National ecosystem assessments supported by scientific and local knowledge

Jeffrey E Herrick^{1*}, Veronica C Lessard², Kenneth E Spaeth³, Patrick L Shaver⁴, Robert S Dayton², David A Pyke⁵, Leonard Jolley⁶, and J Jeffery Goebel⁶

An understanding of the extent of land degradation and recovery is necessary to guide land-use policy and management, yet currently available land-quality assessments are widely known to be inadequate. Here, we present the results of the first statistically based application of a new approach to national assessments that integrates scientific and local knowledge. Qualitative observations completed at over 10000 plots in the United States showed that while soil degradation remains an issue, loss of biotic integrity is more wide-spread. Quantitative soil and vegetation data collected at the same locations support the assessments and serve as a baseline for monitoring the effectiveness of policy and management initiatives, including responses to climate change. These results provide the information necessary to support strategic decisions by land managers and policy makers.

Front Ecol Environ 2010; 8(8): 403-408, doi:10.1890/100017 (published online 27 Jul 2010)

L and degradation has substantiany team liveli-city of global ecosystems to support human liveliand degradation has substantially reduced the capahoods throughout the world (MA 2005). Climate change and human population growth will undoubtedly further reduce the capacity of the land to provide critical ecosystem services (UNEP 1997; MA 2005). Information on types, patterns, and severity of land degradation is urgently needed to support policy and management (McPeak 2003) and to identify those ecosystem processes that must be restored to improve the land (Geist and Lambin 2004). Assessments are currently hindered by the difficulty of determining reference conditions and by a lack of generic protocols that could generate assessment and monitoring data relevant to a broad variety of stakeholder needs (MA 2005; Heinz Center 2008), such as providing early warning of critical degradation thresholds (UNEP 1997) and changes in global biodiversity (Scholes and Biggs 2005). Furthermore, obtaining data for large-scale ecosystem assessments is both timeconsuming and costly (Marsett et al. 2006).

The problems associated with assessment and monitoring are particularly acute in grassland and savanna (rangeland) ecosystems, which are both biologically and physically diverse (Hostert *et al.* 2003). Millions of people depend on the services provided by these ecosystems,

¹US Department of Agriculture (USDA) – Agricultural Research Service, Jornada Experimental Range, New Mexico State University, Las Cruces, NM ^{*}(jherrick@nmsu.edu); ²Resources Inventory and Assessment Division, USDA – Natural Resources Conservation Service (NRCS), Ames, IA; ³Central National Technology Support Center, USDA – NRCS, Fort Worth, TX; ⁴West National Technology Support Center, USDA – NRCS, Portland, OR; ⁵US Geological Survey, Forest and Rangeland Ecosystem Science Center, Corvallis, OR; ⁶Resource Inventory and Assessment Division, USDA – NRCS, Beltsville, MD which cover 18–26% of the Earth's land surface (excluding Antarctica; Groombridge 1992). Satellite-based remotesensing systems have been successfully used to quantify short-term changes in plant cover and forage availability (Marsett *et al.* 2006; Röder *et al.* 2008), but their application to land-degradation assessment is limited by the difficulty of obtaining reliable ground-truth data (Tongway and Hindley 2004), the high variability in rainfall levels that can mask land degradation (Wessels *et al.* 2007), and a reliance on interpretation of reflectance at pixel scales that are often too coarse to interpret indicators of key degradation and recovery processes (Marsett *et al.* 2006).

Here, we present results obtained using a new approach to ecosystem assessment that addresses these challenges (Figure 1). This approach is based on the integration of recent advances across various disciplines: (1) the use of Geographic Information Systems (GIS), remote-sensing imagery, soil surveys, and climate models to stratify landscapes in a way that allows the definition of reference conditions based on the long-term ecological potential of the land (Bestelmeyer et al. 2009; Gilbert 2009; Figure 1a); (2) increased willingness and ability to integrate scientific and local knowledge (MA 2005; Reynolds et al. 2007; Figure 1c; Figure 2) for defining reference conditions (Reed et al. 2008; Fraser et al. 2006); (3) a growing understanding that sustaining ecosystem services necessary for human livelihoods depends on a relatively limited set of ecosystem attributes (NRC 1994; Holling et al. 2002; Tongway and Hindley 2004); and (4) the rapid development of new tools, including field computers and cellular phones, for recording and transmitting geo-referenced data.

