# Desertification, land use, and the transformation of global drylands

**Brandon T Bestelmeyer[1*], Gregory S Okin[2], Michael C Duniway[3], Steven R Archer[4], Nathan F Sayre[5], Jebediah C Williamson[1], and Jeffrey E Herrick[1]**

**Desertification is an escalating concern in global drylands, yet assessments to guide management and policy responses are limited by ambiguity concerning the definition of "desertification" and what processes are involved. To improve clarity, we propose that assessments of desertification and land transformation be placed within a state change–land-use change (SC–LUC) framework. This framework considers desertification as state changes occurring within the context of particular land uses (eg rangeland, cropland) that interact with land-use change. State changes that can be readily reversed are distinguished from regime shifts, which are state changes involving persistent alterations to vegetation or soil properties. Pressures driving the transformation of rangelands to other types of land uses may be low, fluctuating, or high, and may influence and be influenced by state change. We discuss how the SC–LUC perspective can guide more effective assessment of desertification and management of drylands.**

*Front Ecol Environ* 2015; 13(1): 28–36, doi:10.1890/140162

Managing ecosystem state change is among the great challenges humans face in the 21st century (Chapin *et al.* 2010). In the broad sense used in this paper, a state change is any important shift in species abundance, soils, or ecosystem processes in response to disturbance or alterations in environmental conditions (Scheffer and Carpenter 2003; Bestelmeyer *et al.* 2011). State changes may be gradual or abrupt and reversible or persistent. Persistent state changes are known as regime shifts (Scheffer and Carpenter 2003). Given their propensity for generating wide-ranging negative effects, understanding regime shifts is vital to management and policy.

## In a nutshell:

- Desertification in drylands is an important problem worldwide, but the concept is ambiguous in terms of specific processes, conditions, and solutions
- We propose a state change–land-use change (SC–LUC) framework – wherein detailed models of vegetation and soil change (ie state change) are combined with an understanding of land-use change – as a broad, process-oriented way of thinking about the transformation of drylands
- Because some state changes are rapidly or gradually reversible whereas others are effectively permanent, land managers should distinguish among types of state change when prioritizing restoration investments
- Shifts between rangeland, cropland, and urban land uses can cause or be caused by state change, so land-use planners should recognize the potential consequences of state change
- Region-specific information delivery about SC–LUC interactions may be the best hope for mitigating desertification and guiding dryland transformations

[1]*US Department of Agriculture–Agricultural Research Service (USDA-ARS) Jornada Experimental Range, New Mexico State University, Las Cruces, NM* [*]*(bbestelm@nmsu.edu);* [2]*Department of Geography, University of California, Los Angeles, Los Angeles, CA;* [3]*US Geological Survey, Southwest Biological Science Center, Moab, UT;* [4]*School of Natural Resources and the Environment, University of Arizona, Tucson, AZ;* [5]*Department of Geography, University of California, Berkeley, Berkeley, CA*

Drylands are especially prone to state changes as a result of scarce, variable rainfall and low soil fertility (Reynolds *et al.* 2007). Undesirable state changes in drylands are most often described using the term "desertification", the importance of which has been promoted by the United Nations Convention to Combat Desertification. Definitions of desertification emphasize persistent and severe reductions in biological productivity due to unsustainable land uses, often associated with climatic and societal factors such as poverty and migration (Veron *et al.* 2006; Verstraete *et al.* 2009). On the basis of these definitions, desertification is a type of regime shift that occurs in drylands (Scheffer and Carpenter 2003). However, applications of the term desertification to particular cases are fraught with vagueness and inconsistency (Panel 1; Veron *et al.* 2006). There is often no clear statement regarding which ecosystem attributes have changed, the timescales over which degradation or recovery can occur, or the drivers involved (eg grazing, fire, deforestation, cropland agriculture, or infrastructure development; see Geist and Lambin 2004). Desertification (and the broader term "land degradation") is often treated in a qualitative fashion (is/is not or slight/moderate/severe; Veron *et al.* 2006). This lack of specificity explains why assessments of the extent of desertification range from 4–74% globally (Safriel 2007) and, in the case of Mongolia, from 9–90% (Addison *et al.* 2012). The inconsistent relationship between the term "desertification" and specific land conditions limits the implementation of solutions at local to international levels (Reynolds *et al.* 2011).

 

We propose a new conceptual approach, wherein desertification and other land transformations are placed into a "state change–land-use change" (SC–LUC) framework; this would alleviate much of the current confusion. Conceptual models for state change, known as state-and-transition (S&T) models (Westoby *et al.* 1989), have been produced for many ecosystems (Hobbs and Suding 2009). S&T models use box-and-arrow diagrams accompanied by data-supported narratives to describe states (boxes) and the ecological processes driving change within and between states (arrows). These models address both reversible and irreversible changes – the details of which are critical for understanding desertification and communicating in a meaningful way about it – and organize the available information effectively. They are also highly adaptable, improving as new information becomes available. With enough information, S&T models can be extremely detailed, providing clear statements of historical and alternative states alongside mechanisms of change and timescales for potential recovery (eg Miller *et al.* 2011; Rumpff *et al.* 2011). This approach has the advantage of requiring that a so-called "desertified" state must be defined with respect to some other "non-desertified" or historical/reference state (Panel 1). S&T models, however, tend to be developed for particular land uses, primarily rangeland and wildland (hereafter combined as "rangeland"). Consequently, they provide a limited perspective on the causes and consequences of shifts among land uses (Geist and Lambin 2004; Reynolds *et al.* 2007; Sayre *et al.* 2013). Broadening S&T models developed by ecologists to encompass land-use change processes addressed by geographers (eg Quétier *et al.* 2007) is the basis for our proposed SC–LUC framework.

Models relevant to management and policy development must extend across land uses because conversion to unsustainable uses is often (directly or indirectly) associated with desertification, such as when conversion from rangeland to cropland results in an eroded, unproductive state (Herrick *et al.* 2012). Furthermore, state change occurring within rangelands can interact with other land uses spatially and temporally. Rangeland state change can trigger conversion to cropland or urban land uses, and abandoned cropland may revert to certain rangeland states (Cramer *et al.* 2008). Land conversion to cropland or urban uses can also cause state changes in adjacent rangelands (Lambin and Meyfroidt 2011). The combined effects of SC–LUC produce the land-cover mosaics that we seek to manage.

