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LANDSCAPE FUNCTION – CONFUSION OR CLARITY FOR IMPROVED RANGELAND MANAGEMENT?

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ABSTRACT

If the landscape function paradigm is to fulfil expectations, then it must, by definition, make ecological sense at the landscape-scale, as distinct from the patch-scale, and thereby provide meaningful direction to land managers. The approaches so far suggested for interpretation of an ecosystem's functionality do not have universal application. Our overall impression is that the work on developing indicators of landscape function has moved well ahead of the underpinning science. We must do the basic research work necessary to modify our rangeland models so they pass the 'test of predictive power'. We require a good understanding of fluxes of water, soil particles and associated nutrients and litter within and through landscapes and how these fluxes vary in response to, for example, geomorphology, vegetation structure and soil type. Just as we are learning that landscape function cannot be generalised across all landscapes, so we may learn that the interactions of composition, structure and function cannot be generalised to support the management of biodiversity. Our interpretation of the papers presented here is that our ability to interpret impacts of management of rangeland and thereby prescribe alternatives remains elusive.

INTRODUCTION

In the same way that Clementsian succession and then state and transition concepts captured the attention of both theoretical and practicing range ecologists (among others), so has the landscape function approach in recent years. This approach is largely attributable to David Tongway and John Ludwig (e.g. Ludwig and Tongway, 1995) and draws strongly on concepts developed by Noy-Meir (1973), Pickup (1985) and others. State and transition theory forced us to address temporal change and temporal scale but provided little to assist interpretation of spatial change. A major advance provided by the landscape function approach is that it directly addressed spatial scale, especially at the scale important to management. It is an interesting observation that, following the publication of Westoby *et al.* (1989), papers relating to state and transitions were commonly presented at rangeland conferences and congresses, while at this conference there are few if any dealing with this approach. Instead many researchers are now exploring the complimentary approach of analysing landscape function, in order to 'read the rangeland'.

In preparing for this conference session on ecosystem processes we suspected that, as with most new concepts, enthusiasm for applying the approach was racing ahead of the basic process-based research necessary to underpin it. We sought input from researchers who have tested or applied the concepts, or developed spatial perspectives that could contribute to production and biodiversity management. In this overview we explore whether the theory may have been extrapolated too enthusiastically. We also seek to articulate where models of landscape function provide uncertain direction for improved management.

WHAT IS LANDSCAPE FUNCTION?

Many now consider that the fundamental requirement for 'healthy' ecosystems is that the ecosystem functions of nutrient cycling, conservation of soil and water resources and consequent production of biomass remain intact (Ludwig and Tongway, 1997; McNaughton *et al.*, 1989). However, current research does not necessarily address all these functions. Bastin *et al.* (this session) adopt a narrower definition relating to conservation of resources (water, soil and nutrients) and fluxes of these resources through and from landscapes where dysfunction relates to leakage of resources – a 'geomorphic' view. In

this model, nutrient cycling is considered incidentally and not explicitly. We highlight here the need for clarity in defining landscape function. We suggest in this review that the focus be shifted to component parts: both nutrient cycling and geomorphic processes (resource-capture), and the interplay between the two, in determining production of plant biomass.

There seems to be no argument that perennial plants (grasses, shrubs and trees) form the building blocks of functional landscapes. Perennial plants play an essential role in nutrient cycling and nutrient enrichment of surface soils (biogeochemical processes), and vegetated landscape patches moderate both aeolian and fluvial geomorphic processes, assisting retention of water and nutrients within the landscape. However, how each of these processes affects the ability of various landscapes to convert rainfall into biomass remains largely untested (Adams this session), even though conversion of rainfall into biomass is perhaps the most direct measure of landscape function (McNaughton *et al.*, 1989). Indeed Adams suggests that most nutrient enrichment can be readily explained by well-documented processes of nutrient uptake, leaf senescence and litter decomposition, without the need to invoke any contribution from transport mechanisms. On the other hand, quantification of geomorphic processes requires more of a landscape perspective relating to overland flows and ultimate destination of mobilized soil, nutrients and litter. On the assumption that a healthy landscape is efficient in converting rainfall and run-on into biomass, the point of contention is whether this is driven by geomorphic processes or nutrient cycling or a combination of both.

