LAND DEGRADATION & DEVELOPMENT
Land Degrad. Develop. (2013)
Published online in Wiley Online Library (wileyonlinelibrary.com) **DOI**: 10.1002/ldr.2224

TRADEOFFS IN THE REHABILITATION OF A SUCCULENT KAROO RANGELAND

Wiebke Hanke*, Dirk Wesuls, Wiebke Münchberger, Ute Schmiedel
Biodiversity, Evolution and Ecology of Plants, Biocentre Klein Flottbek, University of Hamburg, Ohnhorststr. 18, 22609, Hamburg, Germany

Received: 15 November 2012; Revised: 4 April 2013; Accepted: 4 April 2013

ABSTRACT

Rangeland rehabilitation has multiple, sometimes conflicting goals, such as the reestablishment of the predisturbance vegetation, soil protection, and forage production. The rehabilitation techniques should be also cost-effective and practicable. Given the difficulties and high costs of restoring Succulent Karoo rangelands and the continuously high grazing pressure in the communal lands, tradeoffs should be accepted in the achievement of these goals. We tested the capability of paddock manure redistribution to reverse degradation trends in a heavily grazed Succulent Karoo rangeland in South Africa. Over 3 years, the effects of the manure application were compared with areas planted with mature shrubs as a benchmark for a predisturbance vegetation structure and with four popular rehabilitation techniques: (1) livestock exclusion; (2) brushpacking (coverage of dead shrubs); (3) mineral fertilizing; and (4) microcatchment construction. Manure was, besides planting, the only treatment that resulted in a significant increase in drought-resistant vegetation cover, but it compromised the dominance of native vegetation. In the manure plots, a pasture-like vegetation of non-native forage plants (which germinated mainly from seeds in the dung), developed (foremost *Atriplex semibaccata).* Manure application counteracted erosion as effectively as the planted shrubs and brushpacks. Expected negative side effects such as a decrease in plant species richness or salinization of topsoil were not detected. We also checked the potential of topsoil salinization by the halophytic *A. semibaccata* and found it to be low. For sites where a decrease in grazing pressure is unrealistic under current land tenure, redistribution of manure should be further explored to mitigate acute symptoms of degradation. Copyright © 2013 John Wiley & Sons, Ltd.

KEY WORDS: *Atriplex semibaccata*; communal lands; livestock manure; plant–soil feedbacks; rangeland rehabilitation; Succulent Karoo

INTRODUCTION

The arid to semiarid Succulent Karoo in southwestern Africa is a global biodiversity hotspot hosting one of the richest succulent dwarf shrub vegetation on Earth (Myers *et al*., 2000). The recent land management attaches great importance to the protection of plant diversity. At the same time, about 90% of the area is used for livestock farming in rangelands (Todd *et al*., 2009), which is the traditional type of land use and an important contribution to the livelihoods of the rural population (Hoffman *et al*., 2007). The rangelands are predominantly covered by near-natural vegetation, because intensive agricultural practices such as growing forage plants or fertilization are not common in this arid environment.

Ideally, rangeland productivity is sustained by adapted management practices such as limited stocking densities and rotational grazing (Todd *et al*., 2009). However, as a result of the inequitable land allocation during the colonial and Apartheid eras, in the communal areas of South Africa, livestock numbers are usually high compared with the allocated area (Hoffman *et al*., 2007; Samuels *et al*., 2007). Openly accessible rangelands, for example, in the proximity of settlements, are grazed persistently at high stocking densities (Birch, 2000). This leads to land degradation associated with the replacement of drought-tolerant, long-lived shrubs by a sparse cover of short-lived shrubs and eventually by annuals (Mucina *et al*., 2006; Todd & Hoffman, 2009). In consequence, farmers have to cope with decreased rangeland productivity and severe fluctuations in the forage resource (Vetter, 2005). Further, the land degradation is often associated with topsoil erosion (Beukes and Ellis, 2003), surface compaction (Mills & Fey, 2004; du Toit *et al*., 2009), and an increase in soil salinity (Gröngröft *et al*., 2010).

In the past, researchers have tested different rehabilitation methods to reverse the degradation trends in the rangelands of the Succulent Karoo, pursuing multiple goals: (1) the reestablishment of a perennial, native vegetation; (2) forage production; (3) protection of the soil against erosion, surface compaction, and salinization; and (4) the identification of practicable, cost-effective techniques (compare Botha *et al*., 2008). The prioritization of the goals depends on the specific resource demands and can differ among stakeholders (Choi *et al*., 2008). Conservationists, for example, would usually give priority to the recovery of a predisturbance, native plant cover, whereas farmers mostly focus on forage production.

Achieving all goals simultaneously is difficult. The success of techniques that improve the abiotic soil conditions for plant growth (e.g., brushpacking or digging microcatchments) was mostly confined to an increase of ephemeral species (review by Hanke & Schmiedel, 2010), because long-lived shrubs

*Correspondence to: W. Hanke, Biodiversity, Evolution and Ecology of Plants, Biocentre Klein Flottbek, University of Hamburg, Ohnhorststr. 18, 22609 Hamburg, Germany.
E-mail: w.hanke@biota-africa.org

have often totally disappeared from the soil seed bank (Esler, 1999; De Villiers *et al.*, 2003). At this stage, the reestablishment of perennial vegetation requires the active reintroduction of the desired species (Burke, 2007; Hanke & Schmiedel, 2010). However, even such biotic manipulations do not guarantee recovery. Seeding or the introduction of seedlings repeatedly failed because establishment success is highly dependent on good rainfalls (Beukes & Cowling, 2003; Simons & Allsopp, 2007; Burke, 2008). Transplantation of mature shrubs showed a higher success rate and resulted in a vegetation structure similar to that of nondegraded rangelands (Blignaut & Milton, 2005; Botha *et al.*, 2008; Meyer, 2009), but great expenses of personnel, time, and resources are associated with this method (Carrick & Krüger, 2007). The effort is worthwhile only if the survival of the transplants will be later ensured by a sustainable grazing regime. Given that at current land tenure arrangements a decrease in grazing pressure is not realistic in the surroundings of settlements in communal rangelands (Hoffman *et al.*, 2007), reestablishing the predisturbance vegetation may be a too idealistic goal in these areas.