Here we propose concepts to unite S&T and land-use-change perspectives to define a SC–LUC framework for drylands (Verstraete *et al.* 2009). Our approach begins with a review of the biophysical basis of state change occurring within rangelands and croplands. We then introduce a novel classification of land-use change between rangeland and other land uses, such as cropland and urban development (Peters *et al.* 2015), and highlight the implications of SC–LUC interactions. We argue that a process-specific and integrated SC–LUC perspective will be necessary to guide the stewardship of drylands into the future.

## State change within rangelands and croplands

### *Rangelands*

State change in rangeland systems follows one of three recognized patterns: equilibrium, non-equilibrium, and regime-shift (or threshold) dynamics (Briske *et al.* 2003). In systems that exhibit equilibrium dynamics, the historical or desired state is resilient to disturbance. Changes caused by grazing pressure are readily reversed as a consequence of fertile and erosion-resistant soils alongside plant traits that promote survival and recovery (Cingolani *et al.* 2005). In non-equilibrium systems, vegetation cover can fluctuate widely due to high interannual rainfall variability (von Wehrden *et al.* 2011). In these systems, graz-

**Panel 1. What is desertification?**

Desertification is a controversial term; even the authors of this paper disagreed about what it should mean. The UN Convention to Combat Desertification defines desertification as land degradation, which is the "reduction or loss in arid, semi-arid, and dry sub-humid areas, of the biological or economic productivity and complexity of rain-fed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as: (i) soil erosion caused by wind and/or water; (ii) deterioration of the physical, chemical, and biological or economic properties of soil; and (iii) long-term loss of natural vegetation" (UNCCD 1994). The major problems with this definition are rooted in (1) reference conditions and (2) reversibility of change. Assessing desertification requires a comparison to a non-degraded, reference condition. Establishing the reference condition can fall prey to errors and deliberate manipulation. The characteristics of a historical state for an area can be misrepresented by using dubious historical accounts to assert (or imply) that an area was formerly highly productive when in fact it featured low and variable productivity similar to the current "desertified state" (eg it is a natural desert). Even when there is adequate evidence for state change, historical states existed in the context of broader-scale and longer-term environmental change, so that more productive states may no longer be possible. Desertification is often referred to as a "long-term" phenomenon, but how long that time period should be is often unclear. In equilibrium and non-equilibrium ecosystems, persistent absence of vegetation can be caused by continued land-use pressure or time lags in recovery, yet under the proper conditions recovery can be rapid. The recent rapid greening of parts of the Sahel (Dardel *et al.* 2014), for example, begs the question of whether those areas should have been considered "desertified" in the first place. But narrowing desertification to only difficult-to-reverse (ie regime-shift) change might limit support for instances where change *can* be reversed given adequate policy and financial support. This is critical where interventions can prevent further degradation that might ultimately result in biophysical or societal regime shifts. For these reasons, we argue that greater attention should be given to the specifics of state change, the drivers involved, and the potential for recovery under a broad range of investment scenarios.

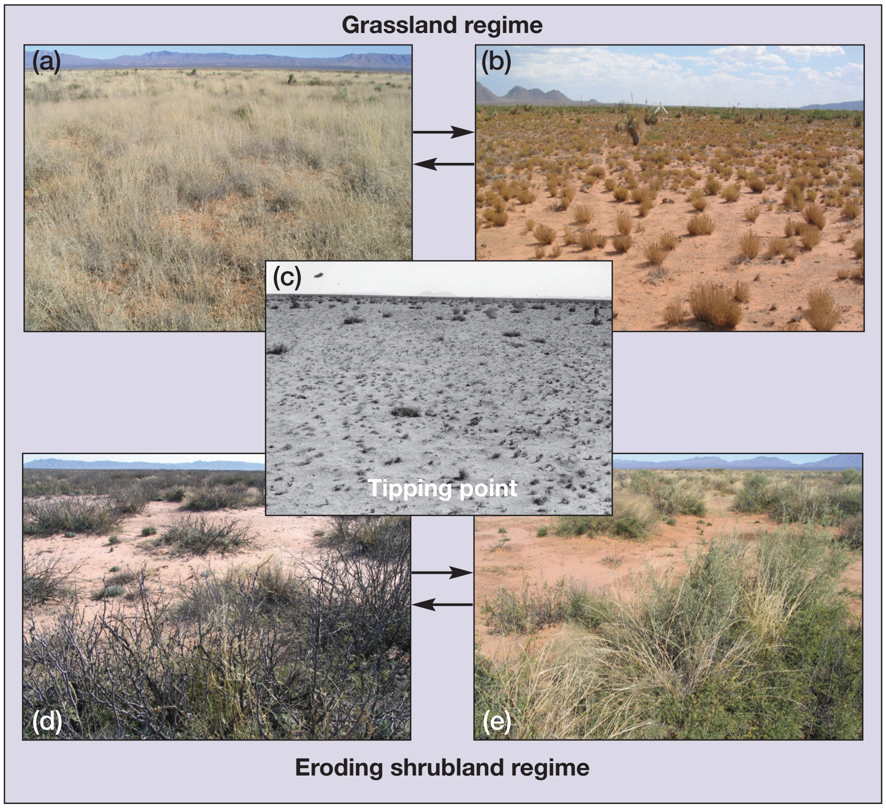


***Figure 1.*** *A regime shift observed in the Chihuahuan Desert of New Mexico, based on observations at the Jornada Experimental Range near the city of Las Cruces. (a) High cover of black grama grass (*Bouteloua eriopoda*) in the historical state can be reduced (b) and subsequently recovered, unless reduced below a tipping point (c). Past this point,* B eriopoda *is locally extirpated and unpalatable shrubs such as mesquite (*Prosopis glandulosa*) become dominant. This results in an eroding shrubland regime (d) that experiences infrequent co-dominance by another perennial grass (*Sporobolus*, [e]) during periods of high rainfall (see Bestelmeyer* et al. *[2011] and references therein).*

ing effects on vegetation may be secondary to weather effects, owing to the inability of livestock to reduce plant population densities when forage and drinking water are periodically limited and livestock migrate or die (Illius and O'Connor 1999). Non-equilibrium systems can be resilient (or "non-equilibrium persistent") because soil degradation and biodiversity loss attributable to livestock impacts are limited (Fernandez-Gimenez and Allen-Diaz 1999).

