Conservation and Optimal Use of Rangelands

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Summary

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. That is, it considers the optimal trajectory of biodiversity change. There are two issues involved in the question of timing. One is the uncertainty associated with the fact that many changes in the flora and fauna of rangelands are 'event-driven'. They depend on stochastic parameters taking particular values before a change of state can occur. A second issue relates to the lag structure of changes. In a system that involves a mix of fast and slow variables, in which the approach to the optimum is not 'most rapid', the optimal trajectory may require the system to remain in an apparently stable intermediate equilibrium for some time before it converges to the optimum state. The paper discusses the role of conservation in the optimal use of rangeland resources.

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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 per cent 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 per cent of area over which a nation has sovereignty (Perrings and Gadgil, 2003). 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 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 explore the implications of the 'speed' of state variables 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. His 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 (Ludwig, Jones and Holling 1978). 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-moving systems are constrained by the large slow-moving systems, but there also occur junctures at which smaller systems are able to disrupt larger systems (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, 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, Gunderson and Peterson (2001) 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, Gunderson and Peterson (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; it's '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 that the fast dynamics of many pests and pathogens can have significant consequences for 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 (Daily and Ehrlich, 1996; Holling, Gunderson and Peterson, 2001; Delfino and Simmons, 2000).

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 of 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 as part of a strategy of optimal use.

2. Resilience and the dynamics of conservation

Ecology works with a rather different set of stability measures than economics. These include measures of 'resistence', 'persistence' and 'resilience' as well as stability. Resistence 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. I 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 states) as we are in the neighbourhood of the stable equilibria. 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.

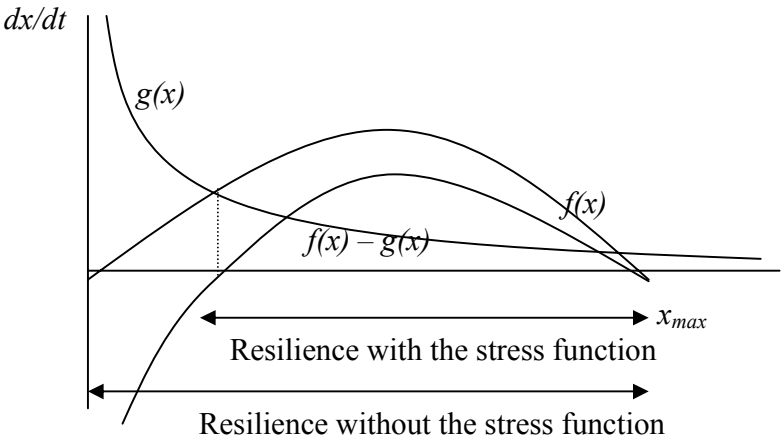
A second and related property of dynamical ecological systems is that their susceptibility to shocks depends on their position in the renewal cycle. Holling describes ecological systems as passing through four phases. A first phase involves the rapid accumulation of both biomass and structure (complexity).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

Figure 1: Resilience with and without stress.