A more practicable alternative, which in other rangeland systems successfully mitigated soil degradation and increased productivity, is the redistribution of livestock manure (Cabrera *et al.*, 2009; Mugerwa, 2009). In traditional livestock husbandry in Southern Africa, livestock is returned to paddocks at night, where the dung concentrates in thick accumulations, whereas soils in the rangelands become depleted due to the persistent removal of plant material (Kizza & Areola, 2010). Mulching intensely grazed rangelands with paddock manure could protect bare soil patches from erosion and increase soil organic matter and fertility. There are, however, also potential problems that might occur. Manure application may alter the native plant composition and decrease biodiversity (Stavast *et al.*, 2005). A further negative side effect might be the raising of salt levels in arid soils, either directly caused by release of salt from the dung (Hao & Chang, 2003), or indirectly caused by an alteration of plant communities, which in turn could affect plant–soil feedbacks. Herbivore dung in Southern Africa often carries seeds of halophytic, non-native *Atriplex* species (Milton & Dean, 2001). Thus, abundance of these species in the vegetation might increase after manure application. This could be problematic because species of the genus *Atriplex* are known to accumulate salts in their tissues and release them onto the soil surface when they die (Sharma & Tongway, 1973). However, manure mulching has hardly been tested in the Succulent Karoo, and scientific proof that such feedbacks really interfere with recovery processes in this ecosystem is lacking.

The present study seeks techniques to combat the loss of vegetation cover and soil erosion in a heavily grazed site in the Richtersveld, the largest communal area in the Succulent Karoo. We compared the impacts of manure mulching with the impacts of popular rehabilitation techniques (livestock exclusion, brushpacking, fertilizing, and microcatchments) and restoration by transplantation of mature shrubs. We evaluated how the treatments affect the soil (erosion, surface compaction, and salinity) and the vegetation (total plant cover, species richness, and perennialness) during 3 years after application, capturing different rainfall scenarios. Further, we assessed the potential of the plant species that increased in response to the manure treatment to induce topsoil salinization.

MATERIAL AND METHODS

Study Site

The experimental site is located in the northern Succulent Karoo, in the Richtersveld, near the village of Eksteenfontein (28°49′S, 17°15′E) on a gently undulating plain. Soils are mainly Epipetric Calcisols consisting of silty fine sands of 10 to 40 cm thickness underlain by calcrete (author's own observation). The mean annual temperature is 16.4 °C and the mean annual precipitation is 93 mm, of which 52 mm falls during the growing season from May to September (Hijmans *et al.*, 2005). This winter rainfall is reliable and mostly occurs as a fine, mist-like drizzle, whereas summer rainfall is erratic and often associated with erosive downpours of thunderstorms (Desmet & Cowling, 1999). Strong winds occur during most days of the year (Odendaal & Suich, 2007). The main type of land use is communal livestock farming with sheep and goats. As a result of continuous high grazing pressure, the original plant communities dominated by the flat cushions of the long-lived, succulent dwarf shrub *Brownanthus pseudoschlichtianus* (Aizoaceae), are largely replaced by a sparse cover of short-lived shrubs and ephemerals (Mucina *et al.*, 2006).

Experimental Layout

The experiment was implemented in October 2007. Five 1-ha-sized blocks were led out randomly within a study area of about 7 km^2 in the vicinity of the village Eksteenfontein. Half of each block was fenced off to exclude livestock. In all blocks, five restoration treatments and a control were randomly applied to 10 × 10 m plots inside and outside the exclosures (Figure 1). So in total, each of the following treatments was applied 10 times.

Brushpacks

Branches picked from *B. pseudoschlichtianus* shrubs were spread in the plots covering 60 to 70% of the soil surface to simulate healthy vegetation patches.

Manure

To cover bare soil, we spread out increase soil organic matter and enhance fertility old sheep and goat dung from nearby paddocks in a layer that was up to 3 cm thick and covered about 90% of the soil.

Fertilizer

To test the impact of nutrient supply alone, we applied a mineral fertilizer. The fertilizer was spread at a rate of 22 kg ha-1 nitrogen, 3·7 kg ha-1 phosphorus, and 18·3 kg ha-1 potassium.

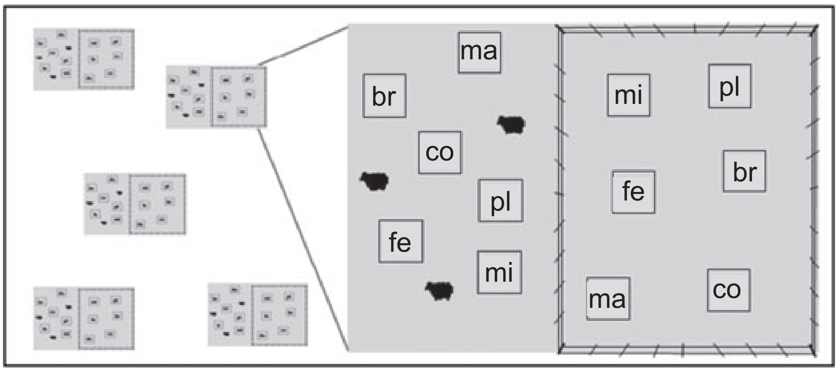

Figure 1. The experimental layout is a split-plot design, repeated in five blocks. Half of each block was fenced off to exclude livestock. Five restoration treatments and one control were randomly applied inside and outside the exclosures to 10× 10 m plots. Thus, each treatment was applied 10 times. co = control, br = brushpacks, ma = manure, fe = fertilizer, mi = microcatchments, pl = planting.

