A degradation threshold for irreversible loss of soil productivity: a long-term case study in China

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Summary

1. During the past three decades, conservation and restoration biologists have increasingly recognized that ecological communities are likely to exhibit threshold changes in structure. However, because long-term monitoring data are generally lacking, little is known about the consequences of such ecological thresholds for the processes of ecosystem degradation and recovery.
2. To identify whether a degradation threshold exists that defines the boundary between the possibility of natural recovery and the need for artificial restoration of an ecosystem and to use this knowledge to support the development of a suitable strategy for environmental restoration, we performed long-term monitoring of vegetation recovery in China’s Changting County since 1984.
3. A major problem was identified, which we refer to as the ‘irreversible loss of soil services’: when vegetation cover decreases below a degradation threshold, this leads to sustained degeneration of the vegetation community, erosion of the surface soil and declining soil fertility. These changes represent a severe and long-lasting disturbance that will prevent ecosystem recovery in the absence of comprehensive artificial restoration measures.
4. Synthesis and applications. We identified a degradation threshold at about 20% vegetation cover suggesting that for some sites, vegetation cover can serve as a simple proxy for more sophisticated approaches to identifying thresholds; restoration must start with the restoration of soil fertility and continue by facilitating vegetation development. Our results support the concept of ecological thresholds (specifically, for soil services in a warm and wet region) and provide a model to inform restoration strategies for other degraded ecosystems.

Key-words: degradation threshold, ecological restoration, landscape disturbance, natural recovery, species richness

Introduction

Ecological communities are most likely to exhibit threshold changes in structure when perturbations cause large changes in (i) limiting soil or other resources, (ii) dominant or keystone species and (iii) attributes of the disturbance regime that influence the recruitment of organisms by a community (e.g. Lamb, Erskine & Parrotta 2005; Srinivasan et al. 2008). There are several types of ecological threshold, and their potential uses in ecosystem management differ accordingly (Bestelmeyer 2006). Restoration ecology could be more effective if environmental managers better understood which type of ecological threshold is most relevant for a given site and their potential roles in ecosystem management. Unfortunately, although the ecological literature contains many conceptual models of thresholds and discussions of ecosystems in which multiple states are possible, there is little guidance about which model is most appropriate for a given situation (Martin & Kirkman 2009). The concept of irreversible soil degradation because of vegetation removal and overgrazing and its link to vegetation cover has been well established in semi-arid regions (e.g. van de Koppel, Rietkerk & Weissing 1997; Rietkerk & van de Koppel 1997), but there has been no study in wetter regions, particularly where harvesting of trees and vegetation is the primary cause of the degradation. Thus, knowledge of how these factors operate in warm and wet regions is required before we can understand the ecological thresholds that determine whether a human-disturbed ecosystem will recover naturally or whether artificial restoration will be required. In the latter case, identification of a suitable strategy for environmental restoration could be based on the nature of the threshold.

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Changes in the environmental variables that determine the distribution and abundance of vegetation, which differ among sites with different ecological histories, must also be understood. Austin (1980, 2002) grouped these environmental variables into three main types: (i) gradients in resources that are consumed by plants (e.g. CO₂, water, light and nutrients); (ii) gradients in resources that are not directly consumed by plants but that have direct physiological influences on growth (e.g. wind, air and soil temperatures, pH) and (iii) indirect gradients that have no direct physiological influence on plant growth but that are correlated with species distribution because of their correlation with variables such as temperature, soil moisture and precipitation (e.g. slope aspect, elevation, longitude and latitude, distance from the coast, relative landscape position). To facilitate the recovery of degraded or damaged ecosystems, it is necessary to understand the state of the original ecosystem and identify what factor or factors have altered that state (Jackson & Hobbs 2009). When multiple factors affect an ecosystem’s state, they may act concurrently (i.e. at the same time) or in series (i.e. one after the other), and these patterns will have different effects on species survival. Where factors act in series, their order can be very important; for example, a species may be able to cope with events A, B and then C, but unable to cope with events A, C and then B.