The second advancement – increased willingness and ability to integrate scientific and local knowledge to define indicators and reference conditions (Figure 2) – is particularly important because empirical reference data for quantitative indicators rarely exist. Reference condi404



Figure 1. General approach to monitoring and assessment ("M and A") applied in this study. See text for detailed explanation.

tions for qualitative observational variables can, however, be developed quickly and draw upon a broader range of information than is available for quantitative indicators. These observational variables can be used to generate an initial assessment of degradation status and to provide a context for interpreting future trends in quantitative monitoring indicators. For example, increased runoff and aeolian (ie wind-borne) deposition (Figure 2c) are both easily observed but difficult to quantify. Many ranchers, farmers, native peoples, and amateur naturalists have intimate knowledge of spatial and temporal variability in observable indicators, which, when combined with a scientific understanding of properties and processes that control ecological potential, can be used to develop reliable descriptions of reference conditions for assessments (Figure 1c; Figure 2).

Several initiatives are taking advantage of the four advances outlined above to reduce the cost and increase the value of assessments in different ways. These include the UN Food and Agriculture Organization's Land Degradation Assessment in Drylands (FAO 2010), the World Overview of Conservation Approaches and Technologies (WOCAT 2010), and a US assessment of non-federal rangelands conducted by the US Department of Agriculture's (USDA) National Resources Inventory (NRI) survey program (USDA 2009). The NRI is designed to assess conditions and trends for soil, water, and related natural resources on non-federal lands, especially at the regional to national level (USDA 2009). Whereas most analyses are based on photo-interpretation of land use and land cover, field measurements are used to address specific objectives. We present key results and conclusions, based on analyses of data from over 10 000 NRI field plots. This represents the first statistically based ecosystem health assessment derived from nationally distributed field measurements and observations.

This assessment is applied at the national level, as part of a broader framework for collecting, organizing, synthesizing, and applying information and knowledge about rangeland ecosystems (Herrick *et al.* 2006a). At this scale, the assessments are used to focus attention on regions where ecological processes associated with different types of ecosystem services have been compromised. The same protocols are then used at the local level, to identify specific issues and support adaptive management (Biggs and Rogers 2004; Herrick *et al.* 2006b).

Methods

The NRI survey program is conducted by the USDA's Natural Resources Conservation Service (NRCS), in cooperation with Iowa State University's Center for Survey

Statistics and Methodology; it is scientifically based, using recognized statistical sampling methods (Nusser and Goebel 1997). Stratification and subsampling, weighting methods used for spatial extrapolation and variance estimation, and other NRI longitudinal survey techniques are described by Nusser and Goebel (1997), Nusser *et al.* (1998), and Breidt and Fuller (1999).

The 10000-plus sample sites used for this assessment are a scientifically selected subset of the 800000 total NRI sample locations; many of these sites have been observed every 5 years since 1982, but the field-data collection protocols used for this assessment were not employed by the NRI survey program until 2003. Interpretation of qualitative and quantitative results (Figure 1f) is based on statistically weighted aggregations of plot-level results into polygons through the use of Level III and IV ecoregions (US EPA 2010) as a template, where estimates for each polygon are based on measurements at a minimum of 45 field plots.

The assessments were generated from unique combinations of 17 easily observed soil and vegetation indicators of three fundamental ecosystem attributes necessary to sustain most ecosystem services: biotic integrity, hydrologic function, and soil and site stability (Pyke *et al.* 2002; Herrick *et al.* 2005; Miller 2008). Many of these indicators were also supported by quantitative measurements of key soil and vegetation properties. Hand-held computers were used to record all data at 10 091 plots, each with an area of 0.164 ha, over a 4-year period, beginning in 2003.

Scientists and managers used a combination of scientific and local knowledge to develop unique reference sheets for each of over 2100 subsets of the approximately 10 000 plots, so that each subset of plots was associated with a single ecological site, as defined by the NRCS (Bestelmeyer *et al.* 2009; Figure 1c). Plots in each subset had a similar soil- and climate-based potential to support particular types of plant communities and levels of net primary production. This means that interpretation of observations (eg of bare ground, water flow patterns, rill – small channel caused by soil erosion density, litter amount) is made locally, relative to the specific potential of a particular type of land. The flexibility of the reference sheet system will also allow climate-change effects to be integrated into future assessments. Onsite soil verification was used for final assignment of each plot to the appropriate ecological site (Figure 1a). A five-class rating system of departure from reference, ranging from "none to slight" to "extreme to total", was used to evaluate each indicator relative to its description on the reference sheet.

