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PARTICIPATORY INDICATOR DEVELOPMENT: WHAT CAN ECOLOGISTS AND LOCAL COMMUNITIES LEARN FROM EACH OTHER?

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Abstract. Given the growing popularity of indicators among policy-makers to measure progress toward conservation and sustainability goals, there is an urgent need to develop indicators that can be used accurately by both specialists and nonspecialists, drawing from the knowledge possessed by each group. This paper uses a case study from the Kalahari, Botswana to show how participatory and ecological methods can be combined to develop robust indicators that are accessible to a range of users to monitor and enhance the sustainability of land management. First, potential environmental sustainability indicators were elicited from pastoralists in three study sites. This knowledge was then evaluated by pastoralists, before being tested empirically using ecological and soil-based techniques. Despite the wealth of local knowledge about indicators, this knowledge was thinly spread. The knowledge was more holistic than published indicator lists for monitoring rangelands, encompassing vegetation, soil, livestock, wild animal, and socioeconomic indicators. Pastoralist preferences for vegetation and livestock indicators match recent shifts in ecological theory suggesting that livestock populations reach equilibrium with key forage resources in semiarid environments. Although most indicators suggested by pastoralists were validated through empirical work (e.g., decreased grass cover and soil organic matter content, and increased abundance of *Acacia mellifera* and thatching grass), they were not always sufficiently accurate or reliable for objective degradation assessment, showing that local knowledge cannot be accepted unquestioningly. We suggest that, by combining participatory and ecological approaches, it is possible to derive more accurate and relevant indicators than either approach could achieve alone.

Key words: Botswana; Kalahari; land degradation; local knowledge; participation; southern Africa; sustainability indicators.

Introduction

Environmental sustainability indicators have been embraced by researchers and policy-makers at local, national, and international scales to monitor progress toward conservation goals (UNCED 1992, UNCCD 1994, Bell and Morse 1999, 2004), and there is growing evidence that local communities can successfully contribute to the identification, evaluation, and selection of relevant indicators (e.g., Fraser et al. 2006, Reed et al. 2006). Indeed, there is evidence of sophisticated monitoring systems based on indicators, developed and applied by local communities around the world to monitor the status of resources (e.g., Berkes and Folke 1998, Berkes et al. 2000). However, despite the recognition that sustainability and conservation goals can only be met with active participation from local communities, the majority of indicators are still developed by academic researchers and/or policy-makers. While often accurate, these indicators are rarely accessible, meaningful, or useful to people who manage the land, who often lack time, money, and specialist training or equipment. For this reason, the results of sustainability monitoring are criticized for being rarely used by local communities to enhance the sustainability of land management (Innes and Booher 1999, Carruthers and Tinning 2003).

If indicators are to influence land management, then they must be easy for local communities to use, in addition to providing accurate assessments of environmental sustainability. There is a growing body of literature suggesting that a combination of local and scientific ecological knowledge may empower local communities to monitor and manage environmental change easily and accurately (e.g., Folke et al. 2002, Thomas and Twyman 2004, Fraser et al. 2006). The goal is to develop an approach to conservation that combines rigor and accuracy with relevance and sensitivity to local perspectives and context, and for conservationists and communities to work together toward shared aims. However, achieving this goal is beset with difficulties. Conservation is no longer the sole domain of ecologists, who must now work with the unfamiliar vocabulary and epistemology of many different disciplines, notably from the social sciences. There are concerns that integrating

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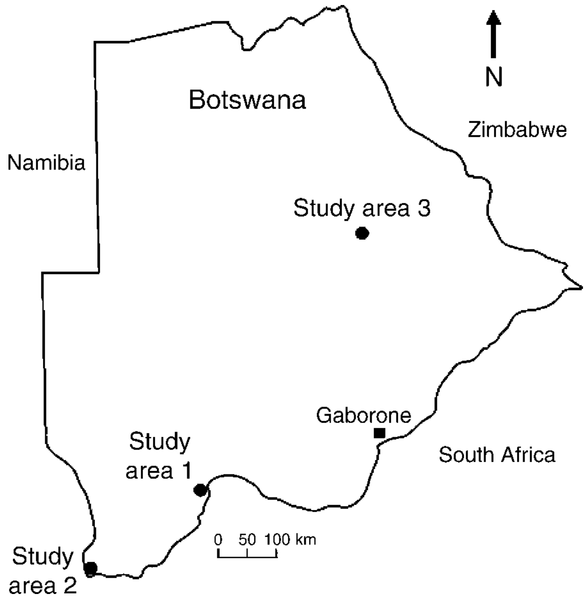

FIG. 1. Map of Botswana showing location of study areas where case study research was conducted. For a color version showing degradation status according to the combined opinions of eight national experts, see Reed et al. (2007).

local and scientific knowledge will inevitably sacrifice rigor and objectivity (Abbot and Guijt 1997). However, there have been few attempts to investigate these claims, and despite the rhetoric, there are few tools available to meaningfully integrate these different knowledge bases.

Developing methods to assess environmental sustainability is particularly important in the semiarid rangelands of sub-Saharan Africa that support the livelihoods of over 25 million pastoralists (Lane 1998), where inappropriate management has been blamed for widespread land degradation (Stiles 1995, Eswaran et al. 2001). Land degradation is one of the world's most pressing environmental problems, and has been described as "an assault on sustainability" (Warren 2002:454). Botswana, in particular, has been described as "one of the most desertified countries in sub-Saharan Africa" (Barrow 1991:191), and there is evidence that the privatization of communal rangeland is further worsening land degradation and deepening already stark social and economic inequalities (e.g., Thomas et al. 2000, Rohde et al. 2006).

This paper therefore aims to develop environmental sustainability indicators for semiarid rangelands in the Kalahari, Botswana that can be used by specialists and nonspecialists alike, by integrating local knowledge with ecological data. To do this, the paper addresses the following research objectives, to:

1) Develop methods that can effectively integrate local and scientific ecological knowledge about environmental sustainability indicators;

2) Identify environmental sustainability indicators from local pastoralists in semiarid Botswana and the literature;

3) Evaluate this indicator knowledge qualitatively in village-level focus groups and quantitatively using ecological and soil-based sampling; and

4) Test the assumption that it is not possible to achieve both meaningful participation and scientific rigor when these different knowledge bases are integrated.

METHODS

Study area selection

To identify and evaluate environmental sustainability indicators, study areas were selected to represent areas within Botswana that were widely perceived to be suffering from significant land degradation. These land degradation "hotspots," identified qualitatively through interviews with an international panel of advisors (Reed et al. 2007), provided a wide choice of potential study areas. To test the transferability of the approach developed, three study areas were selected to represent different biophysical (rainfall, soil, and vegetation type) and cultural settings (Fig. 1, Table 1). Interviews, focus groups, and field data were held and collected between 2001 and 2003, with subsequent production and dissemination of rangeland assessment guides in each area.

Participatory identification and evaluation of indicators

A two-step methodological approach was taken to integrate local and scientific ecological knowledge about environmental sustainability indicators at each of the three study sites. First, indicators were identified from local pastoralists and the literature, and evaluated qualitatively by pastoralists against criteria that they developed. Second, the indicators that emerged from this process were evaluated quantitatively using ecological and soil-based methods.

An essential first step in participatory work on indicators is to find terms for the words "indicator" and "sustainability" that local people understand (Abbot and Guijt 1997). In this study, the word "sign" (and its equivalent in local languages) was found to be the most appropriate. In the absence of consensus over a precise operational definition and meaning of "sustainability" (e.g., Jacobs 1995, Pezzey 1997, Weersink et al. 2002), eliciting sustainability indicators from local communities is problematic. This may explain the lack of public participation in the development of many published farm-level sustainability indicators developed in other regions (e.g., Taylor et al. 1993, Gomez et al. 1996, Rigby et al. 2001). In contrast, definitions of land degradation are better established (Abel and Blaikie 1989, UNEP 1997), and eliciting degradation indicators from communities is easier (Stocking and Murnaghan 2001). As the antithesis of sustainability (Warren 2002), degradation indicators elicited from communities can be reversed to derive sustainability indicators. In this way, it was possible to better elicit sustainability indicators from pastoralists.

TABLE 1. Summary of livelihood information collected in each study area.