In contrast, regime shifts (also known as crossing a tipping point or threshold) involve persistent changes in vegetation structure and soils (ie "bistability"; Figure 1). Recovery of the former state after crossing a tipping point – if possible at all – is largely dependent on active restoration (Briske *et al.* 2003; Scheffer and Carpenter 2003). Dryland regime shifts can be caused by severe, widespread disturbances that limit recruitment of formerly dominant plants, such that competing invaders can persist and come to dominate instead (Seabloom *et al.* 2003). Alternatively, the reduction or cessation of natural disturbances, such as fire in grasslands and savannas, can promote the establishment of woody plants that may expand to a density or size beyond which fire is no longer effective in recovering a grassland/savanna state (Figure 2; D'Odorico *et al.* 2012). These two types of regime shifts involve changes in the dominant vegetation without a collapse of overall vegetation production. While primary production and carbon stocks may be maintained (or even increased) with such transformations (Barger *et al.* 2011), the provision of other ecosystem services (eg forage for livestock production) is dramatically altered (Eldridge *et al.* 2011) and may trigger changes in land use.

Collapse in vegetation production can occur when the loss of dominant perennial plants leads to a reduction in soil water infiltration, accelerated erosion that reduces soil fertility, rising water tables resulting in salinization, or even changes in local climate (D'Odorico *et al.* 2013). Regime shifts associated with soil degradation (Figure 3a) most closely align with current definitions of desertification. The occurrence of regime shifts is more likely when plant–soil feedbacks are important in maintaining alternative states and when soil, chemical, hydrological, and climatic processes are strongly coupled to plants that dominate in the historical state. In contrast to equilibrium/non-equilibrium change, regime shifts associated with soil degradation occur on sites with erodible soil surfaces, as well as in soils characterized by root-limiting horizons at shallow depths, or subsoil and groundwater salinity.

### *Implications*

In equilibrium and non-equilibrium change, vegetation recovery can be initiated by adjustments to existing management practices or in response to weather events that favor plant growth and recruitment (Lewis *et al.* 2010). Asserting that a site is desertified may discourage the initiation of changes in management that could readily achieve recovery (Bestelmeyer 2006). Thus, contentions that a regime shift has taken place (or could take place) should specify the particular mechanism(s) that preclude recovery, including recruitment limitation of dominant plant species, shifts in dominance controlled by plant–environment feedbacks, or changes to soil properties, all of which can be demonstrated experimentally (Bestelmeyer *et al.* 2009). Articulation of the specific state-change mechanisms will help determine what interventions are needed to halt degradation, prevent regime shifts, and begin the process of recovery (where societal conditions permit such interventions; reviewed in Geist and Lambin 2004; Reynolds *et al.* 2007). An understanding of these mechanisms can also be a basis for deciding whether permanent transformation of the ecosystem and its uses (ie to a novel ecosystem) should be acknowledged (Hobbs *et al.* 2011), so that management resources could be directed to areas where they might do the most good.

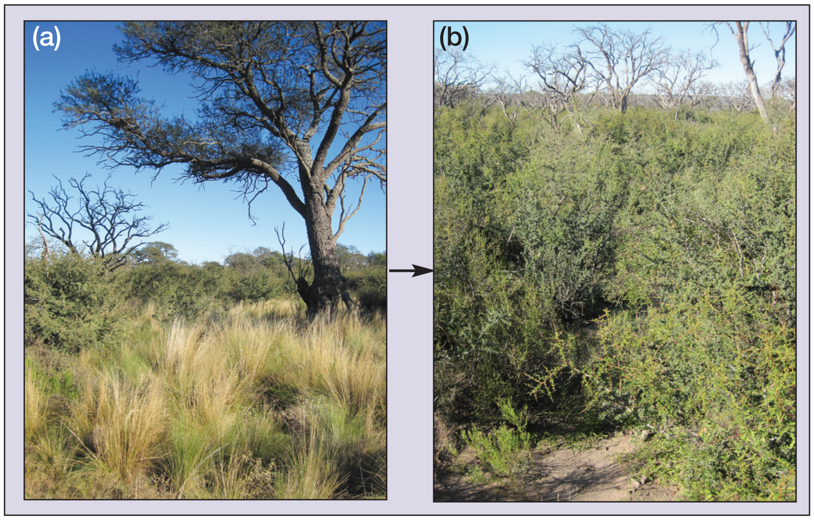


***Figure 2.*** *A regime shift from (a) open forest to (b) dense shrubland following a catastrophic fire in the Calden forests of central Argentina (Dussart* et al. *1998). This regime shift does not involve soil degradation, and restoration is possible.*

### *Croplands*

Vegetation in croplands is directly manipulated, and thus the variables defining state include a suite of soil properties – collectively known as soil quality – that affect crop yields, including soil organic carbon (SOC) or matter, soil structure, and infiltration rates (Seybold *et al.* 1999). In this context, "soil resilience" is the capacity of a soil to recover historical soil quality after disturbances (eg annual cropping; Seybold *et al.* 1999). Recovery of soil quality governs potential crop yields, given rainfall and other inputs (eg fertilizer; Lal 2001). Croplands exhibiting equilibrium dynamics maintain soil resilience through variations in management, such that recommended management practices (such as no-tillage cropping, winter cover crops, residue retention) lead to the recovery of soil quality indicators (eg SOC) toward levels observed in uncropped rangeland (Figure 3b; Tugel *et al.* 2005).

Alternatively, a regime shift can occur under cropland use beyond which soil quality can no longer recover. Regime shifts arise when soil erosion leads to persistent changes in the soil profile, including altered soil texture and reductions in soil depth, water-holding capacity, and nutrient availability. Altered soil-profile properties subsequently constrain plant production, which then limits recovery of SOC and other soil quality indicators, producing feedbacks that further restrict crop production (Lal 2001). Reduced crop yields associated with a regime shift may promote cropland abandonment (ie reverting to rangeland land use) and continued soil erosion (Bakker 2005). Regime shifts in croplands occur under soil conditions similar to those favoring regime shifts in rangelands (Seybold *et al.* 1999).

### *Implications*

Recovery of soil quality can be promoted by adoption of sustainable crop management practices or by conversion back to rangeland vegetation at sites where soil loss is minimal. Soils that are shallow to bedrock, hardpans, or soil horizons high in salts can be permanently altered by soil erosion (Lal 2001). Thus, knowledge of soil-profile characteristics can aid in assessing the potential for recovery of degraded croplands toward historical levels of productivity, as well as in evaluating the risk of a regime shift resulting from rangeland-to-cropland conversion (Herrick *et al.* 2013). There is insufficient information, however, on tipping points in soil variables that cause large changes in crop yields or that limit rangeland recovery (eg tolerable changes in soil depth or SOC content; Arshad and Martin 2002).