A



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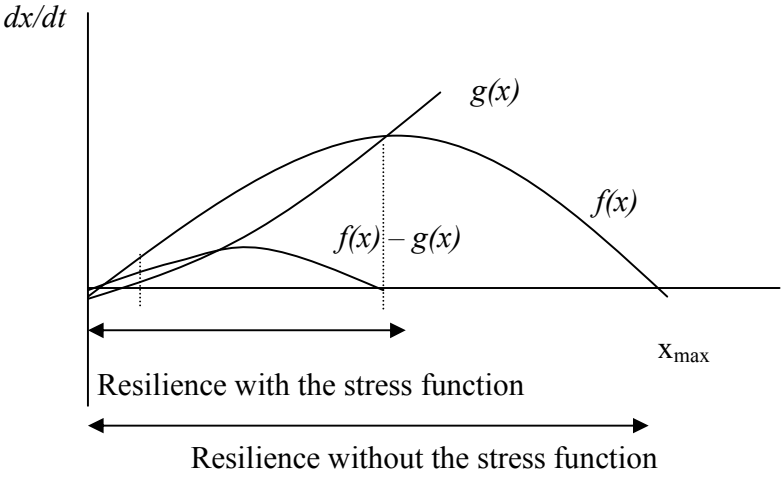
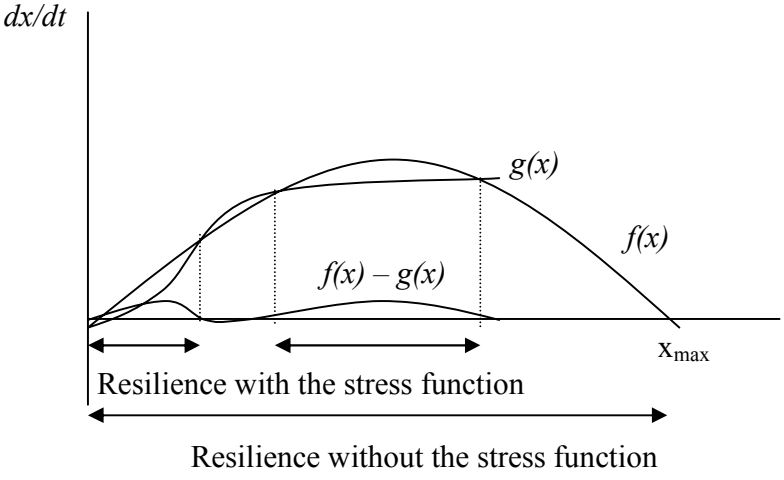


C



Panel A in Figure 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.

To see what this last case implies, let us describe the system as a continuous state space, discrete time Markov process, in which the state variable x_t may exist in one of two basins of attractions. The process is:

$x_{t+1} = f(x_t, u_t)$

with $\{u_t\}$ an IID stochastic process with mean, $\bar{u}$. The two stable equilibria are shown in Figure 2. Suppose the possible realisations of u can be described by the curves, $f(x_t, u_{min})$, $f(x_t, \bar{u})$ and $f(x_t, u_{max})$, $f(x_t, \bar{u})$ being the mean curve of x_{t+1} conditional on x_t, and $f(x_t, u_{min})$ and $f(x_t, u_{max})$.being the lower and upper bounds of the realisations of u. The fixed points of $f(x_t, u_{min})$ and $f(x_t, u_{max})$ define two sets, denoted $x^L_{max} - x^L_{min}$ and $x^U_{max} - x^U_{min}$. The elements of these sets are all possible steady state values for x in the lower and upper basins of attraction respectively.

Now the standard measure of Holling resilience for each set of steady state values of x, is the width of the basin of attraction corresponding to each state. The boundary between the basins depends on the realisation of u, and is indicated by B^L and B^U in Figure 2. It follows that the closer the actual realisation of u is to the lower bound, the larger the basin of attraction corresponding to x^L. Conversely, the closer the actual realisation of u to the upper bound, the larger the basin of attraction corresponding to x^U. For values of x_t between B^L and B^U, the system will converges on either of the two absorbing states, depending on the probability that u_t is above or below $\bar{u}$.

If the upper and lower realisations of u were as in Figure 3, the states corresponding to x^L and x^U might still be 'persistent' but would not belong to distinct basins of attraction. That is, for some values of u there would be a route between the two 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 lower to the upper state.

