest, the opportunity cost (‘speed’) of capital, then it will be optimal to conserve it in some measure.

There are environmental resources whose in situ value grows sufficiently fast for the optimal exploitation rate to be equal to zero. Some of the world’s remaining wildlife refugia—the so-called ‘hot-spots’—are in this category. However, the rate of growth in the value of many renewable resources is such that it is optimal to exploit those resources at a positive rate. In this case, as long as the rate of exploitation is within the carrying or assimilative capacity of the resource and so is consistent with the maintenance of the system in some well-defined state, it may be said to be conserved. If the optimal use of a resource causes a change in its state after some interval, the optimal control sequence can be divided into two

phases: a conservation phase and an exploitation phase. The first phase corresponds to the notion of conservation as preservation, the second to the notion of sustainable use.

To approach the implications of differences in the dynamics of the component resources in our simplified system, we need to be more specific about the functional forms in the model. To do this we first relax the assumption that time is continuous. The decision problem now takes the form:

$$\max_k \sum_{t=0}^T \rho^t (p_t k_t - c(x_t, y_t, z_t))$$

subject to:

$$x_{t+1} = x_t \left( 1 + \alpha \left( 1 - \frac{\psi x_t}{y_t} \right) \right) - k_t$$

$$\begin{aligned}y_{t+1} = & y_t + \beta y_t \left( 1 - c_{yy} \frac{y_t}{y_{\max}} - c_{zy} \frac{z_t}{z_{\max}} \right) \\ & - \sigma_y \phi (y_t - y_{\min}) - \psi x_t\end{aligned}$$

$$\begin{aligned}z_{t+1} = & z_t + \gamma z_t \left( 1 - c_{zz} \frac{z_t}{z_{\max}} - c_{yz} \frac{y_t}{y_{\max}} \right) \\ & - \sigma_z \theta (z_t, y_t - y_{\min})\end{aligned}$$

$x_0, y_0, z_0$  given.

$p_t$	=	the extracted value of wildlife
$u_t$	=	harvest of wildlife
$c(x_t, y_t, z_t)$	=	the cost of ecosystem maintenance
$\alpha$	=	wildlife growth rate
$\beta$	=	grass growth rate
$\gamma$	=	woody biomass growth rate
$\rho$	=	discount factor
$\psi$	=	wildlife consumption of grass
$c_{yy}$	=	competition coefficient: grass/grass
$c_{yz}$	=	competition coefficient: grass/wood
$c_{zz}$	=	competition coefficient: wood/wood
$c_{zy}$	=	competition coefficient: wood/grass
$y_{\max}$	=	maximum potential grass biomass
$z_{\max}$	=	maximum potential wood biomass
$y_{\min}$	=	the minimum fuel load required to sustain a fire
$\sigma_y$	=	proportion of grassy biomass removed by fire
$\sigma_z$	=	proportion of woody biomass removed by fire

The growth function for wildlife is a simple logistic function in which growth is limited by the availability of fodder. For simplicity, and without loss of generality, it is assumed that herbivores graze only. If the grazing requirements of the herd exceed the available fodder, growth is negative. The growth functions for grass and woody biomass respectively have three terms in common. The first is the stock of biomass at the beginning of the period. The second captures the effect of competition on growth during the period. The third captures the effect of fire during the period. Again, without loss of generality, it is assumed that fire occurs with probability one providing that the fuel load exceeds a critical threshold, and that if fire does occur it induces a constant rate of loss in both grass and woody plants. Finally, the equation of motion for grass includes a term,  $\psi x(t)$ , capturing the effect of consumption by herbivores.

The ecological parameters are drawn from the SEESAW rangeland production model (CSIRO). Specifically, the parameter values assumed in this paper are as follows:  $p_t=10$  for all  $t$ ;  $c_x=0.1$ ,  $\alpha=0.2$ ,  $\beta=0.67$ ,  $\gamma=0.15$ ,  $\psi=0.2$ ,  $c_{yy}=1$ ;  $c_{yz}=0.1$ ,  $c_{zz}=1$ ,  $c_{zy}=0.25$ ,  $y_{max}=200$ ,  $z_{max}=200$ ,  $y_{min}=120$ ,  $\sigma_y=0.4$ ,  $\sigma_z=0.3$ . Initial values for the state variables are  $x_0=10$ ,  $y_0=50$ ,  $z_0=50$ .

