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Monitoring ecological processes for restoration projects

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Summary

Restoration of ecological processes is key to restoring the capacity of ecosystems to support social, economic, cultural and aesthetic values. The sustainability of the restored system also depends on processes associated with carbon, nutrient and hydrologic cycles, yet most restoration monitoring is limited to plant community composition. Our research has shown that short-term plant composition monitoring is a necessary but insufficient predictor of long-term restoration success. Long-term (up to 75 years) studies in the western United States show that short-term monitoring of plant community composition alone incorrectly predicted the failure of treatments that were ultimately successful, and the success of treatments that ultimately failed. We propose that vegetation composition monitoring be combined with one or more ecological process indicators reflecting changes in three fundamental ecosystem attributes on which restoration success depends: soil and site stability, hydrologic function and biotic integrity. These simple, rapid, plot-level indicators reflect changes in resource redistribution and vegetation structure. We include a case study involving restoration of mixed grass prairie on mineland in the west-central United States.

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Introduction

The definition of restoration success is commonly based on the presence, density, biomass or cover of one or more plant species at a particular point in time (Ruiz-Jaen & Aide, 2005). There are two significant limitations to this approach. The first is that it fails to reflect today's diverse goals for ecological restoration, which increasingly include the recovery and improvement of landscape function to support multiple ecosystem services (Aronson, Clewell, Blignaut, & Milton, 2006). In some cases, the plant community is restored, but the ecosystem services demanded by society are

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not. In others, a plant community-based approach leads to overpriced restoration efforts by requiring the establishment of late-successional or difficult to establish species early in the recovery process. These species may not be essential to the recovery of the ecosystem services valued by society, particularly if these services are more effectively provided by different species. This is often the case where sustaining high levels of agricultural production is the primary objective (Tilman, Cassman, Matson, Naylor, & Polasky, 2002).

The second limitation of using plant community composition alone is that it ignores the ecological processes on which the persistence of the restored plant communities depends. Most restoration projects emphasise the early stages of recovery through site preparation necessary for plant establishment. It is widely acknowledged that active adaptive management is increasingly required for long-term restoration success (Folke et al., 2004), but the question of what needs to be managed is rarely explicitly addressed.

Our research has shown that short-term plant composition monitoring is a necessary but insufficient predictor of long-term restoration success. Short-term monitoring of plant community composition alone incorrectly predicted the failure of treatments that were ultimately successful and the success of treatments that ultimately failed. Examples of restoration success following apparent failure include the establishment of 7.5 cm high runoff-barriers on a vegetation-free physically crusted loamy soil with a 1% slope in 1975. The dikes were maintained, biosolids were applied once, and the area was seeded at least three times during the next four years. In 1979, following the apparent failure of the fourth seeding, the dikes were abandoned as a restoration failure (Walton, 2005). By 1997, the perennial vegetation which subsequently established was clearly visible from the air (Fig. 1; Walton, Herrick, Gibbens, & Remmenga, 2001). Today, the composition and structure of the vegetation is similar to that of naturally occurring banded vegetation (Tongway, Valentin, & Seghieri, 2001) on similar soils in the area. Walton (2005) attributes the lag in the perennial vegetation response to the slow rate of recovery of infiltration and nutrient cycling processes associated with soil organic matter accumulation. These "slow variables" (Reynolds, 2001) appear to be limiting the recovery of this system. Additionally, periods favorable for grass establishment are relatively rare in this environment in which annual precipitation averages 230 mm, much of which arrives as intense convective storms.

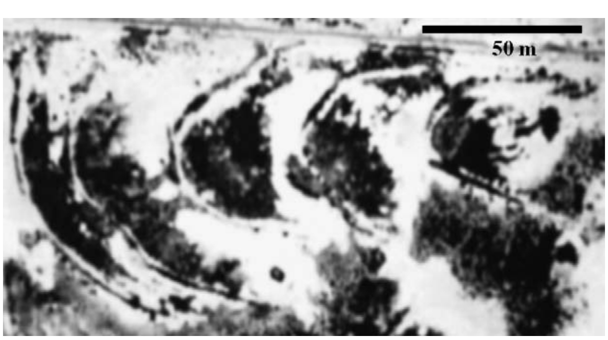

Figure 1. Perennial vegetation recovery 23 years after 7.5 cm high soil berms were constructed in a vegetation-free area in southern New Mexico (reprinted from Walton et al., 2001). Dark areas are vegetated; light areas are bare soil.

In contrast, many restoration projects that are touted as short-term successes do not persist due to the failure to re-establish one or more biophysical processes. In arid and semi-arid regions of the southwestern United States, one-time shrub-removal and grass seeding are rarely sufficient to re-establish desert grasslands on degraded soils (Herrick, Havstad, & Rango, in press; Rango, Huenneke, Buonopane, Herrick, & Havstad, 2005). The initial failure of grass establishment in arid and semi-arid ecosystems is generally associated with inadequate soil moisture (Ethridge, Sherwood, Sosebee, & Herbel, 1997; Gao & Reynolds, 2003) which is related to both weather and ecohydrological processes. Process-related factors that may explain the subsequent loss of grasses and re-establishment of woody invasive species include high rates of runoff and erosion, the persistence of resource islands associated with the woody species, and changes in the fire regime and herbivore populations (Herrick et al., in press). The importance of the relationship between vegetation structure, function and ecological processes has been clearly documented in numerous other systems, including coastal sand dunes (Roze & Lemauviel, 2004) and Mediterranean steppes (Maestre & Cortina, 2004). For example, forest and savanna ecosystems often require reintroduction of a process such as fire (Allen et al., 2002; Falk, 2006). The fire regime, in turn, depends on plant community composition and structure at multiple spatial scales, as well as ignition sources. This understanding has led to the proposal that "reference dynamics" based on an understanding of keystone ecological processes replace "reference conditions" as the standard for restoration success (Falk, 2006).