## Land-use change to and from rangelands

Changes in land use result from interactions between various socioeconomic and cultural pressures and biophysical factors. These interactions can have important direct and indirect effects on state change (including regime

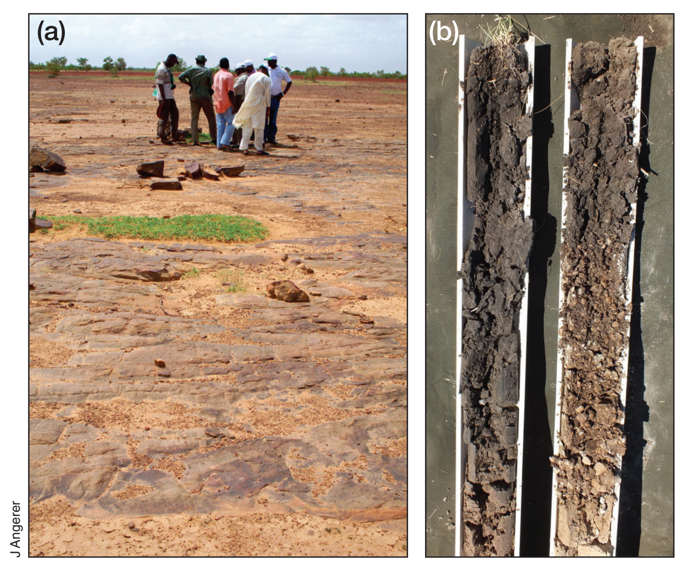


***Figure 3.*** *(a) A rangeland near Bandiagara, Mali, that has experienced soil erosion to bedrock, a regime shift from which recovery of the historical state is unlikely. (b) Two soil cores from adjacent sites on the same soil series (Kube) in southern South Dakota. The core on the left is from a well-managed rangeland, whereas the core on the right is from a long-term (50-yr) hayland; note loss of soil organic matter-darkened soil in the hayland core (which may or may not be associated with a regime shift).*

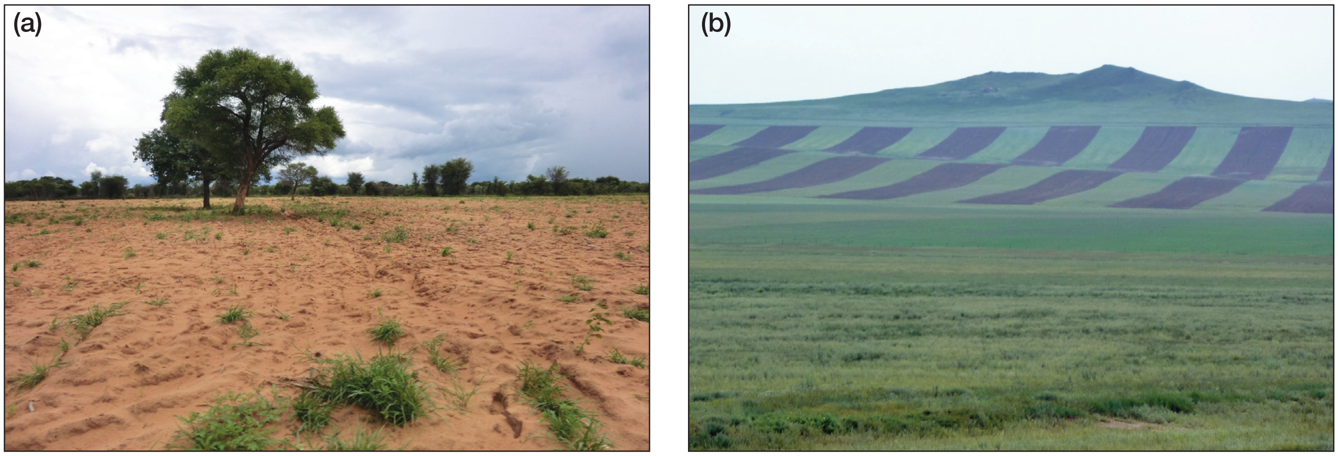


***Figure 4.*** *(a) Rangeland recently converted to cropland in the Kavango region of northeast Namibia, featuring low levels of maize production on sandy soils experiencing extensive wind and water erosion. (b) Recent conversion of rangeland to strip-farming for rapeseed (*Brassica *spp) outside of Ulaanbaatar, Mongolia.*

shifts). To integrate land-use change pressure into our proposed SC–LUC framework, we introduce a new classification scheme that recognizes low, fluctuating, and high pressures to convert rangeland to other uses.

### *Low conversion pressure*

Low pressure to convert from rangeland use can be attributed to inherent low potential for other uses, societal limitations, or institutional barriers. For example, systems featuring very low productivity – due either to natural biophysical limitations (eg low rainfall, shallow and rocky soils, steep slopes) or to soil degradation associated with past land uses (Bakker 2005) – are not economically viable for cropland. Institutional limitations with regard to accessibility and infrastructure, political conflict, and land-tenure issues may also limit conversion to more management-intensive (and often capital-intensive) land uses, even when demand is high (Sayre *et al.* 2013). Increasing land scarcity in the future may overcome some of these limitations as the values of alternative land uses increase (Lambin and Meyfroidt 2011). Finally, legal and regulatory mechanisms, such as conservation easements or protective government status (eg Protected Areas), can preclude land-use change, at least in the short term, given sufficiently strong social institutions.

### *Fluctuating land-use pressure*

Land use can fluctuate in response to socioeconomic factors, including changes in demographics, income and investment opportunities, migration, land-tenure systems, commodity and agricultural input prices, and conservation policies. Decadal-scale climate variability, such as drought or increased rainfall, further contributes to fluctuations. Rangelands converted to cropland agriculture (Figure 4) may be abandoned and revert back to rangeland in areas of marginal productivity, including areas featuring soil limitations or rainfall quantity that is inadequate for dryland farming (< 700 mm $yr^{-1}$), and in which irrigation has not been feasible, available, or sustainable (Lambin *et al.* 2013). Cropland-to-rangeland reversion can result in the recovery of natural vegetation and be followed by cycles of land conversion. For instance, the US Dust Bowl (a decade-long event during the 1930s) resulted from a variety of factors, including new technologies, increased demand for corn and wheat, government policies encouraging cultivation and homesteading, immigration, and high rainfall. The confluence of these factors in the 1920s led to widespread, rapid conversion of rangeland to cropland, much of which was subsequently abandoned during the drought and economic depression of the 1930s (Worster 2004). Most marginal abandoned cropland was allowed to recover naturally to rangeland, or was actively restored through the Conservation Reserve Program (CRP) initiated in 1985 (Munson and Lauenroth 2012). Nevertheless, increasing demand for biofuel production and the expiration of many CRP contracts in 2012 are accelerating the reconversion of grasslands to croplands in the Great Plains (Stubbs 2013; Wright and Wimberly 2013). However, soil degradation in areas of low soil resilience may limit rangeland recovery and preclude reconversion to cropland, leading to novel ecosystem states of relatively low value (Jackson *et al.* 1991; Bakker 2005).