In the presented papers we see conflicting evidence regarding the relative contribution of each of these processes. Ford *et al.*'s (this session) controlled experiments in Pilbara systems showed that applied nutrients did not move laterally in response to rainfall irrespective of whether the sites were disturbed or undisturbed or whether obstructions were removed by fire. On the other hand, Kinloch and Friedel (this session) measured significantly greater movement of soil (and presumably associated soil nutrients) in the arid grasslands of central Australia in the presence of disturbance and when there were fewer active resource traps. These contrasting observations may be attributable to soil and topographic differences.

Just how important is the resource-trapping process vis a vis nutrient cycling? Is the relative importance of each related to structure of vegetation, geomorphology and soil type? Loss of perennial grass on cracking clays, for example, appears to have no effect on loss of resources from the system, although we would assume that nutrient cycling is seriously impaired and the availability of nutrients would decline over time. We are unaware of any data to support these assumptions. Outcomes appear to depend on scale and the type of resource (water, soil or plant nutrients). Processes active at the patch-scale may not be those active or important at the patch-mosaic or landscape scale (Reynolds *et al.*, 1997). For example, accumulation of organic carbon and nitrogen may be more a function of nutrient cycling than physical trapping of wind- and water-blown litter at the scale of individual patches. However, conservation of these nutrients within the landscape may have less to do with nutrient cycling and more to do with resource-trapping. This all highlights the requirement for basic science at the process level to quantify the sources and fluxes of each resource at various scales to establish a firm basis for, what in our opinion, is too loosely referred to as landscape function.

This leads us to suggest that application of the landscape function approach cannot be applied by rote. The approaches so far suggested for interpretation of an ecosystem's functionality do not have universal application. While they have done much to help us think about landscapes differently, they should not constrain our thinking or lead us to assume that 'now we have the answer'.

SCALING UP TO LANDSCAPE SCALE AND RESILIENCE

Our observations are that some landscapes remain highly functional irrespective of losses of perennial plants (that is, they continue to efficiently convert rainfall into biomass) while on others excessive disturbance results in catastrophic losses of soil and significant reductions in landscape function (and thus low rainfall use efficiency). Why is this so? What factors determine a landscape's resilience to disturbance (Walker *et al.*, 1981)? We suspect also that the relevant contributions of these processes may also be related to event size. For example, a one in a hundred-year rainfall event may have a disproportionately greater effect than the accumulated effects of a year or more of modal rainfall events totalling the same amount, and we suggest the concept of threshold events may be a useful in developing landscape function models. We question for example whether the results of Ford *et al.*'s research would be the same following a cyclonic deluge? Tongway and Ludwig (1994) make the distinction between 'plant-moderated' and 'landscape-moderated' landscapes and suggest inter-relations with landscape resilience. Bastin *et al.* present the challenge 'to work out the implications of resource transfer between landscapes - as distinct from the patch scale' - with which we agree wholeheartedly. And, continuing the sentence, 'the extent to which these matter for landscape resilience' - whereas the issue may be rather how landscapes of different resilience respond in terms of redistribution of resources in the face of disturbance. Nevertheless, an *a-priori* assessment of landscape resilience in the Western Australian shrubland provided few insights into the capacity of landscapes to respond to rainfall. Contrary to expectations, herb mass increased on both resilient and non-resilient landscapes as proportional areas occupied by vegetated patches declined (Holm *et al.*, 2002b).

Is resilience a useful categorization for landscapes, which will enable better understanding and management strategies? We don't know. There are few established procedures to classify landscapes according to resilience (see Holm *et al.*, 2002b for details of one approach). To quote Illius and Hodgson (1996) 'The most widely practiced [way of establishing an ecosystem's stability and resilience] is to look out of the window periodically and see if it is still there after all these years and, if so, conclude that it must be persistent, is probably stable, and therefore resilient'.