Consistent with this proposal, we propose that vegetation composition indicators be supplemented with one or more ecological process indicators reflecting changes in three fundamental ecosystem attributes on which restoration depends: soil and site stability, hydrologic function and biotic integrity. This approach should allow managers to more accurately predict long-term restoration success based on short-term data. Because these attributes also serve as the foundation for nearly all ecosystem services (Fig. 2), the approach can also be used to help explain the importance of restoration projects to diverse groups in society.

The specific objective of this paper is to present and illustrate a cost-effective approach for monitoring restoration success in arid and semi-arid upland ecosystems. The approach uses a combination of soil and vegetation indicators that reflect the status of key ecological processes. A qualitative assessment protocol is used to select quantitative monitoring indicators. Indicators of vegetation cover, composition and structure are interpreted together with soil surface and near-surface characteristics to monitor changes in ecological status. A unique reference is used for each site, based on its soil- and climate-defined ecological potential. Guidance for flexibly interpreting individual indicators across different ecosystems is provided in a separate document (Herrick, Van Zee, Havstad, Burkett, & Whitford, 2005b).

Monitoring system

The ten-step, iterative approach includes three basic modules (Fig. 3). The first module, monitoring programme design, is completed prior to the initiation of the restoration project. The second module, short-term monitoring, is then used to document and, where possible, adjust factors that may affect restoration success. The third module involves more intensive long-term monitoring of restoration success. This information is used to adjust management strategy. Some of the long-term monitoring indicators may also be used for short-term monitoring.

Monitoring programme design

Monitoring programme design should be an integral part of the restoration planning process, beginning with the definition of restoration objectives and monitoring objectives (Step 1), which are developed iteratively with a landscape stratification (Step 2) and the assessment of the current status of key ecosystem attributes and processes (Step 3). The landscape stratification (Step 2) is particularly important for projects covering large areas where the current status and probable

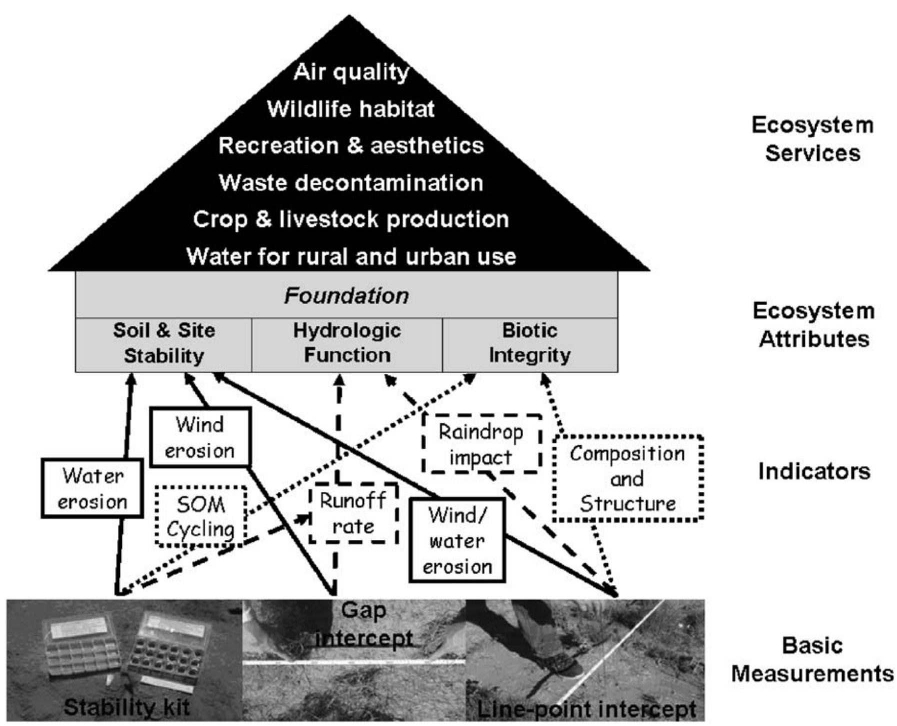

Figure 2. The three basic measurements are used to generate indicators of the processes shown in the boxes which, in turn, are related to the three ecosystem attributes which serve as the foundation for most ecosystem services and are essential to the success of nearly all restoration projects.