It was applied once in May 2008 (half a year later than the other treatments) at the start of the rainy season.

Microcatchments

To create soil patches favorable for plant growth, we dug 25 crescent-shaped pits (10 cm deep, 40 cm wide, and 60 cm long) per plot. They were dug perpendicular to the slope to trap water, seeds, and organic material. Excavated soil was piled to form a berm on the downslope side of each pit. The pits were arranged in a staggered fashion, 2 m apart from each other.

Transplanting

Twenty mature individuals of *B. pseudoschlichtianus* were transplanted from the surrounding areas to the designated plots and watered for 2 weeks. The transplants were arranged in a regular fashion with at least 2 m between them. Three years after planting, 65% of the transplanted individuals had survived. Transplanting served as a benchmark for the optimum outcome – a vegetation of long-lived shrubs similar to that of nondegraded rangelands.

Field Sampling and Analyses

Rainfall data were recorded at an automatic weather station (MC Systems, Cape Town, South Africa) in the center of the study site. Annual rainfall was calculated for hydrologic years (1 October to 30 September). After treatment application in the plots, projected vegetation cover per vascular plant species was estimated in percent with an accuracy of 0·01. This sampling was conducted annually during the vegetation period in three consecutive years (2008–2010).

Whereas the effects on vegetation were assessed for the whole plot, soil related measurements were taken at points within the plot likely to experience the full impact of the treatment (i.e., points that were covered by brushpacks, transplants, or manure or located in the microcatchments). Soil surface compaction and electrical conductivity (EC) were sampled once 2 years after treatment application. Five subsamples per plot were taken, one in each of the four corners and one in the center of the plot. EC was measured in a composite of the five subsamples from the upper soil layer (0–10 cm) in a suspension of soil and distilled water with a ratio of 1:2·5. Soil surface compaction was measured as the physical resistance using a pocket penetrometer (Model 06·03, Eijkelkamp®, Eijkelkamp Agrisearch Equipment, Giesbeek, the Netherlands). The median of the five subsamples per plot was used for statistical analyses.

Soil erosion (or soil deposition) was measured by metal pins driven into the ground during treatment establishment. They were arranged in transects trough, the plots placed in unvegetated soil, except in the transplanting treatment, where the pins were installed in the periphery of the transplants. The number of installed pins per plot varied among the treatments. Valid pins (i.e., not bent by livestock trampling) per plot were 2 to 3 in the controls, 4 to 6 under the brushpacks, 9 to 12 under the transplants, one in the manure, and one in the fertilizer plots. One year later, in 2008, after the disturbance of the soil surface, caused by treatment establishment and pin installation, had abated the pin height above soil surface was recorded. The height was remeasured in 2009 and the difference taken as a measure of soil erosion (or deposition) in that year. Means per plot were used for the subsequent analyses. Because sedimentation in the microcatchments occurred at a much higher magnitude, soil deposition for this treatment was estimated visually using the pit's initial depth of 10 cm as reference.

The paddock dung used for the manure treatment was assessed with regard to viable seed material in 2007. In each of the 10 manure plots, a sample of 80 ml (composites of five subsamples collected at the corners and center of a plot) was taken for germination trials in the greenhouse according to the methods described in Dreber & Esler (2011).

In the first year after treatment, we observed that in the manure plots, two species of genera well-known for topsoil salinization had considerably increased: the pauciennial *Atriplex semibaccata* (Chenopodiaceae) and the annual *Mesembryanthemum hypertrophicum* (Aizoaceae). We assessed their potential to induce topsoil salinization in a supplementary study in 2008. For this purpose, five adult individuals of each of the two species were selected randomly in the vicinity of the experimental blocks. Topsoil samples (0–5 cm depth) were collected in vegetated patches under the canopies and in bare soil at about 2 m distance to the vegetated patches. Then, the EC was determined as a measure for salt content.

All statistical analyses were carried out separately for each year. The treatment effects on total plant cover, species richness, and the soil parameters were analyzed by split-plot ANOVAs using Statistica 8 (Statsoft, Tulsa, Oklahoma, USA). We followed the procedure laid out in a paper on split-plot designs by StatSoft (2003). Treatment and livestock exclusion were included as fixed factors and block as random factor. All variables were normally distributed except soil deposition, which had to be log-transformed to achieve normal distribution. In case of significant differences, the treatments were compared with the control using the Dunnett's test.

In the second step, the impact of the treatment on the cover of functional plant groups was analyzed by conducting four separate tests for each group. For this purpose, all plant species found in the plots were assigned to four groups on the basis of their life form and their life span: (1) long-lived shrubs; (2) short-lived shrubs; (3) pauciennial forbs; and (4) annuals (Table I). Perennial grasses and geophytes were excluded from the groupwise analyses because their occurrences were negligible. Because of the high number of zero values, we employed a nonparametric method, that is, the Wilcoxon matched-pairs test. The livestock exclusion effect was analyzed by matching the treatment pairs inside and outside the exclosures, which resulted in 30 matched pairs (six fenced and six unfenced plots in each of the five blocks). The treatment effect was analyzed by matching each treatment to the control. This resulted in 10 treatment-control pairs per treatment consisting of one pair inside and one pair outside each exclosure.