For these reasons, one of the greatest challenges in environmental biology is to predict the effects of human activity on the complex webs of interactions among species and systems (Berlow et al. 2009). A common example is when vegetation degradation leads to a loss of topsoil and a reduction in soil fertility, thereby impeding recolonization of a site by many of the original species (Lamb, Erskine & Parrotta 2005; Ludwig, Wilcox & Breshears 2005). If the site degradation becomes sufficiently severe, a threshold is crossed; beyond that threshold, the ecosystem cannot recover to its original state without human intervention or the passage of years or even decades without further disturbance. Human impacts can widen the range of habitats in which such threshold dynamics occur, and these impacts can shift communities into new states that are difficult to reverse (Suding & Hobbs 2008). Therefore, the concept of a degradation threshold can guide the development of targets for sustaining landscape functions and developing management regimes for variables such as habitat loss, habitat fragmentation, connectivity changes and soil erosion (Huggett 2005).

In the present paper, we discuss key issues associated with the degradation threshold concept, and the management implications, using a case study of vegetation loss and soil erosion in China. Our goal is to show how an improved understanding of thresholds can guide the restoration of a degraded ecosystem. The high intensity of monsoonal rainfall in our study area occurs at the end of a fairly long dry period, which means that herbaceous vegetation cover has decreased and the risk of severe soil erosion is higher than at other times of year. Soil erosion is a particular problem in China because of the country’s huge population and rapid economic development, which have led to unsustainable agricultural land uses, combined with adverse climatic changes that have led to the expansion of deserts in many parts of arid and semi-arid China (Cao, Chen & Yu 2009a). Large-scale restoration projects have been implemented in an effort to combat these problems, but for ecological restoration to succeed, we must recognize the potentially severe consequences caused by crossing a degradation threshold, their effect on afforestation and other forms of artificial vegetation restoration, and the resulting consequences for traditional restoration activities.

Recent theoretical advances have emphasized that the existence of thresholds and the alternative stable states that result when thresholds are exceeded are key factors that influence the outcomes of ecological restoration efforts. Abiotic thresholds, such as those associated with changes in soil or water conditions, are widely recognized as crucial factors in efforts to restore degraded ecosystems (Norton 2009), because these factors partly determine which species can survive at a given site. Although many studies have examined the concept of ecological thresholds in ecological restoration, the complexity of coupled systems is not well understood (Lamb, Erskine & Parrotta 2005; Sasaki et al. 2008; Jackson & Hobbs 2009; Norton 2009), and most of the previous work has been theoretical rather than observation based (Radford, Bennett & Cheers 2005; Liu et al. 2007; Sasaki et al. 2008). Moreover, the lack of long-term field research means that there are insufficient data to confirm the various theoretical models of ecological thresholds (Andersen et al. 2009). In addition, most efforts to overcome ecosystem degradation have involved interventions such as tree planting (McVicat et al. 2007, 2010) or protecting sites from impacts such as grazing so that they can undergo natural recovery (Suding & Hobbs 2008).

In the management of degraded land, thresholds reflect changes in vegetation and soils that are expensive or impossible to reverse. Uncritical use of thresholds may lead to the abandonment of management efforts in areas that would otherwise benefit from an intervention (Bestelmeyer 2006). Because of the paucity of field studies, we lack a sophisticated understanding of the ecological degradation threshold that determines whether a human-disturbed ecosystem has the potential for natural recovery if it is protected against further disturbance or whether artificial restoration will be necessary. To provide empirical evidence for the existence of an ecological ‘degradation threshold’ and to demonstrate how this knowledge can be used to guide successful ecological restoration, we have carried out long-term monitoring of vegetation recovery in a warm, wet region of China’s Changting County since 1984. We found a severe problem in this area, which we refer to as the ‘irreversible loss of soil services’: when vegetation cover decreases below an ecological degradation threshold, leading to sustained degeneration of the vegetation community, erosion of the surface soil and declining soil fertility occur. These changes represent a severe and long-term disturbance of the vegetation, the soil and the landscape.