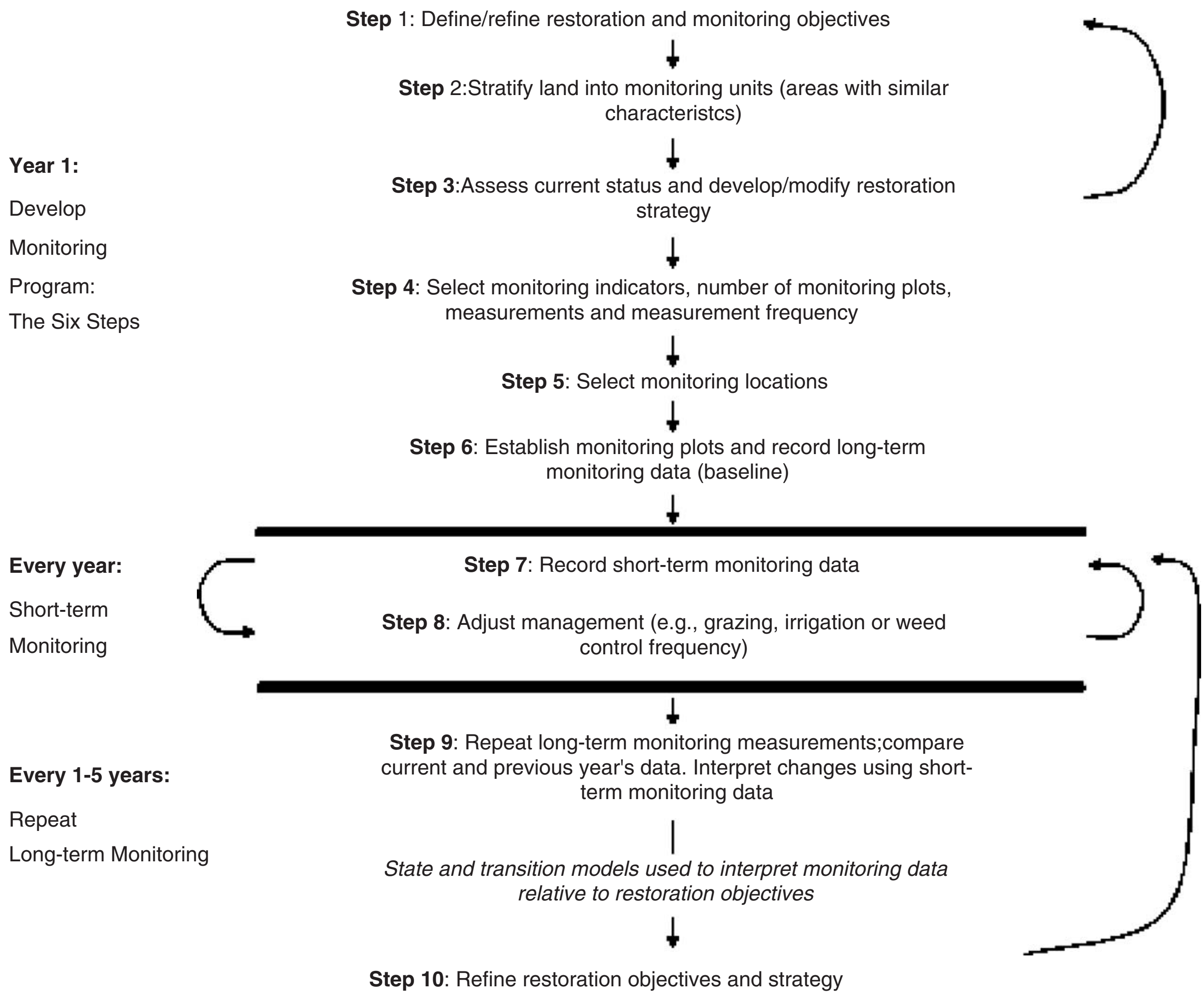

Figure 3. Restoration monitoring programme design and implementation (modified from Herrick et al., 2005b, Volume II).

treatment response are likely to vary spatially. Ecological sites are widely applied in the United States. An ecological site is a type of land with similar soil, topography and climate that has the potential to support particular types and amounts of vegetation (Herrick, Bestelmeyer, Archer, Tugel, & Brown, 2006). Only relatively static soil properties (Tugel et al., 2005), like texture and depth, are used to define ecological sites. An ecological site therefore includes areas with similar long-term potential to support ecosystem services. Each ecological site may occur at multiple locations in a region based on soil and climate patterns.

For each ecological site or strata defined based on soil, topography and climate (Step 2), a unique conceptual "state and transition model" (Bestelmeyer et al., 2003; Stringham, Krueger, & Shaver, 2003) is used together with assessment tools (Step 3) to define realistic restoration objectives (Step 1). State and transition models consist of one or more ecological states. The boundaries between states are marked by relatively irreversible transitions or thresholds. Relatively reversible transitions occur within states. Current understanding of the status of processes within states and factors associated with transitions among states is described in the associated text. Because these models are conceptual, they are easily updated by both scientists and land managers.

Within the restoration planning process, state and transition models can be used to define what is possible and what is realistic. Moving among plant communities within a state is generally possible without significant external inputs. Shifting from an undegraded state to a degraded state is also relatively easy. Moving from a degraded state to a less degraded state requires significantly more inputs because ecological processes have been altered to the point that simply removing a stress is insufficient to promote recovery.

A qualitative assessment protocol (Step 3), "Interpreting Indicators of Rangeland Health", is widely applied throughout North America to evaluate the status of the land relative to its ecological potential for three attributes: soil and site stability, hydrologic function and biotic integrity (Pellant, Shaver, Pyke, & Herrick, 2005; Pyke, Herrick, Shaver, & Pellant, 2002). The protocol uses 17 easily observed soil and vegetation indicators (Table 1).

Applying this protocol prior to project implementation can help identify the processes that are likely to limit restoration success. Many of these processes (e.g. infiltration and runoff) are very expensive to measure, but can be easily evaluated using a combination of indicators including water flow patterns, rills, soil compaction, and plant community composition and spatial distribution. In order to increase consistency, a reference sheet describing the range of variability expected for each indicator is developed for soils with similar ecological potential. For example, loamy soils on gently sloping terrain in a mid-latitude region with 700 mm annual precipitation would be expected to have short (less than 1 m) water flow patterns, no rills and no compaction layer. The soils should support relatively uniform vegetative cover that would slow the movement of any runoff that did occur. Reference sheets are developed using all available sources of information, including scientific literature and local knowledge. For ecological sites with a state and transition model, the reference sheet describes the range of variability associated with the undegraded or reference state (Herrick et al., 2006). A typical reference sheet describing all 17 indicators for an ecological site is approximately 1–2 pages long.