Livelihood asset	Study area: Area 1	Study area: Area 2	Study area: Area 3
Natural capital			
Use communal rangeland	73%	85%	90%
Own rangeland (fenced)	27% (19%)	15% (<1%)	10% (10%)
Rent or own arable land	none	53% (growing maize, beans, sorghum, watermelon, and/or pumpkin on small scale)	none
Average no. cattle	165 ± 492	34 ± 52	20 ± 43
Average no. goats	70 ± 77	17 ± 26	66 ± 87
Average no. sheep	42 ± 95	2 ± 9	88 ± 201
Main rangeland products used	firewood, fruit, hunting, medicine, vegetables, gums	building poles, thatching grass, firewood, fruits, vegetables, medicine, Mopane worms and hunting	firewood, building materials, fruit, vegetables, and medicine
Livelihood significantly constrained by natural capital	52%	23%	39%
Social capital			
Members of farming groups	41%	25%	39%
Access to farming magazines, TV, or radio programs	85%	42%	45%
Frequent, some or no contact with agricultural extension (in the order Site 1, 2, and 3)	58%, 32%, and 10%	32%, 35%, and 32%	21%, 58%, and 21%
Livelihood significantly constrained by social capital	15%	45%	24%
Physical capital			
Access to motor vehicle	89%	23%	31%
Access to donkey cart	53%	36%	55%
Sole or family owners of borehole or well	39%	37%	24%
Distance to sell livestock	mainly local, or ~200 km to BMC abattoir on tar road	mainly local, or ~200 km to BMC abattoir on tar road via quarantine	mainly local, or ~400 km to BMC abattoir, half on sand road
Distance to buy supplies	local	local	local
Access to telephone	60%	63%	31%
Livelihood significantly constrained by physical capital	21%	53%	38%
Human capital			
Average no. of family laborers	0.8 ± 1.4	2.5 ± 2.3	4.9 ± 4.5
Average no. of paid laborers	4.5 ± 7.3	0.7 ± 1.1	1.1 ± 1.5
Average years in formal education	6.4 ± 5.9	5.6 ± 3.9	6.2 ± 4.5
Main sources of informal education	parents, extension workers, South African farmers they worked for	parents, extension worker through Farmers Association	parents, South African farmers they worked for, training courses run by extension service
Constrained by health	16%	17%	40%
Livelihood significantly constrained by human capital	32%	25%	35%
Financial capital			
Main income sources	livestock, government welfare support, small business	government welfare support, diamond mines, livestock, small business	livestock, government welfare support
Access to savings	90%	50%	29%
Access to credit	94%	44%	35%
Livelihood significantly constrained by financial capital	38%	65%	43%

Notes: Percentages are based on a total of 67 respondents in Study Area l, 40 respondents in Area 2, and 53 respondents in Area 3. Data including error terms are means ± SD. BMC = Botswana Meat Commission.

Indicators were identified during semistructured interviews. Respondents were asked to identify signs they would see in rangeland that had declined in productivity over the long term, due to inappropriate use, as opposed to drought. Where necessary, prompts were used to elicit indicators representing different ecosystem components (e.g., soils, plants, and animals). Respondents were then asked to identify which of these signs they would expect to appear first, which might provide an early warning that detrimental change was likely in the future. (See Reed and Dougill [2002] for preliminary results from this part of the analysis for Study Area 1.) Longer-term historic environmental changes were also explored qualitatively through a series of oral histories in each site, focusing, where possible, on changes in potential indicators.

During these interviews, respondents were asked to identify characteristics of useful indicators that could be used as selection criteria, and "accuracy" and "ease of use" were considered most important. Local participation in the development of evaluation criteria is essential in selecting appropriate indicators (Krugmann 1996). Indicators identified in interviews were also compared and combined, where relevant, with indicators from the literature that had been developed in comparable semiarid rangeland areas. All these possible indicators were then evaluated against community-derived criteria in at least three village-level focus groups in each study area, using Multi-Criteria Evaluation (MCE) approaches (Banville et al. 1998). For this, all indicators were printed on cards in local languages and supported with images. Indicators were then ranked against the two criteria (accuracy and ease of use) by assigning stones, as counters, to cards (the two criteria equally weighted). Participants were given the same number of stones as there were indicator cards, and they placed stones on cards to rank the *accuracy* of each indicator and then the *ease* with which each indicator could be used. Finally, the results of this exercise were discussed with participants, to elicit reasons for differently ranked indicators. Information about early-warning indicators was also checked in these focus groups by asking participants to select and rank indicators that could provide an early warning that detrimental change was imminent. This was essentially a qualitative exercise, using ranks to initiate group discussions about the accuracy and ease with which different indicators could be used.

Empirical evaluation and selection of indicators

To test the assumption that it is not possible to develop indicators that can achieve both meaningful participation and scientific rigor, land degradation indicators, perceived to be accurate and easy to use by at least two out of three focus groups in each site, were tested empirically using ecological and soil-based sampling techniques. Pastoralists were involved in the collection of ecological data in each study area, and provided expert assistance with species identification, local plant names, and uses (cf. Huntington 2000, Oba and Kotile 2001, Mapinduzi et al. 2003), as there are many benefits to community involvement in ecological fieldwork. Notably, it was possible to collect valuable ethnobotanical data, including the palatability of certain plants for specific livestock species.

Indicators were measured along grazing gradients of distance away from waterpoints as a surrogate for degradation, based on the assumption, corroborated through interviews, that rangeland degradation was primarily grazing induced. The grazing gradient or "piosphere" approach (Andrew 1988) assumes that grazing intensity declines exponentially with distance from a fixed water point, with the most distant, underutilized areas acting as a pseudo-control, representing nondegraded rangeland. Although the decline is rarely spatially uniform, it is possible to use dung and cattle track frequency to corroborate assumed changes in grazing intensity, and this approach has been widely used in semiarid rangelands with point water sources (e.g., Hardy and Hurt 1989, Perkins and Thomas 1993, Jeltsch et al. 1996, Pickup et al. 1998, Thomas et al. 2000, van der Westhuizen et al. 2005).

Sample sites were located at exponentially increasing distances from each borehole (water source), starting at 200 m, to reflect the decrease in grazing intensity. Measurements along grazing gradients rarely extend beyond 3 km from water points in ecological literature. However, encroachment by thorny bushes (of very limited forage value to cattle) extended beyond this distance in some of the areas selected for this research. Therefore, to ensure that all study areas included the full utilization gradient, local knowledge was used to stratify vegetation around each water point. The distance of each gradient was then determined by the extent of ecological zones around each water point, ranging from 3.2 km in Study Area 2 to 12.8 km in Study Area 1. To do this, we used a participatory mapping approach (cf. Rocheleau 1995, Roth 2007), asking communities to draw maps of ecological assemblages and resource use patterns in the sand. These were transcribed to paper, checked and amended with key informants, using a combination of aerial photography and/or remotely sensed imagery, with Global Positioning System readings taken on rangeland drives (cf. Suyanto et al. 2004). Fig. 2 shows an example of land use and vegetation maps developed in Study Area 3.

Boreholes with clear grazing gradients were selected for sampling, avoiding areas where gradients were interrupted by fences or overlapped with gradients from other boreholes. In Study Area 2 this was problematic, as numerous unauthorized wells had been dug between boreholes, creating an overlapping mosaic of grazing gradients. However, there were clear grazing gradients radiating from villages, and so these settlements were used in place of water sources as the origin of degradation gradients. The approach had to be modified in Study Area 3 because of the length of the grazing gradients (up to 18 km) and logistical difficulties of following gradients across fields of parallel dunes. In this case, discrete areas with contrasting grazing intensities were defined from participatory mapping. Areas distant from water sources that were rarely used by livestock were used as nondegraded sites (13–18 km from water), sample sites close to water sources (0.75–1.5 km) were used to represent degraded land, and a number of sample sites (6–8 km from water) were located in between these areas to represent land in intermediate condition. This approach reduced the number of sites that could be sampled compared to the other areas (14 sites in Study Area 3 compared to 44 and 32 in Study Areas 1 and 2, respectively).