The data from the supplementary study on topsoil salinization by *A. semibaccata* and *M. hypertrophicum* were analyzed by paired *t*-tests comparing the EC of soil samples from under the canopies and from the bare patches nearby.

RESULTS

The annual rainfall was 80 mm in 2007/08, 145 mm in 2008/09, and 68 mm in 2009/10. The last year can be considered a drought year due to the lack of rain in August (Figure 2). Two treatments, transplanting and manure, resulted in a significant and lasting increase of the total plant cover. This effect also withstood the general decline in cover in the drought year (Table II, Figure 3a). The increase in the transplanting plots almost exclusively consisted of the *B. pseudoschlichtianus* transplants; the increase in the manure plots was mainly formed by the pauciennial *A. semibaccata (*Figure 3b). In the good rainfall year 2008/09, the plant cover in the manure plots clearly exceeded that of the other treatments. In that year, *A. semibaccata* became the dominant species in the manure plots, covering more than 15%. Manure mulching was the only treatment that increased species richness (Figure 3a).

Other significant effects of the treatments on the vegetation were restricted to the group of annuals (mainly the Asteraceae *Foveolina dichotoma)* and fluctuated between the years *(*Figure 3b). The cover of annuals significantly increased in the fertilized plots in the first year, in the transplanting plots in the second year, and under livestock exclusion in both years. Also in the manure plots, the annual *M. hypertrophic*um tended to increase in the first 2 years (mean cover of about 1% compared with $<0{\cdot}1\%$ in the control). However, because of a high variability among the plots, this increase was not reflected in a significant increase of the group "annuals".

All treatments, except livestock exclusion, had significant effects on the soil parameters (Table II, Figure 4). They significantly increased the deposition of soil compared with the control plots in which we found a slight topsoil loss of $-0{\cdot}2$ mm in the measurement period. Soil deposition was about 4 to 5 mm under the plantings, the manure and the brushpacks and about 2 mm in the fertilizer plots. Soil deposition was highest in the microcatchments (about 36 mm), but the deposited material might partly come from the berms. Also in the microcatchments, soil surface compaction was significantly decreased. The penetration resistance was $2{\cdot}2\,\mathrm{kg\,cm^{-2}}$ in the pits compared with $3{\cdot}5\,\mathrm{kg\,cm^{-2}}$ in the control plots. EC was significantly increased under the brushpacks ($620\,\mu\mathrm{S/cm^{-1}}$) and under the transplants ($491\,\mu\mathrm{S/cm^{-1}}$) as compared with the control ($211\,\mu\mathrm{S/cm^{-1}}$). Also in the manure plots, EC was found to be slightly increased ($362\,\mu\mathrm{S/cm^{-1}}$), although not significantly.

Generally, no significant interactions of the active treatments and livestock exclusion were found; neither for the effect on vegetation nor for the effect on soil parameters.

Table I. Definition of the plant groups used in this study. The estimation of palatability is based on Hendricks *et al*. (2002) and personal communications with the local herders

Plant group	Ecological strategy	Palatability	Most abundant taxa
Long-lived shrubs	Slow growing	Moderate	*Brownanthus pseudoschlichtianus, Zygophyllum prismatocarpum, Zygophyllum retrofractum, Salsola zeyheri*
Short-lived shrubs	Opportunistic	Moderate	*Galenia sarcophylla, Psilocaulon spp., Drosanthemum inornatum*
Pauciennials	Pioneers, biennial to triennial	High	*Atriplex semibaccata, Atriplex lindleyi*
Annuals	Pioneers, annual to biennial	High	*Foveolina dichotoma*, other Asteraceae, Fabaceae, Poaceae, *Mesembryanthemum hypertrophicum*

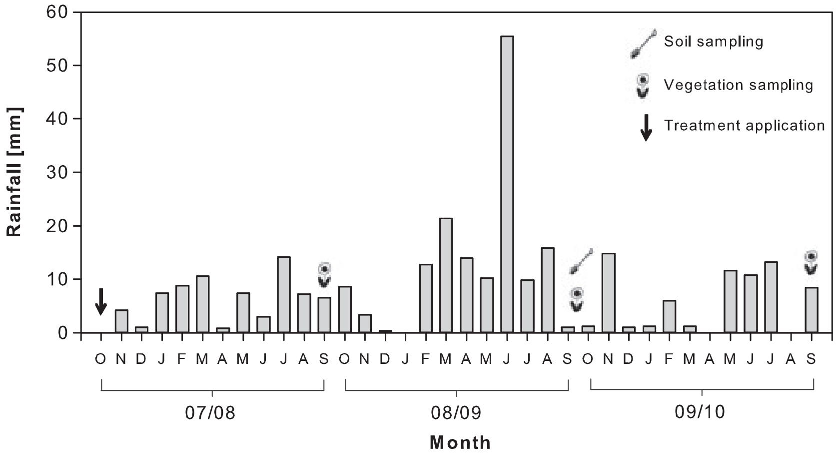

Figure 2. Monthly rainfall from a weather station at the study site. The braces indicate the hydrologic years.

The supplementary study on plant-induced topsoil salinization showed that topsoil EC was significantly increased ($p < 0.05$) under *M. hypertrophicum* ($463 \pm 146\,\mu S/cm^{-1}$, mean ± SD) compared with the bare patches beside ($183 \pm 70\,\mu S/cm^{-1}$). EC under *A. semibaccata* ($406 \pm 59\,\mu S/cm^{-1}$) was not significantly different ($p = 0{\cdot}2$) from the bare patches beside ($293 \pm 156\,\mu S/cm^{-1}$).