Each of the three attributes is evaluated independently to ensure that serious problems in one set of processes are not masked by unrelated indicators. For example, invasive species have negative effects on biotic integrity but sometimes improve hydrologic function. This is one of several reasons that invasive species removals can have unintended consequences (Zavaleta, Hobbs, & Mooney, 2001).

Another key feature of the protocol is that a "preponderance of evidence" approach allows indicators to be variably weighted for different soils (Pyke et al., 2002). For example, the absence of rill formation on a lakebed soil is a less important indicator of soil and site stability than it would be on the loamy soils described above. Similarly, water flow patterns and large vegetation gaps are expected in areas where woody vegetation is expected to dominate and their mere presence would have no effect on the evaluation. These indicators would serve as indicators of degraded hydrologic function in areas where the ecological potential is for perennial grassland, or in shrublands where the herbaceous component was lost.

Table 1. Ecological attribute assignments for 17 indicators used in the qualitative assessment protocol

Indicator		Attribute		
		Soil & site stability	Hydrologic function	Biotic integrity
1	Rills	S	H	
2	Water-flow patterns	S	H	
3	Pedestals and/or terracettes	S	H	
4	Bare ground	S	H	
5	Gullies	S	H	
6	Wind-scoured, blowouts, and/or deposition areas	S		
7	Litter movement	S		
8	Soil surface resistance to erosion (soil stability in water test; Herrick et al., 2001)	S	H	B
9	Soil surface loss or degradation	S	H	B
10	Plant community composition and distribution relative to infiltration and runoff		H	
11	Compaction layer	S	H	B
12	Functional/structural groups			B
13	Plant mortality/decadence			B
14	Litter amount		H	B
15	Annual production			B
16	Invasive plants			B
17	Reproductive capability of perennial plants			B

See Pellant et al. (2005) for detailed explanation of each indicator.

The use of an ecological site-specific reference sheet (Herrick et al., 2006) and the independent evaluation of three ecosystem attributes are two of the critical differences between this assessment protocol and others which generate one or more indices that are independent of site potential (e.g. Tongway, 1995; Tongway & Hindley, 2004). While indices are attractive (they are easily compared) and can be quite effective where they have been calibrated, universal application can result in lost opportunities to apply local knowledge of the relationships between ecosystem properties and processes. The reference sheet approach is also distinguished from index-based approaches in that it does not assume a linear relationship or, in fact, any defined relationship between individual properties, processes and functions. For example, critical dynamic properties for limiting soil erosion are different for wind and water, and vary with soils, climate and hydrology. Soil physical crusts limit wind erosion in low rainfall zones, but increase water erosion because they increase runoff and are highly dispersible. Microbiotic crusts nearly always limit both wind and water erosion, and may increase or decrease water infiltration (Warren, 2001). Consequently, both physical and microbiotic crusts may be desirable at some stages of the restoration process in some ecosystems, and undesirable at other stages in other ecosystems.

In addition to identifying the processes that must be addressed by the restoration treatments, the assessments are used to select monitoring indicators (Step 4; see "Long-term monitoring") and locations within the area to be restored (Step 5) where baseline measurements are completed (Step 6). A stratified random sampling design is generally superior to randomly locating plots or transects because it focuses monitoring efforts on the most important areas.

Short-term monitoring

The greatest interest in monitoring often exists immediately after the restoration treatments have been applied. In arid ecosystems, however, complete recovery takes decades and annual monitoring of long-term indicators like perennial plant cover and soil structure is generally not cost-effective. However, monitoring the factors which contribute to restoration success (Step 7) can be extremely valuable *if* it is possible to adjust management to modify these factors (Step 8). For example, in grazed systems, forage utilisation and residual cover can be recorded to guide seasonal grazing management. Where fire is a critical process, fuel loads should be monitored, while soil moisture should be measured or at least observed in areas that can be irrigated or where evapotranspiration can be modified. Unless the restoration project has a research component, however, it is irresponsible to recommend allocating resources to soil moisture monitoring if there is no way the manager can modify moisture availability. Finally, documenting and being prepared to respond to rare but potentially critical events, such as floods, should be a part of any short-term monitoring programme.

Long-term monitoring

The three primary objectives of repeating long-term monitoring measurements (Step 9) are to evaluate restoration success, to adjust restoration objectives and to adapt the management strategy. To the extent possible, long-term monitoring indicators should reflect both the stated restoration objectives and the ecosystem properties and processes required to achieve those objectives. In order to make the monitoring system as cost-effective as possible, each measurement should generate multiple indicators that are relevant to one or more restoration objectives and processes.In arid and semi-arid ecosystems, there are three basic measurements which together generate a set of key indicators that reflect critical changes in the three key ecosystem attributes: line-point intercept, gap intercept, and a soil stability test (Fig. 2; Herrick, Van Zee, Havstad, Burkett, & Whitford, 2005a). These measurements also provide sufficient plant community composition and structure data to allow vegetation-based objectives to be evaluated. By quantifying the changes in both the plant community and soil structure, they indirectly reflect nutrient cycling processes which are expensive and difficult to monitor directly.

