

## Indicator patches: exploiting spatial heterogeneity to improve monitoring systems

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**Abstract.** When choosing which environmental attributes to monitor in assessing disturbance, it is important to consider not only which metric will provide the most sensitive indicator of disturbance, but also the spatial considerations of where in the landscape that metric will be most responsive to change. Degradation in landscapes is often unevenly expressed because: (i) disturbance is spatially localised, (ii) landscape elements differ in their sensitivity to disturbance, and (iii) degradation following localised disturbance is spatially contagious. The spatial heterogeneity of degradation has proven to be a key obstacle to rangeland monitoring (e.g. where the initiating processes of broad-scale degradation are concentrated in landscape locations that are not detected by surveys) but can also provide opportunities to focus monitoring efforts. We propose that the effectiveness of monitoring could be enhanced by identifying and selectively monitoring ‘indicator patches’, i.e. specific landscape locations that provide the most management-relevant and timely information about the consequences of a monitored disturbance. We tested and demonstrated the utility of the ‘indicator patch’ concept in the rangelands of the Succulent Karoo in southern Africa. We contrasted the grazing response of dominant ‘representative’ vegetation, with responses of interspersed patches of distinct vegetation associated with zoogenic mounds. Since mound vegetation is more palatable and preferentially grazed by sheep, we tested whether mounds could serve as ‘indicator patches’ in providing a sensitive measure of grazing disturbance. Percentage canopy cover measurements in dominant off-mound vegetation provided a poor indicator of grazing disturbance (although more intensive plant size measurements did reveal grazing impacts on plant population dynamics). In contrast, vegetation on mounds displayed patterns of changes in species abundances that were easier to detect and useful for interpreting and quantifying the effects of grazing. Mound vegetation could, therefore, be used as ‘indicator patches’ and targeted for exclusive sampling as a sensitive method for monitoring rangeland condition and detecting early warnings of vegetation change. This approach could be widely employed to better harness the extensive knowledge base regarding the patchy, spatially localised nature of degradation-initiating processes in numerous other landscapes. Routinely incorporating this understanding into the design of monitoring programs could improve the effectiveness of sampling effort, allow detection of more subtle trends (changes), and provide earlier warning of impending degradation so remedial action can be taken before degradation becomes severe and widespread.

**Additional keywords:** Karoo, plant demographics, rangeland.

### Introduction

Degradation in landscapes is often unevenly expressed (i.e. heterogeneous) because (i) disturbance is spatially localised, (ii) landscape elements differ in their sensitivity to disturbance, and (iii) degradation following localised disturbance is spatially contagious (Bisigato and Bertiller 1997; Pastor *et al.* 1998; Adler

*et al.* 2001; Peters *et al.* 2004). This can create obstacles as well as opportunities for collecting and interpreting ecological data. Over the past few decades, a considerable body of knowledge has been amassed regarding the interrelationships between landscape patterns and spatial localisation of biological and physical processes (e.g. Senft *et al.* 1987; Swanson *et al.* 1988; McAuliffe

1994; Ludwig *et al.* 1997; Aguiar and Sala 1999; Bestelmeyer *et al.* 2006b), but we suggest that this understanding has yet to be fully utilised in the design of ecosystem monitoring programs to improve the sensitivity in detecting ecological changes. This is evidenced, for example, by the observation that detailed (one-off) assessments of landscape processes often provide evidence that ecological degradation is occurring (e.g. Pringle and Tinley 2003; Pringle *et al.* 2006), yet long-term monitoring programs (which consist of repeated-measurements over time) may fail to detect substantial negative trends in measured indicators of ecological health (e.g. Watson *et al.* 2007). In the present work we consider the challenges presented by each of the three cases of spatial heterogeneity in degradation processes listed above, and then suggest the opportunities these present for improving the design of monitoring programs.

First, different patch types experience different frequencies and intensities of disturbance. For example, patch grazing may develop due to the differential palatability of plants. Grazing is subsequently concentrated on those patches and diminished on adjacent patches (Senft *et al.* 1987; Adler *et al.* 2001). In some cases, underlying soil heterogeneity or previous disturbance may initially generate differences in plant species composition to which grazers respond. Thus, monitoring in non-preferred patch types would fail to register impacts until after large changes had already occurred in preferred patches.

Second, patch types often differ in their responses to more-or-less uniform disturbance. For example, plant communities may differ in their sensitivity to disturbance due to the characteristics of plants comprising the community, or even in similar plant communities due to differences in soil properties that may be difficult to detect (Stokes 1994, 1999; Stroh *et al.* 2001; Bestelmeyer *et al.* 2006b). In aggregated data, the responses of one patch type may be obscured by contrasting responses of other patches.

Third, and most insidious, the process of landscape degradation may initially be confined to localised portions of a landscape, but expand over time to affect broader areas (Stokes 1999; Bestelmeyer *et al.* 2006a; Pringle *et al.* 2006). This process can be driven by expanding populations of invasive species, soil erosion, or by broad-scale hydrological or eolian processes that interact with localised disturbance. The localised 'trigger sites' for such cross-scale interactions may be insufficiently sampled or missed altogether if sampling designs target the dominant landscape elements that initially only weakly express effects of disturbance.