The assessment of the viable seed material in the manure gave the following results: of 415 germinated seedlings, 46% were annuals, 27% pauciennials, 7% short-lived shrubs, and 5% long-lived shrubs. The other 17% could not be identified. The most abundant species were the annual forb *Chenopodium murale* (Chenopodiaceae, 101 individuals), the pauciennial forb *A. semibaccata* (Chenopodiaceae; 99 individuals), the annual grass *Karoochloa schismoides* (Poaceae; 41 individuals), and the annual forb *Lotononis spec.* (Fabaceae; 16 individuals).

DISCUSSION

Treatment Effects

We evaluated the capability of five active restoration treatments combined with livestock exclusion, to reverse degradation trends in a communal rangeland in the Succulent Karoo. Within the 3 years of this study, the reestablishment of a drought-resistant vegetation cover was achieved only by the treatments transplanting and manure application. Both treatments involved biotic manipulations: (1) the transplanting trough; (2) the introduction of mature shrubs; and (3) the manure contained seeds of preferred forage plants. The increase in vegetation cover in the manure plots consisted particularly of the non-native herb *A. semibaccata*, which benefitted from the seed introduction through the dung. *A. semibaccata* belongs to a group of forage plants, which were introduced to South Africa from Australia in the late 18th century, have been naturalized, and are now widespread pioneers in degraded Succulent Karoo rangelands (Milton *et al*., 1999). Seeds of non-native *Atriplex* species are a typical feature in herbivore dung in the different dryland biomes of Southern Africa (Milton & Dean, 2001), suggesting that our findings might be transferable to other sites. Because of its pauciennial nature, *A. semibaccata* created a type of vegetation, which was more resistant to drought than the annual communities in the fenced or fertilized plots. In fact, the total plant cover values in the manure plots during the 3 years, despite varying rainfall conditions, never fell below that of the plots with the transplants (Figure 3). The *Atriplex* cover thus buffered fluctuations in forage resources, overcoming one of the main challenges for farmers to cope with in degraded arid rangelands (compare Vetter, 2005).

The treatments that relied on abiotic manipulations were successful in ameliorating the soil conditions by decreasing the erosion potential and the soil surface compaction but similar to other studies in the Succulent Karoo, their effects

Table II. Results of split-plot ANOVAs testing the effects of the factors block, treatment and livestock exclusion, on total plant cover, species richness, soil deposition, surface compaction, and electrical conductivity (EC). The *p* values in bold denote statistically significant effects. $n = 10$

Source of variance		Total plant cover			Species richness			Soil compaction	Soil deposition	Soil EC
	df	2008	2009	2010	2008	2009	2010	2009	2009	2009
Block	4	**0·007**	**<0·001**	**0·028**	**0·040**	**0·007**	0·075	0·576	0·676	0·260
Livestock exclusion	1	0·074	0·534	0·875	0·576	0·847	0·577	0·613	0·744	0·990
Treatment	5	**<0·001**	**<0·001**	**<0·001**	**<0·001**	0·079	**0·008**	**<0·001**	**<0·001**	**<0·001**
Livestock exclusion × treatment	5	0·587	0·333	0·881	0·475	0·511	0·818	0·665	0·120	0·850