All measurements are completed along transects. Guidance on the number of transects and measurements required to detect change at plot and landscape scales is provided in Herrick et al. (2005b). Line-point intercept is used to quantify plant canopy, basal cover and composition, and ground and plant litter cover. Plant canopy cover is an important indicator of production, while ground cover is the most important indicator of erosion resistance. Litter cover is an important indicator of decomposition and nutrient cycling. Where the potential for wind erosion exists or vegetation structure is an important restoration objective (i.e. for wildlife habitat), height measurements can be completed on a subset of the points. Toledo (2004) found that height measurements at just 10 points per 50-point transect were sufficient to generate an indicator that was highly correlated with visual obstruction values measured with a cover pole.

The gap intercept method (Herrick et al., 2005a, 2005b) is used to rapidly measure the proportion of soil surface covered by functionally significant gaps between plants. In this modification of the continuous line intercept method, only gaps (between plant canopies or plant bases) longer than a minimum length (e.g. 30 cm) are measured. Canopy gaps are measured where wind erosion (Okin, Gillette, & Herrick, 2006) or microsite modification for seedling establishment (O'Connor, 1996) are important processes. Plant basal gap intercepts are measured where increasing runoff resistance is critical to restoration success because the distribution of these gaps is related to the "leakiness" of the system (Ludwig, Eager, Bastin, Chewings, & Liedloff, 2002; Ludwig, Wilcox, Breshears, Tongway, & Imeson, 2005). Independent estimates of under canopy and interspace recovery can be generated by combining line-point intercept data with gap intercept and soil stability data. Together, the methods provide a spatially explicit snapshot of cover, composition, structure and stability, and generate indicators that relate to hydrologic, erosion and vegetation recovery processes. A soil stability kit (Herrick et al., 2001) is used to rapidly measure soil surface and sub-surface macroaggregate stability in water. This method, which allows 20 samples to be tested in 10 min, is sensitive to soil organic matter inputs and decomposition.

These three methods were selected because each provides indicators that can be independently interpreted while yielding information that can be used to help interpret indicators provided by the other measurements, as illustrated by the case study below. The spatially explicit data collection system used for the first provides many opportunities to develop relationships among indicators, which can help elucidate the status of different processes, particularly when the measurements are interpreted together with the qualitative indicators used in the assessment process. Another important selection criterion was measurement cost relative to information provided. However, the cost of even these simple measurements can exceed many restoration monitoring budgets. Careful review of restoration and monitoring objectives can be used to eliminate one or more measurements. Where time is limited, alternative semi-quantitative methods that generally do not require transect establishment can be applied (Herrick et al., 2005a, 2005b). Additional measurements described in Herrick et al. (2005b), Elzinga, Salzer, Willoughby, and Gibbs (2001) and in soil science references (e.g. Klute, 1986; McKenzie, Coughlan, & Resswell, 2002), can be used to monitor processes not specifically addressed by these three measurements.

For example, where trends in rare or exotic plant species need to be monitored, a belt transect can be added. Exotic plant species invasion is one of the most important process in many arid and semi-arid ecosystems, such as the North American Great Basin (Pyke & Knick, 2003). The line-point intercept, while more precise, repeatable and versatile than plot-based methods, is notoriously poor for early detection of plant invasion, and for monitoring relatively rare species of interest. Plot-based methods can be expensive due to the time required for plot establishment. The belt transect takes advantage of the existing transect. By focusing only on those species of interest, time costs are reduced and precision increased. A 'virtual plot' is created by walking along the transect with a pole of defined length: one-half the width of the desired plot. The pole is extended perpendicularly from the transect as necessary to determine whether or not an individual falls within the transect. This is much more rapid than establishing formal plots.

Data analysis

Data that are not immediately summarised, analysed and interpreted are unlikely to be used. A number of user-friendly databases and field data entry systems are now available. Many of these systems (e.g. USDA-ARS Jornada Experimental Range, 2006) automatically generate key indicators so that interpretations can be made as soon as data collection is complete.

Indices and additional indicators

This approach does not generate an index because the relationship between ecological properties, patterns and processes varies across ecosystems. Different processes are critical for the functioning of tallgrass prairie ecosystems (fire and grazing; Collins, Knapp, Briggs, Blair, & Steinauer, 1998) than for arid and semi-arid savanna ecosystems dominated by high-intensity summer rainfall (runoff and erosion; Davenport, Breshears, Wilcox, & Allen, 1998). Also, the relationship between structure, processes and functions varies among systems, and may even be different during degradation and different types of recovery trajectories (Cortina et al., 2006). However, the indicators can be used to develop ecosystem-specific indices. For example, Tongway and Hindley (2004) present an index-based approach that has been successfully applied to a number of ecosystems. Many of the indicators and much of the conceptual basis for the approach described here reflect Tongway's earlier work (e.g. Tongway, 1995). The predictive power of this approach can be enhanced by integrating it with ecosystem- and site-specific indicators of drivers (Herrick et al., 2005b) and the dynamics of specific processes (Falk, 2006).

Case study: Mineland restoration in Wyoming, USA

In June 2005, we applied the long-term protocol to evaluate restoration success at three sites relative to native reference areas on similar soils at two surface coal mines in northeastern Wyoming (total of five sites; Table 2). Treatments were implemented 4–10 years prior to measurement. This area has traditionally been used almost exclusively for livestock production. A rapidly growing urban and exurban population, however, increasingly values these lands for other uses, including recreation and watershed protection.