Figure 2 Distinct states of nature

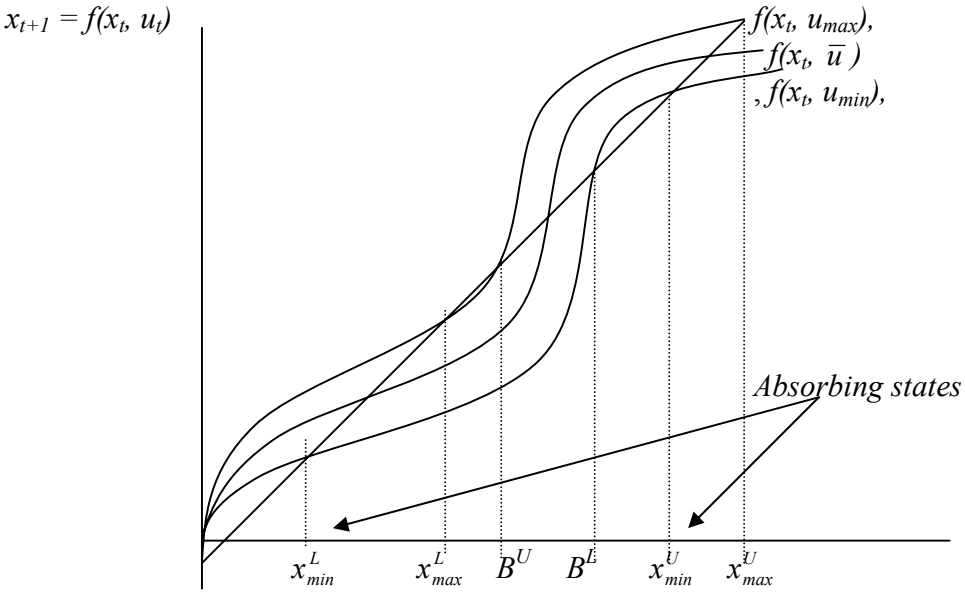
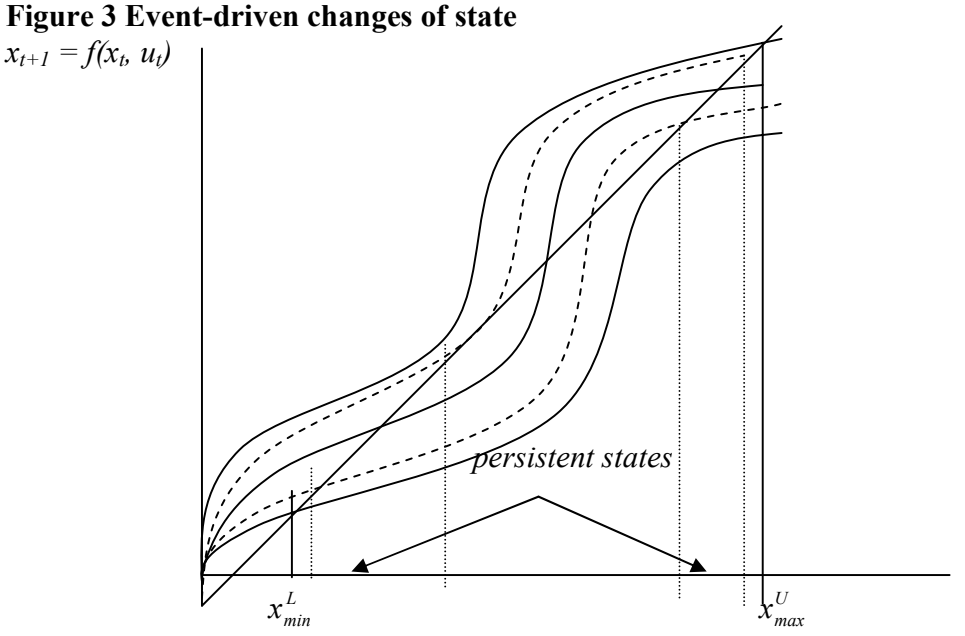


Figure 3 Event-driven changes of state



3. The rangeland model

The model system here is the semi-arid savannas of Central and Southern Africa. A stylised description of this system follows. They have 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 both soils, the rainfall regime and 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 of 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 takes 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. It focuses 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. The paper is concerned less with the steady state, than with the control trajectory. That is, it is 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, $u(t)$, to maximise the net benefits from the use of the ecosystem, where this is the difference between the revenues from harvest, $p(t)u(t)$, and the costs of maintaining the system, $c(x(t), y(t), z(t))$:

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

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

$$\begin{aligned}
\dot{x} &= f(x(t), y(t), z(t)) - u(t) \\
\dot{y} &= g(x(t), y(t), z(t)) \\
\dot{z} &= h(x(t), y(t), z(t)) \\
x(0) &= x_0, y(0) = y_0, z(0) - z_0
\end{aligned}$$

$u(t) \geq 0$.