Quantitative measurements included vegetation cover and composition based on line-point intercept, a field test of soil-surface aggregate stability, and the size distribution of plant intercanopy gaps greater than 30 cm in length (Herrick *et al.* 2005). These measurements were selected because they are rapid, repeatable, and can be used to calculate a large number of indicators for monitoring the status of multiple ecosystem services, including those that depend on soil stability (air

<figure>

Figure 2. Illustration of the integration of local and scientific knowledge to define land potential as a reference for assessment and monitoring. (a) Deep soil profile with high water-holding capacity suggested potential for high productivity grassland with well-developed A horizon under current climate (scientific knowledge), but no records exist. (b) Kenyan tribal elders, when asked whether soil color had always been red, said "yes", but when asked to compare soil 20 years ago with current surface ([a] and inset right soil) versus remnant A horizon soil from under a tree (left center of [c] and inset left), they said "even darker than left". (c) Land potential to support a highly productive savanna was defined based on this knowledge, together with their other observations, and scientific knowledge. This definition of potential was used to support an objective assessment of the current landscape as degraded. (d) The process and results were similar to those used to develop the national assessment references in the US.

and water quality), plant-community composition and structure, and the water cycle.

Results

The qualitative assessments revealed that over 21.3 \pm 1.3% of the 158 786 000 ha of rangelands included in this study showed at least moderate departure from reference conditions for at least one of the three attributes, and 9.7 \pm 1.1% showed at least moderate departure for all three attributes (Figure 3a–c). Biotic integrity showed the most widespread departure from reference conditions, with moderate departure recorded on 18.2 \pm 1.1% of the land. Hydrologic function was second at 14.9 \pm 1.4%, followed by soil and site stability at 12.0 \pm 1.4%.

The spatial patterns in Figure 3a–c provide general information on the extent to which different types of ecosystem services from rangelands have been modified. Those services that depend on minimizing soil degradation, including soil erosion, should be relatively intact across much of the northern US (Figure 3c), whereas greater changes are likely to have occurred in those services that depend on a diverse, productive, native plant

community (Figure 3a). In the more arid US Southwest, degradation of both soils and vegetation have important implications for the capacity of the land to provide a wide variety of ecosystem services, including those related to water (Figure 3b).

These patterns are comparable to those reported in the only other published, broad-scale study based on this assessment protocol – that of Miller (2008), which revealed that 44.6% of plots distributed across 760 000 ha of federally owned arid and semiarid rangelands in southern Utah showed at least moderate departure from reference conditions, with the biotic integrity attribute showing the greatest amount of change. These federal rangelands are located in an area where relatively high levels of departure from reference conditions were also found on the non-federal rangelands (Figure 3).

Many of the general conclusions, which are based on qualitative assessments, are further supported by quantitative data, which also provide a more precise baseline for monitoring. The qualitative assessments showing that biotic integrity (Figure 3a) is compromised across larger areas than soil and site stability (Figure 3c) are supported by quantitative data. The quantitative data show that





Figure 3. Results of land-degradation assessment relative to reference conditions (a-c) and status of key quantitative indicators (d-i) for non-federal rangelands in the US. Proportion of rangeland where (a) biotic integrity, (b) hydrologic function, and (c) soil and site stability were rated moderately degraded or worse, relative to the reference. (d) Proportion of land where non-native species are present and (e) comprise over 50% of plant cover. (f) Proportion of land where non-native annual Bromus species are present. (g) Bare ground, (h) proportion of soil surface in large (>1m) intercanopy gaps, and (i) proportion of soil surface covered by soil aggregates with low stability in water (field test < 3; Herrick et al. 2005).

non-native species, which negatively affect biotic integrity, are now present on $48.5 \pm 1.4\%$ of the land (Figure 3d) and represent over 50% of total plant cover on $5.3 \pm 0.5\%$ (Figure 3e). Non-native species often negatively affect biotic integrity (Figure 3a) by modifying plant-community structure, vegetation production, and nutrient cycling, and, in many cases, by making arid and semiarid ecosystems less resilient through increased fire frequency and intensity (Brooks et al. 2004). These results are of particular interest because strategies to combat land degradation in US rangelands have largely focused on soil stabilization beginning prior to the establishment of the Soil Conservation Service and the Civilian Conservation Corps in the 1930s (Salmond 1967), and efforts to control soil erosion, increase rangeland productivity, and stabilize roadsides often included

lack a systematic sampling design.