Indicators from our preliminary, qualitative assessment (Pellant et al., 2005) suggested that while plant cover had been successfully restored at all three sites, ecological processes were not functioning at their full potential at two of the sites. We observed differences relative to the reference sites in each of the three ecosystem attributes. Based on these observations, we selected the three basic measurements described above and used them to generate indicators that are relevant to each of the attributes: line-point intercept, gap intercept, and soil surface and sub-surface aggregate stability in water. Each measurement was completed along six randomly located 25 m transects at each of the three reclaimed mine sites and two native reference sites.

Line-point intercept was used to quantify the cover and composition of the dominant plant structural groups: shrubs, perennial grasses and annuals. We also used the data to calculate native species canopy cover. All gaps between perennial plant canopies longer than 25 cm were recorded. Soil stability was rated on a scale from 1 to 6 (Herrick et al., 2001) on 18, 6–8 mm diameter, 2–4 mm thick, soil surface fragments collected at random locations along the transects at each of the five sites. For more detail on the methods used, see the "Long-term monitoring" section above and Herrick et al. (2005a, 2005b).

Results and discussion

At Mine 1, the traditional vegetation measurements indicated partial recovery at the more recently restored site (Reclaimed 1) and near-complete recovery at the older site (Reclaimed 2) (Table 3). The spatial distribution of the perennial vegetation, however, was more heterogeneous in Reclaimed 2 than in the native reference site (Native), resulting in a higher proportion of the soil surface exposed in large (>50 cm) intercanopy gaps in Reclaimed 2 (15%) than in Native (6%), indicating incomplete recovery. Over 85% of the soil surface was exposed in >50 cm long intercanopy gaps at Reclaimed 1 due primarily to low perennial plant cover. Average soil stability was also lower at both reclaimed sites than at the native site. Soil stability under perennial plant canopies, however, was virtually identical at all three sites, suggesting that soil structure recovers relatively rapidly where there are sufficient litter inputs. Litter cover was higher under plant canopies. The results suggest that while vegetation recovery is complete at Reclaimed 2, the site remains more susceptible to soil erosion and associated negative feedbacks than at Native. Qualitative indicators, including litter movement, water flow patterns and pedestalling, indicated that runoff and erosion were occurring, but at relatively low rates. Management practices, such as mulch application, that increase intercanopy soil aggregate stability and reduce gap size by promoting perennial plant establishment could further reduce resource loss from these sites in the future.

Table 2. Site characterisation data of native rangeland and reclaimed mine sites in northeastern Wyoming

	Mine 1			Mine 2	
	Native	Reclaimed 1	Reclaimed 2	Native	Reclaimed
Elevation (m)	1521	1436	1458	1463	1486
Slope (%)	2–3	3–4	3–10	1–5	3–15
Recovery (years)	n/a	4	10	n/a	10
Texture	Sandy loam	Fine sandy loam	Loam	Sandy loam	Sandy loam

Table 3. Soil and vegetation indicators for the native and reclaimed sites in northeastern Wyoming (average ± standard error)

	Sample size (n)	Mine 1			Mine 2	
		Native	Reclaimed 1	Reclaimed 2	Native	Reclaimed
		Canopy cover				
Total (%)	6	61(8)	51(5)	59(6)	71(7)	51(4)
Native species (% of total plant vegetation)	6	100(0)	28(6)	94(3)	85(4)	70(4)
Perennial grass (% of total vegetation)	6	80(6)	22(6)	83(5)	69(5)	70(2)
Shrubs (% of total vegetation)	6	11(7)	1(1)	6(5)	9(1)	0(0)
		Soil surface in canopy gaps (%)				
25–50 cm	6	14(3)	4(2)	18(1)	13(1)	14(2)
50–100 cm	6	6(4)	7(2)	9(3)	4(1)	7(3)
100–200 cm	6	0(0)	22(7)	4(2)	1(1)	2(2)
> 200 cm	6	0(0)	59(11)	2(2)	0(0)	0(0)
		Soil stability (value from 1–6)				
Average	18	3.2(0.7)	1.8(0.7)	2.3(0.6)	3.3(0.6)	2.7(0.6)
Under canopy	0–9[a]	2.9(0.7)	2.9(0.8)	3.0(0.7)	3.7(0.6)	n.d.[b]
Intercanopy	9–18	3.6(0.8)	1.0(0.0)	1.6(0.3)	3.2(0.6)	2.7(0.6)

[a]Samples were randomly located, resulting in fewer under canopy than intercanopy samples at most sites.
[b]None of the sampling points fell completely beneath a plant canopy.

At Mine 2 reclaimed site, total canopy cover had not yet recovered to levels recorded at the corresponding native site (Table 3). This appeared to be largely due to the absence of shrubs from the site. At this reclaimed site, however, the vegetation was much more uniformly distributed than at the Mine 1 reclaimed sites, resulting in a virtually identical gap size distribution. This is largely due to the dominance of western wheatgrass (*Pascopyrum smithii*) at the reclaimed site. Since western wheatgrass is rhizomatous, it is highly dispersed and does not form a significant canopy. As a result, no under canopy soil stability samples were collected at this site. Between-plant-canopy samples at the reclaimed site were similar to those at the native site, suggesting that significant recovery of soil structure had occurred. This may be due in part to the more dispersed plant spacing, which increased uniformity of microclimate and litter inputs, and provided better protection from wind. These results suggest that while traditional indicators of restoration success showed incomplete recovery at the Mine 2 reclaimed site, the more process-based indicators showed near-complete recovery.