Materials and methods
Our research area is located in a warm and wet part of China’s Changting County that covers 309 720 ha in western Fujian
Based on meteorological data collected at a soil and water conservation monitoring station established in 1940 in the study area, the mean annual precipitation is 1721 mm year⁻¹, the average annual potential evapotranspiration is 892 mm year⁻¹, and the mean annual temperature is 18.3 °C, with a minimum temperature of −7.9 °C and a maximum temperature of 39.4 °C based on historical climate records from 1952 to 2004 (Yang, Zhong & Xie 2005). Planting of crops is the primary agricultural activity, and there is little or no livestock grazing other than to feed domestic animals. However, because a half-century of China’s Grain First Strategy led to widespread replacement of forests with agricultural land, much of the agriculture occurred on unsuitable land, leading to severe erosion and long-term site degradation in many areas. These conditions have contributed to the inability of the degraded sites to recover naturally. Moreover, the region’s monsoon climate means that there is a relatively dry season, when vegetation cover decreases, followed by a season with heavy rainfall. The combination of degraded land, a lack of vegetation cover and high inputs of rain have increased the frequency and scale of water erosion of the soil and the severity of floods, leading to further degradation of the county’s forests and landscape. Another problem is that the region’s poor farmers have little money to buy coal for cooking, leading to harvesting of trees and woody vegetation for use as cooking fuel; this unsustainable exploitation of the remaining vegetation is a primary contributor to the environmental damage (Cao et al. 2009b).

To examine the ecosystem’s capacity for natural recovery, we randomly selected 30 representative hilly plots, each 400 m² (20 × 20 m) in size, in four towns (Cewu, Hetian, Sanzhou and Zhuotian). The plots were established at similar topographic positions, at mid-slope positions on slopes ranging from 20% to 30%. Before 1984, none of the plots was formally managed, and local residents harvested trees for use as firewood or construction materials. Based on discussions with local government officials, there appears to have been only minor differences among study sites in the historical or cultural factors that might have affected the locations of the most severely degraded plots. We grouped these plots into six groups (with five plots per group) based on the original vegetation cover (both annual and perennial species) in 1984: (i) ≤20%, (ii) 20%–25%, (iii) 25%–30%, (iv) 30%–35%, (v) 35%–40% and (vi) >40%. To restore the ecological environment, tree harvesting and grazing of domestic livestock were prohibited throughout Changting County starting in 1984. To ensure that this prohibition would be effective, the purpose of the plots was explained to local residents, and incentives were provided, including fuel and support for the purchase of livestock fodder, thereby making it unnecessary to collect firewood or to graze domestic animals in the study plots (Cao et al. 2009b).

To better understand the specific reasons (mechanisms) behind the existence of any ecological thresholds at these sites, to propose a restoration approach in the plots where threshold dynamics were occurring and to determine whether it was possible to develop a new, rapid way to perform ecological restoration, we randomly selected an additional 28 representative plots (for a combined total of 58 plots), each 400 m² (20 × 20 m) in size, for a new study that was conducted from 1999 to 2009. In each of these 28 plots, the initial cover of annual vegetation was <30%; in 14 of the plots, we allowed natural recovery to occur, but in 14 plots, we performed artificial restoration using local species starting in 1999. In each of the two groups of 14 plots, seven plots had initial vegetation cover of <20%, and the remaining seven plots had initial vegetation cover >20% (actually 20%-1% or higher, but initial cover values ranging from 20% to 30%). Artificial restoration was performed by planting 600 trees (Schima superba, Morella rubra, Liquidambar formosana or Castanopsis fissa) and 2400 shrubs (Lespeceola davidiiflora) per hectare. In addition, Paspalum wettsteinii was seeded at a rate of 105 kg ha⁻¹, or Paspalum notatum was seeded at a rate of 45 kg ha⁻¹, with 900 kg ha⁻¹ of oil cake fertilizer (the organic matter that remains after extraction of the edible oils from rapeseed) added to improve the soil’s organic matter content.