Spatial heterogeneity in disturbance responses and processes has proven to be an important issue for rangeland monitoring, and is exemplified by contrasting observations of degradation in Western Australian rangelands (Pringle and Tinley 2003; Pringle *et al.* 2006; Watson *et al.* 2007). A recent analysis of the Western Australian Rangeland Monitoring System (WARMS), based on long-term on-ground monitoring sites, found little evidence of degradation on flat interflaves away from permanent water (Watson *et al.* 2007). In contrast, aerial assessments of rangeland condition in the same region have indicated that drainages in floodplains are becoming more deeply incised, base levels are becoming lower and the landscape is increasingly desiccated (Pringle and Tinley 2003; Pringle *et al.* 2006). A possible explanation for this apparent contradiction has been given in

terms of spatial localisation of degradation processes that begins with channel incision in drainages cutting back upslope: as landscapes desiccate and intervening areas of intact interflaves contract, portions of uplands away from drainages may initially remain unchanged, or even improve (Pringle *et al.* 2006; Watson *et al.* 2007). These observations highlight the critical need to consider where in the landscape the effects of disturbance are most strongly expressed, and where and how these localised responses can be effectively measured to give earlier forewarning of impending changes.

Rather than adding complication, the consideration of spatial heterogeneity in degradation and recovery processes represents an opportunity: the effectiveness in monitoring sampling designs and analyses could be improved by considering which locations within the landscape provide the most useful information on the initiating processes of degradation. The subdivision of rangeland plants into grazing-response categories has served a similar purpose in rangeland monitoring in the past: for example, the segregation of grazing-intolerant 'decreasers' from disturbance-adapted 'increasers' (Dyksterhuis 1949; Heard *et al.* 1986) as well as targeting 'key species' as the most sensitive indicators of grazing impacts (Willis and Trollope 1987). We suggest that landscape elements may be used in a similar way by formally recognising 'indicator patches', which we define as those particular elements of the landscape that provide the most management-relevant and timely information about the consequences of a monitored disturbance. By identifying 'indicator patches' during the design phase of a monitoring program, sampling could be strategically targeted on those specific portions of the landscape where degradation processes are initiated, effects of disturbance are most strongly expressed, and temporal trends in monitored attributes could be most readily distinguished. This strategy could improve the effectiveness of monitoring effort in detecting early signs of degradation, particularly where resources are limited or areas to be monitored are large. Monitoring the ultimate impacts of disturbance in the dominant/representative parts of the landscape may only provide indicators of degradation after-the-fact, when those impacts are already representatively (and often irreversibly) expressed across the whole landscape. However, monitoring indicators of the proximate causal processes in localised 'indicator patches' (that could quite possibly be unrepresentative of the rest of the landscape and its condition at the time of sampling) has the potential to provide a more sensitive indicator of the future condition of the rest of the landscape, and an early forewarning of impending change. Early warnings provide important additional opportunities to avoid degradation through timely corrective management intervention. Furthermore, the segregation of sensitive from non-sensitive elements can reduce the 'noise' (where spatial and temporal factors other than grazing disturbance strongly affect variations in the abundance and distribution of species), which often precludes effective interpretation of monitoring data, by ensuring that disturbance has a strong influence on trends of the monitored attributes in the selectively-sampled landscape locations.

Here, we demonstrate the utility of the 'indicator patch' concept by presenting a case study in the rangelands of the Succulent Karoo in southern Africa. We contrast the grazing response of dominant 'representative' vegetation, with responses

of interspersed patches of distinct vegetation associated with zoogenic mounds. Since mound vegetation is more palatable than non-mound vegetation, and, thus, preferentially grazed by sheep (Armstrong and Siegfried 1990), we tested the hypothesis that mounds could serve as 'indicator patches' in providing a sensitive indicator for monitoring grazing disturbance [illustrating cases (i) and (ii) above]. Similar opportunities for 'indicator patches' occur in rangelands throughout the world, including Mima mounds (e.g. Cox 1984; Cox and Roig 1986; Cox and Gakahu 1987), patches generated by biopedurbation of desert rodents (e.g. Bestelmeyer *et al.* 2006b), grazing-sensitive areas such as riparian areas (West 2003), intermediate zones along grazing gradients that most clearly indicate important changes in land condition (in contrast to patches near to water points or patches far away from water points) (e.g. Pickup *et al.* 1998; Pringle and Landsberg 2004), and patches generated by landforms, geology, and physical or chemical soil properties (e.g. Denbow 1979; McAuliffe 1994; Stroh *et al.* 2001; Augustine 2004).

### Materials and methods

The present study was conducted in the dwarf shrublands of the Succulent Karoo, which form the south-west, winter rainfall portion of the extensive semi-arid rangelands in southern Africa (Mucina *et al.* 2006). It has long been suggested (and disputed) that the dominant land use – extensive grazing by sheep – has strongly influenced vegetation and land resources in the region (Acocks 1953; Bond *et al.* 1994; Dean *et al.* 1995; Hoffman *et al.* 1995). However, it has been difficult to quantify the effects of grazing on Succulent Karoo vegetation due to the diverse flora, which has localised and patchy species distributions (Hilton-Taylor 1987; Cowling *et al.* 1989), and 'demographic inertia' of long-lived woody shrubs, which can dilute the apparent influence of recent grazing disturbance on vegetation. The occurrence of ancient zoogenic mounds (Lovegrove and Siegfried 1986; Midgley *et al.* 2002), locally known as 'heuweltjies' (which translates literally as 'little hills' or 'mounds'), produces clearly-defined vegetation patches. The mounds, which are  $\approx 20$  m in diameter and less than 1 m high, are associated with ancient (thousands of years old) termitaria, and are focal points for animal activities that have helped maintain and enhance the modified soil and vegetation characteristics (Lovegrove and Siegfried 1986; Milton and Dean 1990; Moore and Picker 1991). These patches exist within a spatially-dominant matrix of unpalatable species, usually a *Pteronia* sp. (asteraceous dwarf shrub) and/or a mesemb (e.g. *Ruschia* spp.), a diverse, monophyletic group of leaf succulents that is largely endemic to southern Africa (Chesselet *et al.* 1995). Matrix vegetation has chemical and/or physical defences against herbivores (e.g. Kellerman *et al.* 1988) and, therefore, does not respond strongly to grazing. The contrasting behaviours of these patch types has complicated the interpretation of traditional extensive rangeland measurements (which sample across patch types or target the dominant matrix vegetation) but also provides an ideal opportunity to test the 'indicator patch' concept.