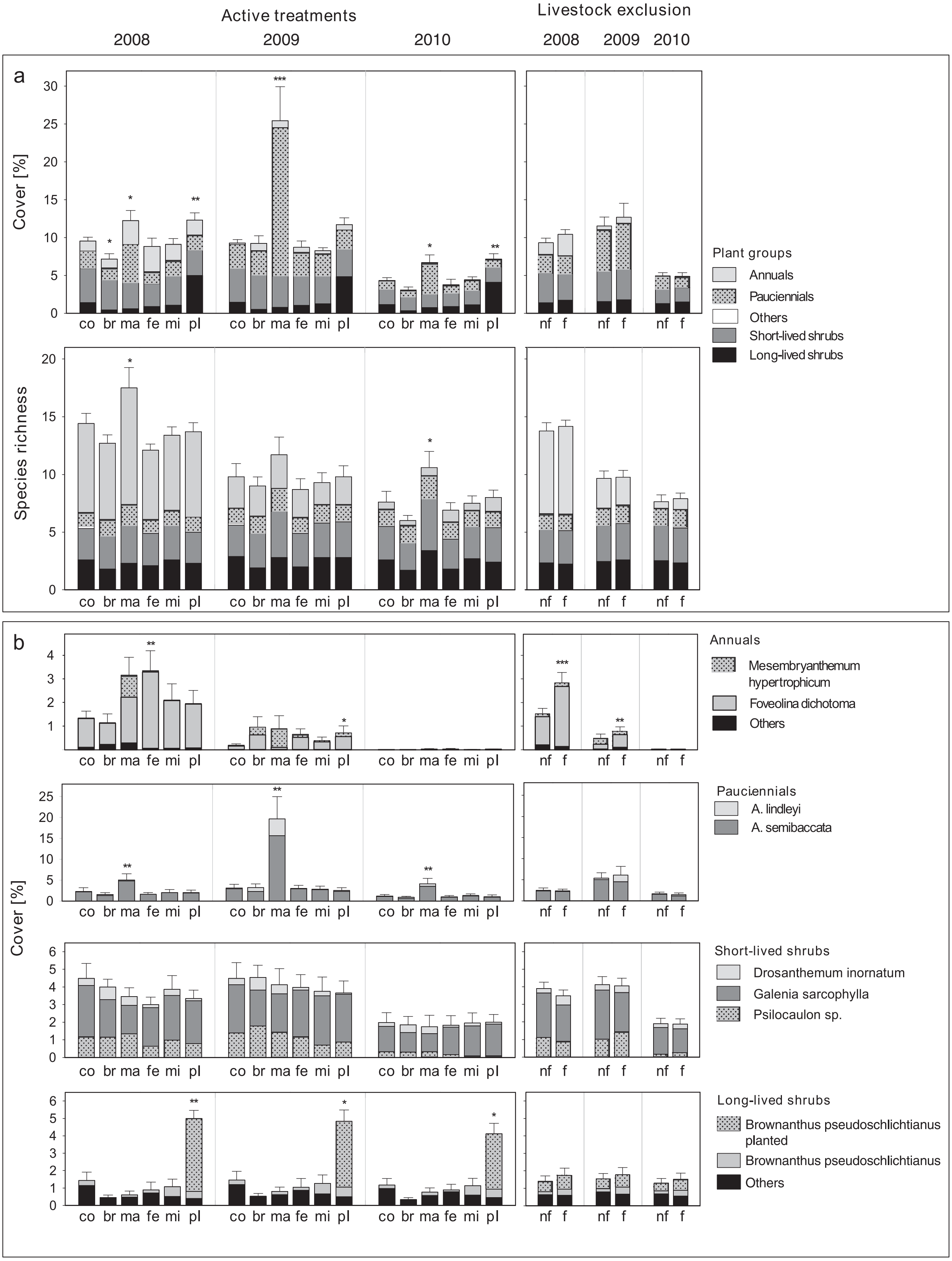

Figure 3. (a) Means (+1 SE) of total plant cover and species richness. Asterisks indicate significant differences from the control within the respective year as derived from Dunnett's tests following the split-plot ANOVAs ($p < 0{\cdot}05$). For illustration purposes, the bars are staggered according to plant groups. (b) Means (+1 SE) of plant group cover. Asterisks indicate significant differences of the total cover of the plant groups from the control based on Wilcoxon matched-pairs tests. For illustration purposes, the bars are staggered according to species. The most abundant species are given by name, the remaining species subsumed under "others". (a and b) The number of replicates was 10 for the treatment effect and 30 for the effect of livestock exclusion. co = control, br = brushpacks, ma = manure, fe = fertilizer, mi = microcatchments, pl = transplanting, nf = not fenced, f = fenced. (* $p < 0{\cdot}05$; ** $p < 0{\cdot}01$; *** $p < 0{\cdot}001$).

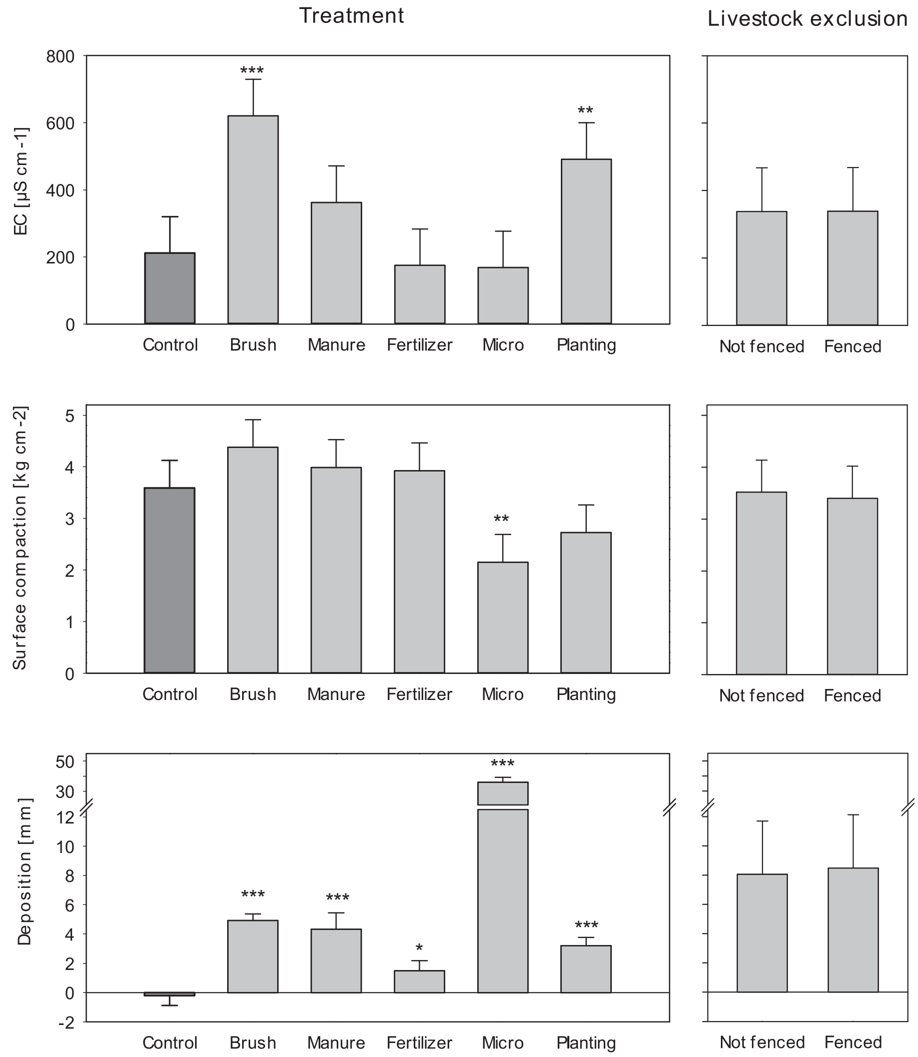

Figure 4. Means (+1 SE) of soil electrical conductivity (EC), soil surface compaction, and fine soil deposition in 2009, that is, 2 years after treatment application. The control is shown in dark and the treatments in light gray. Asterisks indicate the significant differences from the control derived from Dunnett's tests following the split-plot ANOVAs (* $p < 0{\cdot}05$; ** $p < 0{\cdot}01$; *** $p < 0{\cdot}001$). The number of replicates was 10 for the treatment effect and 30 for the effect of livestock exclusion. Micro = microcatchment, Brush = brushpacks, Planting = transplanting.

