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RESEARCH ARTICLE

Short-Term and Long-Term Effects of Soil Ripping, Seeding, and Fertilization on the Restoration of a Tropical Rangeland

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Abstract

Rangeland degradation is a serious problem in semiarid Africa. Extensive areas of bare, compacted, nutrient-poor soils limit the productivity and biodiversity of many areas. We conducted a set of restoration experiments in which all eight combinations of soil tilling, fertilization, and seeding with native perennial grasses were carried out in replicated plots. After 6 months, little aboveground biomass was produced in plots without tilling, regardless of seeding or fertilization. Tilling alone tripled plant biomass, mostly of herbaceous forbs and annual grasses. Perennial grasses were essentially limited to plots that were both tilled and seeded. The addition of fertilizer had no significant additional effects. After 7 years, vegetation had declined, but there were still large differences among treatments. After 10 years, one tilled (and seeded) plot had reverted to bare ground, but the other tilled plots still had substantial vegetation. Only one seeded grass (*Cenchrus ciliaris*) was still a contributor to total cover after 10 years. We suggest that restoration efforts on these soils be directed first to breaking up the surface crust, and second to the addition of desirable seed. A simple ripping trial inspired by this experiment showed considerable promise as a low-cost restoration technique.

Key words: degradation, East Africa, infiltration, Kenya, Laikipia, overgrazing, rehabilitation, ripping.

Introduction

Approximately one-quarter of the Earth's surface is devoted to managed grazing (Asner et al. 2004). In semiarid rangelands, the interaction of heavy grazing and climatic variability can cause dramatic ecological degradation (Asner et al. 2004; Wessels et al. 2007). Despite extensive research on the causes and consequences of rangeland degradation, studies on rangeland restoration are less common, and findings are often anecdotal or context dependent (King & Hobbs 2006). In East Africa, rangeland degradation is serious and pervasive, but investigations of rangeland restoration have been especially rare (Descheemaeker et al. 2006; King & Stanton 2008). Here, we present an experimental investigation of the relative ability of three common restoration techniques to generate long-term rangeland improvements in a semiarid Kenyan savanna.

Heavy grazing initially alters vegetation composition and decreases primary productivity, especially of palatable species (Yates et al. 2000; Simons & Allsopp 2007). By reducing native species diversity and increasing the exposure of bare ground, heavy grazing can decrease community resilience and initiate damaging positive feedbacks (van de Koppel et al. 1997). For example, reduced vegetation cover can lead to reduced microtopography, increased runoff, and increased erosion, which in turn can lead to reduced seed retention, water availability, nutrient retention, and plant establishment (Jones & Esler 2004; Descheemaeker et al. 2006; Mati et al. 2006). Once bare ground is exposed, livestock and raindrops can cause soil compaction and reduce soil aggregate stability. Eventually, such structural degradation can lead to the formation of a surface seal that further reduces infiltration and hinders seed germination (Beukes & Cowling 2003). In Eastern and Southern Africa, many rangelands are now pockmarked by large bare areas with minimal organic matter and a smooth, sealed surface crust (van der Merwe & Kellner 1999). This study focuses on the restoration of such severely degraded areas, in particular on the goal of increasing vegetative cover.

Semiarid African rangelands are often characterized by threshold dynamics and alternate stable states that are highly resilient (Ellis & Swift 1988; Milton et al. 1994; Todd 2006). In this context, severely degraded rangelands can be viewed as lands that have undergone dramatic state shifts or threshold transitions (Milton & Dean 1995; van de Koppel et al. 1997; Bestelmeyer 2006). For the restoration of these lands, passive methods (e.g., the removal of livestock) are usually

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insufficient, because the degraded system is relatively stable and resilient in its undesirable state. Several studies from South Africa and the United States suggest that without active intervention, vegetation regeneration does not occur on timescales that are practical for land managers (Milton & Dean 1995; Wiegand & Milton 1996; Valone et al. 2002). In response to this problem, studies have explored the effectiveness of different active restoration techniques for severely degraded lands. Much of the research to date has focused on three key strategies: soil disturbance (i.e., breaking the surface seal), seeding of desirable species, and soil amendment.