In the southwestern US, widespread loss of hydrologic function (Figure 3b) was reflected in observed indicators of bare ground, increased susceptibility to soil physical crusting associated with a loss of soil-aggregate stability, and replacement of perennial grasses with shrubs and trees, which increases hydrologic connectivity in these ecosystems (Ares et al. 2003; Turnbull et al. 2008; Okin et al. 2009). These qualitative indicators were supported by quantitative data for the same region that reflect a combination of lower ecological potential associated with low rainfall and land degradation. These indicators include the proportion of bare ground (Figure 3g), the proportion of the land exposed in large intercanopy gaps (Figure 3h), and soil-aggregate stability based on a rapid field test (Figure 3i). Gap size distribution is an index of spatial

the use of non-native vegetation (Forman 2003).

Discussion

biotic integrity and quantitative vegetation cover and composition data provide new information about the extent to which non-native species have modified ecosystems in different parts of the country (Sakai et al. 2001). However, biotic integrity (Figure 3a) cannot be entirely explained by non-native species dominance (Figure 3e). In some cases, such as the arid to semiarid southwestern US. loss of biotic integrity is associated with increased dominance of native invasive species, such as honey mesquite (Prosopis glandulosa) and juniper (Juniperus spp). More detailed information on the spatial distribution of individual species or groups of species can also be extracted from the data (eg Bromus spp in Figure 3f), to provide the additional information necessary to prioritize management efforts at the national level and to update existing databases. For example, species distributions reported by both the National Institute of Invasive Species Science (NIISS 2010) and the USDA PLANTS Database (USDA 2010) are based on anecdotal observations that are generally not standardized at the national level. Also, other invasive species databases either are not comprehensive or

407

vegetation pattern, which is increasingly cited as a sensitive indicator of critical threshold transitions (Scheffer *et al.* 2009), in part because larger gaps are more likely to be hydrologically connected. These uninterrupted gaps increase the rate of water movement and create runoff during intense storms, effectively reducing infiltration and water available to plants (Turnbull *et al.* 2008). These spatial patterns are also related to wildlife habitat structure (Toledo *et al.* 2010) and wind erosion susceptibility (Okin *et al.* 2009). Soil-aggregate stability reflects soil resistance to erosion. Quantitative data based on standardized methods also provide a more precise baseline for monitoring; the data are used as inputs for wind and water erosion models and may also be used to predict the spread of invasive species and impacts of climate change.

Because they are based on aggregations of assessments relative to site-specific potential, the maps in Figure 3a–c can be used to identify those parts of the country where policy and management interventions may have the greatest impact, based on current degradation status. They can also be used to support continuation of policies and general management practices that are being applied in parts of the country where little or no degradation has occurred. This type of data allows a science-based discussion of policy and management objectives and comparisons of potential tradeoffs among different ecosystem services, such as the relative costs (to biodiversity conservation) versus the benefits of using non-native species for soil stabilization and to promote water infiltration.

Modelling based on these quantitative data (Figure 3g-i and especially bare ground) can then be used to predict the effects of different types of interventions, and to support cost-benefit analyses prior to policy implementation. The US NRI Rangelands study illustrates how assessments of land degradation and recovery that integrate local and scientific knowledge can be completed across large areas through the application of a spatially unbiased statistical design that includes qualitative assessments and quantitative data. Spatially unbiased designs facilitate scaling while allowing for integration with remote-sensing-based approaches, which are currently being considered for the NRI and other assessment and monitoring programs. The process of integrating local and scientific knowledge in the development of reference information for assessments also increases local involvement and commitment (Stafford Smith et al. 2007), and provides opportunities for adapting assessment and monitoring to local degradation and recovery processes and information needs (Figure 2).

Although the example presented here relied on many dedicated data collectors and therefore had a relatively high cost, the basic approach could be adapted for application by a diverse network of land managers. Recent advances in cellular phone and Global Positioning System technologies provide the opportunity for individuals with limited formal training to collect and transmit data on soil-surface conditions and vegetation composition and structure at specific locations in the course of their daily activities. Geolocated data and photographs facilitate data verification and quality control, whereas the ability to make local knowledge spatially explicit and electronically searchable through annotated photographs opens the door to a new source of metadata for interpretation (FAO 2010). Spatially explicit local knowledge is particularly important for identification of thresholds or tipping points (UNEP 1997; Gillson and Hoffman 2007; Bestelmeyer et al. 2009), because these thresholds often depend on spatially and temporally variable management systems, which are rarely documented. Local knowledge is also critical for determining the relative importance and relevance of different ecological thresholds and for defining thresholds of potential concern, which can be used to strategically adapt management techniques (Biggs and Rogers 2004).