Summary and conclusions

The pilot study supports the need for process-based indicators. Vegetation cover and composition indicators suggested complete recovery at one of the reclamation sites at Mine 1 (Reclaimed 2), and incomplete recovery at Mine 2. The process-based indicators suggest the opposite. Applying the two types of indicators together can help reclamationists, regulatory agencies, and land managers develop more effective restoration and post-restoration strategies. While not directly addressed in this case study, the data can also be used to evaluate restoration success relative to other societal values, such as wildlife habitat (Herrick et al., 2005b).

Future directions

The primary limitation of the monitoring approach described here is that it ignores landscape level processes (Herrick et al., 2006). New indicators reflecting landscape-scale processes are being developed and can be adapted to monitor restoration project success. Landscape-scale indicators are necessary for both restoration project design and monitoring, especially in arid and semi-arid ecosystems where there is often a high level of resource redistribution among landscape units due to high rates of runoff, and wind and water erosion. These indicators will increasingly need to integrate landscape ecology approaches that emphasise spatial pattern with more mechanistic indicators of landscape processes including soil erosion (Tongway & Hindley, 2004). Animal activity and restoration treatments themselves can lead to further modifications in resource redistribution processes and in both the wind and water vectors themselves (Peters, Havstad, Herrick, Huenneke, & Schlesinger, in press). Cost-effective monitoring of these patterns and processes will increasingly require ultra-high resolution remote sensing data which will increasingly be collected using aerial vehicles that can autonomously photograph restoration sites and associated controls (Rango et al., in press). This approach also fails to address episodic events, such as fire and flooding. These dynamics are essential to the recovery of many systems (Falk, 2006). Episodic events, as well as current and potential future drivers (Herrick et al., 2005b) should be identified and included in restoration monitoring programmes. Process- and ecosystem-specific monitoring must be developed.

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References

Allen, C. D., Savage, M., Falk, D. A., Suckling, K. F., Swetnam, T. W., Schulke, T., et al. (2002). Ecological restoration of southwestern ponderosa pine ecosystems: A broad perspective. *Ecological Applications*, *12*, 1418–1433.

Aronson, J., Clewell, A. F., Blignaut, J. N., & Milton, S. J. (2006). Ecological restoration: A new frontier for nature conservation and economics. *Journal for Nature Conservation*, *14*, 135–139.

Bestelmeyer, B. T., Brown, J. R., Havstad, K. M., Alexander, R., Chavez, G., & Herrick, J. (2003). Development and use of state-and-transition models for rangelands. *Journal of Range Management*, *56*, 114–126.

Collins, S. L., Knapp, A. K., Briggs, J. M., Blair, J. M., & Steinauer, E. M. (1998). Modulation of diversity by grazing and mowing in native tallgrass prairie. *Science*, *280*, 745–747.

Cortina, J., Maestre, F. T., Vallejo, V. R., Baeza, M. J., Valdecantos, A., Pérez-Devesa, M. Ecosystem structure, ecosystem function, and restoration success: Are they related? *Journal for Nature Conservation*, 2006, *14*, 152–160.

Davenport, D. W., Breshears, D. D., Wilcox, B. P., & Allen, C. D. (1998). Viewpoint: Sustainability of pinon-juniper ecosystems – a unifying perspective of soil erosion thresholds. *Journal of Range Management*, *51*, 231–240.

Elzinga, C. L., Salzer, D. L., Willoughby, J. W., & Gibbs, J. P. (2001). *Monitoring plant and animal populations*. Malden, MA: Blackwell Scientific.

Ethridge, D. E., Sherwood, R. D., Sosebee, R. E., & Herbel, C. H. (1997). Economic feasibility of rangeland seeding in the arid southwest. *Journal of Range Management*, *50*, 185–190.

Falk, D. A. (2006). Process-centered restoration. *Journal for Nature Conservation*, *14*, 140–151.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., et al. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology Evolution and Systematics*, *35*, 557–581.

Gao, Q., & Reynolds, J. F. (2003). Historical shrub-grass transitions in the northern Chihuahuan Desert: Modeling the effects of shifting rainfall seasonality and event size over a landscape gradient. *Global Change Biology*, *9*, 1475–1493.

Herrick, J. E., Bestelmeyer, B. T., Archer, S., Tugel, A., & Brown, J. R. (2006). An integrated framework for science-based arid land management. *Journal of Arid Environments*, *65*, 319–335.

Herrick, J. E., Havstad, K. M., & Rango, A. (in press). Remediation research at the Jornada: Past and future. In: *Structure and function of a Chihuahuan Desert ecosystem: The Jornada Basin LTER*. Oxford: Oxford University Press, in press.

Herrick, J. E., Van Zee, J. W., Havstad, K. M., Burkett, L. M., & Whitford, W. G. (2005a). *Monitoring manual for grassland, shrubland and savanna ecosystems. Vol. I: Quick start*. USDA-ARS Jornada Experimental Range, Las Cruces, NM: Distributed by University of Arizona Press.

Herrick, J. E., Van Zee, J. W., Havstad, K. M., Burkett, L. M., & Whitford, W. G. (2005b). *Monitoring manual for grassland, shrubland and savanna ecosystems. Vol. II: Design, supplementary methods and interpretation*. USDA-ARS Jornada Experimental Range, Las Cruces, NM: Distributed by University of Arizona Press.

Herrick, J. E., Whitford, W. G., deSoyza, A. G., Van Zee, J. W., Havstad, K. M., Seybold, C. A., et al. (2001). Soil aggregate stability kit for field-based soil quality and rangeland health evaluations. *CATENA*, *44*, 27–35.