Breaking sealed soil can increase plant establishment by increasing seed retention, water infiltration, and the presence of favorable microsites (van der Merwe & Kellner 1999). Seeding is important in sites that have lost connection (via dispersal or seed banks) to extant populations of desirable species (Sheley et al. 2006). Soil amendments such as fertilizer, mulch, salts, topsoil, or litter can improve the nutrient balance of degraded sites (e.g., Beukes & Cowling 2003; van den Berg & Kellner 2005; Belgacem et al. 2006). In some cases (e.g., topsoil or litter), the added material can also provide favorable germination sites, improved water retention, and higher seed retention (Rotundo & Aguiar 2005).

Despite evidence that each of the three active restoration approaches outlined above can improve the condition of degraded lands, few studies have explored their relative importance. At least two studies compared tillage and seeding (Snyman 2003; Cox & Anderson 2004), and one study compared seeding and soil amendment (Beukes & Cowling 2003). Of these, only Snyman (2003) monitored treatments for more than 4.5 years. To our knowledge, only three studies have compared all three approaches. Two of these were located in South Africa (Visser et al. 2004; van den Berg & Kellner 2005), and one was located in California (Sheley et al. 2006). All three of these studies were monitored for 3 years or less.

Rare long-term monitoring has shown that initial restoration successes may be transient (Snyman 2003). Long-term monitoring is crucial for restoration experiments, especially when results can be influenced by strong year effects (e.g., Cox & Anderson 2004; Visser et al. 2004). Interannual variability in precipitation is a major feature of arid and semiarid East African rangelands, where few active restoration techniques have been rigorously evaluated, either separately or in combination.

In this study, we followed a full factorial experiment for more than 10 years to determine the relative importance of ripping, seeding, and fertilization for long-term vegetation restoration in a degraded, semiarid Kenyan savanna. The major goals of this study were 2-fold:

(1) to identify the factor (or combination of factors) that most limit plant germination, establishment, and long-term persistence in a severely degraded rangeland, and

(2) to identify effective and efficient (in terms of both cost and effort) management strategies for creating long-term improvements in grass cover and plant species richness.

Study Site

Laikipia district is an important contributor to Kenya's ranching economy, having both large private ranches and communally owned group ranches. The communal ranches bordering the private ranches are widely denuded, with large bare patches (Augustine 2003). Although the private ranches appear to be well-managed, large patches of bare compacted red soil also deny the ranchers much needed livestock forage. Some ranch managers have attempted to treat these areas by opening up (ripping) the soil or covering it with cut bushes, with mixed success (J. Wreford-Smith 2003, Mpala Farm, personal communication). Nearby communal ranches urgently require remedial action to restore plant cover, and the local management knowledge of the ranchers needs be put on a more scientific standing.

This research was carried out at Mpala Research Centre in central Laikipia. The area had been moderately stocked with cattle for several decades before our study began. Preliminary reconnaissance identified a site on Mpala with large, uniform degraded patches. This site was located within 200 m of a cattle-dipping facility that received particularly high livestock impact. Degraded patches were all characterized by smooth, compacted soil and virtually no vegetation cover. The soil surface was close to level ($<2\%$ slope). The vegetation surrounding these bare patches was Acacia scrub, dominated by *A. etbaica* and *A. mellifera*, and an understory (when present) of various grasses and forbs, including *Eragraostis tenuifolia* and *Cynodon dactylon* (Augustine 2003). The study site ($35^\circ 52'$E, $0^\circ 23'$N) was located at an elevation of 1,680 m. Rainfall is weakly trimodal, with a distinct dry season lasting from December to March, and with an average annual rainfall of approximately 500 mm/yr, with great interannual variation. The initial research was carried out over the course of one rainy season, March–May 1997, with additional surveys in 2004, 2006, and 2007.

Methods

The study plots were located and marked in May 1996. Pretreatment soil compaction tests were done in October 1996. The treatments were established on 6 March 1997, just before the rainy season began. Thirty-two 2×5–m plots were established, all at least 3 m apart. Three treatments were applied: (1) tilling (ripping) the upper soil; (2) seeding with perennial grasses; and (3) fertilization with NPK fertilizer.

All eight combinations of these three treatments were applied (a complete $2 \times 2 \times 2$ factorial design) in four replicate blocks. Treatments were randomly assigned to the plots in each of four blocks. Initial conditions among the various plots were similar. All had hard, compacted soils and supported very little vegetation (essentially bare ground). Plots were first tilled, then fertilized, then seeded.