The particular problem we wish to consider is the effect of differences in the ‘speed’ of the components of the rangeland system on the state of the system as it converges on the optimal path. All three state variables summarise distinct communities in the system, and are characterised by different intrinsic rates of growth. It is assumed that  $\beta > \alpha > \gamma$ —i.e., that the rate of growth of grassy biomass is greater than the rate of growth of wild herbivore biomass which is greater than the rate of growth of woody biomass. Woody biomass is the slow variable in the system. But all three state variables are also interdependent. In the absence of herbivores, grasses dominate and the system is regulated by fire. In the presence of herbivores, woody plants dominate, although this depends on the level of grazing pressure. Importantly, fire is excluded from the system.

The model is optimised (numerically) by choice of a ‘steady state’ optimal level of harvest that is then applied in all periods. While this offers a slower convergence to the optimal path than an MRAP strategy, it is qualitatively similar in its dynamic

effects and helps to clarify the conservation element in the optimal policy. The time horizon is assumed to be 20 years. The initial values for the state variables reflect an assumption that the system is far from equilibrium, but an alternative set of assumptions are explored in the discussion.

Simulations of the optimal time path of the system over a range of discount rates is shown in Fig. 3. This offers a straightforward way to see the conservation

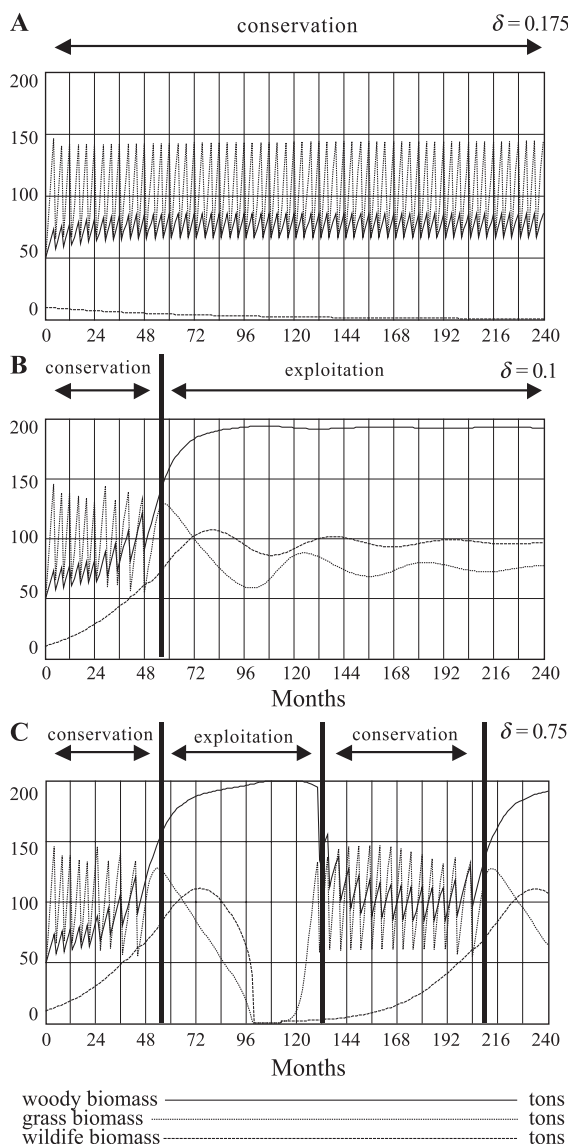


Fig. 3. Optimal balance between woody plants, grasses and wildlife.

implications of variations in the opportunity cost of rangeland use or, put another way, the ‘speed’ of natural rangeland assets relative to that of produced capital. Since private discount rates have been shown to be correlated with incomes (Holden et al., 1998), by the way, it also provides a straightforward way to see the conservation implications of relative poverty/affluence. This figure reports values for the three state variables: woody plants, grasses and wild herbivores for three rates of discount: 17.5% (Fig. 3A), 1% (Fig. 3B), and 7.5% (Fig. 3C). Costs are assumed to be increasing in the stock of herbivores.

In Panel A, the selected discount rate is close to the maximum natural rate of growth of wild herbivores. If the discount rate were equal to the maximum natural rate of growth of wild herbivores, it would be optimal to treat them as a non-renewable resource—and hence to remove them from the system in the first period. In this case the stock of wild herbivores is run down over time. Note that the high frequency of fire is a function of the structure of the model. It is assumed, for simplicity, that the probability of fire is the same in every period. The panel also shows the effect of herbivores on the balance between woody vegetation and grasses. In the absence of herbivores, woody vegetation is excluded, and the system converges on a state at which it has the characteristics of a fire-regulated grassland.