on vegetation were restricted to annuals (compare review by Hanke & Schmiedel, 2010). Because the increase in annual cover was confined to good rainfall years and to the peak of the vegetation period, we agree with Simons and Allsopp (2007) that an effect on annuals only can be counted as a partial success in Succulent Karoo rehabilitation. Our results thus again confirm that beyond a certain threshold, degraded states of Succulent Karoo vegetation cannot be shifted back by the amelioration of the abiotic conditions – at least, not within a time frame relevant for farming practice (compare Hoffman & Rohde, 2007). The stability of the degraded state is ascribed to the depletion of the soil seed bank (De Villiers *et al.*, 2003; Burke, 2007; Dreber & Esler, 2011).

Synergisms and Tradeoffs Among the Rehabilitation Goals

The increase in vegetation cover in the manure and the transplanting plots as well as the simulated vegetation cover in the brushpacking plots were associated with positive effects on the soil conditions. First, in these plots, fine soil deposition occurred, whereas the control plots showed the tendency of topsoil erosion. The prevention of erosion by vegetative cover is a widely known phenomenon in drylands (Cerdà, 1998; Zheng, 2006) and can be ascribed to the protection of the soil surface from heavy rains and wind (Mucina *et al.*, 2006). In the manure plots, soil deposition of about 4 mm occurred in the sampling year, which is comparable to the deposition underneath the *B. pseudoschlichtianus* transplants and the brushpacks. This shows that the cover of manure mulch and the herbaceous *Atriplex* combats erosion similarly successfully as a perennial shrub cover. Second, a previous study conducted in the framework of this experiment showed that depletion of soil water was retarded in covered soils. After an average winter rainfall event, soils under the manure mulch had reached the wilting point 4 days, underneath the transplants 5 days, and underneath the brushpacks 10 days later compared with the control (Hanke *et al.*, 2011).

The facilitation of soil recovery through the increase of vegetation cover shows that these two goals are synergistic. However, with regard to quality of the plant cover, there are different, partly conflicting, objectives. The reestablishment of a predisturbance state of the vegetation, for instance, might interfere with the restoration goals' practicability and fast forage production. The transplanting resulted in the maximum outcome rehabilitation is likely to achieve from the perspective of nature conservation within the given time – a plant cover of native, long-lived shrubs (i.e., *B. pseudoschlichtianus*) similar to nondegraded rangelands of the area. However, the palatability of *B. pseudoschlichtianus* is only moderate (Hendricks *et al.*, 2002), and thus, from the farmers' viewpoint, the treatment insufficiently achieved the goal of forage production. Another disadvantage of the transplanting treatment may arise from its financial implications. In a review on mine soil restoration, Carrick & Krüger (2007) pointed out that transplanting and the possibly required watering during the first weeks are quite time, labor, and cost-intensive. Furthermore, the acquisition of the transplants from surrounding vegetation could cause disturbance, and pregrowing them in nurseries would involve further expenses.

The manure treatment made, by contrast, use of a resource, which would otherwise be wasted (Kizza & Areola, 2010). Moreover, the manure mulching was the best treatment with regard to forage production. However, by shifting the natural species' composition toward a "pasture of herbaceous forage plants", the dominance of the native vegetation was compromised. A further conflicting outcome of the manure treatment was expected from a possible soil salinization by the dung, as observed in an experiment in the western Succulent Karoo (Meyer, 2009) or after repeated applications in a semiarid rangeland in Southern Alberta, Canada (Hao & Chang, 2003). However, salt content in our study was not significantly increased in the manure plots. According to a rangeland study in New Mexico by Cabrera *et al.* (2009), heavy applications of manure increased the soil salinity, whereas light applications combated erosion and improved soil moisture. It is likely that our moderate application rate and also the avoidance of fresh dung, which has a high content of soluble salts, limited the increase in soil salinity. Nevertheless, further studies on long-term effects of manure applications and the impact of repeated applications or variations in doses are needed.

The tested treatments attained different rehabilitation goals. Hence, combining the techniques could increase the chance of achieving a broader set of goals. For example, synergisms could arise from microcatchments lowering soil compaction, filled with manure increasing the vegetation cover. In the Sahel, such pits (called **zaïs**) are traditionally used as planting pits to increase the yield in cropped fields (Critchley *et al.*, 1994). Another promising approach could be the installation of a mosaic of manure patches and succulent shrub transplants to provide success concerning both goals: reestablishing of native, perennial vegetation and forage production. Offering *Atriplex* as forage, which is preferred by livestock over the less desirable succulent shrubs, may protect the transplants from browsing damage. Preferential grazing (compare Hussain & Durrani, 2009) might increase the survival probability of transplants under high grazing pressure and make transplanting projects more sustainable.