Tilling

Immediately prior to seeding, we opened the top 5 cm of the soil manually using a fork "jembe" (a locally available tool; see Appendix 2), on each designated plot.

Fertilization

A compound NPK fertilizer, 17:17:17 was applied on each designated 2 × 5–m plot on 6 March 1997. The rate of fertilization was 220 g of fertilizer in 2 × 5–m plots to achieve an effective application rate equivalent to 75 kg N, 75 kg P, and 75 kg K/ha (standard local agricultural rates for wheat).

Seeding

Four native grass species of high nutritional value and palatability were used: *Chloris gayana* Kunth. (var. Malaba and Elba), *Eragrostis superba* Peyr., *Chloris roxburghiana* Schult., and *Cenchrus ciliaris* L. Although introduced (and even invasive) in other parts of the world, these four species are native to throughout East Africa and specifically are native to Laikipia (Ibrahim & Kabuye 1987). The *C. gayana* seed was obtained from Kenya Seed Company (Box 553, Kitale, Kenya). All other grass seed was obtained from Kenya Agricultural Research Institute (KARI) National Range Research Station at Kiboko, Kenya. Seeds of the four grass species were evenly mixed and broadcast in each designated 2 × 5–m plot on 6 March 1997, at a rate equivalent to approximately 19 kg/ha (1.9 g/m^2). The approximate seed densities per species were: *Eragrostis superba* (2000/m^2), *Chloris gayana* (1200/m^2), *Cenchrus ciliaris* (100/m^2), and *Chloris roxburghiana* (20/m^2).

1997 Survey

Soil Compaction and Infiltration. In May 1997, we measured soil compaction (crust hardness) and infiltration. A soil penetrometer with a sharp point of 100.3 N force was used 15 times in every plot. The penetrometer registers the force needed to penetrate the surface of the soil. Infiltration rate was measured in every plot using a ring infiltrometer. This consisted of an outer cylinder 20.5 cm in diameter, and an inner cylinder 15.7 cm in diameter, both 20 cm tall. The infiltrometer was pressed firmly onto the soil and water was poured into both the inner space and the space between the cylinders. The height of the water in the inner space was measured every two minutes for 42 minutes.

Standing Biomass. On 31 May 1997 (12 weeks after seeding), a subsample of aboveground standing biomass was sampled in every plot. A 0.5 × 0.5–m quadrat was placed randomly in every plot, and the aboveground vegetation within the quadrat clipped to within 1 cm of the ground, separated by guild (perennial grasses, annual grasses, and forbs), bagged, oven-dried at 70°C for 24 hours and weighed.

2004 Survey. In July 2004 (7 years and 4 months after the treatments were put in place), we surveyed the plots again. Many of the original stone plot markers had been removed (by elephants and people), but we were able to relocate 17 of the plots. These were remarked with steel posts. In the intervening time, the plots had been moderately to heavily grazed by cattle. We measured percent aerial cover of each plant species in each plot. Because this was a dry-season measurement, absolute cover was low and this made pin frame estimates inefficient. For less vegetated plots, we measured every plant in the 2 × 5–m plots. For more vegetated plots, we measured each plant in two 1 × 1–m quadrats. We measured the basal diameter for grasses and the crown area for forbs. All individuals were identified to species or morphotype. Even in their dry, nonreproductive state, we could assign a species name to all but 28 of 743 individual plants in the plots.

2007 Survey. In July 2007 (10 years and 4 months after the treatments were put in place), we surveyed the plots a third time. We were able to relocate 11 plots. There had been recent rainfall, and the vegetation was green and often in flower. We measured plant cover by placing a 10-pin point frame at four regular intervals within each plot. We recorded the first pin hit by each species ("aerial cover"). We also recorded the identity of all plant species found in each plot.

2006 Survey of a 2003 Management Ripping. Based on our initial results (see below), the ranch management ripped soil in the vicinity of the study plots in 2003. A subsoil implement was dragged behind a tractor throughout the study area that had compacted soil devoid of vegetation. The implement consists of two teeth set 1.5 m apart. Each tooth is 6 cm wide and 32 cm long. The soil was ripped to a depth of approximately 20 cm (deeper than the experimental tilling).