Seven sites were used for the study, consisting of the Worcester Veld Reserve (33°39'S, 19°26'E  $\approx 100$  km ENE of Cape Town, South Africa) and a contiguous set of small peri-urban sheep-grazing properties and a block of municipal land to

the east. The sites ranged in area from 2 to 30 ha, and spanned 2 km of south-facing slopes along an east-west spur in the foothills of the Hex River Mountains (part of an anticline in the Cape Fold Belt). Soils were skeletal and derived from Malmsbury shales, but those on zoogenic mounds were more fertile, moister and more alkaline (Midgley and Musil 1990). Mounds covered  $\approx 10$ –15% of the site by area at a density of 3–4 ha<sup>-1</sup>, and were uniformly distributed among sites and slope positions. The mean annual rainfall is 269 mm most of which falls in winter between June and August (Riginos *et al.* 2005). The lowest mean monthly minimum temperature of 5.7°C occurs in June, and the highest mean monthly maximum temperature of 34.5°C occurs in February (Smitheman and Perry 1990).

The seven sites examined (designated A–G) had strongly contrasting long-term sheep grazing histories, while the close proximity of the sites to one another (within 2 km of each other) within the same landform and sharing the same aspect ( $\approx 20\%$  slopes facing south situated along the same mountain spur) ensured that other factors, such as climate, soils and vegetation, would originally have been similar (before differences in land management). There was sufficient information on past land management activities to rank the sites into four categories of long-term (>20 years) grazing utilisation (Table 1). Sheep had been excluded from the Worcester Veld Reserve (two separate hill slopes, sites A and B) and municipal land (site C) for the past 50 years. Two properties had historically been moderately grazed (sites D and E) and one property had been heavily stocked (site F), but these sites had been left ungrazed for the past 3 years. The most utilised site (site G), used as a small ( $\approx 2$  ha) holding area, had been very heavily grazed in the past and was still being lightly grazed. None of the sites was being heavily grazed at the time of sampling which ensured that plant measurements would reflect the long-term effects of sheep grazing (rather than the immediate, short-term effect of browsing in reducing shrub canopy dimensions).

Dominant matrix vegetation was sampled using five 20-m transects (orientated upslope) approximately evenly spaced along the south slopes ( $\approx 300$  m in length) of each site using the ellipse intercept method (Stokes and Yeaton 1994), avoiding rocky outcrops, mounds and fence lines. Similarly, 4–6 mounds (depending on availability) were sampled along the hill slope at each site by running sample lines across the diameter (15–25 m) of each mound in the direction of the slope. Samples were more

**Table 1. Categorisation of the seven study sites into four levels of grazing utilisation according to past histories of sheep grazing**

The altitude column indicates the range of slope positions sampled along the hill slope at each site. For convenience, the letters used to designate the seven sites (A–G) have been allocated in order of increasing utilisation score

Site	Land use history	Altitude (m)	Utilisation category
A	Protected reserve, hill slope 1	300–360	0
B	Protected reserve, hill slope 2	310–380	0
C	Ungrazed municipal land	360–430	0
D	Moderately grazed small holding	320–370	1
E	Moderately grazed small holding	320–370	1
F	Heavily grazed small holding	320–370	2
G	Very heavily grazed holding area	300–340	3

closely spaced in the smaller holding area (site G, ≈2 ha, ≈200 m slope length). The ellipse intercept method involves approximating the canopy of each plant to an ellipse and, where this ellipse is intercepted by the sampling line, the lengths of the two perpendicular axes (largest diameter and then perpendicular to that) of the projected canopy ellipse are measured. Canopy cover (*C*, percentage area) and plant density (*D*, plants m<sup>-2</sup>) are then calculated as (Stokes and Yeaton 1994):

$$C = 100 \frac{\pi}{4l} \cdot \sum \left( \frac{a_i \cdot b_i}{W(a_i, b_i)} \right) \% \quad (1)$$

$$D = \frac{1}{l} \cdot \sum \left( \frac{1}{W(a_i, b_i)} \right) \quad (2)$$

where

$$W(a_i, b_i) = a_i \left( \frac{2}{\pi} + \left( 1 - \frac{2}{\pi} \right) \cdot \left( \frac{b_i}{a_i} \right)^{1.45} \right),$$

*l* is the length of sampling transect, *a<sub>i</sub>* is the length of the longest canopy ellipse axis for plant *i*, and *b<sub>i</sub>* is the length of the shortest, perpendicular canopy ellipse axis (all measures of length should use the same units, e.g. m).

The ellipse intercept method was chosen because it provides information on plant sizes and densities, and can reveal changes in vegetation that are not evident from canopy cover alone (Stokes and Yeaton 1994). However, it takes extra sampling effort to gain this information, so monitoring approaches would typically use less intensive sampling methods that rely on a single measure of abundance (such as percentage canopy cover alone) for each species.