In Panel B, the discount rate is very low, and leads to a regime based on sustainable long-term use or exploitation. The initial phase—the conservation phase—involves a fire-regulated regime. During this phase herd sizes are built up to the point where grazing pressure begins to dominate fire as the regulating mechanism. The second phase—the exploitation phase—is one in which the system moves through damped oscillations towards a steady state at which woody plants are dominant, and grasses are controlled through grazing pressure. Fire is absent from the system.

Panel C considers an intermediate discount rate of 7.5%. The optimal use of the rangeland in this case is cyclical, with conservation and exploitation phases following each other in sequence. This may be thought of as equivalent to a fallow system, in which the availability of forage determines when rangeland use switches from regulation by fire to regulation by grazing pressure.

The length of time the system remains in one or other state depends on the relative ‘speed’ of the variables. For example, if the maximum rate of growth of grassy biomass increases substantially, the system will remain in the conservation phase for longer. This is because of the effect of the increased incidence of fire on woody biomass. Until herbivores increase in number by enough to replace fire as the regulating mechanism, the rangeland behaves as if it were a fire-regulated grassland. A secondary effect of the higher growth potential of grass is an increase in the speed at which the system in the exploitation phase converges on the steady state.

#### 4. Discussion

Many rangelands exist in two states: as a fire-regulated grassland and as a grazing-regulated savanna dominated by woody vegetation. An optimally managed rangeland can exist in both states sequentially. When it is in the first state, it is referred to as being in a conservation phase. When it is in the second state, it is referred to as being in an exploitation phase. This reflects two assumptions. The first is that the range in its natural state is closer to a fire-regulated grassland than to a grazing-regulated woody savanna. The second is the assumption that fire is a natural regulator whereas grazing pressure is a direct consequence of offtake. Of course, fire may be used as a management tool, but this is only feasible where there is a sufficient fuel load anyway. In this case, ‘management’ merely increases the probability that a range with sufficient fuel load will burn.

The notion that there may be a conservation phase in the exploitation of ecosystems is integral to the theory of optimal renewable natural resource management. We have seen that any problem for which the Hamiltonian is linear in the control variable will support a most rapid approach to the steady state. If the initial values of the state variables are below their optimum values, this implies a period of zero-exploitation or conservation. In this paper the optimisation algorithm chosen selects a steady state level of offtake to be applied in all periods. Hence, the conservation phase is not a ‘no take’ phase, but it is a ‘low pressure’ phase. That is, the management regime is such that the system can function as if it were in the natural state—



at least for some period. More importantly, it is an implication of the management regime that the system will flip from a fire-regulated to a grazing-regulated state at some point, and that the dynamics of the system will be very different in each state.

The numerical example used to illustrate these characteristics of optimal control in a system with both fast and slow variables assumes initial values of the state variables below the optimum values. In the ‘bang bang’ control problem, it is this that favours an initial ‘no-take’ phase. In the steady state optimal control problem, it is this that leads to an initial ‘low pressure’ phase. As might be expected, however, the dynamics of the system are sensitive to initial conditions. But even if the initial conditions favour a grazing-regulated state, the optimal trajectory of the system may still include a sequence of states. For example, a change in the initial conditions of the state variables in this problem, such that all three are relatively high, produces an optimal trajectory with the reverse sequence of states. If the range is initially overgrazed, the optimal policy involves a very high rate of extraction (a rate above the maximum growth rate of wild herbivores), which leads eventually to the depletion of the herd. This in turn leads a grazing-regulated wooded savanna to be succeeded by a fire-regulated grassland. This, and the other examples used in this paper are illustrative only. Moreover, the numerical model used has not been calibrated for any given rangeland. Nevertheless, the existence of a conservation phase in the use of ecosystem goods and services turns out to be a quite general property of the optimal exploitation of renewable resources.

While the economic theory of conservation is relatively poorly developed, it is latent in the theory of renewable resource extraction. The work has yet to be done to explore this formally, but it is quite intuitive that the optimal conservation of distinct resources at different points in time should reflect their relative rates of renewal, as well as their (initial) condition. What this paper seeks to show is that optimal conservation does not necessarily imply a once and for all commitment to preservation. For most resources, conservation is part of a strategy of optimal use-sustainable over some given planning horizon. An optimal strategy may imply a greater or lesser commitment to conservation at different times, and this will reflect both the initial status of the

resources to be conserved, the objectives of the decision-maker, and the state of the natural and the economic environment.

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