In July 2004, these long linear rips were still devoid of vegetation. In July 2006, however, many had considerable colonizing vegetation. Virtually all sections either had more than 25% cover or were essentially unvegetated. Although there may have been unvegetated, ripped sections that had filled in with soil and were no longer recognizable, it appeared to us that we were able to find the great majority of the rips. We surveyed 1,050 m through seven of these rips, scoring measured lengths as vegetated or not. We then placed 42 10-pin frames regularly throughout the rips, half in the vegetated sections, and half in the bare sections. We counted the number of pin hits by each species. We also surveyed all the vegetated sections, recording all species present.

Statistics. For the 1997 data, in which all 32 plots were measured, all soil and vegetation variables were analyzed using three-way fixed factor analyses of variance (ANOVAs), with all interaction terms, after averaging values within each of the four replicate blocks. Tilling, seeding, and fertilization were the independent variables. In 2004 and 2007, we had fewer identified plots, and did not include a blocking factor. Based on the 1997 analysis (where all untilled plots were similar, and fertilization had nonsignificant effects), we analyzed these later surveys with one-way ANOVAs, lumping plots into three treatment classes (sample sizes in 2004 and 2007): untilled plots (6, 4), tilled plots without seeding (4, 3), and tilled plots with seeding (7, 3). Data collected in 2004 and 2007 were ln-transformed to ensure homogeneity of variances. All statistics were done using JMP software (SAS 2007).

Results

1997 Survey

Compactness and Infiltration. Surface-tilled soils were half as compact as untilled soils ($F = 617.3$, $N = 32$, $p < 0.0001$) and water infiltrated twice as quickly into tilled soils as into untilled soils ($F = 24.0$, $N = 32$, $p < 0.0001$). These variables were unaffected by seeding or fertilization ($p > 0.18$).

Standing Biomass. Tilling and seeding had strong effects on plant biomass accumulation and community composition (Table 1, Fig. 1, Appendix 3). The dry biomass of vegetation on plots that were not tilled was limited to a few annual grasses and not significantly different from the control plots, regardless of other treatments. In contrast, all tilled plots show large increases in biomass compared with the controls. In unseeded plots, the plant community was dominated by annual grasses and forbs. In seeded plots, the seeded perennial grasses gained dominance (hence the significant Tilling × Seeding interactions). Fertilization had no significant effects on vegetation.

2004 Survey. Total vegetation cover in 2004 was much lower than after original treatments in 1997, in part because of a prolonged dry period prior to the May 2004 survey, compared to the wet weeks before the 1997 survey. This meant that we were effectively surveying the basal parts of dessicated, dormant individuals. Nonetheless, treatment effects were still evident for *Cenchrus ciliaris* ($F = 72.7$, $p < 0.001$), other (nonseeded annual) grasses ($F = 12.6$, $p < 0.001$), and forbs ($F = 13.0$, $p < 0.001$) (Fig. 2). Untilled plots were essentially bare. The grass cover values were based on basal diameters of heavily grazed bunches, and these would be considerably greater during wet periods (see the 2007 Survey results). The only seeded grass evident after 7 years was *Cenchrus ciliaris*, which accounted for nearly one-third of the basal grass cover in the tilled and seeded plots.

2007 Survey. More than 10 years after the experimental treatments, tilled plots still had cover, and untilled plots still were essentially bare (Figs. 3 & 4; total cover: $F = 8.94$, $p < 0.02$). Treatments had significant effects on nonseeded annual grasses ($F = 12.1$, $p < 0.003$), and forbs ($F = 4.48$, $p < 0.05$), but not on *Cenchrus ciliaris* ($F = 1.55$, $p = 0.26$). Recent wet weather produced relatively high cover in the tilled plots, compared to the dry season survey in 2004. Although individual *C. ciliaris* plants had grown since 2004, they accounted for a minority of the total grass cover, which was dominated at the time of the survey by the annuals *Eragrostis tenuifolia* and *Tragus berteronianus.* A few individuals of *Chloris roxburghiana* were found. One tilled plot (out of seven) had reverted to bare ground. This plot (#5) was more vegetated in 2004, but even then had far less cover than other tilled plots. More than 40 species of plants were found in the other six tilled plots (see Appendix 1), which had a mean species richness of 16.