At each sampling point, each of the indicators suggested by local pastoralists were quantified along 30-m line intercepts (for tree-based indicators) and 5-m line intercepts (for all other indicators). The line intercept method is well suited to sampling sparse dryland vegetation communities (Kent and Coker 1996). At all sites, as many as possible of the indicators perceived to be accurate and easy to use by the communities were tested empirically. Trees and bushes were identified, and the length of their canopies measured, to derive percentage cover. The height of all trees intercepted was also measured. Dead trees were also identified. Height of exposed roots and nebkha dunes (around the base of bushes) were measured. Ground layer species were measured along three 5-m intercepts located at the ends and middle of each 30-m intercept. Evidence of harvester termite (Isoptera: Hodotermitidae) activity was identified from grass tillers that had been cut cleanly at variable heights. Soil measurements were also made at the center of each 5-m intercept (three samples per 30-m intercept). Soil was classified as consolidated or unconsolidated. The number of cattle tracks crossing 30-m intercepts, and the amount of dung (presence/absence at 1-m intervals) was counted to estimate grazing intensity, following the method of Perkins and Thomas (1993).

A number of additional measurements were made to evaluate indicators that pastoralists had only considered relevant for certain study areas. For example, in Study Area 1, the condition of cattle encountered at each intercept was assessed by using a subjective scale based on the prominence of ribs and shoulders, and coat condition (cf. Krugmann 1996); however, the low density of cattle resulted in little data from this approach. As an alternative approach, livestock populations from each borehole ("crush data") were analyzed to look for evidence of long-term declines in herd size that could indicate land degradation. Tree girth and presence of flowers and fruit were also noted in Study Area 1. Insect specimens were also collected in Study Area 1 using pitfall traps located along a grazing gradient, to test ant indicators suggested by pastoralists (Jew 2005). Insect sampling was undertaken in: (1) a sacrifice zone adjacent to a water point; (2) a bush-encroached zone, dominated by *A. mellifera* and *Grewia flava* bushes; (3) a grazing reserve dominated by palatable perennial grasses; and (4) an intermediate zone between the grazing reserve and bush-encroached zone. At each sample location, 49 traps were set in a 35 $\times$ 35 m grid and left for 72 hours prior to collection, count, and identification of all insects in the pitfall traps.

In Study Area 2, 10 leaves were picked from each intercepted *Colophospermum mopane* tree, and checked for edible insect cases. Rocks (>5 cm) and plant litter were also measured along three 5-m intercepts located at the ends and middle of each 30-m intercept. Presence of edible fruits on all plants was noted. Evidence of a white crust or crystals formed by salt was assessed at each soil sample point. Evidence of diarrhea was noted during dung measurements. Due to the flat topography, visibility at each 30-m intercept was determined by measuring the distance to the point where a person disappeared from sight, which was one of the indicators suggested by pastoralists in this area. In Study Areas 2 and 3, 10-g samples were taken from the soil surface at each sample point (Study Area 2, $n = 31$; Study Area 3, $n = 25$) excluding leaf litter where present, and mixed to obtain a representative sample for each 30-m intercept. These were tested for organic carbon and conductivity, using laboratory procedures (as outlined by Anderson and Ingram [1993]) at the Botswana Government Soils Laboratory.

In Study Area 3, distance from each intercept to the crest of the nearest two sand dunes was measured, and the height of each sand dune was measured. At each measurement site, a 30-m line intercept was made at both dune valley and dune crest locations, as each have distinct soil and vegetation types. Availability of firewood along each intercept was also noted at Study Area 3 sites.

Statistical analyses

To determine the nature of grazing-induced environmental change, linear regression analysis was used to examine the relationships between environmental variables, selected to represent proposed degradation indicators, and distance from water points (Study Area 1) and villages (Study Area 2). In Study Area 3, independent t tests were used to determine if there were significant differences between indicator values in different degradation zones.

To quantify the composition of vegetation samples, an indirect ordination was performed on the floristic data from each site using Detrended Correspondence Analysis (DCA) with the DECORANA program (Hill 1979, Hill and Gauch 1980). DCA arranges line intercepts in a multidimensional ordination space according to their floristic differences. Vegetation samples that are floristically similar are grouped together, and floristically dissimilar samples are separated in the ordination space along multiple axes. DCA has been used elsewhere to evaluate local ecological

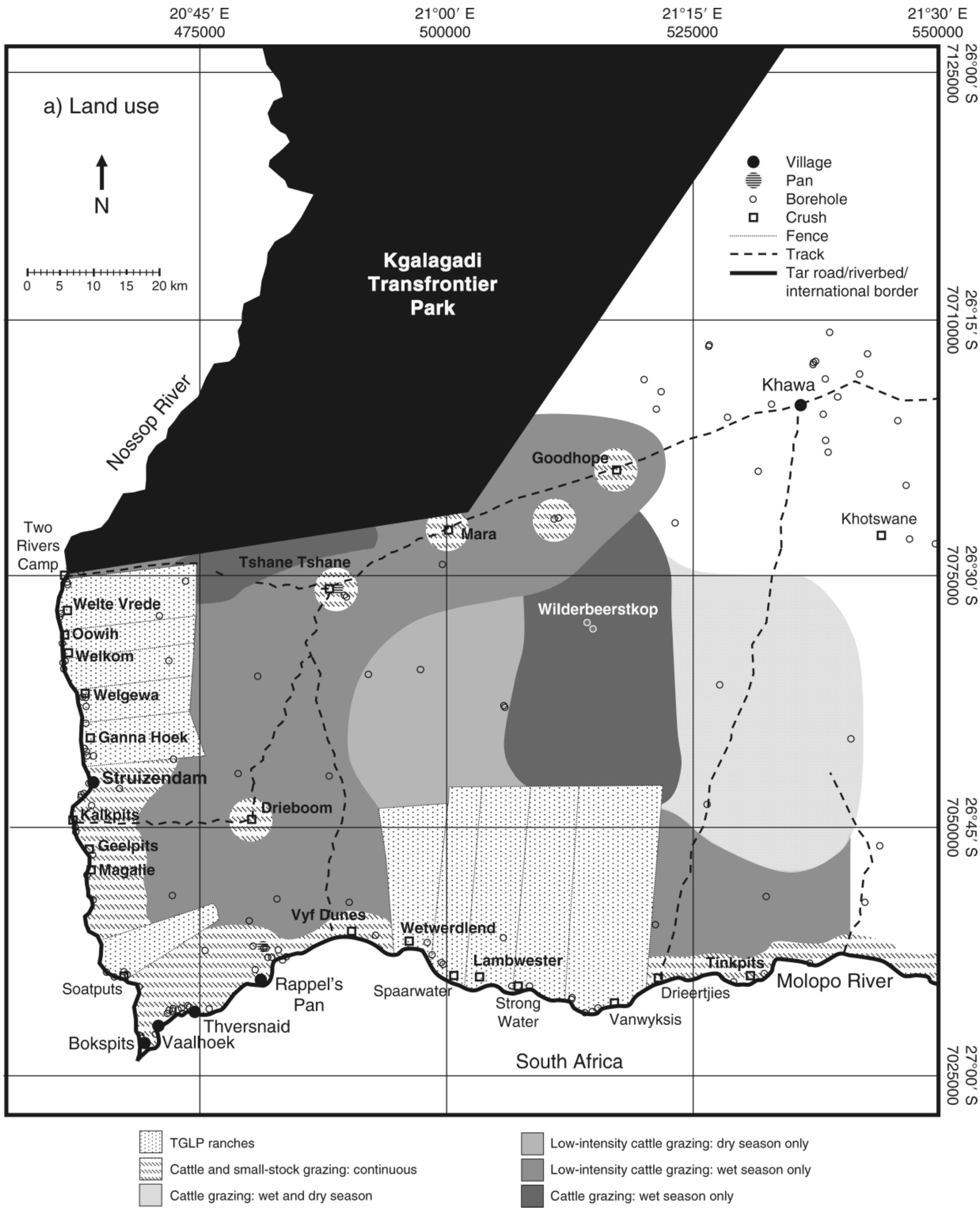

FIG. 2. Study Area 3: (a) land use and (b) ecological zones, developed through a participatory mapping exercise.

knowledge; for example, Verlinden and Dayot (2005) showed how an ordination of Namibian vegetation samples correlated with local perceptions of the functional differences between different units of land. Percentage cover data was used as the dependent variable and four ordination axes were calculated. As outliers can significantly affect the results of DCA, these were omitted during the analytical procedure, and rare species were down-weighted. Axis scores from DCA were then used to test the hypothesis that distance from