Widespread availability of information and communication technologies, an increased understanding of the value of local knowledge, and a willingness to standardize methods across regions to realize the benefits of spatial data integration are key to emerging approaches to assessment and monitoring. Data from programs such as the NRI can be used alone or as future inputs for integrated assessment models, as part of ongoing efforts to develop local, national, and international land-degradation assessment and monitoring systems. Future efforts must continue to combine remote-sensing and field-based approaches to biophysical data collection with increased understanding of socioeconomic and cultural patterns and processes (Reynolds *et al.* 2007; Verstraete *et al.* 2009), to focus attention on areas at or near a threshold or tipping point.

Acknowledgements

We thank D Oman for creating the maps and the large number of NRCS employees who helped with field-data collection and interpretation. H Mooney, J Reynolds, D Peters, M Miller, J Belnap, S Phillips, and D Thompson provided helpful comments on the paper.

References

- Ares J, Bertiller M, and Bisigato A. 2003. Modeling and measurement of structural changes at a landscape scale in dryland areas. *Environ Model Assess* 8: 1–13.
- Bestelmeyer BT, Tugel AJ, Peacock GL, et al. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Rangeland Ecol Manag 62: 1–15.
- Biggs HC and Rogers KM. 2004. An adaptive system to link science, monitoring and management in practice. In: du Toit J, Rogers KM, and Biggs HC (Eds). The Kruger experience: ecology and management of savanna heterogeneity. London, UK: Taylor and Francis.
- Breidt FJ and Fuller WA. 1999. Design of supplemented panel surveys with application to the National Resources Inventory. *J Agric Biol Envir St* **4**: 391–403.
- Brooks ML, D'Antonio CM, Richardson DM, et al. 2004. Effects of invasive alien plants on fire regimes. BioScience 54: 677–88.
- FAO (Food and Agriculture Organization). 2010. Land degradation assessment in drylands. Rome, Italy: FAO. www.fao. org/nr/lada/. Viewed 10 Jun 2010.

- Fraser EDG, Dougill AJ, Mabee WE, et al. 2006. Bottom up and top down: analysis of participatory processes for sustainability indicator identification as a pathway to community empowerment and sustainable environmental management. J Environ Manage 78: 114–27.
- Geist HJ and Lambin EF. 2004. Dynamic causal patterns of desertification. *BioScience* **54**: 817–29.
- Gilbert N. 2009. Digital soil map for Africa launched. NatureNews. www.nature.com/news/2009/090113/full/news. 2009.17.html. Viewed 7 Feb 2010.
- Gillson LG and Hoffman MT. 2007. Rangeland ecology in a changing world. Science 315: 53–54.
- Groombridge B. 1992. Global biodiversity: status of the Earth's living resources. London, UK: Chapman and Hall.
- Heinz Center. 2008. Report of the state of the nation's ecosystems. Washington, DC: Heinz Center.
- Herrick JE, Bestelmeyer BT, Archer S, *et al.* 2006a. An integrated framework for science-based arid land management. *J Arid Environ* **65**: 319–35.
- Herrick JE, Schuman GE, and Rango A. 2006b. Monitoring ecological processes for restoration projects. J Nat Conserv 14: 161–71.
- Herrick JE, Van Zee JW, Havstad KM, *et al.* 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Vol 1: quick start. Tucson, AZ: University of Arizona Press.
- Holling CS, Gunderson LH, and Peterson GD. 2002. Sustainability and panarchies. In: Gunderson LH and Holling CS (Eds). Panarchy: understanding transformations in human and natural systems. Washington, DC: Island Press.
- Hostert P, Röder A, and Hill J. 2003. Coupling spectral unmixing and trend analysis for monitoring of long-term vegetation dynamics in Mediterranean rangelands. *Remote Sens Environ* **87**: 183–97.
- MA (Millennium Ecosystem Assessment). 2005. Ecosystems and human well-being: current state and trends. Washington, DC: Island Press.
- Marsett RC, Qi J, Heilman P, *et al.* 2006. Remote sensing for grassland management in the arid Southwest. *Rangeland Ecol Manag* **59**: 530–40.
- McPeak JG. 2003. Analyzing and addressing localized degradation. Land Econ **79**: 515–36.
- Miller ME. 2008. Broad-scale assessment of rangeland health, Grand Staircase–Escalante National Monument, USA. *Rangeland Ecol Manag* **61**: 249–62.
- NIISS (National Institute of Invasive Species Science). 2010. http://ibis-live.nrel.colostate.edu/cwis438/Browse/BrowseData. php?WebSiteID=1. Viewed 30 Aug 2010.
- NRC (National Research Council). 1994. Rangeland health: new methods to classify, inventory, and monitor rangelands. Washington, DC: National Academy Press. http://www.nap.edu/openbook.php?record_id=2212. Viewed 10 Jun 2010.
- Nusser SM, Breidt FJ, and Fuller WA. 1998. Design and estimation for investigating the dynamics of natural resources. *Ecol Appl* 8: 234–45.
- Nusser SM and Goebel JJ. 1997. The National Resources Inventory: a long-term multi-resource monitoring programme. *Environ Ecol Stat* 4: 181–204.