We concur with Illius and O'Connor's summary (1999) that theoretical concepts, including metaphors for landscape expression such as resilience, may be popular but do little to assist defining underlying mechanisms responsible for different responses of landscapes to perturbation, a point well made by Adams. A proper basis for understanding concepts such as resilience and landscape function will only be achieved through studies that quantify the interplay between patch heterogeneity, geomorphic processes affecting movement of materials around landscapes, and biogeochemical processes of nutrient cycling. The work of Snyman and Van Rensburg (1987) in South African grasslands and Burke *et al.* (1999) in grassland ecosystems in the central Great Plains of the USA are examples known to us, however the work is at the small-plot scale and Snyman himself makes the point that it is difficult to scale up these results to the landscape-scale. Kinloch and Friedel and Roth *et al.* (poster session), both consider processes at the landscape-scale, and we see a need for more investigations at this scale.

INDICATORS OF LANDSCAPE FUNCTION

In the context of this conference session, Bastin *et al.* provide an admirable definition of an indicator as 'a piece of information, easily acquired, that links scientific measurements to practical interpretations of landscape function'. If the landscape function paradigm is to fulfil expectations, then it must, by definition, make ecological sense at the landscape-scale, as distinct from the patch-scale, and thereby provide meaningful direction to land managers. Our overall impression is that the work on developing indicators of landscape function has moved well ahead of the underpinning science.

The landscape function analysis (LFA, Tongway and Hindley, 2000) approach is being enthusiastically adopted, on the assumption that it can be applied in all landscapes. Clearly this has not been sufficiently tested. In a study by one of us, proportional areas of vegetated patches were found to be positively correlated with oxidizable C and total N in surface soil in low-woodland, but not in low-shrubland communities. Furthermore, suggested indices of soil-surface condition and derived ratings of landscape function were positively related to soil fertility but less clearly with proportional areas of vegetated patches, and these relationships were community-type specific (Holm *et al.*, 2002a). Webb *et al.* (this session) reported that 15 years of excluding cattle from Burdekin sites produced large improvements in biomass, cover and LFA indicators of stability, infiltration and nutrient cycling, but resulted in little improvement in soil nutrients.

Returning to the point made earlier, we require a good understanding of fluxes of water, soil particles and associated nutrients and litter within and through landscapes and how these fluxes vary in response to, for example, geomorphology, vegetation structure and soil type, before we can begin to unravel relationships between indicators of landscape function.

Scaling up inevitably leads to loss of fine scale detail, which is incorporated into some broader-scale response. Plant biomass responses to rainfall at the patch-scale may vary widely according to many patch-specific factors (run-on/run-off, plant composition, nutrient availability etc.) yet at the landscape-scale, this variability is subsumed and may be represented, for example, as a single index of greenness as measured by NOAA satellites (Tucker *et al.*, 1985). This information is useful for clients with an interest in such broad-scale issues as global warming, drought and carbon sequestration but may be of limited use if questions being asked, for example, relate directly to impacts of land-use on biodiversity. However clients have different information requirements depending on values they ascribe to the landscape (West *et al.*, 1994) and for a range of indicators that address these needs.

LANDSCAPE FUNCTION AND BIODIVERSITY

While researchers are well on the way to defining the unanswered questions and areas of doubt with regard to factors affecting how landscapes respond to rainfall, we appear to know less about landscape function and biodiversity. The relationship can be viewed in both directions as in these two example questions. A) Is there a change in resource retention when single-stemmed plants without mounds replace multi-stemmed shrub mounds? In other words, what is the effect of change in biodiversity on landscape function? Or considering the effect in reverse: B) Is there concomitant decline in biodiversity as landscapes degrade? As with all these processes, the answers to these questions have both temporal and spatial dimensions. On most landscapes, degradation is accompanied by loss of perennial plants and an increase in annual plants in the short-term. Over much longer time-frames dislodged soil materials may be re-deposited, and colonised by alternative suites of perennial species, and often at broader spatial scales (e.g. Archer, 1995).