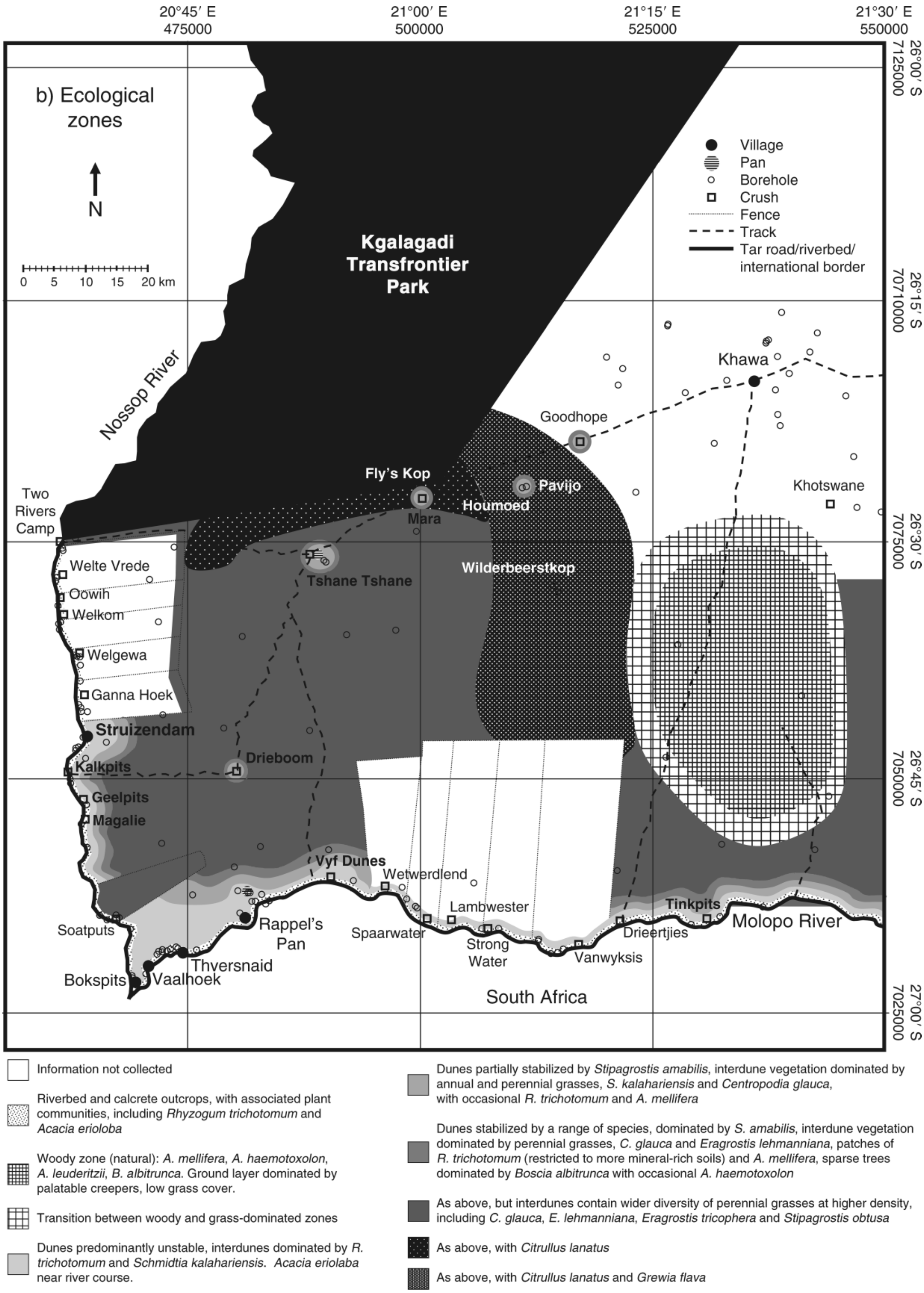

FIG. 2. Continued.

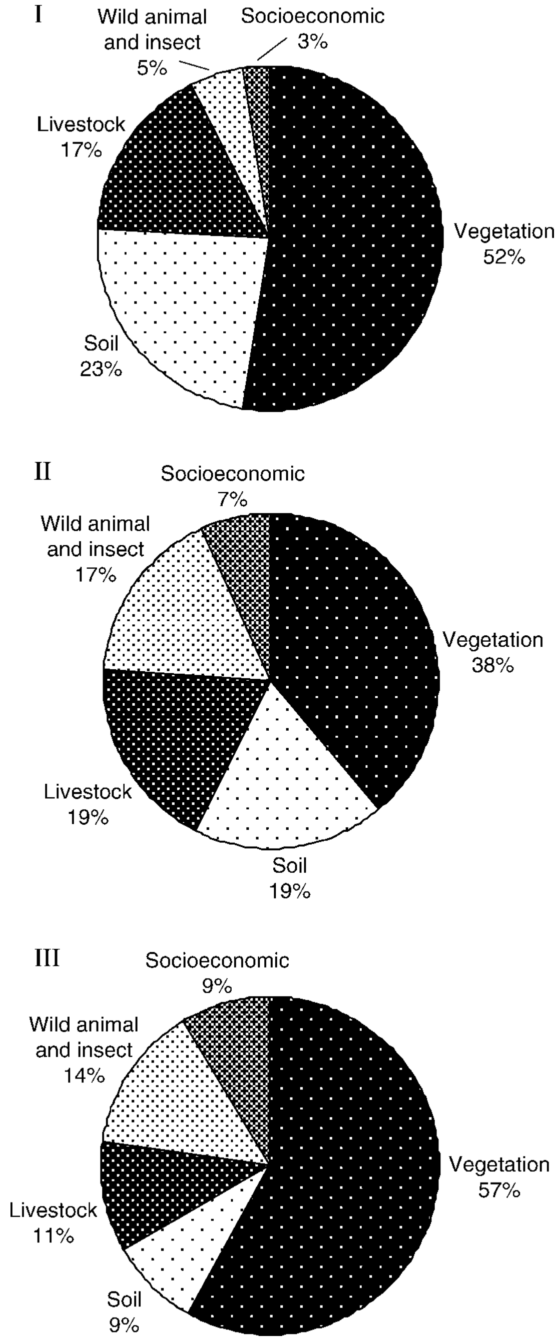

FIG. 3. Percentage of indicators cited at each site (for each category) for vegetation, soil, livestock, wild animal and insect, and socioeconomic factors cited by pastoralists in (I) Study Area 1, (II) Study Area 2, and (III) Study Area 3.

water was the primary factor determining compositional change along grazing gradients, and to validate the degradation zones identified from participatory mapping. Ordination axes were correlated against distance from water (Study Area 1) or village (Study Area 2) using linear regression to determine which, if any, axis represented a degradation gradient. Dung frequency was used to test the validity of degradation zones in Study Area 3. Finally, indicator measurements were correlated against the ordination axes that were related to degradation to determine which indicators accounted for most change in degradation status. Indicator measurements were also correlated against the other ordination axes with high eigenvalues to see if they could account for any of the variation these axes represented. Due to the large number of comparisons being made, $P \leq 0.01$ was used as the statistical significance level.

Following analysis, the results of ecological and soil-based indicator testing were presented to pastoralists for evaluation in village-level focus groups. Discussion focused on indicators for which no empirical evidence could be found, and these discussions informed the final indicator selection.

RESULTS

Indicator identification and evaluation

A significant number of potential land degradation (and by reversal, sustainability) indicators representing a wide range of agro-ecological system components were elicited from local pastoralists. A total of 84, 79, and 64 indicators were elicited in Study Areas 1, 2, and 3, respectively (from 67, 40, and 53 interviews, respectively), with a total of 140 different indicators.