2006 Survey of 2003 Management Ripping. These data were from a management trial, not a controlled experiment, and are included here as an informal example of the kind of results one might get from this real-world management practice. Of the 1,050 m of surveyed rips, 41% (± 5%, SE) were vegetated (Appendix 4). The vegetated sections averaged 34% (± 3%) cover. The unvegetated areas had less than 1% cover. The vast majority of the plant cover was split between the annual grass *Eragrostis tenuifolia* (24.3 ± 3.4% cover) and the perennial grass *Cynodon* sp. (5.7 ± 2.0% cover). *Eragrotis* was essentially limited to the rips, but *Cynodon* was spreading beyond the rips via stolons, many of which were over 1 m long, and beginning to root at some of the nodes (see Appendix 4). No other plant species accounted for more than 1% cover. Additional plant species encountered in the 620 m of vegetated rips included seven grass species, 16 forb species, and the seedlings of two *Acacia* spp.

Discussion

This study was designed to investigate the relative importance of tilling, seeding, and fertilization for the restoration of severely degraded (denuded) rangelands in a semiarid Kenyan savanna. Large bare patches like the ones studied here are susceptible to soil degradation and erosion (Mati et al. 2006), have vegetative cover and diversity, provide no forage for livestock and wildlife, and rarely recover without active intervention (Milton & Dean 1995). Thus, it is imperative to find effective and efficient restoration techniques for such

Table 1. Results of three-way ANOVAs to evaluate the effects of tilling, seeding, and fertilization on the aboveground biomass of seeded perennial grasses, annual grasses, and forbs in May 1997.

		Seeded Perennial Grasses		*Annual Grasses*		*Forbs*	
Source	*df*	*F*	*p*	*F*	*p*	*F*	*p*
Tilling	1	43.76	<0.001	5.64	0.025	44.00	<0.001
Seeding	1	45.82	<0.001	12.02	0.002	11.99	0.002
Fertilization	1	1.45	0.24	0.14	0.70	4.02	0.056
Tilling × Seeding	1	43.76	<0.001	35.08	<0.001	25.28	<0.001
Tilling × Fertilization	1	1.85	0.19	0.40	0.53	0.13	0.72
Seeding × Fertilization	1	1.46	0.24	0.65	0.43	0.60	0.44
Tilling × Seeding × Fert.	1	1.85	0.19	0.08	0.78	0.29	0.60

Figure 1. Initial dry biomass accumulation of three plant functional groups in a dry Kenya rangeland as affected by experimental tilling, seeding, and fertilization. Error bars are one standard error of four replicate blocks.

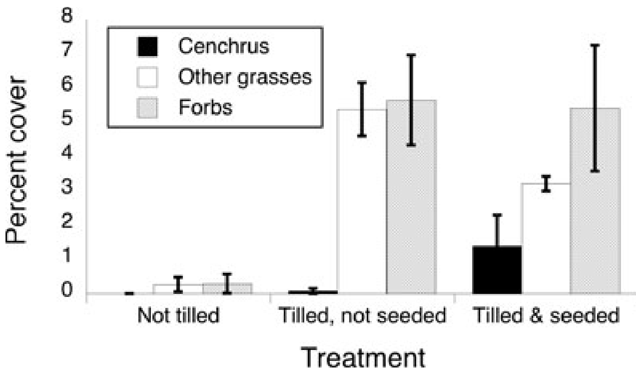

Figure 2. Dry season cover in 2004 of planted *Cenchrus ciliaris*, and nonplanted grasses and forbs as affected by tilling and seeding, 7 years after the intervention.

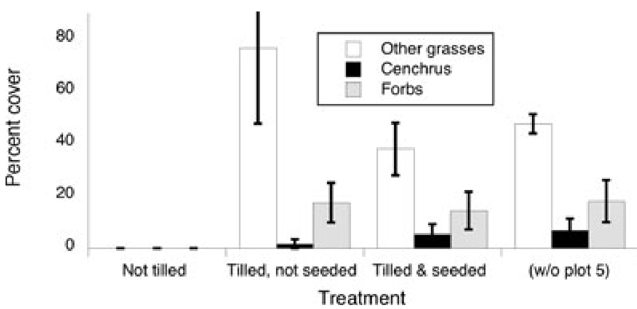

Figure 3. Wet season cover in 2007 of planted *Cenchrus ciliaris*, and nonplanted grasses and forbs as affected by tilling and seeding, 9 years after initiation. One tilled and seeded plot (#5) had reverted to bare ground, and the values for this treatment are presented with and without this plot.