Some contributors to this session consider type B questions – that is, the effect of landscape degradation on biodiversity. Ayers' (poster session) evidence from western New South Wales is that, at a single point in time, patches supporting trees and perennial grass scored slightly better than woody shrub patches, and consistently better than run-off areas, on Tongway and Hindley's (1995) soil surface condition assessment. She also found that most taxa examined (which included plants, vertebrates and invertebrates), were unresponsive to differing shrub cover, although a small proportion were. This is consistent with Landsberg *et al.* (1997) who examined the response of various taxa to gradients of grazing in a range of sites across inland Australia. They found that a few species were restricted to the least grazed (most 'functional') areas, while most others persisted to a greater or lesser extent.

Landman *et al.* (this session) reported that trees in Pilbara creeklines depend on both soil and groundwater while trees on floodplains and hillslopes do not access groundwater and survive without it. Industrial use

of groundwater will lead to differential loss of trees in creeklines, as a consequence. While this will mean a substantial change at landscape scale, what do we know about likely changes to functionality of the creeklines and impact on other taxa? Chilcott *et al.* (this session) provide us with some insights into the impact of biodiversity on function. They tell us that retaining strips of native vegetation, including trees, in paddocks being cleared for grazing benefits the soil's physical, chemical and biological properties and brings about some changes to soil and surface fauna, arguably through reduction in evapotranspiration.

A body of theory has been developed on Type A questions – that is, the effect of loss of biodiversity on landscape function. Concepts of 'rivets' (Erlach and Walker, 1998) where the system fails when one too many (species) falls out, 'keystones' (Hurlbert, 1997) where the system fails when a key species goes and 'ecosystem engineers' (Power *et al.*, 1996) species which shape environments to the benefit of others, remain tantalizing ideas, as yet unproven, and provide few insights into the relationship between biodiversity and landscape function in rangelands. We need to refine our concepts and test them more thoroughly before we can be confident that they will lead to improved management.

CONCLUSIONS

Our concepts of retention and loss of resources in rangeland systems are dependent on scale. Losses from a single shrub mound may be captured by others close by, so that at a 100 sq m scale, no loss occurs. Losses from several hundred sq m may be captured by a nearby closed drainage system, so that there is still no net loss. In the case of livestock production systems, overall plant production may remain the same despite spatial rearrangement, or it may be radically reduced if the system is open and susceptible to resource loss at the whole system level. We may need to think in terms of nested scales to understand the different processes at work.

It is perhaps relatively simple to think about landscape function in relation to livestock production systems because we can relate function to the scale of management units, perhaps paddocks or properties. Biodiversity is less immediately amenable to neat scale limits because it can be conceptualised in many ways, from diversity at genetic to landscape level, and from perspectives of structure, function and composition (Noss, 1990). Right now, our understanding of biodiversity in rangelands is largely limited to structure and composition at community to landscape scale, while we are beginning to deal with function and its interactions with structure and composition.

We suspect that increasing understanding will show that we should not be looking for a single model which will work for all landscapes or ecosystems. Just as we are learning that landscape function cannot be generalised across all landscapes, so we may learn that the interactions of composition, structure and function cannot be generalised to support the management of biodiversity. We will also need to grapple with spatial and temporal scale and recognise that we may require different models to deal with aspects from local up to regional scale and apply these models not just over a few years as is the norm, but over decades or more.

The concept of landscape functional analysis (Tongway, 1991) has been influential in the last decade of rangeland science and has stimulated us to re-define our understanding of ecological processes. However, it is important that we continue to test and challenge our concepts. We believe that enthusiasm for adopting and applying this new thinking has indeed moved ahead of the underpinning science and we are hopeful that the next decade will see significant growth in our understanding of ecological processes in relation to how nutrients are distributed and redistributed within and between landscapes. This is a clear challenge to those distressingly fewer and fewer researchers among us – especially to those who have 'got about more', to quote Adams. We must do the basic research work necessary to modify our rangeland models so they pass the 'test of predictive power'.

We have made the point elsewhere that most rangeland monitoring systems have attempted to assess impact of management through interpretation of change in botanical composition, but this has been fraught with uncertainty (Friedel *et al.*, 1993; Holm *et al.*, 2002a). We acknowledge that these monitoring systems should also address processes that modify growth and productivity especially at landscape scales, but our interpretation of the papers presented here is that our ability to interpret impacts of management and thereby prescribe alternatives remains elusive.

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