This wealth of local knowledge was thinly spread. Of the 140 indicators suggested, on average, individuals could only describe 6 ± 2.8 indicators (mean $\pm$ SD) (range = 1–18), 9 ± 4.4 indicators (range = 2–18), and 7 ± 3.2 (range = 3–16) indicators in Study Areas 1, 2, and 3, respectively. For this reason, focus groups that shortlisted potential indicators acted as a valuable learning opportunity, providing the pastoralists with collective knowledge beyond that known to any individual. Although certain indicators were cited by many (e.g., grass cover was cited by 67, 35, and 21% of respondents in Study Areas 1, 2, and 3 respectively), there was little overlap between the knowledge of individual pastoralists. In Study Areas 1 and 3, pastoralists were more reliant on vegetation indicators (52 and 57% of those elicited compared to 38% in Study Area 2). People used fewer soil indicators in Study Area 3 (9% compared to 23% and 19% in Study Areas 1 and 2) (Fig. 3).

In Study Area 1, linear regression showed that formal education was a predictor of indicator conceptualization ($P < 0.01$; $r^2 = 0.25$). In addition to knowing more indicators, better-educated respondents cited proportionately fewer vegetation and more wild animal indicators than less educated farmers, who relied more on vegetation and livestock indicators. Tests (t tests) showed that men knew significantly more indicators than women in Study Area 2 (on average 12 and 7 indicators, respectively; $P < 0.01$); however there was no difference in the kind of indicators they knew. In Study Area 3, there was no relationship between indicator conceptualization and any of the factors that were assessed.

TABLE 2. Indicators considered accurate and easy to use by rangeland pastoralists in all study areas showing evidence from the literature and results from empirical testing.

Indicator	Supported by literature?	Study Area 1	Study Area 2	Study Area 3
Decreased grass cover	yes	**	**	**
Increased abundance of grass unpalatable to cattle	yes	**	**	**
Decreased abundance of grass palatable to cattle	yes	**	**	**
Decreased availability of thatching grass	···	†	NS	**
Decreased abundance of medicinal plants	···	‡	‡	‡
Decreased abundance of trees	no	NS	NS	NS
Stunting of trees and bushes	no	NS	NS	NS
Tree canopy die-off	···	**	†	†
Increased abundance of *Boscia albitrunca*	···	**	†	†
Decreased abundance of *Grewia flava*	yes	NS	‡	‡
Increased abundance of *Acacia mellifera*	yes	**	†	**
Decreased vegetation cover/increased bare ground	yes	**	**	**
Decreased soil organic matter content	yes	†	**	**
Increased soil looseness	no	NS	NS	**
Increased density of cattle tracks	···	NS	NS	†

Note: Ellipses (···) in the "Supported by literature?" column denote that there is no literature for the indicator.
** Significant at $P \leq 0.01$; NS, not significant.
† Not short-listed in this site.
‡ Insufficient data.

Using Multi-Criteria Evaluation at focus groups, short-lists were developed of 38, 63, and 42 (out of the original 84, 79, and 64) indicators that were perceived as being both accurate and easy to use by pastoralists in each area. Table 2 lists the indicators that were considered both accurate and easy to use by rangeland pastoralists in all study areas. (Since this selection only shows indicators that overlapped between sites, it is a small fraction of the total indicators.) Short-listed indicators from each site were then tested using ecological and soil-based techniques.

Empirical indicator evaluation

A total of 19, 33, and 26 indicators were evaluated using ecological and soil-sampling methods in Study Areas 1, 2, and 3, respectively. This is equivalent to 49, 53, and 62% of the indicators that were deemed accurate and easy to use by pastoralists. Of these, empirical field evidence was found to support 67, 26, and 60% of indicators in each area, respectively (excluding indicators for which there was insufficient data to draw reliable conclusions). Given the differences in findings for each site, these are now considered for each Study Area in turn.

Empirical evidence for indicators in Study Area 1 (Tsabong-Werda).—Floristic variation in Study Area 1 was determined primarily by a degradation gradient represented by axis 1 of the ordination in Fig. 4a, which correlated significantly with distance from a water point ($P < 0.01$; $r^2 = 0.48$). Intensively grazed sites closer to water sources had lower vegetation cover ($P < 0.01$; $r^2 = 0.35$) and were dominated by more (correlation with axis 1; $P < 0.01$; $r^2 = 0.40$) and taller ($P < 0.01$; $r^2 = 0.29$) encroaching *Acacia mellifera* bushes, which had been suggested as a degradation indicator. Pastoralists also suggested a number of tree-based indicators. However, there was no evidence of a decreased abundance of trees or tree stunting in degraded sites. Although not statistically significant, trees at degraded sites did appear to have less girth at breast height ($P = 0.02$; $r^2 = 0.13$), and there was a weak correlation between the proportion of tree canopies that were dead and degradation along axis 1, as suggested by pastoralists ($P = 0.02$; $r^2 = 0.16$).

The secondary axis in Study Area 1 represented ground layer responses to land degradation (Fig. 4a). It was significantly correlated with a reduction in grass cover ($P < 0.01$; $r^2 = 0.61$) and a shift toward less palatable grass species. These had been suggested as degradation indicators by pastoralists. Degraded sites were characterized by increasing abundance of the unpalatable sedge *Fimbristylis hispidula* ($P = 0.01$; $r^2 = 0.16$) and *Senna italica* (a creeping medicinal plant) ($P < 0.01$; $r^2 = 0.21$), and decreasing abundance of the grasses *Schmidtia pappophoroides* (high grazing value) ($P < 0.01$; $r^2 = 0.40$), *Eragrostis lehmanniana* (intermediate grazing value) ($P < 0.01$; $r^2 = 0.22$) and *Aristida stipitata* (low grazing value) ($P < 0.01$; $r^2 = 0.23$) (palatability according to van Oudtshoorn [1999]). A reduction in the abundance of *S. pappophoroides* and *E. lehmanniana* has been previously associated with rangeland degradation in this region (e.g., Makhabu et al. 2002, Skarpe 2002).

Although soils were significantly more likely to be consolidated under bush canopies than between bushes ($P < 0.01$), there was no evidence for a decrease in soil consolidation along degradation gradients, as proposed in the soil looseness indicator (and associated capacity to use two-wheel-drive vehicles and bicycles). This is consistent with other data collected in Study Area 1 indicating significant soil crusting under *A. mellifera* bushes (Berkeley et al. 2005). There was no evidence that there were fewer wild fruits or flowers in degraded land, or that trees had become stunted, as suggested by pastoralists.

Livestock herd sizes only showed a significant trend over time at one borehole in this study area (Lebubeng),