areas. In this study, although differing vegetation conditions required different sampling methods across surveys, the results were very consistent. It appears that the combination of tilling (breaking the surface of compacted soil) and broadcast seeding of desirable perennial grasses (particularly *Cenchrus ciliaris*) is the most effective restoration prescription for these degraded sites. Studies from South Africa and the western United States have shown similar patterns, emphasizing the necessity of soil tilling, and the desirability of seeding (Snyman 2003; Visser et al. 2004; Cox & Anderson 2004; van den Berg & Kellner 2005).

The only other semiarid restoration trials that simultaneously tested tilling, seeding, and fertilization all had durations of 3 years or less (Visser et al. 2004; van den Berg & Kellner 2005; Sheley et al. 2006). Our 10-year study showed signs of an overall decline in restoration success through time (Snyman 2003), but also consistent long-term treatment effects. This suggests that short-term studies may anticipate long-term answers to treatment questions, at least qualitatively.

It appears that the most limiting feature of these soils is their surface compactness. Only plots with ripped soils had more than minimal cover, and these had considerable plant biomass of both seeded and nonseeded species. It is clear that soil ripping can greatly decrease soil compaction and increased infiltration (Osunbitan et al. 2005). This is a likely contributor to the success of plants in tilled plots. Tilling may also have added in the capture of seeds transported along the soil surface by wind and runoff water (Visser et al. 2004) in the ability of roots to penetrate the soils once germinated (Snyman 2003), and in protecting seeds from predators.

Although ripping alone had a large effect on plant biomass, perennial grasses only established in seeded plots. It appears that the local seed rain of perennial species at our site was too low (at least in 1997) to produce appreciable recruitment without direct seeding. We note, however, that the desirable perennial grass *Cynodon* did successfully colonize the management scratches initiated in 2003. *Cenchrus ciliaris* has been identified as a preferred restoration species in African rangeland (Mnif & Chaleb 2006; King & Stanton 2008), perhaps in part because of the kind of establishment success we also found in our study. It has been shown to do particularly well on sandy soils (Snyman 2003). It is also possible that the greater success of seeded *Cenchrus* in our experiment was