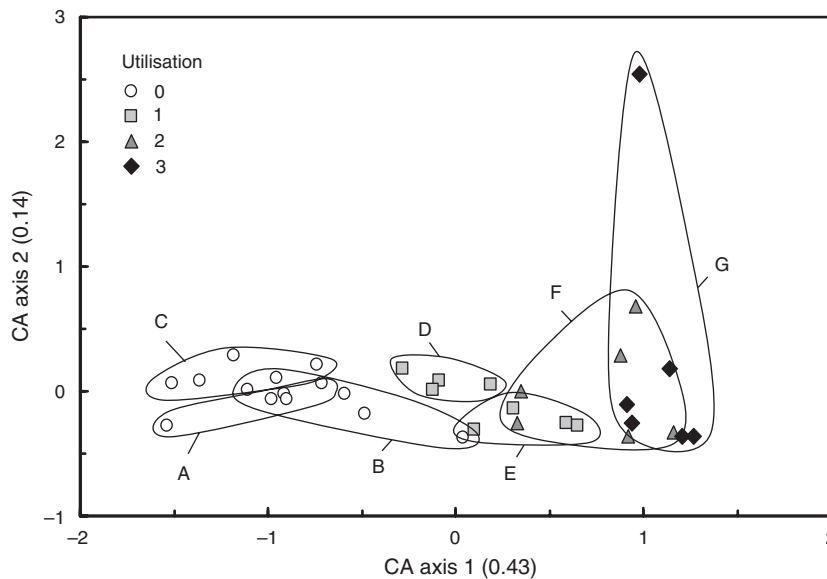
We analysed the two datasets for mound and off-mound surveys separately, both using the same following approach.

Species' canopy covers (% area) were calculated for each sample (individual ellipse intercept transect) and were then subjected to ordinations using correspondence analysis (Pielou 1984). The set of samples from each site was then combined to calculate the average first and second axis ordination scores. The relationship between site utilisation categories (Table 1) and ordination scores was then tested based on Spearman's rank correlation coefficient using R (v.2.01) statistical analysis software (R Development Core Team 2008). To interpret the ordinations, grazing responses were quantified for dominant species and other potential indicator species. We then plotted trends of changes in plant canopy cover with increasing grazing utilisation for each of these species, and tested these relationships statistically using rank correlations as described above.

For the dominant off-mound species (*Pteronia paniculata* L.f. and *Ruschia caroli* Schwantes) and dominant mound species (*Pteronia incana* (Burm.) DC., *Euphorbia burmannii* E. Mey. ex Boiss., *Euphorbia mauritanica* L. and *Galenia africana* L.), size distributions within each utilisation category were generated by calculating plant densities in the canopy size classes <40 cm<sup>2</sup>, 40–160, 160–640, 640–2560 and >2560 cm<sup>2</sup> (equivalent to circular canopy diameter classes of <7 cm, 7–14, 14–29, 29–57, and >57 cm, respectively). A log scale was used to offset the increase in variation that accompanies increases in canopy size/age (Yeaton 1984). Responses of plant size and plant density of each species to increasing grazing utilisation across the seven sites were tested using Spearman's correlation coefficient as before.

### Results

Ordination of mound vegetation surveys (based on canopy cover measurements) revealed a strong gradient with the first ordination axis accounting for 39% of variation (Fig. 1). The ranking of study sites along the first ordination axis corresponded with the rank order of historical utilisation categories. Statistical analyses



**Fig. 1.** Correspondence analysis (ordination) of surveys of plant canopy cover for mound vegetation. Eigen values are presented with their respective axis. Labels for sites and grazing utilisation categories (0, protected–3, very heavily grazed) follow Table 1.

confirmed a strong and significant correlation between utilisation level and the first ordination axis score (Table 2). The influence of grazing on vegetation was reflected in progressive changes in canopy cover of the dominant mound species (Fig. 2). On mounds, increases in utilisation level were associated with the gradual replacements of: (i) *P. incana* by *G. africana* (woody shrubs), (ii) *E. burmannii* by *E. mauritanica* (stem succulents), and (iii) patches of *R. caroli* and *P. paniculata* (the dominant off-mound vegetation) by patches of bare ground or annual species. However, in analyses by individual species, only the decline in *P. incana* with increasing utilisation was statistically significant (Table 2). The first ordination axis score, which is based on an integrated 'weighted average' of species abundances, provided a better indication of the overall effects of utilisation level (higher correlation) than any individual species alone (Table 2), possibly because of spatial variation in the abundance of the individual species among mound patches. The increase in abundance of species such as *G. africana* and *E. mauritanica* at higher utilisation levels compensated for the decrease in other species, so that the overall decline in vegetation cover was not statistically significant (Table 2).

Size distributions of mound species (Fig. 3) provided further evidence for the sequential replacement of species that had already been shown by percentage canopy cover data (Fig. 2), but emphasised that these changes did not just involve changes in the canopy sizes of individual plants but also changes in plant density, i.e. mortality of 'decreaser' species (Fig. 3a, b) and recruitment of 'increaser' species (Fig. 3c, d).