To assess the amount of vegetation cover at the study site, we measured both the crown area of the trees and coverage of the ground by understory vegetation (only green plants, therefore photosynthetically active vegetation). Using a steel tape, we measured the crowns of 20 randomly selected trees in each plot each year during the middle of the growing season (between the last 10 days of June and the end of August) to determine crown area, which we used to represent the mean crown cover per tree. We measured the maximum and minimum crown radii and modelled the crown as an ellipse, with these radii representing the semi-major and semi-minor axes, and calculated the mean canopy area for each species using geometric mean values to account for extreme values. Total tree canopy cover (the proportion of the total site area accounted for by a vertical projection of the elliptical crowns of the trees, including the leaves plus the stems and branches) was calculated by multiplying the mean crown area in a given year by the number of trees that were present in that year, then dividing this total by the total area planted with that species. Where canopies overlapped, we carefully determined the extent of the overlap and calculated its area; we then divided this area equally between the two trees to avoid double-counting.

In each portion of the plot that was not covered by tree or shrub canopies or where grass was growing below trees, we performed line-intersect sampling using two 10-m transects at right angles to each other to survey herbaceous vegetation (ground cover). We calculated the net vegetation cover for the grass and woody cover by averaging the two cover values (i.e. for woody and herbaceous vegetation). We identified the vegetation cover every year at the same time (between the last 10 days of June and the end of August). Total vegetation cover (the combined cover of trees and herbaceous vegetation such as grasses, forbs and herbs) for a given area was calculated by multiplying the mean cover value for a given vegetation type (woody vs. non-woody vegetation) by the proportion of the total plot area occupied by that type of vegetation. To describe the plant species richness in the study plots, we collected samples of all plant species annually in each plot in August. The samples were brought to Fujian Normal University for identification if we could not confirm their identity in the field.

To monitor soil erosion at each of the 30 selected representative natural restoration plots, a sand sedimentation pond (a run-off pond) was established at each plot. We selected 20-m-long by 5-m-wide observation sections along the slopes in each test plot and constructed a stone and concrete sand sedimentation pond with a 15-m³ capacity at the bottom of the slope. We calculated the total water input by multiplying the amount of rainfall (mm) by the surface area of the observation section (100 m²); we then estimated the volume of water collected by the sedimentation pond (i.e. the run-off) and expressed this volume as a proportion (5%) of the total input of water. In addition, all the soil was removed from the bottom of each pond within 24 h after the rain, and three random samples of this soil were dried for 12 h at 105 °C and were weighed to determine the water content of the sediments. This was then used to determine the total oven-dry quantity of soil eroded by the rain event. We used these data to calculate the erosion modulus (kt soil per km² per rain event).
We also sampled the uppermost 30 cm of the soil, where the majority of the seedling and grass roots would be found, using an auger at three randomly selected locations at 5-year intervals, in October of 1984, 1989, 1994, 1999, 2004 and 2009, to measure the soil nutrient contents. All other plot parameters were measured in the same years. Study parameters were measured annually in each plot, but for simplicity, we have reported data only at 5-year intervals, corresponding to the interval used for the soil nutrient measurements. The uppermost 5 cm of the soil was also sampled on these dates using the same auger at three randomly selected locations in each plot, and the samples were passed through a series of sieves to determine the content of sand and coarser materials (>1 mm). Soil organic matter content was determined by means of oxidation with potassium dichromate in a heated oil bath. Total nitrogen was measured by means of alkali distillation. Total phosphorus was measured by means of atomic absorption spectrophotometry (with a Varian spectrophotometer; Varian Inc., Palo Alto, CA, USA). Total potassium was determined by digestion with hydrofluoric acid and perchloric acid.

The data from the 30 natural recovery plots in the initial year of the study and the values collected every 5 years thereafter are expressed as means ± SD, as are the results for the additional 28 plots that were monitored starting in 1999. We performed repeated-measures ANOVA to identify whether significant differences among treatments existed, and when they did, we used the least-significant-difference (LSD) test to determine which specific combinations of values were significantly different. All tests were performed using version 12.0 of the SPSS software (SPSS Inc., Chicago, IL, USA).