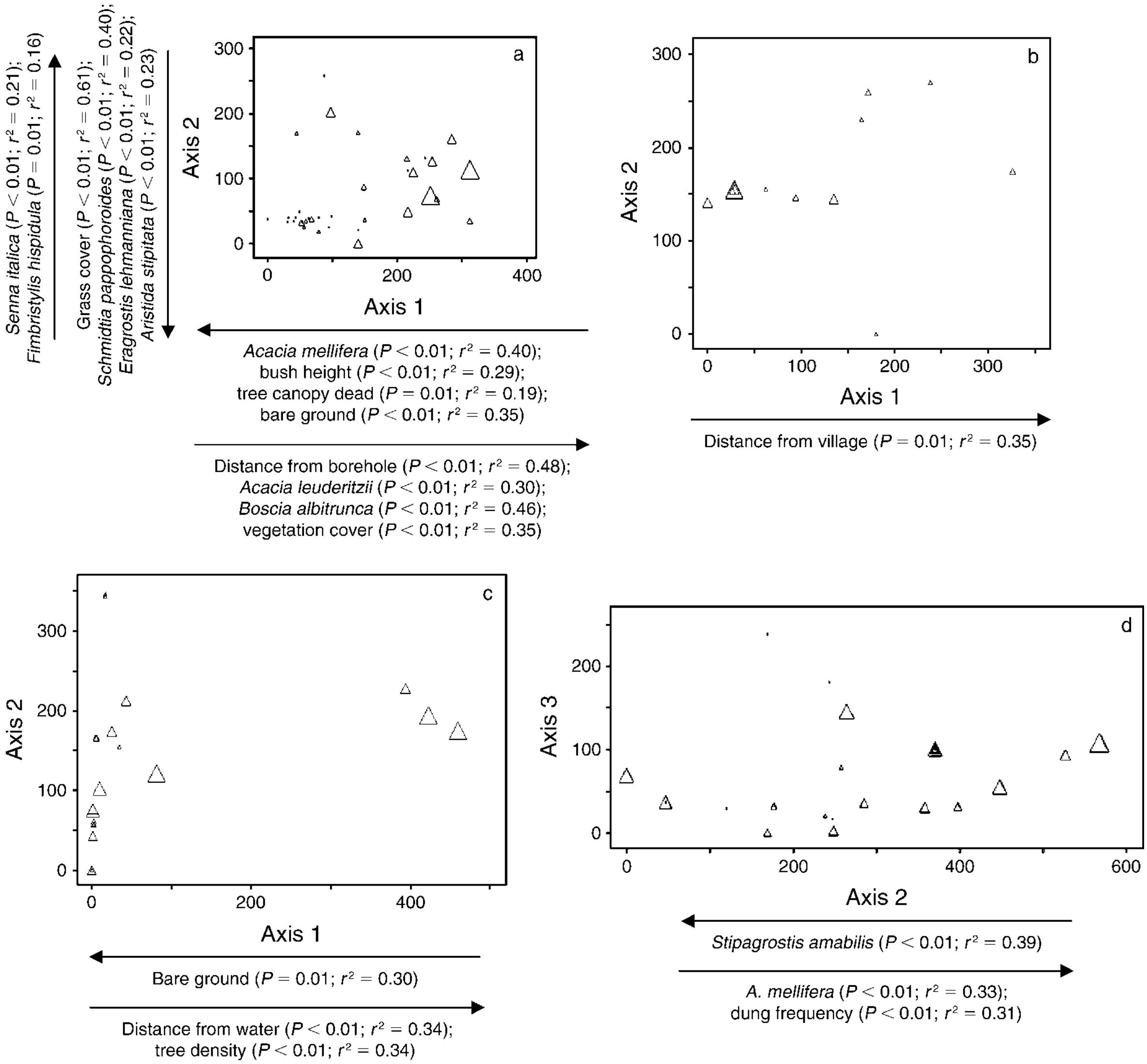

FIG. 4. DCA ordination plots for: (a) all Study Area 1 intercepts (eigenvalues, axis 1 = 0.77; axis 2 = 0.46), with triangle size representing distance from borehole; (b) Study Area 2 grassland intercepts (eigenvalues, axis 1 = 0.88; axis 2 = 0.40), with triangle size representing distance from village; (c) Study Area 2 intercepts in the *C. mopane*-dominated area (eigenvalues, axis 1 = 0.79; axis 2 = 0.62), with triangle size representing distance from water; and (d) all Study Area 3 intercepts (eigenvalues, axis 2 = 0.89; axis 3 = 0.41), with triangle size representing dung frequency.

where Spearman rank correlation tests showed that cattle ($P < 0.01$) and goats ($P < 0.01$) had significantly increased between 1994 and 2002. There was no significant correlation between cattle herd size and rainfall with a one-year lag ($P = 0.02$) or without a lag ($P = 0.08$), but there was a significant positive correlation between goat herd size and rainfall with a one-year lag ($P < 0.001$).

A positive identification could only be made for one ant species that had been suggested as a degradation indicator by pastoralists: *Pachycondyla*: sp. *Ponerinae*. This species was only caught in degraded sites (primarily sacrifice zones next to boreholes, devoid of vegetation cover), suggesting that it may indeed be a useful indicator (Table 3) (Jew 2005).

Empirical evidence for indicators in Study Area 2 (Mid-Boteti).—The principal floristic differences between sample sites in Study Area 2 were between sites in *C. mopane*-dominated areas and grassland sites on the floodplains (Fig. 4b). These represent two distinct ecosystems within the study area. An inverse relationship between the encroacher, *C. mopane,* and grass cover is known to exist (Timberlake 1999, Smit 2004).

Floristic variation between sample sites in both grassland and *C. mopane*-dominated sites was primarily determined by proximity to village and water, respectively; there was a significant correlation between first

TABLE 3. Total number of ants caught in pitfall and bait traps in Study Area 1, in vegetation zones with increasing distance from a borehole (Jew 2005).

Genus	Sacrifice zone	Bush encroachment zone	Intermediate zone	Grazing reserve
Pachycondyla	11	1		
Brachymyrmex				7
Crematogaster	65	21	217	1338
Cerapachys				325
Forelius	175	1078	257	11
Neivamyrmex	55	901	532	542

axis and distance to village/water in grassland ($P = 0.01$; $r^2 = 0.35$; Fig. 4c), and *C. mopane*-dominated sites ($P < 0.01$; $r^2 = 0.34$) (Fig. 4d).

No *Grewia* species were found in any of the Study Area 2 sample sites, despite the fact that the ranges of a number of species extended into the study area (Van Wyk and van Wyk 1997). This may reflect a significant decline in the abundance of *Grewia* species in the study area, which would support the validity of *Grewia* species as indicator species (supported by Moleele and Chanda 2003). Reductions in the abundance of a number of other species were also suggested as degradation indicators by pastoralists, but were not found in the sample sites: *Ximenia* spp., *Cencrus ciliaris, Acacia hebeclada,* and *Cleome gynandra*. The current absence or low abundance of these species means they are unlikely to be particularly sensitive to change or easy to use. In addition to this, some species were found at abundances too low to conduct statistical analyses: *Dichrostachys cinerea, Sporobolus fimbriatus, Boscia albitrunca, Boscia foetida,* and *Acacia tortilis*. *A. tortilis* and *D. cinerea* are well-known encroacher species [Moleele et al. 2002]), so an increase in the abundance of these species may usefully be applied as degradation indicators. Given the importance of *Boscia* spp. for pastoralists' drought-coping strategies, a significant decrease in their abundance would reduce the resilience of the livestock production system. As a valuable source of browse, *Boscia* spp. growing under intense browsing pressure will inevitably become stunted and regenerate poorly. However, given the taboo associated with felling them, any browsing-induced decline in the number of *Boscia* trees in the landscape is a slow process, so this species is not a particularly sensitive or easy-to-use indicator.

There is evidence for the validity of the indicator "decreased abundance of grasses palatable for cattle." In grassland areas, there was a significant positive correlation between the abundance of the most palatable grass (e.g., *C. dactylon*) and distance from village. There was also a negative correlation between the abundance of the thatching grass *E. pallens* and distance from village, contradicting local suggestions that a decrease in thatching grass indicates land degradation.

Tree density decreased significantly along the degradation gradient, axis 1 ($P < 0.01$; $r^2 = 0.34$; Fig. 4d), with highest densities in most degraded areas. This is at variance with pastoralist perceptions that tree density declines with degradation. However, the conclusion is supported by literature suggesting *C. mopane* is an encroacher species (Timberlake 1999, Smit 2004). The local perception that there are fewer trees in degraded areas may be influenced by fuelwood shortages in this study area.

In grassland sites, soils were increasingly less consolidated with proximity to water sources. Although this result was not significant ($P = 0.02$; $r^2 = 0.34$), there were significantly lower levels of soil organic carbon (SOC) closer to water ($P < 0.01$; $r^2 = 0.49$), suggesting that "increased soil looseness" is a valid indicator of land degradation in grassland sites. There was no correlation between the proportion of soil samples that were consolidated or percentage of SOC, and proximity to water in *C. mopane*-dominated sites. Despite claims by some pastoralists that soil salinization mainly occurs in degraded sites, there was no correlation between soil conductivity and either distance from water or any of the ordination axes.

Empirical evidence for indicators in Study Area 3 (Bokspits).—The most significant differences between sample sites in Study Area 3 were between degraded interdune sites (dominated by *Schmidtia kalahariensis* and *Rhigozum trichotomum*) and all other sample sites, represented by the first ordination axis (eigenvalue: 1.00). To explore other differences between sample sites, axes 2 and 3 were analyzed in more detail (Fig. 4e).