### *High conversion pressure*

Strong pressure for conversion from rangeland to cropland, or from rangeland or cropland to urban uses, is associated with increases in land values adjacent to urban areas (including peri-urban and exurban areas), proximity to infrastructure that facilitates development or resource exportation (eg irrigation water for croplands, powerlines for energy development, roads), or changes in technology, policies, and market prices (Figure 5; York *et al.* 2011). High rangeland-to-cropland conversion rates occur when institutional, local economic, and cultural

 

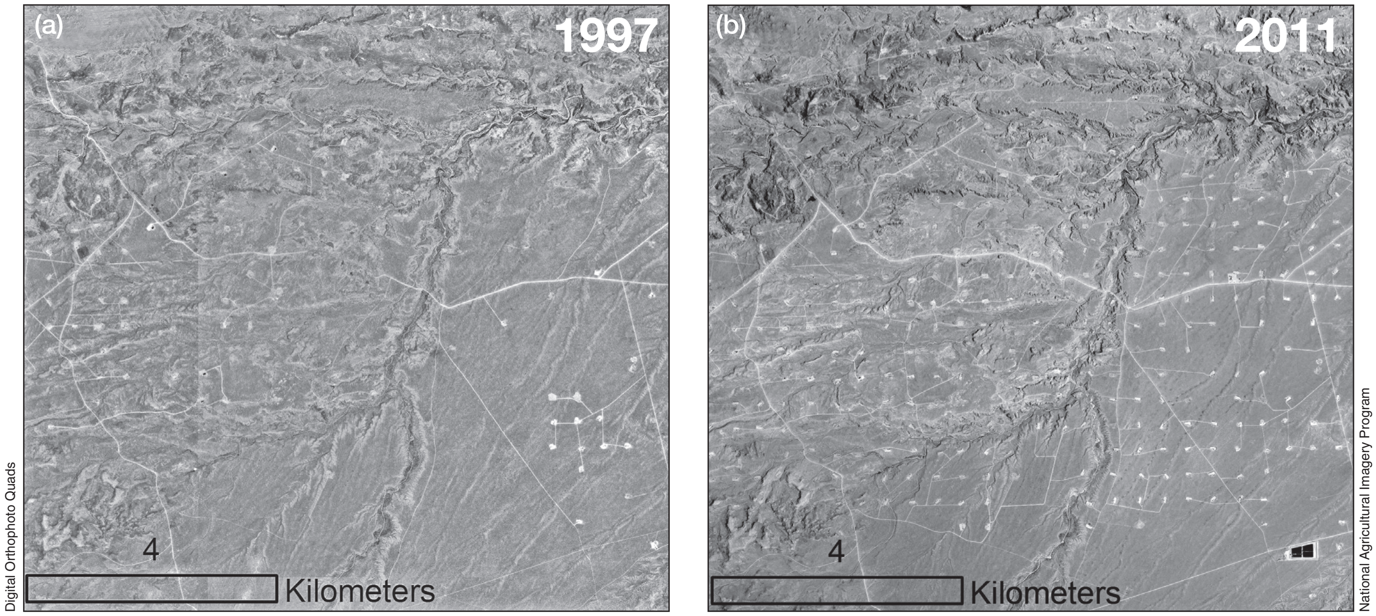


***Figure 5.*** *Aerial photographs circa 1997 (a) and 2011 (b) from the Uintah Basin (near Vernal, Utah) documenting increased oil and natural gas development in a previous rangeland landscape.*

limitations are overcome through large investments from foreign countries where agricultural productivity can be enhanced by technology (eg "land grabs"; Rulli *et al.* 2013). Once investments are made to convert land to cropland or urban uses, strong socioeconomic feedbacks often increase conversion rates and inhibit reconversion to rangeland (Brunson and Huntsinger 2008; Lambin and Meyfroidt 2011). Reid *et al.* (2008) estimated that 35–50% of mesic (semi-arid and dry subhumid) rangelands had been converted to cropland worldwide, with another 2–4% being urbanized.

### *Implications of land-use change for desertification*

Land-use change from rangeland to other uses is expected to accelerate as land scarcity increases in future decades (Lambin and Meyfroidt 2011). The recently released Cropland Data Layer (USDA NASS 2013) reveals that rapid land-use change is already taking place in drylands of the western US. According to these data, a net total of 21 488 km$^2$ of rangelands – roughly equivalent to the areal extent of the state of New Jersey – were converted to cropland in US drylands (primarily in semi-arid areas) between 2008 and 2013 (WebPanel 1). This includes reconversion of restored rangelands to cropland (fluctuating land use) as well as conversion of rangelands never before cropped (Wright and Wimberly 2013; Clay *et al.* 2014). A similar acceleration in the conversion of semi-arid areas to cropland has occurred in Chaco forests in Argentina since the 1990s. The absence of strong land tenure among the indigenous pastoralists in the Chaco has contributed to high conversion rates (Zak *et al.* 2008). In both the US and Argentina, new technologies and management practices – such as no-tillage cropping and genetically engineered, herbicide-resistant crop varieties – have promoted extensive conversion in semi-arid climates, a process accelerated by high grain prices. There is concern that predicted increases in the frequency of extreme weather events in North America (Clay *et al.* 2014) or reversal of recent increases in rainfall in the Argentine Chaco (Zak *et al.* 2008) may result in cropland abandonment and state changes/regime shifts.

Even when local land-use changes are sustainable, however, they may induce off-site effects or "cascades" that accelerate similar transitions in adjacent lands (Lambin and Meyfroidt 2011). As shown by the Cropland Data Layer (WebPanel 1), 1477 km$^2$ of rangeland and cropland were converted to urban uses from 2008–2013, mostly in the vicinity of arid cities. There was negligible conversion from urban to other uses (21 km$^2$), a finding that is consistent with high conversion pressure. Cascading transitions can indirectly cause state change, for instance by increasing grazing pressure in rangelands adjacent to croplands or urban areas (Galvin *et al.* 2007) and conversion of marginal rangelands to cropland (Lambin and Meyfroidt 2011). Conversion pressure on lands adjacent to urban areas can be managed through policy tools such as land zoning or conservation easements. Areas of low resilience to cropland uses can be identified based on soil properties (Bakker 2005).