**Results**

Vegetation cover differed significantly among the initial cover classes (ANOVA). LSD tests revealed that the sites with an initial cover of <20% had significantly (\( P < 0.05 \)) lower initial vegetation cover than all other sites throughout the study period. By 1989, sites with an initial vegetation cover >30% had significantly higher vegetation cover than sites with an initial vegetation cover between 20% and 30% (\( P < 0.05 \)); by the end of the study period, only the site with an initial vegetation cover of 20–25% had significantly lower vegetation cover than the other sites (\( P < 0.05 \)), and vegetation cover was increasing so rapidly in these plots that we expect it to catch up with the other sites within a few years (Fig. 1). At a vegetation cover of <20%, erosion of the surface soil accelerated greatly during rain events, and the resulting loss of fertile topsoil resulted in a decrease in soil nutrient contents (Fig. 2) that led to a sustained loss of vegetation cover and very slow recovery of the vegetation community in the absence of human intervention (Fig. 3). These changes represent a severe and long-term disturbance of both the vegetation and the soil. When the vegetation cover was higher than 20% in the natural restoration plots, the vegetation cover and the soil characteristics in the study plots were both able to recover significantly (\( P < 0.001 \)) in the absence of human disturbance and without requiring artificial restoration, although the recovery was faster at higher initial vegetation cover values. The differences compared with the <20% vegetation cover were significant for all other vegetation covers throughout the study period (\( P < 0.05 \)).

In the absence of an improvement in vegetation cover (initial cover <20%), the soil underwent continuing degradation, indicated by ongoing erosion of the surface soil and depletion of soil fertility (Figs 2 and 3), although the number of plant species increased over time even in the most severely degraded plots (Fig. 3). As a result, soil erosion and run-off in the plots with an initial vegetation cover of <20% increased throughout the study period, leading to a continuing loss of fine particles and enrichment of sand and coarser particles (>1 mm) in the exposed surface soil (Fig. 3). In contrast, when the initial vegetation cover was >20%, soil nutrient properties improved continuously (Fig. 2), and soil erosion and run-off decreased continuously (Fig. 3), leading to a gradual decrease in the proportion of coarse materials in the surface soil (Fig. 3). All these changes were statistically significant (\( P < 0.05 \)).

Our 25 years of monitoring data also revealed strong and significant relationships between vegetation cover and the soil characteristics after human disturbance of the sites was prohibited (Fig. 4). The increase in vegetation cover (\( y \)) was significantly positively correlated with the following independent variables (\( x \)): soil organic matter (\( y = 0.0958x + 1.373; R = 0.948, P < 0.001 \)), total N (\( y = 0.0056x + 0.0811; R = 0.940, P < 0.001 \)), total K (\( y = 0.1525x + 3.379; R = 0.946, P < 0.001 \)), total P (\( y = 0.0029x + 0.0992; R = 0.929, P < 0.001 \)) and the number of plant species (\( y = 0.3338x + 0.4621; R = 0.890, P < 0.001 \)). It was significantly negatively correlated with soil erosion (\( y = -0.0832x + 8.3222; R = -0.837, P < 0.001 \)), with run-off (\( y = -0.2612x + 54.761; R = -0.921, P < 0.001 \)), and with the proportion of sand and coarser material (>1 mm) in the surface soil (\( y = -0.1144x + 47.311; R = -0.823, P < 0.001 \)).

In contrast to the results for the 30 natural restoration plots, artificial restoration was able to reverse the ecosystem degradation even when the original vegetation cover was <20%; the recovery became significantly faster than natural recovery within 5 years for all parameters ($P < 0.05$; Table 1). The results also indicate that ecosystem properties, including the vegetation cover and soil characteristics, recovered significantly ($P < 0.05$) faster in plots with an initial vegetation cover >20% as a result of the artificial restoration, and that all properties were significantly better at the sites with artificial restoration ($P < 0.05$; Table 2). In the 14 plots with vegetation cover <20% and where only natural restoration occurred, the results were similar to those in the 30 natural restoration plots: little change or continued degradation.