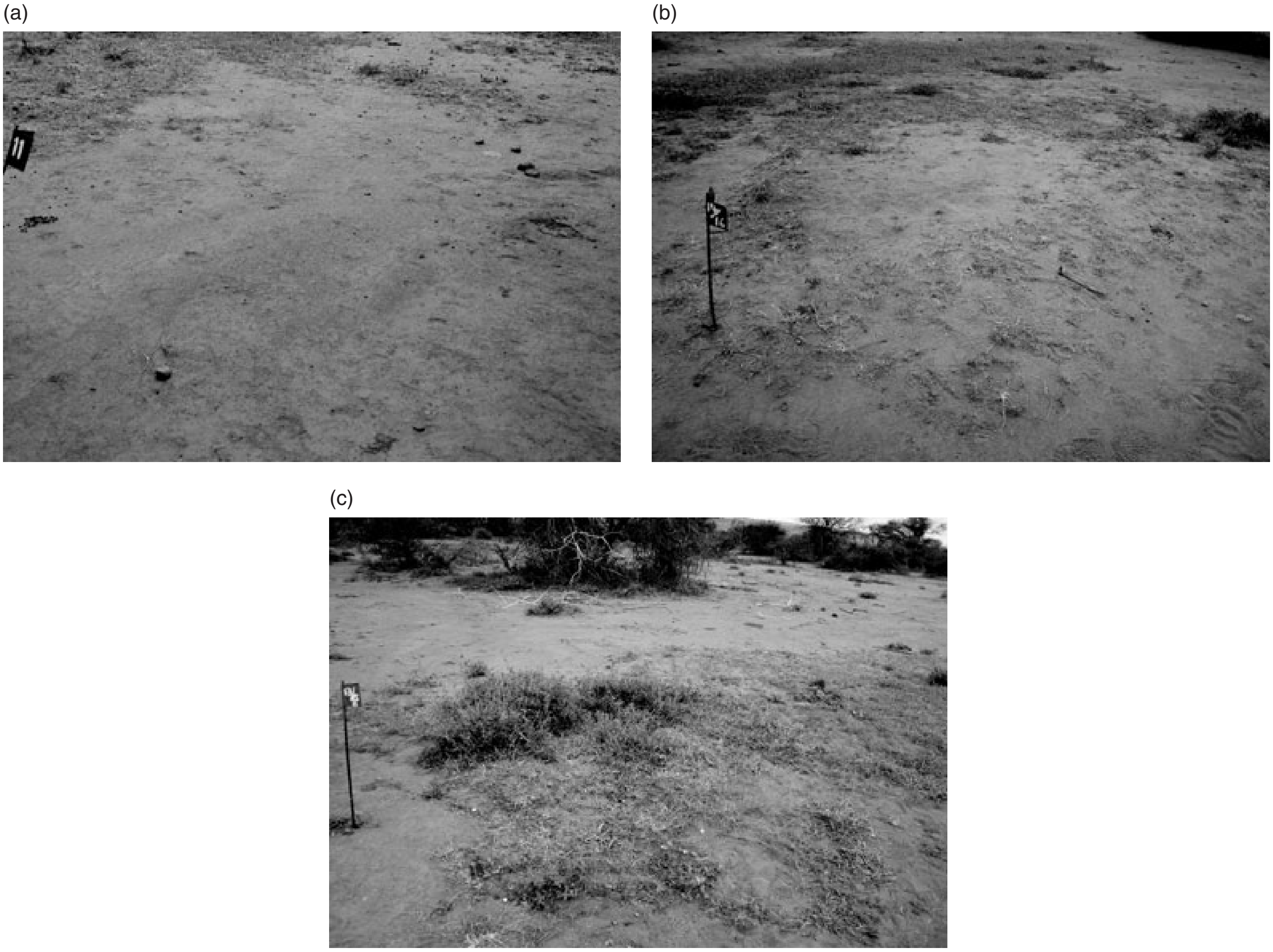

Figure 4. Photos of three of the study plots in July 2007. Top: untilled plot; Middle: tilled, unseeded plot; Bottom: tilled and seeded plot.

due to some unique characteristics of the seeding year. Given the strong year-to-year variability in arid and semiarid systems, it seems prudent to actively seed restoration sites, and with multiple species.

The large biomass accumulation of seeded grasses in 1997 was associated with a decline in the cover of spontaneous annual grasses and forbs in the seeded plots, apparently through competitive exclusion. In 2004, and to a lesser extent in 2007, the increase in *Cenchrus ciliaris* in seeded plots was associated with a decrease in spontaneous grasses but not forbs, suggesting more guild-specific competition at lower plant abundances.

Although there is some evidence that plant growth on these soils is N and P limited (Augustine et al. 2003), the addition of an NPK fertilizer had little or no effect on any measured plant abundances in our study. Although we may have missed some subtle short-term effects of fertilization, it appears that fertilization at these sites is unlikely to be cost effective.

We are intrigued by the observation that at least one of our tilled and seeded plots reverted to compacted bare ground, even after a promising initial response. It is also possible that some of the other plots that we were unable to relocate in 2004 and 2007 also reverted to bare ground. We were not able to identify any biotic or abiotic factors that led this particular plot to revert to bare ground. In any case, this reversion suggests the existence of possible multiple stable states in this system—in this case, degraded and bare versus restored and vegetated (cf. Ellis & Swift 1988; Milton et al. 1994; Todd 2006).

Certainly, the current landscape appears to be a mosaic of vegetated and unvegetated patches (Augustine 2003), a situation not uncommon in semiarid ecosystems (Aronson et al. 1993; van der Merwe & Kellner 1999; Ludwig et al. 1999). Only some of these patches seem to be related to obvious external factors, such as greater herbaceous cover beneath low tree canopies, and bare ground associated with harvester ant and termite mounds. The persistence for nearly a decade of substantial vegetation in many of our tilled (and seeded) plots suggests a relatively stable restoration response, but it is always possible that the reversion of the one plot to bare ground is a harbinger of a slow reversion of all the restored plots. If this is the case, any estimate of the cost-effectiveness of this kind of rangeland restoration would need to take into account the duration of the range improvement as well as the degree.