Areas that have already undergone regime shifts are logical sites to which urban growth could be directed. For example, Stoms *et al.* (2013) produced a spatial model to evaluate the potential for solar energy development in the deserts of southern California based on land degradation, land tenure, and accessibility to infrastructure. In this model, lands that were experiencing persistent loss of vegetation or that had been invaded by non-native annual grasses were identified as preferred areas for development. Information and assumptions regarding desertifi-

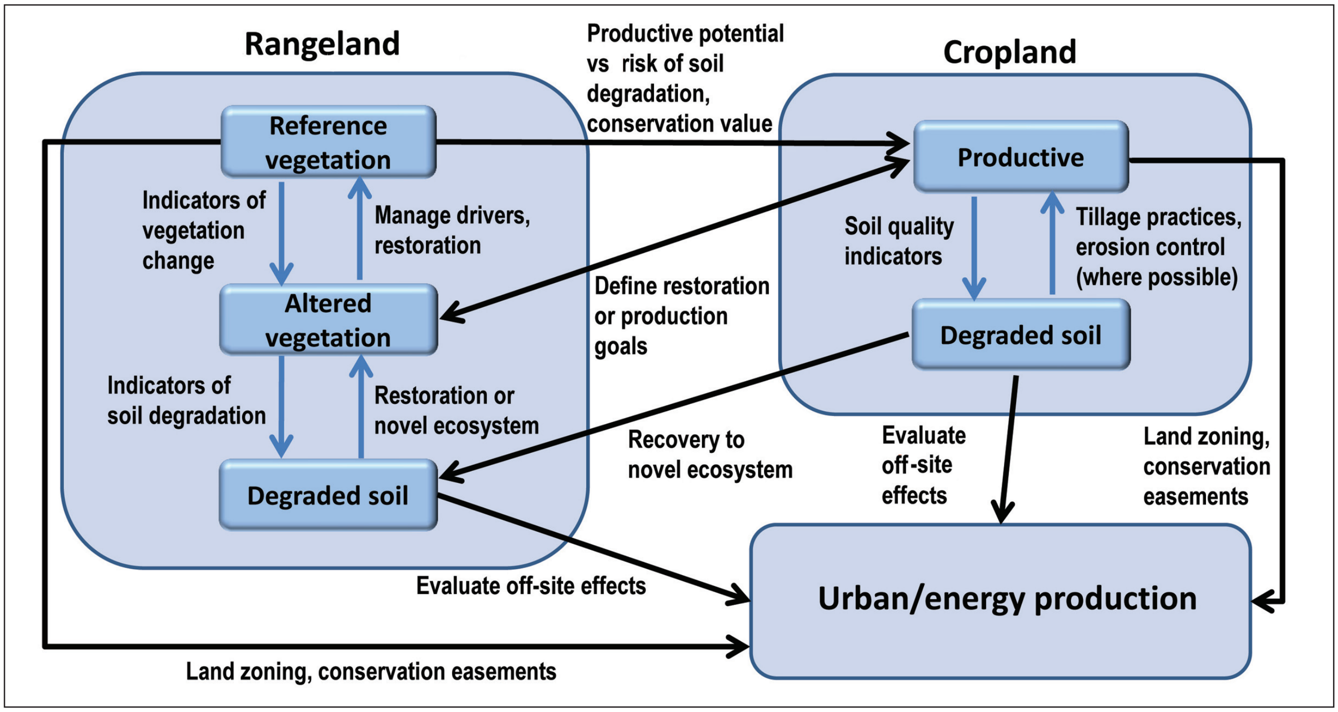


***Figure 6.*** *General conceptual model of state change in drylands, featuring prominent land uses (large boxes), generalized vegetation/soil states (small boxes), considerations for managing state change and regime shifts within a land use (blue arrows), and considerations for managing land-use change (black arrows).*

cation are likely to become increasingly important for land-use planning in drylands.

## ■ Applications to desertification assessment

The term "desertification" – as a general phenomenon – has considerable international importance because it highlights the complexity and urgency of management challenges in drylands (Reynolds *et al.* 2007). Yet use of the term as a catch-all for diverse types of state change obscures the underlying causes, as well as potential solutions. We argue that a framework that distinguishes between equilibrium, non-equilibrium, and regime-shift state changes, and that integrates state change with land-use change, can provide context-specific analyses and point to useful management and policy responses (Table 1).

Several activities will be needed to implement this approach. First, broad conceptual models, similar to S&T models, could be developed to summarize existing information on the nature of state change, land-use change, and their interactions and drivers for different regions (Figure 6). Such models could be based on existing S&T models, alongside land-use/land-cover-change models (NRC 2014) that are currently separated in distinct academic disciplines (ecology and geography, respectively). These integrated models could provide region-specific, comprehensive information on the processes causing state change that may be interpreted as desertification, includ-

**Table 1. Response strategies with respect to different state-change dynamics and land uses**

| *Change type* | *Response strategy in rangeland use* | *Response strategy in cropland use (including recently abandoned)* |
|---|---|---|
| **Equilibrium** | Managed grazing; recovery predictable | Conservation tillage and cover crops; restore to rangeland; recovery predictable |
| **Non-equilibrium** | Management opportunities to promote recovery confined to periods of favorable climatic conditions; recovery episodic and dependent on weather | Low-density planting, staggered planting dates, changing crops when replanting; recovery to rangeland episodic and dependent on weather |
| **Regime shift (vegetation)** | Restoration or management as novel ecosystem; intensive restoration | Abandonment without proactive restoration leads to novel ecosystem |
| **Regime shift (soil)** | Manage as novel ecosystem, convert to urban uses; intensive restoration | Conversion to low-productivity novel ecosystem or urban use; intensive restoration |
| **Land-use change cascades** | Land zoning, conservation easements | Land zoning, conservation easements |

 

ing (1) the characteristics of states and early-warning indicators, (2) realistic options for either preventing or reversing undesirable state change, and (3) the effects of land use on state change and vice versa (Geist and Lambin 2004; Verstraete *et al.* 2009). In regions where well-supported S&T models are lacking – including most drylands – vegetation/soil inventories, experimentation (eg vegetation recovery experiments), and historical reconstructions must be research priorities (Bestelmeyer *et al.* 2009).