Plant-Induced Feedbacks

Our results do not support the hypothesis that the increase in halophytic species in response to manure mulching will cause topsoil salinization. Although we found significantly raised salinity in the topsoil underneath the canopies of *M. hypertrophicum* tested in the vicinity of the experimental sites, we do not expect a salinization problem in the manure plots from the increase in *M. hypertrophicum* cover. First, because the increase was not very strong, and second, it was a short-term effect, which abated in the second year. However the phenomenon of increased topsoil salinity under the canopies of *M. hypertrophicum* should be further investigated in another context. It could be an explanation for the large monodominant stands of *M. hypertrophicum* in degraded plains throughout the Richtersveld reported in Mucina *et al*. (2006). Our sampling design does not allow a definite discrimination of cause and effect (i.e., has *M. hypertrophicum* salinized the topsoil or has it colonized salty patches?), but the high concentrations of NaCl found in the leaves of *M. hypertrophicum* (Von Willert, 1980) suggest that it is responsible for increased salt levels in the topsoil. This deposition of salts on the soil surface might inhibit less salt-tolerant species (see studies on other *Mesembryanthemum* species by Vivrette & Muller, 1977; El-Ghareeb, 1991; Wentzel *et al*., 1994).

Under the canopies of the *A. semibaccata*, we found no raised topsoil salinity, suggesting that no topsoil salinization has to be expected from the increased abundance in the manure plots. Although long-term accumulation cannot be excluded, the presumption that non-natives induce topsoil salinization and inhibit indigenous plants (Milton *et al*., 1999; Botha *et al*., 2008) so far could not be corroborated for *A. semibaccata. Atriplex* species have been popular in dryland restoration worldwide because they are good forage plants, tolerant to water shortages and soil salinity, and they are able to colonize bare soils (Le Houerou, 1992). In Southern Africa, *Atriplex* species have been tested for the revegetation of soils disturbed by mining (De Villiers *et al*., 1992), but there are doubts whether the colonization by *Atriplex* facilitates the reestablishment of indigenous Karoo vegetation (Milton *et al*., 1999). Also in our study, recovery of long-lived shrub cover was not initiated in the manure plots. However, neither did the dominance of *A. semibaccata* negatively affect species richness nor the cover of the other plant groups. Future monitoring has to show how persistent the dominance of *A. semibaccata* is and how it will impact natural vegetation and soils in the long-term.

Implications for Practice

The rehabilitation goals have been best attained by the transplanting and the manure treatment. In different ways, both were able to increase drought-persistent vegetation cover and to combat erosion. Whereas transplanting reestablished a predisturbance vegetation of native shrubs, the manure application created a pasture-like vegetation of non-native forage plants.

The two treatments represent two approaches set against each other in the context of the paradigm shift restoration ecology has undergone in the last decades: restoration of original communities versus rehabilitation of ecosystem functions (Temperton, 2007). Because restoration success has often lagged behind expectations, acceptance has grown that reassembling of historical plant communities may have become too difficult in systems in which global change and land use have irreversibly impacted the environment (Hobbs, 2007). Therefore, goal setting has become less idealistic and is more strongly orientated to the needs of land users and socioeconomic limitations (Choi, 2007; King & Hobbs, 2007).

Given the challenges of the slow dynamics of arid ecosystems and the high costs involved in restoring transformed land in the Succulent Karoo and that the high grazing pressure in the open-access parts of communal lands, tradeoffs have to be accepted in the achievement of rehabilitation goals. Whereas transplanting of key species can be recommended as a revegetation technique at sites that are less accessible to livestock (e.g., protected areas), manure mulching might be a practicable and effective measure to increase forage production and to combat the acute symptoms of desertification at the degradation hotspots of communal lands. For these areas, harvest and redistribution of old paddock manure should be further explored as a management practice, even if – or precisely because – it is associated with the introduction of seeds of *Atriplex* spp. or other fast growing forage plants.

CONCLUSIONS

The results of this study confirm that the achievement of all rehabilitation goals (i.e., to reestablish a perennial, native vegetation, to enhance soil protection, forage production, practicability) by only one treatment is unlikely. Microcatchments and brushpacks have a good potential to combat erosion but failed to increase the vegetation cover. The release from grazing and the application of mineral fertilizer promotes the abundance of palatable annuals, but the effect is only a short-term benefit that depends on rainfall. The reestablishment of a drought-persistent vegetation cover required biotic manipulations. Transplanting of the desired shrub species is the safest option, but the high effort is worthwhile only if the survival of the transplants will be ensured afterwards by a sustainable grazing regime. For the rehabilitation of severely degraded areas, where a decrease in grazing pressure is unrealistic under the prevailing conditions, tradeoffs should be accepted. The redistribution of the hitherto unused paddock manure is a practicable alternative. The resulting pasture-like vegetation of non-native forage plants, of which the manure carried seeds, compromised the “nativeness” of the vegetation but increased the forage resource for livestock. Managing the multiple and sometimes conflicting restoration goals could be facilitated by applying spatial mosaics of different rehabilitation techniques.

ACKNOWLEDGEMENTS

This study was conducted within the framework of the BIOTA Southern Africa project sponsored by the German Federal Ministry of Education and Research under the promotion number 01 LC 0624A2. Northern Cape Nature Conservation, South Africa, kindly issued a research permit. We thank the management of the Richtersveld Community Conservancy and members of the Eksteenfontein community for support during the experiment. We thank Randall Joseph, Janne Weber, Sarah Meyer, Anne Schulz, and Matthias Rös for field assistance and Jürgen Dengler for his comments on the manuscript. We gratefully acknowledge the valuable comments and suggestions of one anonymous reviewer and from Matthew Bowker and Artemi Cerdà. The development of this paper has benefited substantially from their comments.

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