**Table 2. Correlations of site utilisation scores with ordination axis scores and measures of canopy cover for potential indicator species**

All scores were first combined into an average for each site,  $n = 7$ , Spearman's rank correlation coefficient; \* $P < 0.05$

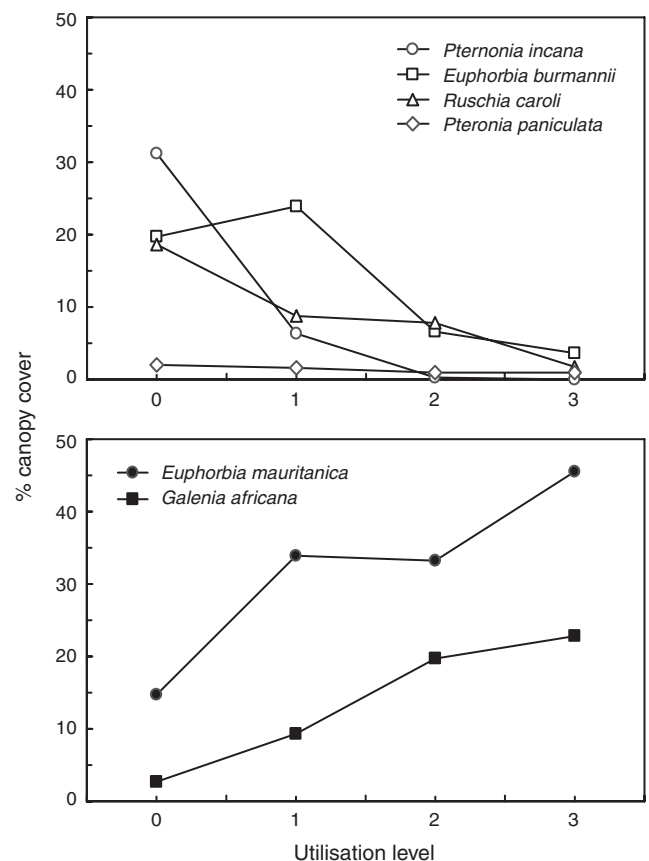
Species	Correlation with utilisation score	Average % canopy cover <sup>A</sup>	Change in % canopy cover <sup>B</sup>
<i>Mound vegetation</i>			
Axis 1	+0.95*	—	—
Axis 2	+0.37	—	—
Total cover	-0.47	97	-21
<i>Pteronia incana</i>	-0.77*	10	-31
<i>Euphorbia burmannii</i>	-0.11	13	-16
<i>Ruschia caroli</i>	-0.43	9	-17
<i>Pteronia paniculata</i>	-0.3	1	-1
<i>Euphorbia mauritanica</i>	+0.62	32	+31
<i>Galenia africana</i>	+0.67	14	+20
<i>Off-mound vegetation</i>			
Axis 1	+0.37	—	—
Axis 2	-0.6	—	—
Total cover	-0.6	67	-46
<i>Pteronia paniculata</i>	+0.07	27	-19
<i>Ruschia caroli</i>	-0.51	20	-16
<i>Tetragonia hirsuta</i>	-0.77*	3	-5
Subcanopy succulents	-0.58	3	-6
<i>Euphorbia mauritanica</i>	+0.49	1	+3

<sup>A</sup>Canopy cover was averaged across all sites as an indication of the general abundance of each species.

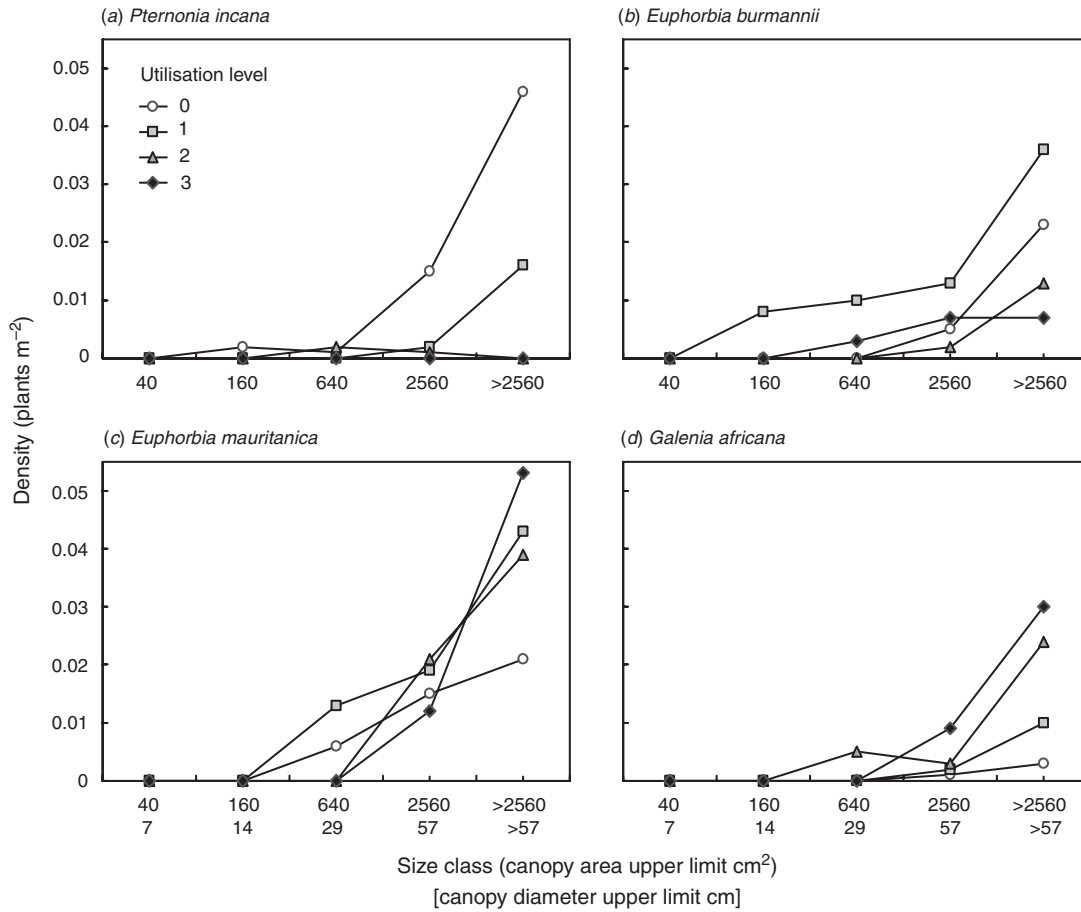
<sup>B</sup>The change in % canopy cover from the least utilised to the most utilised sites indicates the magnitude of response for each species.