Klute, A. (1986). *Methods of soil analysis. Part 1: Physical and mineralogical methods* (2nd ed.). Madison, USA: American Society of Agronomy.

Ludwig, J. A., Eager, R. W., Bastin, G. N., Chewings, V. H., & Liedloff, A. C. (2002). A leakiness index for assessing landscape function using remote sensing. *Landscape Ecology, 17*, 157–171.

Ludwig, J. A., Wilcox, B. P., Breshears, D. D., Tongway, D. J., & Imeson, A. C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology, 86*, 288–297.

Maestre, F. T., & Cortina, J. (2004). Insights into ecosystem composition and function in a sequence of degraded semiarid steppes. *Restoration Ecology, 12*, 494–502.

McKenzie, N., Coughlan, K., & Resswell, H. (2002). *Soil physical measurement and interpretation for land evaluation*. Colingwood, Australia: CSIRO Publishing.

O'Connor, T. G. (1996). Hierarchical control over seedling recruitment of the bunch-grass *Themeda triandra* in a semi-arid savanna. *Journal of Applied Ecology, 33*, 1094–1106.

Okin, G. S., Gillette, D. A., & Herrick, J. E. (2006). Multiscale controls on and consequences of aeolian processes in landscape change in arid and semiarid environments. *Journal of Arid Environments*, *65*, 253–275.

Pellant, M., Shaver, P., Pyke, D., & Herrick, J. E. (2005). *Interpreting indicators of rangeland health, Version 4. Interagency Technical Reference*. Denver, CO: Bureau of Land Management.

Peters, D. P. C., Havstad, K. M., Herrick, J. E., Huenneke, L. F., & Schlesinger, W. M. (in press). Future directions in Jornada research: Developing and applying an interactive landscape model to solve old and new problems. In *Structure and function of a Chihuahuan Desert ecosystem: The Jornada Basin LTER*. Oxford: Oxford University Press, in press.

Pyke, D. A., Herrick, J. E., Shaver, P., & Pellant, M. (2002). Rangeland health attributes and indicators for qualitative assessment. *Journal of Range Management*, *55*, 584–597.

Pyke, D. A., & Knick, S. T. (2003). Plant invaders, global change and landscape restoration. In *Paper presented at proceedings of the IVth international rangeland congress*.

Rango, A., Huenneke, L., Buonopane, M., Herrick, J. E., & Havstad, K. M. (2005). Using historic data to assess effectiveness of shrub removal in southern New Mexico. *Journal of Arid Environments*, *62*, 75–91.

Rango, A., Laliberte, A., Steele, C., Herrick, J., Bestelmeyer, B., Schmugge, T., Roanhorse, A., & Jenkins, V. (in press). UAV utilization for rangelands: Current applications and future potentials. *Environmental Practice*, in press.

Reynolds, J. F. (2001). Desertification. In S. Levin (Ed.), *Encyclopedia of biodiversity*, Vol. 2. San Diego, CA: Academic Press.

Roze, F., & Lemauviel, S. (2004). Sand dune restoration in North Brittany, France: A 10-year monitoring study. *Restoration Ecology, 12*, 29–35.

Ruiz-Jaen, M. C., & Aide, T. M. (2005). Restoration success: How is it being measured? *Restoration Ecology, 13*, 569–577.

Stringham, T. K., Krueger, W. C., & Shaver, P. L. (2003). State and transition modeling: An ecological process approach. *Journal of Range Management*, *56*, 106–113.

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., & Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature*, *418*, 671–677.

Toledo, D. (2004). *Vegetation composition and structure as an indicator of rangeland wildlife habitat quality*. MS Thesis, New Mexico State University, Las Cruces, NM.

Tongway, D. (1995). Monitoring soil productive potential. *Environmental Monitoring and Assessment*, *37*, 303–318.

Tongway, D. J., & Hindley, N. L. (2004). *Landscape function analysis: Procedures for monitoring and assessing landscapes*. Canberra, Australia: CSIRO Sustainable Ecosystems.

Tongway, D. J., Valentin, C., & Seghieri, J. (Eds.). (2001). *Banded vegetation patterning in arid and semiarid environments: Ecological processes and consequences for management* Ecological Studies, 149.

Tugel, A., Herrick, J. E., Brown, J. R., Mausbach, M. J., Puckett, W., & Hipple, K. (2005). Soil change, soil survey, and natural resources decision making: A blueprint for action. *Soil Science Society of America Journal*, *69*, 738–747.

USDA-ARS Jornada Experimental Range. (2006). Rangeland database and field data entry system. http://usda-ars.nmsu.edu/JER/Monit_Assess/monitoring.php. Accessed 6 April 2006.

Walton, M. (2005). *Spatial patterning of resource accumulation in a 22 year-old water harvesting project in the Chihuahuan Desert*. Ph.D. Thesis, University of Dayton, Dayton, OH.

Walton, M., Herrick, J. E., Gibbens, R. P., & Remmenga, M. (2001). Persistence of biosolids in a Chihuahuan Desert rangeland 18 years after application. *Arid Land Research and Management*, *15*, 223–232.

Warren, S. D. (2001). Synopsis: Influence of biological soil crusts on arid land hydrology and soil stability. In J. Belnap, & O. L. Lange (Eds.), *Biological soil crusts: Structure, function and management* (pp. 349–360). Berlin: Springer.

Zavaleta, E. S., Hobbs, R. J., & Mooney, H. A. (2001). Viewing invasive species removal in a whole-ecosystem context. *Trends in Ecology and Evolution*, *16*, 454–459.