**Discussion**

Traditional ecosystem restoration efforts have focused on re-establishing historical disturbance regimes or abiotic conditions and have relied on successional processes to assist the recovery of biotic communities. However, strong feedback between biotic factors and the physical environment can alter the efficacy of these succession-based management efforts (Suding, Gross & Houseman 2004). In our study, an initial
vegetation cover of about 20% represented such a threshold: at a vegetation cover below 20%, the vegetation cover continued to decline slowly throughout the 25-year study period despite a lack of human disturbance, whereas at higher initial vegetation covers, natural recovery led to complete restoration of the ecosystem (to 100% vegetation cover), even in the absence of artificial restoration by planting and seeding combined with soil remediation; recovery was slower at an initial vegetation cover of 20–25%, but vegetation cover in these plots nonetheless reached 81% by the end of the 25 years and showed signs of exponentially approaching 100% within only a few more years (Fig. 1). Many soil services are closely associated with the degree of ecosystem resilience – the amount of change a system can undergo whilst retaining the same structure, functions and feedbacks (Suding & Hobbs 2008; McVicar et al. 2010). If this resilience declines, the ecosystem services can generally be expected to decline (Myers 1996). Degraded systems resist traditional restoration efforts owing to constraints such as changes in landscape connectivity and structure, species loss, changes in the dominant species, interactions among different trophic levels, and simultaneous changes in soil and other biogeochemical processes (Sasaki et al. 2008). Exceeding a disturbance threshold leads to a loss of ecosystem functions, and it may become impossible for these functions to recover naturally, even after periods as long as 25 years. When the vegetation cover was >20%, the vegetation cover in the study plots was able to recover naturally in the absence of human disturbance (Fig. 1) because the soil fertility and other
Identifying threshold behaviour is difficult in terrestrial ecosystems because the main components of the systems change slowly before the threshold is reached (Sasaki et al. 2008). In addition, some changes are short-term changes caused by climate variability, which affects the life cycle (e.g. recruitment strategies) of individual plant species. Therefore, ecologists have been eager to develop complex predictive tools and a broader conceptual framework capable of identifying thresholds before they are reached, thereby helping managers to prevent irreversible degradation from occurring or guiding the restoration of degraded ecosystems (Garten & Ashwood 2004; Suding, Gross & Houseman 2004). Unfortunately, these frameworks have been too complex for many managers and local residents, who can seldom use these tools to predict when thresholds will be reached (Chapin et al. 2006). Rapid environmental change renders this task even more daunting, so a major challenge for ecologists will be to develop effective means of quickly assessing the status of, and prognosis for, ecosystems that are undergoing various alterations (Jackson & Hobbs 2009). Our results suggest that for sites similar to those in our study, vegetation cover may be a useful proxy for more complex threshold calculations.
Natural recovery means no human intervention and artificial restoration comprised planting and soil remediation. Values of a parameter followed by different letters differ significantly between the artificial restoration and natural recovery treatments ($P < 0.05$). Values represent means and standard deviations ($n = 7$). Run-off represents the estimated proportion of rainfall that was recovered in sedimentation ponds at the base of the sample plots (see the Materials and methods section for details).

Table 2. Change of vegetation cover and soil parameters in 14 plots where the original vegetation cover was >20% under natural recovery and artificial restoration since 1999

<table>
<thead>
<tr>
<th></th>
<th>First year</th>
<th>5 years after</th>
<th>10 years after</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>Natural recovery</td>
<td>Artificial restoration</td>
<td>Natural recovery</td>
</tr>
<tr>
<td>Vegetation cover (%)</td>
<td>25.0 ± 2.8ab</td>
<td>25.4 ± 4.2a</td>
<td>29.2 ± 3.7b</td>
</tr>
<tr>
<td>Organic matter (g kg$^{-1}$)</td>
<td>3.9 ± 0.5a</td>
<td>2.5 ± 0.3b</td>
<td>5.3 ± 1.1c</td>
</tr>
<tr>
<td>Total N (g kg$^{-1}$)</td>
<td>0.26 ± 0.07a</td>
<td>0.18 ± 0.06b</td>
<td>0.31 ± 0.01ad</td>
</tr>
<tr>
<td>Total P (g kg$^{-1}$)</td>
<td>0.20 ± 0.04a</td>
<td>0.16 ± 0.02b</td>
<td>0.22 ± 0.05a</td>
</tr>
<tr>
<td>Total K (g kg$^{-1}$)</td>
<td>8.1 ± 3.1a</td>
<td>6.7 ± 1.0b</td>
<td>9.6 ± 2.4ac</td>
</tr>
<tr>
<td>Erosion modulus (kt km$^{-2}$)</td>
<td>5.37 ± 1.21a</td>
<td>5.98 ± 1.17b</td>
<td>4.39 ± 1.05a</td>
</tr>
<tr>
<td>Run-off (%)</td>
<td>48.2 ± 1.9ab</td>
<td>50.6 ± 2.1a</td>
<td>44.4 ± 3.4b</td>
</tr>
<tr>
<td>Sand and coarser (&gt;1.0 mm) (%)</td>
<td>43.9 ± 1.8a</td>
<td>46.2 ± 2.7b</td>
<td>43.3 ± 2.8a</td>
</tr>
</tbody>
</table>