In terms of canopy cover data, responses of off-mound vegetation to grazing were less clear, and became apparent only under the heaviest level of utilisation (Fig. 4). The correlation of utilisation with the first ordination axis score was weak, and although the correlation with the second axis was slightly stronger, this correlation was not significant (Table 2). Responses of individual species in off-mound vegetation were generally weak and masked by the dominant unpalatable matrix of *P. paniculata* and *R. caroli* (together accounting for 70% of the vegetation cover), which declined only under very heavy utilisation (Fig. 5, Table 2). Indicators of grazing responses were confined to species with low abundances. *Tetragonia hirsuta* L.f., the most widely distributed palatable plant, provided the only statistically significant indicator of utilisation level in off-mound vegetation (Fig. 5, Table 2). Small succulents [e.g. *Adromischus filicaulis* (Eckl. & Zeyh.) C.A. Sm., *Crassula* spp., and *Senecio radicans* (Dinter) Dinter & Schwantzi.] that grow under the canopies of woody dwarf shrubs (e.g. *P. paniculata*) showed a weak negative correlation with utilisation, and the slight increase in *E. mauritanica* with increasing utilisation showed an even weaker relationship.

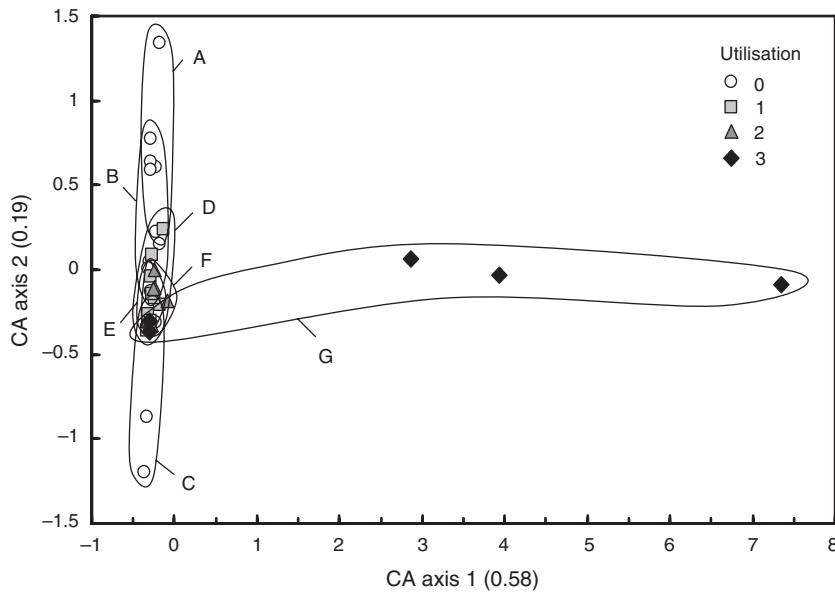
Data on plant sizes, however, revealed that population dynamics of off-mound species had been affected by long-term



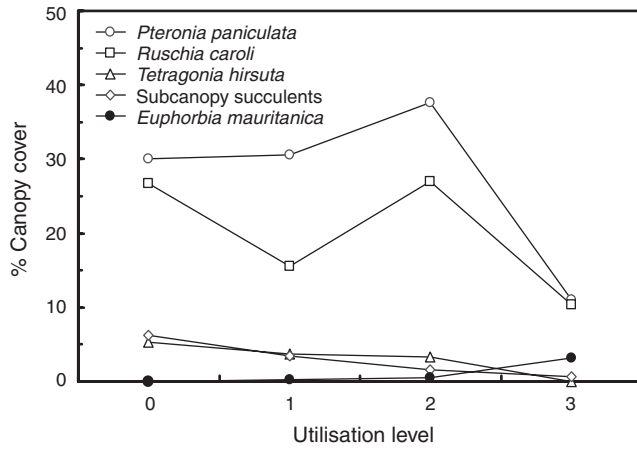
**Fig. 2.** Responses of dominant mound species to increasing levels of grazing utilisation. Utilisation categories (0, protected–3, very heavily grazed) follow Table 1. Data from sites within the same utilisation category have been averaged together.



**Fig. 3.** Influence of grazing utilisation on size distributions of dominant mound species. Utilisation categories (0, protected–3, very heavily grazed) follow Table 1. Data from sites within the same utilisation category have been averaged together.



**Fig. 4.** Correspondence analysis (ordination) of surveys of plant canopy cover for off-mound vegetation. Eigen values are presented with their respective axis. Labels for sites and grazing utilisation categories (0, protected–3, very heavily grazed) follow Table 1.