Dung frequency correlated significantly with the second ordination axis ($P < 0.01$; $r^2 = 0.31$), suggesting that this does represent a utilization gradient that can be used as a proxy for a degradation gradient. The palatable perennial grasses *Centropodia glauca* and *E. lehmaniana* were significantly less abundant in degraded areas ($P = 0.01$ for both species). In addition to this, there was a negative correlation between the degradation axis (2) of the ordination (Fig. 4e) and the palatable dune grass *Stipagrostis amabilis*, which is used for thatching, suggesting that this species is indicative of less-degraded sites ($P < 0.01$; $r^2 = 0.39$). Abundance of the encroacher *A. mellifera* correlated significantly with the degradation gradient (axis 2) in the ordination (Fig. 4e), suggesting that it is also a degradation indicator ($P < 0.01$; $r^2 = 0.33$). This indicator was suggested by one member of the pastoralist community; however, others suggested that *A. mellifera* was in fact less abundant in degraded areas due to overbrowsing by goats. Focus group discussions about this indicator agreed with the latter assessment, which contradicts the ecological data. Smallstock were much more important in this community than in the other two study areas, and the majority of local observations by smallstock owners focused on the 2–3 km area around settlements where most browsing occurs. In this area, browsed areas may have less *A. mellifera*. However, ecological measurements were taken over a much wider area, up to 20 km from settlements, and participatory maps show that the most distant areas that were measured experience very little livestock activity and have a much lower abundance of *A. mellifera* (Fig. 2). So in the context of the wider landscape, *A. mellifera* is an effective degradation indicator.

Dune soils were significantly less consolidated than interdune soils ($P < 0.01$). Both dune and interdune soils in degraded sample sites were significantly less consolidated than soils from intermediate and nondegraded sites ($P < 0.01$). For both dune crests and interdunes, there was significantly more SOC in nondegraded sample sites, compared to degraded and intermediate sample sites ($P < 0.01$). Dune crest soils in degraded and intermediate sites also had significantly higher conductivity than nondegraded sites ($P < 0.01$).

Discussion

These results show that local knowledge can be a rich source of information about indicators of land degradation and environmental sustainability. However, these indicators may not always be sufficiently reliable or sensitive for accurate land degradation assessment. The following sections first discuss the identification and evaluation of local knowledge, and then discuss the results of ecological and soil-based sampling that was designed to further evaluate the validity of the suggested indicators.

Indicator identification and evaluation

In Study Area 1, formal education was a good predictor of people's ability to conceptualize indicators. This suggests that better-educated respondents were able to conceptualize and articulate indicators more easily than less-educated respondents. The difficulty of conceptualizing and articulating indicator knowledge may also account for the apparently thin spread of knowledge across the community. The majority of those who took part in the Multi-Criteria Evaluation felt able to comment on the accuracy of most indicators, suggesting familiarity with the information they had been presented with, even when such knowledge had not been included in their own individual indicator lists. Less-educated pastoralists relied more on livestock and vegetation indicators. This may be a reflection of their management objectives, which were more likely to focus on improving herd size and quality, and income generation. Better-educated pastoralists cited a more diverse range of objectives, including identification of optimal rotational grazing regimes, livestock breeds, and the grasses most suitable for different breeds. Perhaps as a consequence, this group tended to be able to conceptualize a more diverse range of indicators.

The range of indicators elicited was far broader than published indicator lists, encompassing vegetation, soil, livestock, wild animal, and socioeconomic indicators (Fig. 3). Livestock, wild animal, and socioeconomic indicators tended to differ between sites and hence are not included in Table 2, but included, for example: livestock spending more time eating bushes and foraging farther from water points; increased abundance of *Pachycondyla* spp. ants; increased distance to collect firewood; and increased household expenditure on products formerly obtained from rangeland and decreased income from range products. The majority of rangeland-monitoring manuals aimed at pastoralists focus entirely on vegetation and/or soil indicators (e.g., Field 1978, Foran et al. 1978, Vorster 1982, Tongway 1994, Milton et al. 1998, Esler et al. 2005). However, there is evidence that reliance on a narrow range of indicators may produce misleading results for degradation assessment, and pastoralists typically used a combination of indicators to diagnose problems in their rangeland. The breadth of indicators used by pastoralists in the Kalahari matches the call by the UN Convention to Combat Desertification for "integrated sets of physical, biological, social and economic indicators" (UNCCD 1994). It should be noted however, that different kinds of indicators were cited in different study areas. In Study Areas 1 and 3, pastoralists were more reliant on vegetation indicators than those in Study Area 2, and people used fewer soil indicators in Study Area 3 (Fig. 3).

Pastoralists' preference for vegetation-based indicators in all study areas matches that of previous farm-level assessment manuals for southern African drylands (e.g., Field 1978, Foran et al. 1978, Vorster 1982, Milton et al. 1998, Esler et al. 2005). However these previous assessments have been predominantly species based, an emphasis brought into question by this research, as we found pastoralists tended to group vegetation by morphology and palatability, rarely mentioning specific species.

Kalahari pastoralists generally downplayed soil-based indicators, something which is at variance with the focus of manuals produced for other dryland regions (e.g., Tongway 1994, NRC 2000). This is, however, consistent with scientific evidence that physical and hydrochemical soil degradation processes are not widely evident in the Kalahari (Dougill et al. 1999). This is interesting in relation to contemporary theoretical debates on semiarid ecological change and degradation (e.g., Illius and O'Connor 2000, Vetter 2005). Pastoralists' focus on vegetation and livestock indicators is at variance with the nonequilibrium concept that livestock populations are not coupled to their forage resources, as their numbers are regulated in a non-density-dependent manner by stochastic rainfall events (Scoones 1995). Contrary to nonequilibrium claims that "the risks of environmental degradation in nonequilibrium environments are limited, as livestock populations rarely reach levels likely to cause irreversible damage" (Scoones 1995:iv), pastoralists claim that livestock (there is no longer active herding in Botswana) are capable of causing permanent damage to forage resources, inducing a transition to a less productive ecological state, as predicted by state-and-transition models for the Kalahari (Dougill et al. 1999). If livestock are capable of degrading their forage resources, then it stands to reason that changes in vegetation can indicate the onset of land degradation, which may account for the pastoralist focus on vegetation indicators. The recognition that livestock-induced vegetation degradation is possible in dryland environments is consistent with recent challenges to nonequilibrium theory suggesting that livestock can reach equilibrium with the key forage resources they depend on during the dry season or drought, leading to degradation of the resource (Illius and O'Connor 1999, 2000).

The absence of livestock indicators from existing rangeland condition assessment manuals also contrasts with information provided by Kalahari pastoralists. Previous attempts to identify livestock indicators tended to be highly specialized and could not be assessed by pastoralists. For example, there are references to declining livestock production (e.g., Abel 1993, White 1993), the most frequently used index of which is the energy contained in the output of calves (Abel 1993), whereas Grant et al. (1996) refer to reduced mineral status in fecal and milk samples. The only exception is work showing that Massai in Kenya monitor livestock condition to inform their rangeland management (Kipuri 1996).

The majority of indicators elicited were "state" and "impact" indicators, according to the Driving Force-Pressure-State-Impact-Response terminology (EEA 1998). In addition to this, pastoralists were asked to identify more process-based indicators that could provide early warning of detrimental change: "pressure indicators" according to EEA's (1998) framework. A total of 14, 14, and 12 of these "early-warning" indicators were identified in each study area, respectively. Many people found this distinction difficult to make, and cited only state and impact indicators. This is consistent with Kipuri's (1996) findings from work with pastoralists in Kenya, and may be related to the apparency of state and impact indicators. However, the extra information available in early-warning indicators makes them vital to developing effective indicator-based management tools and enhancing extension advice. Wider dissemination of such indicators may facilitate timely adaptation to environmental change and potentially enhance the sustainability of rangeland management. Early-warning indicators tended to focus on vegetation and soils.

By building on local knowledge, the indicators developed in this research are familiar to pastoralists who have the capacity to apply them without any need for specialist training or equipment. Although most of the indicators cited by pastoralists are found in the literature (Table 2), pastoralists can often provide more meaningful interpretations of existing indicators, with nontechnical means of measuring complex variables. Rain use efficiency is an example of an indicator that would conventionally require too much specialist training and equipment for pastoralists to use (Joyce 2000). However it was used in a simplified form by a number of pastoralists, who defined it as "plants responding to rain with greater growth." Similarly, some pastoralists used the "dirtiness" of the sand as a surrogate for soil organic matter. Pastoralist experience and empirical analyses here show that the information provided by these surrogates is sufficiently accurate to support management decisions.