Implications for Practice

- Supplemental seeding on bare, compacted soils is unlikely to be effective without first breaking the soil surface.
- Although tilling alone produced large increases in plant cover and biomass, this was principally by annual plants, which have limited value for grazing.
- Fertilizer application had little discernable benefit.
- Our results suggest that a combination of soil tilling and broadcast seeding of desirable perennial grasses (particularly *Cenchrus ciliaris*) produced the best results. Indeed, this very combination is now being used with success in other parts of Laikipia (King & Stanton 2008; Young & McGeoch 2008, personal observation)

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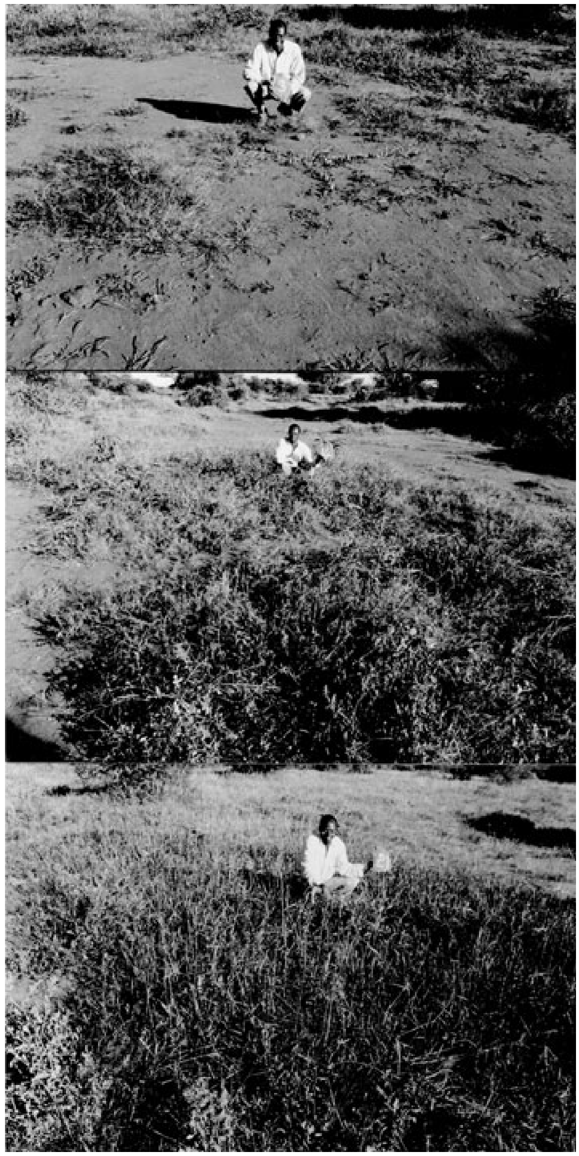
Appendix 1. Species found in the experimental plots in July 2007.

Common species (>2.5% of relative cover, in order of abundance):
- *Eragrostis tenuifolia*
- *Cenchrus ciliaris*
- *Barleria spinisepala*
- *Cyathula polycephala*
- *Melhania ovata*
- *Indigofera volkensii*
- *Tragus berteronianus*
- *Cynodon dactylon*
- *Ipomoea kituiensis*
- *Hypoestes forskahlii*
- *Solanum incanum*
- *Chloris virgata*

Uncommon species (≤1% of relative cover):
- *Chloris roxburghiana*
- *Ipomoea* sp.
- *Portulaca* sp.
- *Barleria* sp.
- *Aristida keniensis*
- *Microchloa kunthii*
- *Phyllanthus* sp.
- *Dactyloctenium aegyptium*
- *Gutenbergia* sp.
- *Hibiscus* sp.
- *Acacia mellifera* (seedling)
- *Acacia etbaica* (seedling)
- *Acacia nilotica* (seedling)
- Four unidentified forb spp.

Appendix 2. Photo of a jembe, used to break the sealed soil.



Appendix 3. Photos of three of the study plots in May 1997. Top: untilled plot (fertilized, no seeds added); Middle: tilled plot (fertilized, no seeds added); Bottom: Tilled and seeded plot (fertilized).

Appendix 4. Parallel vegetated rips in 2006 from the 2003 management trial. The area in the lower center shows the spread of *Cynodon* sp. well beyond the initial rips.