Second, mapping based on integrated SC–LUC models could be used to assess the likelihood of state change with respect to soil resilience (eg potential-based land classification; Herrick *et al.* 2013), current ecological state (Steele *et al.* 2012), and land-use change pressures (Hansen *et al.* 2013). Digital data on land use such as the Cropland Data Layer (WebPanel 1) and on soils via the National Cooperative Soil Survey in the US could be intersected to produce maps showing the relationships between land-use change and soil resilience. At a global level, the Global Soil Map (www.globalsoilmap.net), along with remote-sensing products from the Global Land Cover Facility (www.landcover.org) and the Global Forest Change dataset (Hansen *et al.* 2013), will have similar uses. Yet for most drylands, maps of rangeland and cropland states do not exist. This absence is due to the difficulty in using satellite-based technologies to distinguish dryland states as well as to the imprecise application of the concepts and methods used to differentiate between states (Veron *et al.* 2006). However, intensive mapping of ecological states based on sub-meter resolution imagery (available globally via Google Earth) and region-level S&T models are promising approaches for project-level applications (Steele *et al.* 2012). The combination of maps and models (eg Wylie *et al.* 2012) could be applied to direct land-zoning policies or incentives (such as CRP) to promote sustainable land uses and to direct restoration investments.

Finally, the development of web-based applications, such as the Land Potential Knowledge System (Herrick *et al.* 2013), could provide a mechanism for crowd-sourced model development and site-specific information delivery via mobile devices (eg smartphones). Site-specific information on soil and vegetation (http://websoilsurvey.sc.egov.usda.gov), ecological states, and land use could educate land users anywhere in the world about best management practices to maintain or restore desired states. The proposed Global Drylands Observing System (Verstraete *et al.* 2009) could also benefit by reporting desertification to governments according to the specific processes causing and constraining state change. Collectively, these multi-institutional and multiscale activities based on an SC–LUC concept could vastly improve the effectiveness of desertification assessment and the sustainable management of drylands.

## Acknowledgements

This contribution was supported by Jornada Basin Long Term Ecological Research Program (DEB-1235828) and the USDA-ARS Jornada Experimental Range. SRA acknowledges support from the Arizona Agricultural Experiment Station (ARZT- 1 360540-Hl 2- 99). We thank DPC Peters for comments that improved the paper.

## References

Addison J, Friedel M, Brown C, *et al.* 2012. A critical review of degradation assumptions applied to Mongolia's Gobi Desert. *Rangeland J* **34**: 125–37.

Arshad MA and Martin S. 2002. Identifying critical limits for soil quality indicators in agro-ecosystems. *Agr Ecosyst Environ* **88**: 153–60.

Bakker MM. 2005. Soil erosion as a driver of land-use change. *Agr Ecosyst Environ* **105**: 467–81.

Barger NN, Archer S, Campbell J, *et al.* 2011. Woody plant proliferation in North American drylands: a synthesis of impacts on ecosystem carbon balance. *J Geophys Res-Biogeo* **116**: G00K07.

Bestelmeyer BT. 2006. Threshold concepts and their use in rangeland management and restoration: the good, the bad, and the insidious. *Restor Ecol* **14**: 325–29.

Bestelmeyer BT, Ellison AM, Fraser WR, *et al.* 2011. Analysis of abrupt transitions in ecological systems. *Ecosphere* **2**: art129.

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.

Briske DD, Fuhlendorf SD, and Smeins FE. 2003. Vegetation dynamics on rangelands: a critique of the current paradigms. *J Appl Ecol* **40**: 601–14.

Brunson MW and Huntsinger L. 2008. Ranching as a conservation strategy: can old ranchers save the New West? *Rangeland Ecol Manag* **61**: 137–47.

Chapin FS, Carpenter SR, Kofinas GP, *et al.* 2010. Ecosystem stewardship: sustainability strategies for a rapidly changing planet. *Trends Ecol Evol* **25**: 241–49.

Cingolani AM, Noy-Meir I, and Díaz S. 2005. Grazing effects on rangeland diversity: a synthesis of contemporary models. *Ecol Appl* **15**: 757–73.

Clay DE, Clay SA, Reitsma KD, *et al.* 2014. Does the conversion of grasslands to row crop production in semi-arid areas threaten global food supplies? *Global Food Security* **3**: 22–30.

Cramer VA, Hobbs RJ, and Standish RJ. 2008. What's new about old fields? Land abandonment and ecosystem assembly. *Trends Ecol Evol* **23**: 104–12.

D'Odorico P, Bhattachan A, Davis KF, *et al.* 2013. Global desertification: drivers and feedbacks. *Adv Water Resour* **51**: 326–44.

D'Odorico P, Okin GS, and Bestelmeyer BT. 2012. A synthetic review of feedbacks and drivers of shrub encroachment in arid grasslands. *Ecohydrology* **5**: 520–30.

Dardel C, Kergoat L, Hiernaux P, *et al.* 2014. Re-greening Sahel: 30 years of remote sensing data and field observations (Mali, Niger). *Remote Sens Environ* **140**: 350–64.

Dussart E, Lerner P, and Peinetti R. 1998. Long term dynamics of 2 populations of *Prosopis caldenia* Burkart. *J Range Manage* **51**: 685–91.

Eldridge DJ, Bowker MA, Maestre FT, *et al.* 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecol Lett* **14**: 709–22.

Fernandez-Gimenez ME and Allen-Diaz B. 1999. Testing a non-equilibrium model of rangeland vegetation dynamics in Mongolia. *J Appl Ecol* **36**: 871–85.

Galvin KA, Reid RS, Behnke Jr RH, and Hobbs NT. 2007. Fragmentation in semi-arid and arid landscapes: consequences for human and natural systems. New York, NY: Springer.

Geist HJ and Lambin EF. 2004. Dynamic causal patterns of desertification. *BioScience* **54**: 817–29.

Hansen MC, Potapov PV, Moore R, *et al.* 2013. High-resolution global maps of 21st-century forest cover change. *Science* **342**: 850–53.

Herrick JE, Brown JR, Bestelmeyer BT, *et al.* 2012. Revolutionary land use change in the 21st century: is (rangeland) science relevant? *Rangeland Ecol Manag* **65**: 590–98.

Herrick JE, Urama KC, Karl JW, *et al.* 2013. The global Land-Potential Knowledge System (LandPKS): supporting evidence-based, site-specific land use and management through cloud computing, mobile applications, and crowdsourcing. *J Soil Water Conserv* **68**: 5A–12A.

Hobbs RJ, Hallett LM, Ehrlich PR, and Mooney HA. 2011. Intervention ecology: applying ecological science in the twenty-first century. *BioScience* **61**: 442–50.