The current value Hamiltonian for this problem is:

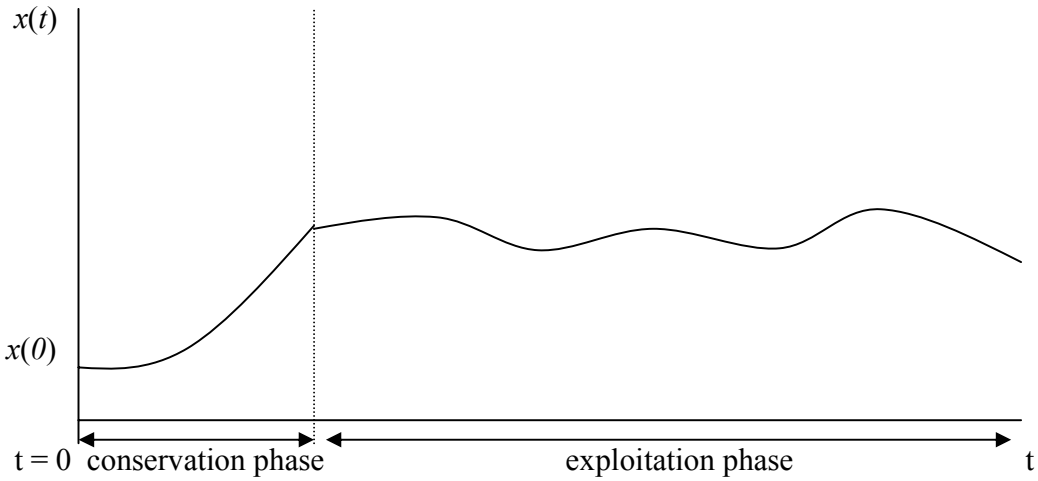
$$\tilde{H} = [p(t)u(t) - c(x(t), y(t), z(t))] + \lambda[f(x(t), y(t), z(t)) - u(t)] + \mu[g(x(t), y(t), z(t))] + \zeta[h(x(t), y(t), z(t))]$$

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 $u(t) = 0$. That is, the optimal control obeys the law:

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

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. This can be thought of as a conservation phase in the optimal exploitation of the resource. In the steady state, the optimal level of harvest implies a particular value for the optimal stock of both the directly exploited resource, and the components of the ecosystem on which it depends. This can be thought of as the steady-state level of conservation of those resources. The optimal control sequence in this case 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 conservation as sustainable use. This is illustrated in Figure 4.

Figure 4: The conservation phase in the exploitation of depleted renewable resources



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:

$$Max_{U,V,S} \sum_{t=0}^{T} \rho^t \left(p_t u_t - c(x_t, y_t, z_t)\right)$$

subject to:

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

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

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

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
α = wildlife growth rate
β = grass growth rate

γ = woody biomass growth rate
ρ = discount factor
ψ = wildife 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
σ_y = proportion of grassy biomass removed by fire
σ_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.

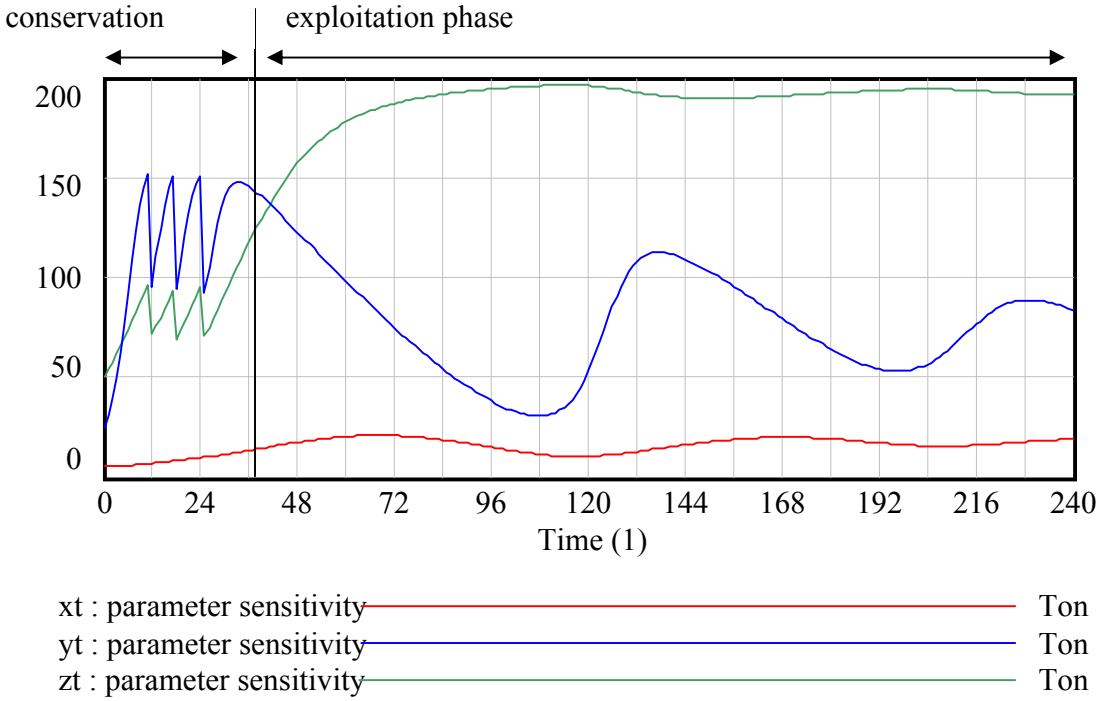
The ecological parameters are drawn from the SEESAW rangeland production model (CSIRO).[1] They are assumed to be constant. The system dynamics are, however, tested for their sensitivity to variation in specific parameter values. The particular problem I wish to consider is the effect of differences in the 'speed' of the components of the rangeland system. 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