- Okin GS, Parsons AJ, Wainwright J, et al. 2009. Do changes in connectivity explain desertification? *BioScience* **59**: 237–44.
- Pyke DA, Herrick JE, Shaver P, and Pellant M. 2002. Rangeland health attributes and indicators for qualitative assessment. *J Range Manage* **55**: 584–597.
- Reed MS, Dougill AJ, and Baker TR. 2008. Participatory indicator development: what can ecologists and local communities learn from each other? *Ecol Appl* 18: 1253–69.
- Reynolds JF, Stafford Smith DM, Lambin EF, *et al.* 2007. Global desertification: building a science for dryland development. *Science* **316**: 847–51.
- Röder A, Udelhoven T, Hill J, et al. 2008. Trend analysis of Landsat-TM and -ETM+ imagery to monitor grazing impact in a rangeland ecosystem in northern Greece. *Remote Sens Environ* **112**: 2863–75.
- Sakai AK, Allendorf FW, Holt JS, et al. 2001. The population biology of invasive species. Annu Rev Ecol Syst **32**: 305–32.
- Salmond JA. 1967. The civilian conservation corps, 1933–1942: a new deal case study. Durham, NC: Duke University Press.
- Scheffer M, Bascompte J, Brock WA, et al. 2009. Early-warning signals for critical transitions. Nature 461: 53–59.
- Scholes RJ and Biggs RA. 2005. Biodiversity intactness index. Nature 434: 45–49.
- Stafford Smith DM, McKeon GM, Watson IW, et al. 2007. Learning from episodes of degradation and recovery in variable Australian rangelands. P Natl Acad Sci USA 104: 20 690–95.
- Toledo DP, Herrick JE, and Abbott LB. 2010. Assessment and monitoring of habitat structure: a comparison of cover pole with standard, widely applied vegetation monitoring methods. *J Wildlife Manage* **74**: 600–04.
- Tongway DJ and Hindley NL. 2004. Landscape function analysis: procedures for monitoring and assessing landscapes. Canberra, Australia: CSIRO Sustainable Ecosystems.
- Turnbull L, Wainwright J, and Brazier RE. 2008. A conceptual framework for understanding semi-arid land degradation: ecohydrological interactions across multiple space and time scales. *Ecohydrology* 1: 23–34.
- UNEP (UN Environment Programme). 1997. World atlas of desertification, 2nd edn. London, UK: Edward Arnold.
- USDA (US Department of Agriculture). 2009. Summary report: 2007 National Resources Inventory. Washington, DC: Natural Resources Conservation Service. Ames, IA: Center for Survey Statistics and Methodology, Iowa State University.
- USDA (US Department of Agriculture). 2010. USDA PLANTS database. http://plants.usda.gov/. Viewed 7 Feb 2010.
- US EPA (US Environmental Protection Agency). 2010. Western Ecology Division, ecoregion, maps and GIS resources. www.epa.gov/wed/pages/ecoregions.htm. Viewed 7 Feb 2010.
- Verstraete MM, Scholes RJ, and Stafford Smith M. 2009. Climate and desertification: looking at an old problem through new lenses. *Front Ecol Environ* **7**: 421–28.
- Wessels KJ, Prince SD, Malherbe J, et al. 2007. Can humaninduced land degradation be distinguished from the effects of rainfall variability? J Arid Environ 68: 271–97.
- WOCAT (World Overview of Conservation Approaches and Technologies). 2010. www.wocat.net/. Viewed 10 Jun 2010.