Natural recovery means no human intervention and artificial restoration comprised planting and soil remediation. Values of a parameter followed by different letters differ significantly between the artificial restoration and natural recovery treatments ($P < 0.05$). Values represent means and standard deviations ($n = 7$). Run-off represents the estimated proportion of rainfall that was recovered in sedimentation ponds at the base of the sample plots (see the Materials and methods section for details).

In addition, most degraded landscapes are a mosaic of land uses that may include patches of intact natural vegetation and productive agricultural lands as well as degraded lands. It is rarely possible to revegetate the whole landscape, especially if it includes many small areas of human land use such as farms (Garten & Ashwood 2004). Hence, a simple predictive tool or index that would let managers and other stakeholders predict when an ecosystem is nearing a threshold condition is urgently needed to facilitate restoration of such ecosystems by allowing interventions to occur before the threshold is reached. In our study area, we identified the threshold based on the observed difference in trends for plots with different initial levels of vegetation cover (i.e. we identified the level of vegetation cover at which the trend changed from deterioration to improvement). This analysis is important in China because by the early 1990s, 3.67 million km$^2$ (about 38% of the land area) was experiencing the kind of soil erosion and vegetation loss described in the present study; this comprised 1.79 million km$^2$ of water erosion and 1.88 million km$^2$ of wind erosion (Anonymous 1993). In our study area, 238 km$^2$ has been affected by water erosion and vegetation loss, accounting for 8% of Changting County’s total land area; 16% of this land has a vegetation cover of <20% and will therefore require artificial restoration instead of only being protected in the hope that natural recovery will occur (Cao et al. 2009b).

Our results should only be generalized to other regions with great care. Our study was conducted in a specific warm and rainy region of China, with its own unique history and vegetation and environmental conditions; therefore, researchers in other regions must study the unique vegetation recovery processes in their regions to identify the level of vegetation cover that represents a degradation threshold for their ecosystems. That is, even when a simple index such as vegetation cover can be identified, a more complex analysis such as that of Sasaki...
et al. (2008) or long-term research such as that in the present study will be required to identify the actual threshold value for that index.

Ecosystem restoration sometimes fails because ecological interactions are more complex or human intervention is more difficult than anticipated (Byers et al. 2006); factors other than human disturbance, such as climate variability (e.g. a drought shortly after planting of vegetation), can result in failure of a strategy that would succeed under better conditions. Some degraded ecosystems can only be sustained through ongoing management, but many conservation efforts preclude such interventions (Liu et al. 2007). Although ecologists can recognize many of the species changes that are likely to precipitate threshold changes in community composition, biotic interactions can be unexpected, and because responses often depend strongly on local conditions, they cannot be broadly generalized (Chapin et al. 2006). For example, complex ecosystems with multiple interacting species may have a variety of thresholds (Garten & Ashwood 2004; Hunt et al. 2008). Desertification is another example and has been shown to result from strong biogeomorphic feedbacks that operate across several spatial scales (Suding & Hobbs 2008). When overgrazing of arid grasslands reduces vegetation cover, water infiltration decreases, further limiting plant growth and leading to persistent desertification (Byers et al. 2006). Such spatial discontinuities, called ecotones, can be detected using multivariate data ordered in one dimension through comparisons of measures of dissimilarity computed between the systems on either side of the discontinuity (Anderson et al. 2009).

The ‘irreversible loss of soil services’ described in our study, whether at national, regional or local scales, will have a variety of thresholds, and it will be necessary to calibrate this index for different regions before it becomes a useful management tool. However, as our results show, it is possible to identify useful proxies for thresholds and use them to guide subsequent management of degrading sites.

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