[1] The parameter values assumed in this paper are as follows: $p_t = 10$ for all t; $c_{x=0.1,}\alpha = 0.15$, $\beta = 0.5$, $\gamma = 0.1$, $\psi = 0.8$, $c_{yy} = 1$; $c_{yz} = 0.1$, $c_{zz} = 1$, $c_{zy} = 0.25$, $y_{max} = 200$, $z_{max} = 200$, $y_{min} = 150$, $\sigma_y = 0.4$, $\sigma_z = 0.3$.

interdependent. In the absence of herbivores grasses dominate. It is a fire-regulated grassland. In the presence of herbivores, woody plants dominate, depending on the level of grazing pressure, and 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. The (constant) discount rate is assumed to be 5 per cent. 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 initial 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. A simulation of the time path for the system under a profit-maximising strategy is shown in Figure 5. It reports values for the three state variables, woody plants, grasses and wild herbivores. Costs are assumed to be increasing in the stock of herbivores.

Figure 5: Optimal stocks of woody plants, grasses and herbivores, 20 yr horizon.

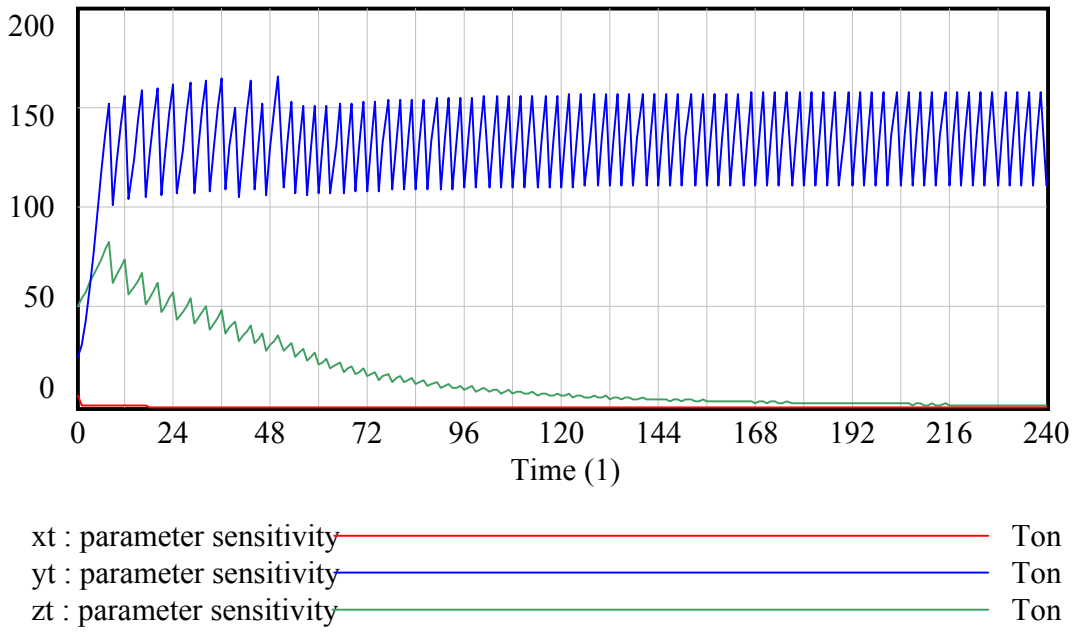


The initial phase in the optimal trajectory of the system – the conservation phase – involves a fire regulated regime, dominated by fast-growing grasses. During this phase

herd sizes are optimally 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.