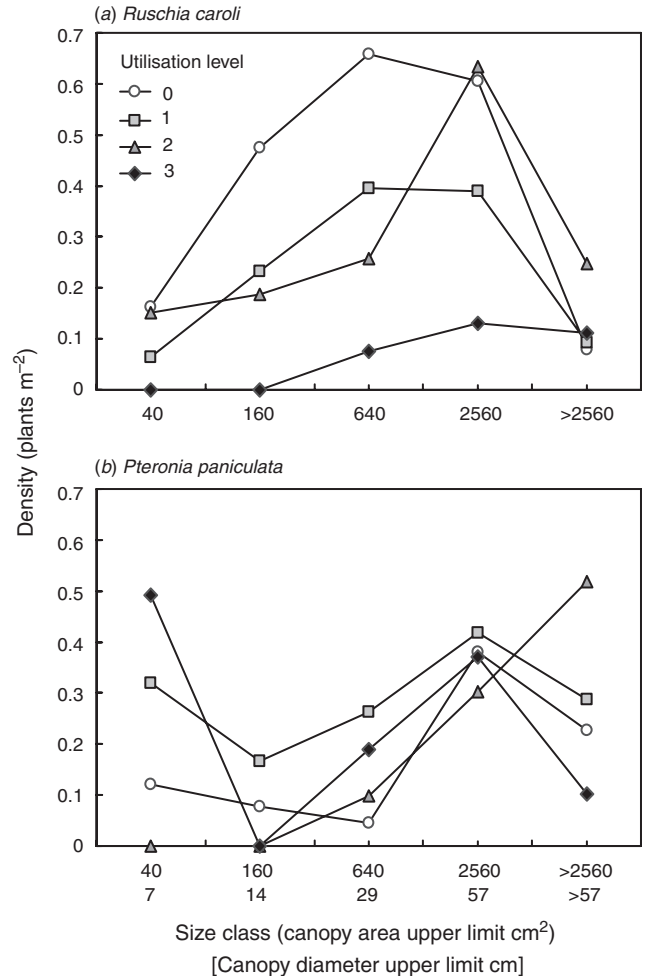
**Fig. 5.** Responses of potential indicator species in off-mound vegetation to increasing levels of grazing utilisation. Utilisation categories (0, protected–3, very heavily grazed) follow Table 1. Data from sites within the same utilisation category have been averaged together.

grazing. There were qualitative differences in size distributions of *R. caroli* plants across utilisation levels (Fig. 6a), and quantitatively these responses were evident as changes in plant size and density (Table 3). With increasing utilisation there was a significant decline in the density of *R. caroli* plants (Table 3). However, this was offset by an increase in plant sizes (Table 3) so that changes in percent canopy cover were small and not statistically significant (Table 2). In *P. paniculata* there was no obvious qualitative trend in plant size distributions across grazing utilisation levels (Fig. 6b). Neither sizes nor densities of *P. paniculata* plants showed significant correlations with utilisation level (Table 3).

**Discussion**

We set out to test the proposition that those landscape elements where initiating processes of degradation are most strongly expressed, here termed ‘indicator patches’, can provide a useful means of focusing sampling, thereby improving the effectiveness of long-term environmental monitoring in providing early forewarnings of degradation. Indicator patches can be distinguished: (i) by harnessing an existing understanding of where and how degradation processes are initiated and propagate in landscapes, and (ii) by empirically assessing disturbance responses of different landscape elements to identify those patch types where responses to disturbance are most pronounced (as we have tested in this case study example). The results from our case study show clearly that vegetation on zoogenic ‘heuweltjie’ mounds in the Succulent Karoo provides a much more sensitive indicator of grazing effects than the dominant surrounding matrix vegetation, demonstrating the utility of the ‘indicator patch’ concept.

On ‘heuweltjie’ mounds – our proposed indicator patches – measurements of percentage canopy cover alone were sufficient to show strong responses to grazing. A clear grazing gradient was apparent, with well defined changes in abundance and replacement of dominant species in response to increasing grazing pressure represented by different decades-long grazing



**Fig. 6.** Influence of grazing utilisation on size distributions of dominant off-mound species. Utilisation categories (0, protected–3, very heavily grazed) follow Table 1. Data from sites within the same utilisation category have been averaged together.

**Table 3.** Correlations of site utilisation scores with plant densities and sizes for the dominant matrix species in off-mound vegetation

All measurements were first combined into an average for each site,  $n = 7$ , Spearman’s rank correlation coefficient; \* $P < 0.05$

Species attribute	Correlation with utilisation
<i>Ruschia caroli</i>	
Density	-0.84*
Size	+0.95*
<i>Pteronia paniculata</i>	
Density	+0.32
Size	-0.3

management histories. The aggregate index of species canopy covers represented by the first ordination axis score clearly discriminated grazing-induced vegetation changes across a wide range of utilisation levels. This effect is likely to be due to both the

intrinsic low tolerance of mound-associated plant species to grazing and the foraging behaviour of sheep that selectively graze mound vegetation (Armstrong and Siegfried 1990). Off-mound plants, though, may be less preferred or more resistant to grazing, and measurements of percentage canopy cover alone were insufficient to distinguish the progression of grazing impacts.