Many indicators from the literature were not cited by pastoralists. Discussions in focus groups showed that most of these were considered too difficult to measure. This included soil crusts, which have been used as indicators of rangeland condition in manuals targeted at pastoralists elsewhere in Southern Africa (Milton et al. 1998). Some indicators from the literature were considered irrelevant to the study area, such as soil compaction, which is not a problem in Kalahari soils due to their consistently high proportion of fine sands (Dougill et al. 1998). In some instances, pastoralists took direct issue with indicators from the literature. For example, in Study Area 1, unsustainable livestock practices are likely to lead to increased fuelwood availability due to bush encroachment. This contrasts with the other study areas and literature based on areas where deforestation is a threat to sustainability (e.g., Ottichilo 1990). In addition, contrary to evidence in the literature citing decreased soil infiltration rate as a degradation indicator (e.g., Tongway 1994, Bellows 1995, Weixelman et al. 1997, Sharma 1998), pastoralists viewed this as a positive sign, indicative of more consolidated sand with higher organic matter content. This is due to differences in soil type between this study area (dominated by fine sands) and those from other dryland regions. Unconsolidated soils with high infiltration rates tend to have low biological soil crust cover, can cause "Long Claw" in cattle (a condition where hooves become deformed due to walking on soft sand), and make travel difficult without a four-wheel-drive vehicle (each of these were considered to be degradation indicators).

Tree-based indicators tended to be cited frequently as early-warning indicators: notably tree stunting, decreased abundance of trees, and an increased proportion of trees dropping branches and leaves. A decline in total grass cover was widely cited as the best early-warning indicator of changes in rangeland condition. This is indicative of the increased stresses imposed on rangelands by intense grazing, especially during drought events (Illius and O'Connor 2000). It is at such times that effectively permanent changes in ecological communities of the Kalahari have been predicted (Dougill et al. 1999), and therefore early-warning indicators need to be tied to advice on drought-coping strategies that aim to retain some grass cover.

Empirical indicator evaluation

It has been suggested that the use of indicators by nonspecialists will inevitably involve a trade-off between meaningful participation and scientific rigor. The considerable overlap between scientific literature and local knowledge (Table 2), and the results of empirical testing suggests that such a trade-off is by no means inevitable. This study shows that there are a considerable number of indicators representing a wide range of system components that have a clear empirical basis and can be used effectively by nonspecialists.

In this study, the majority of vegetation indicators suggested by pastoralists were validated through ecological analysis. For example, reduced grass cover and increased bare ground were identified by pastoralists and supported by field observations at all study areas, as were shifts toward less palatable forage species. This is consistent with indicators used by pastoralists in other drylands (e.g., Oba and Kaitira 2006) that are supported by evidence from the literature (e.g., de Soyza et al. 1998, Whitford et al. 1998, Manzano and Navar 2000). Soil looseness had been suggested as a potential indicator by pastoralists in all sites, but was only validated in Study Area 3, where both dune and interdune soils from degraded areas were less consolidated and had lower SOC than soils from intermediate and nondegraded sites. Reductions in SOC have been observed elsewhere in degraded drylands (e.g., Dougill et al. 1999, Hill and Schütt 2000).

There are very few tree-based indicators in the literature, and yet a number were suggested by pastoralists as early-warning indicators. A reduction in the density of trees was suggested as an indicator of land degradation in all study areas, but this was not supported by measurements (trees had less girth at breast height [1.3 m]) in degraded sites in Study Area 1, but this was only significant at $P = 0.02$; there were fewer trees in degraded compared to nondegraded sites in Study Area 3, but this was only significant at $P = 0.04$). Despite these results, pastoralists in Study Area 1 continued to support the validity of this indicator, suggesting sample size as a potential reason for the absence of a statistically significant relationship. Given the sparse tree cover (average 6-m tree spacing recorded along intercepts), 30-m intercepts may not have been long enough, and 44 intercepts may have been too few. In Study Area 2 there was actually an increase in tree density along degradation gradients in *C. mopane* woodland ($P < 0.01$; $r^2 = 0.34$). This is supported by literature suggesting *C. mopane* is an encroacher species, favored by intense grazing (Smit 2004). There have been similar suggestions, from elsewhere in Africa, that human activity can lead to increased tree cover (e.g., Reid and Ellis 1995, Fairhead and Leach 2001). Although a reduction in tree density may accurately reflect degradation processes, it lacks sufficient sensitivity to be useful for land management, due to the long time scales over which density changes.

Although woody plants were significantly shorter in degraded parts of Study Area 3, this was probably due to differences in species composition, with degraded sites dominated by the naturally dwarf *R. trichotomum* and nondegraded areas dominated by the naturally taller *A. haemotoxolon*. Due to the mutual exclusivity of these species, it was difficult to assess the extent to which individuals of the same species were stunted by browsing. However, there was no significant difference between the height of *A. haemotoxolon* individuals growing in intermediate and nondegraded sites ($P = 0.30$). This supports the findings of Oba and Post (1999) who found evidence that browsing stimulates twig production in some *Acacia* species and as such significantly increases biomass accumulation.

Although it was not possible to adequately assess livestock condition in the field, herd size was analyzed using secondary data in Study Area 1. There was no evidence of declining herd sizes at any of the boreholes that were analyzed that could be used to infer degradation. The only significant change over time (increasing goats) correlated with rainfall (with a one-year lag), suggesting that herd size is an unreliable indicator of land degradation, as it is primarily affected by drought events.

It should be noted that it was not possible to collect sufficient data to test the validity of some indicators (e.g., abundance of wild fruits due to season; abundance of certain species that were not found in the sample sites that may have been found in a larger sample). Given the lack (or seasonality) of available data, many of these indicators would be difficult for pastoralists to use. For example, although pastoralists in all study areas cited a reduction in the abundance of medicinal plants, it was not possible to substantiate this. Although some species with known medicinal properties were found to be less common in degraded sites (e.g., *Boscia albitrunca* and *Cynodon dactylon*), many medicinal species were degradation indicators (e.g., *Senna italica* and the bush encroachers, *Acacia mellifera* and *Dichrostachys cinerea*). It is difficult to assess this indicator without knowledge of the medicinal properties of all the species found, and more ethnobotanical research would be required to determine these properties.

Finally, it is important to point out that in some cases, the same environmental changes were perceived as indicators of land degradation by pastoralists in some sites but not in others. This emphasizes the highly contextual nature of land degradation (Warren 2002), and one of the key benefits of participatory indicator development. Depending on the objectives of the land manager, an environmental change might be perceived as land degradation or might alternatively represent the emergence of a new resource: degradation is in the eye of the beholder. In Study Area 1, the livelihoods of the primarily cattle-owning population were significantly constrained by the replacement of palatable grasses with thorn bushes that were inedible to cattle (Table 1). As such, bush encroachment was perceived to be the primary form of land degradation. In Study Area 3, where smallstock ownership far outweighed cattle, a decrease in the abundance of the same thorn bush (*A. mellifera*) was perceived to be an indicator of land degradation, due to its high value for smallstock. Only by developing indicators with land managers is it possible to ensure that indicators are sensitive to the context in which they are to be applied

Conclusions

Environmental sustainability indicators developed through integrated participatory and ecological research are highly familiar to pastoralists, who have the capacity to apply them without any need for specialist training or equipment. However, not all pastoralist indicators could be used accurately or reliably to monitor land degradation. By testing local indicator knowledge empirically, it was possible to help pastoralists make an informed short-list of the indicators that could be used most sensitively and reliably to detect long-term rangeland degradation. In this way, it was possible to combine qualitative insights from participatory research with insights from more top-down empirical research to produce more accurate and relevant results than either approach could achieve alone.

Local knowledge was more holistic than many published indicator lists for monitoring rangelands, encompassing vegetation, soil, livestock, wild animal, and socioeconomic indicators. Reliance on single or few indicators can provide misleading results. More reliable interpretations can be derived from a greater number of indicators representing different system components. Pastoralist preferences for vegetation and livestock indicators match recent shifts in ecological theory, suggesting that livestock populations may reach equilibrium with dry-season or drought forage resources in semiarid environments. Early-warning indicators tended to focus on vegetation and soils, including tree-based indicators (which are rare in the literature).

Despite considerable overlap between indicators elicited from each of the Study Areas (30 out of 140 were elicited in all study areas), there were still significant differences between the indicators proposed for each Study Area. For this reason, it is essential for indicator-based monitoring tools and decision support tools to be site specific.

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