To see the effect of the regulatory role of wild herbivores in the system, consider a simulation for the same problem, but with the discount rate increased to 15 per cent. This is equal to the maximum natural rate of growth of wild herbivores, and implies that it will be optimal to treat herbivores as a non-renewable resource – that they will be removed from the system in the first period. The result is shown in Figure 6.

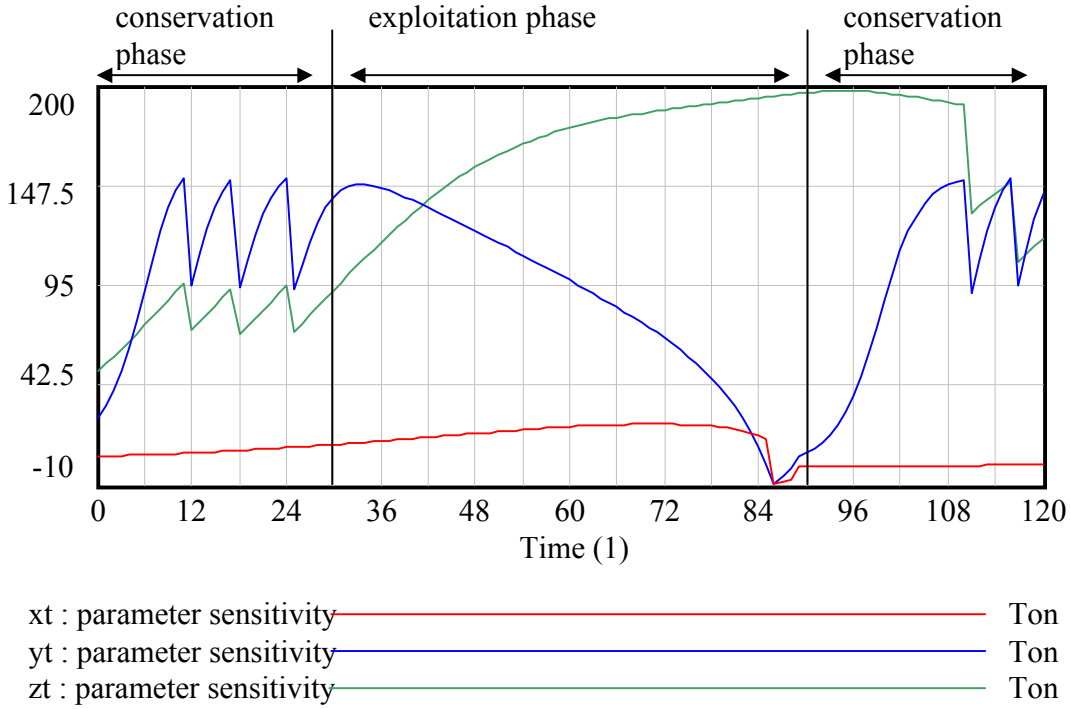
Figure 6: : Optimal stocks of woody plants, grasses and herbivores, 20 yr horizon (discount rate = 0.15)



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 figure does, however, serve to show 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. The length of the decision-maker's planning horizon is also important for the optimal trajectory of the state variables. Figures 7 and 8 show the optimal trajectory for a 10 and 30 year horizon respectively (with no change in the rate of discount). While extending the horizon does not affect the trajectory, shortening it does. Over the shorter horizon it is optimal to harvest at a lower rate, allowing a more rapid build up of herbivore stocks – to the point where overgrazing induces a collapse in both stocks. If repeated, this leads to a cycle of conservation and exploitation phases.

Figure 7: Optimal stocks of woody plants, grasses and herbivores, 10 yr horizon.