Nonetheless, our more intensive measurements of plant size distributions (rather than canopy cover measurements alone) were able to detect subtle grazing-induced changes in off-mound vegetation that might influence productivity of grazable plants, ecological function, biodiversity and animal habitat [showing the value of this additional information (Stokes and Yeaton 1994)]. One relatively rare species, *T. hirsuta*, was responsive to grazing, but its rarity makes it difficult to monitor directly. Changes in the size and density of dominant *R. caroli* plants would be similarly costly to monitor, although these changes represent an ecologically important process. This unpalatable, prostrate, creeping mesemb fragments and propagates vegetatively as the mesemb hummock is colonised by dwarf shrubs in a similar manner to mesembs elsewhere in the Succulent Karoo that colonise bare soil, trap and accumulate litter, and act as nurse plants to woody shrubs (Yeaton and Esler 1990; Milton 1992; Esler and Cowling 1993). The progressive increase in *R. caroli* size and decrease in plant density with increasing grazing pressure imply: reduced colonisation of *R. caroli* hummocks by other species, increased mortality/reduced recruitment of young *R. caroli* plants (reduced establishment), and/or competitive release of surviving *R. caroli* plants, all of which suggest a progressive disruption of plant recruitment processes with increased grazing pressure. However, because increases in plant size compensated for the decline in plant density at higher utilisation levels, percentage canopy cover measurements (alone) of *R. caroli* remained a poor indicator of grazing impacts in off-mound vegetation over the decades-long period represented by our utilisation levels (Tables 2 and 3). Thus, although our intensive, time-consuming vegetation measurements were able to detect subtle effects of grazing in off-mound vegetation, it would be difficult to detect these changes with extensive, rapid survey methods (based on measures of species abundance alone) that are typically used in rangeland monitoring in the region. Over a longer term, the consequences of these changes could become more profound, manifesting in forms such as reduced plant cover (particularly for rare, palatable species), declining land productivity, losses of biodiversity, and soil erosion. Detecting the early signs of ecological dysfunction, which were difficult to measure off-mound but clearly evident on mounds, would allow timely management interventions to be taken before changes became widespread, irreversible or too costly to remediate.

Thus, although grazing impacts were more easily detectable on 'heuweltjie' mounds, they were not confined to there, and changes in our mound indicator patches reflected landscape-wide grazing effects. Our assessment across utilisation levels shows that future monitoring efforts to evaluate grazing management could focus on mound patches to provide a sensitive, rapid assessment of the early development of grazing-induced changes in these landscapes.

Broadly defined, we propose that the 'indicator patch' approach should have widespread relevance and application in monitoring rangelands and other environments. Heterogeneity

in biological, chemical or physical characteristics of soils and landforms produce patchiness in vegetation, degradation and recovery processes, and sensitivity to disturbance in many landscapes (Cox and Gakahu 1987; Friedel *et al.* 1993; McAuliffe 1994; Stroh *et al.* 2001; Ludwig *et al.* 2005; Bestelmeyer *et al.* 2006b). For example, Bestelmeyer *et al.* (2006c) showed that different patch types within a Chihuahuan Desert grassland in North America would be likely to respond to degradation processes at different rates, mediated by differences in soil aggregate stability associated with bare patch size and connectivity. Numerous other landscape features and processes could provide a basis for selecting indicator patches (e.g. as listed in the 'Introduction'). Such understanding of spatial degradation and recovery processes could be more fully harnessed to improve monitoring programs if it were explicitly incorporated into initial planning of sampling designs and analytical approaches.

In considering how indicator patches could be routinely incorporated into monitoring approaches, it is first necessary to recognise what types of landscape heterogeneity could be exploited. The patterns of landscape patchiness that could serve as indicator patches will depend on the scale of sample units to be used for individual measurements and the scale of the sites (true replicates) being measured. Patches would need to be larger in size than sample units and, ideally, should still be small enough for multiple sampling patches to be distributed across each site. Indicator patches would not need to be as discrete as the 'heuweltjie' mounds in our example. Continuous patterns of heterogeneity could be exploited so long as indicator patches could be defined in such a way that they could be repeatedly identified (e.g. defining a band within a continuous catena gradient). The approach can accommodate a wide range of scales from fine scale point or plant measurements to larger patches via remote sensing, and is therefore capable of exploiting the hierarchical nature of patchiness in landscapes (Klijn and Dehaes 1994; Bestelmeyer *et al.* 2006a). In addition, (i) there should be a demonstrated link between degradation in indicator patches and synchronous or future changes more broadly in the (unsampled portions of the) landscape, and (ii) measurements from indicator patches should be able to distinguish a broad range of sequential changes in land conditions, rather than being so sensitive to disturbance that measurements cease to continue providing useful information on the progress of degradation after the initial stages (e.g. if water points or other 'sacrifice areas' were used as indicator patches). Furthermore, it is important to recognise that spatial sampling designs are but one of many considerations for planning monitoring (e.g. Novelly *et al.* 2008), and there may be competing reasons to target sampling on landscape elements that are not necessarily the most sensitive indicators of disturbance. Our main concern is that the potential from the extensive knowledge base of spatial degradation/recovery processes has not yet been fully harnessed because such considerations have not routinely received the explicit attention they deserve in designing monitoring systems.

There is a growing demand for sensitive monitoring procedures that can demonstrate the effectiveness of land management interventions in maintaining and improving land condition. This is coming from two quarters. First, land managers are increasingly being held accountable for their environmental stewardship in maintaining the condition of natural resources



(e.g. for renewal of leases on leasehold properties). Second, community groups such as regional natural resource management bodies and Landcare groups are increasingly being asked to account for publically funded on-ground activities by demonstrating improvements in land condition. In both cases, routinely incorporating 'indicator patch' considerations into monitoring designs could improve their sensitivity, (i) allowing for more rapid detection of positive trends in land condition, (ii) increasing the robustness of monitoring in being able to demonstrate that land condition is not deteriorating, and (iii) providing earlier forewarnings in cases where remedial intervention is required.

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