Hobbs RJ and Suding KN. 2009. New models for ecosystem dynamics and restoration. Washington, DC: Island Press.

Illius AW and O'Connor TG. 1999. On the relevance of nonequilibrium concepts to arid and semiarid grazing systems. *Ecol Appl* **9**: 798–813.

Jackson LL, McAuliffe JR, and Roundy BA. 1991. Desert restoration. *Ecol Restor* **9**: 71–80.

Lal R. 2001. Soil degradation by erosion. *Land Degrad Dev* **12**: 519–39.

Lambin EF, Gibbs HK, Ferreira L, *et al.* 2013. Estimating the world's potentially available cropland using a bottom-up approach. *Global Environ Change* **23**: 892–901.

Lambin EF and Meyfroidt P. 2011. Global land use change, economic globalization, and the looming land scarcity. *P Natl Acad Sci USA* **108**: 3465–72.

Lewis T, Reid N, Clarke PJ, and Whalley RDB. 2010. Resilience of a high-conservation-value, semi-arid grassland on fertile clay soils to burning, mowing and ploughing. *Austral Ecol* **35**: 464–81.

Miller ME, Belote RT, Bowker MA, and Garman SL. 2011. Alternative states of a semiarid grassland ecosystem: implications for ecosystem services. *Ecosphere* **2**: art55.

Munson SM and Lauenroth WK. 2012. Plant community recovery following restoration in semiarid grasslands. *Restor Ecol* **20**: 656–63.

NRC (National Research Council). 2014. Advancing land change modeling: opportunities and research requirements. Washington, DC: National Academies Press.

Peters DPC, Havstad KM, Archer SR, and Sala OE. 2015. Beyond desertification: new paradigms for dryland landscapes. *Front Ecol Environ* **13**: 4–12.

Quétier F, Thébault A, and Lavorel S. 2007. Plant traits in a state and transition framework as markers of ecosystem response to land use change. *Ecol Monogr* **77**: 33–52.

Reid RS, Galvin KA, and Kruska RS. 2008. Global significance of extensive grazing lands and pastoral societies: an introduction. In: Galvin KA, Reid RS, Behnke Jr RH, and Hobbs NT (Eds). Fragmentation in semi-arid and arid landscapes. Dordrecht, The Netherlands: Springer.

Reynolds JF, Grainger A, Smith DMS, *et al.* 2011. Scientific concepts for an integrated analysis of desertification. *Land Degrad Dev* **22**: 166–83.

Reynolds JF, Stafford Smith DM, Lambin EF, *et al.* 2007. Global desertification: building a science for dryland development. *Science* **316**: 847–51.

Rulli MC, Saviori A, and D'Odorico P. 2013. Global land and water grabbing. *P Natl Acad Sci USA* **110**: 892–97.

Rumpff L, Duncan DH, Vesk PA, *et al.* 2011. State-and-transition modelling for adaptive management of native woodlands. *Biol Conserv* **144**: 1224–36.

Safriel UN. 2007. The assessment of global trends in land degradation. In: Sivakumar MVK and Ndiang'ui N (Eds). Climate and land degradation. Berlin, Germany: Springer.

Sayre NF, McAllister RRJ, Bestelmeyer BT, *et al.* 2013. Earth Stewardship of rangelands: coping with ecological, economic, and political marginality. *Front Ecol Environ* **11**: 348–54.

Scheffer M and Carpenter SR. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends Ecol Evol* **18**: 648–56.

Seabloom EW, Harpole WS, Reichman OJ, and Tilman D. 2003. Invasion, competitive dominance, and resource use by exotic and native California grassland species. *P Natl Acad Sci USA* **100**: 13384–89.

Seybold CA, Herrick JE, and Brejda JJ. 1999. Soil resilience: a fundamental component of soil quality. *Soil Sci* **164**: 224–34.

Steele CM, Bestelmeyer BT, Burkett LM, *et al.* 2012. Spatially explicit representation of state-and-transition models. *Rangeland Ecol Manag* **65**: 213–22.

Stoms DM, Dashiell SL, and Davis FW. 2013. Siting solar energy development to minimize biological impacts. *Renew Energ* **57**: 289–98.

Stubbs M. 2013. Conservation Reserve Program (CRP): status and issues. Washington, DC: Congressional Research Service.

Tugel AJ, Herrick JE, Brown JR, *et al.* 2005. Soil change, soil survey, and natural resources decision making. *Soil Sci Soc Am J* **69**: 738–47.

UNCCD (United Nations Convention to Combat Desertification). 1994. Elaboration of an international convention to combat desertification in those countries experiencing serious drought and/or desertification, particularly in Africa. Paris, France: United Nations. A/AC.241/27 (www.unccd.int/Lists/SiteDocumentLibrary/conventionText/conv-eng.pdf).

USDA NASS (US Department of Agriculture National Agricultural Statistics Service). 2013. CropScape – Cropland Data Layer. Washington, DC: USDA. http://nassgeodata.gmu.edu/CropScape. Viewed 1 Apr 2014.

Veron SR, Paruelo JM, and Oesterheld M. 2006. Assessing desertification. *J Arid Environ* **66**: 751–63.

Verstraete MM, Scholes RJ, and Smith MS. 2009. Climate and desertification: looking at an old problem through new lenses. *Front Ecol Environ* **7**: 421–28.

von Wehrden H, Hanspach J, Kaczensky P, *et al.* 2011. Global assessment of the non-equilibrium concept in rangelands. *Ecol Appl* **22**: 393–99.

Westoby M, Walker B, and Noy-Meir I. 1989. Opportunistic management for rangelands not at equilibrium. *J Range Manage* **42**: 266–74.

Worster D. 2004. Dust Bowl: the Southern Plains in the 1930s. New York, NY: Oxford University Press.

Wright CK and Wimberly MC. 2013. Recent land use change in the western Corn Belt threatens grasslands and wetlands. *P Natl Acad Sci USA* **110**: 4134–39.

Wylie BK, Boyte SP, and Major DJ. 2012. Ecosystem performance monitoring of rangelands by integrating modeling and remote sensing. *Rangeland Ecol Manag* **65**: 241–52.

York A, Shrestha M, Boone C, *et al.* 2011. Land fragmentation under rapid urbanization: a cross-site analysis of southwestern cities. *Urban Ecosystems* **14**: 429–55.

Zak MR, Cabido M, Caceres D, and Diaz S. 2008. What drives accelerated land cover change in central Argentina? Synergistic consequences of climatic, socioeconomic, and technological factors. *Environ Manage* **42**: 181–89.

 