Over a longer planning horizon, the cycle takes the form of damped oscillations. Unlike the case of the short horizon, however, the system remains regulated by grazing pressure throughout. The length of time the system remains in one or other state depends on the relative 'speed' of the variables. In this case, if the maximum rate of growth of grassy biomass increases by 50 per cent, it doubles the time the system remains in the conservation phase. This is because of the suppressive 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.

Figure 8: Optimal stocks of woody plants, grasses and herbivores, 30 yr horizon.

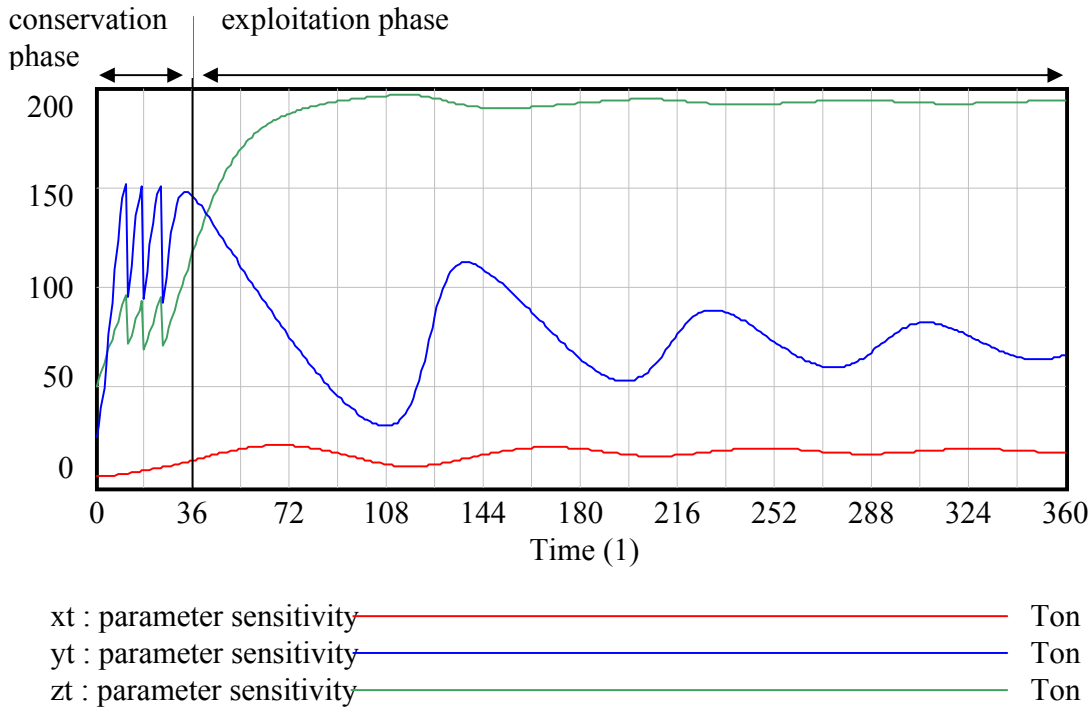
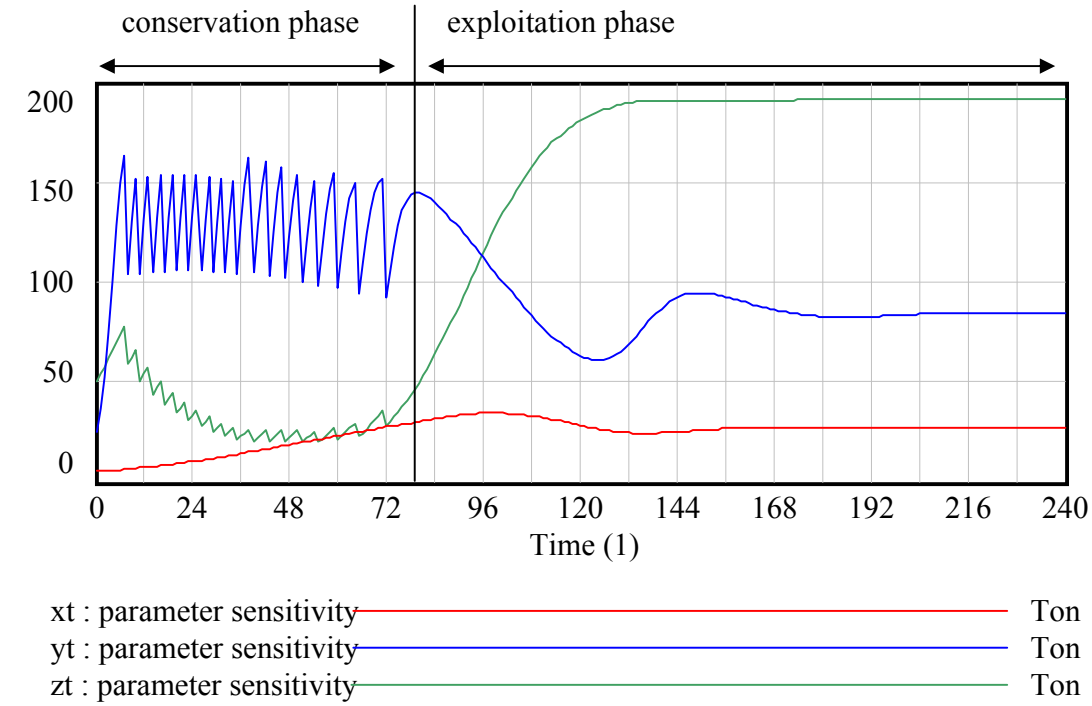


Figure 9: Optimal stocks of woody plants, grasses and herbivores, 20 yr horizon (enhanced rate of grass growth)



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. Both things are illustrated in Figure 9.

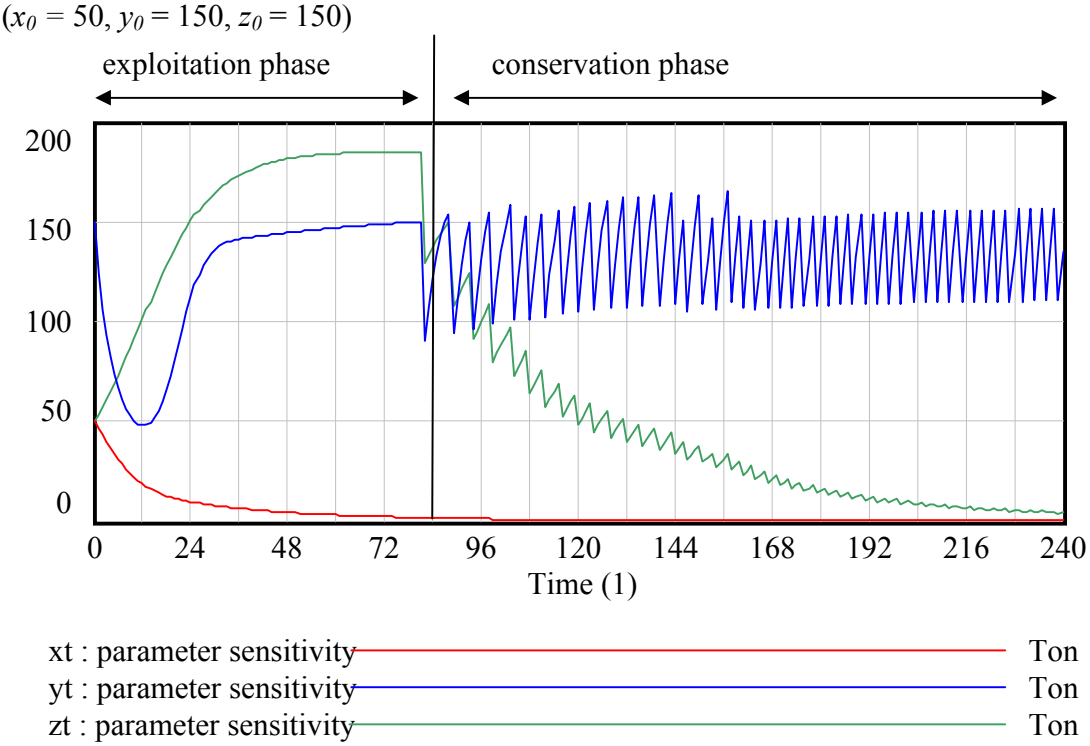
4. Discussion

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.

Figure 10: Optimal stocks of woody plants, grasses and herbivores, 20 yr horizon

(x_0 = 50, y_0 = 150, z_0 = 150)



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 (see Figure 10). 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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