Plant Ecol (2017) 218:7–22
DOI 10.1007/s11258-016-0644-3

Assessing the efficiency and effectiveness of rangeland restoration in Namaqualand, South Africa

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Received: 15 March 2016 / Accepted: 9 August 2016 / Published online: 5 January 2017

Abstract South Africa's Succulent Karoo is home to unmatched numbers of dryland plant species. Unfortunately, decades of overstocking these rangelands with small livestock and historical ploughing for fodder have led to extensive degradation. Some areas are severely degraded, negatively affecting both agricultural livestock productivity and ecosystem health. Land degradation reduces land use options and leaves land users, and the ecosystems on which they depend, more vulnerable to environmental and economic stressors. Ecological restoration is promoted as an effective and cost-efficient option for building the resilience of local and regional ecosystems. However, dryland restoration confronts many environmental challenges that have limited its success to date. Here, we present the results of a local-scale participatory restoration trial and an assessment of the costs of regional-scale ecological restoration in the Nama Khoi area in Namaqualand, South Africa. In combination, these analyses are useful for identifying opportunities and barriers for the improved efficiency and effectiveness of dryland restoration. In Namaqualand, we find that ecological restoration is difficult and expensive. The expected impacts of climate change will only exacerbate these challenges. However, we argue that a holistic suite of land management actions that include sound management, the prevention of further degradation, and prudent investments in restoration even where costs are high is likely to be the only real option for sustaining land-based livelihoods in this region over the longer term.

Keywords Ecological restoration · Dryland ecosystems · Economic analysis · Degradation · Rangelands

Communicated by Dr. Olga Kildisheva, Dr. Lauren Svejcar and Dr. Erik Hamerlynck.

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Introduction

The Succulent Karoo in western South Africa is a globally unique Biodiversity Hotspot (Myers et al. 2000; Sloan et al. 2014) and one of only two Hotspots in the drylands. The Succulent Karoo is characterised by high diversity, shrubby vegetation, and dwarf succulents (Cowling et al. 1997). Home to exceptional plant biodiversity and extraordinary levels of endemism (Desmet 2007), it is highly threatened by the pressures of global change (Myers et al. 2000).

The area has been populated by hunter-gatherers for at least 100,000 years and by nomadic sheep farmers for between 1000 and 2000 years prior to colonisation by Europeans in the eighteenth and nineteenth centuries (Smith 1994). It has one of the longest records of grazing by domesticated livestock in Southern Africa (Webley 1986). As a result of colonisation, wild herbivores were largely exterminated and replaced with sheep and goats on a large scale; land was fenced; and, in the twentieth century, an exponential growth in the permanent farming population occurred (Milton et al. 1994). Commercial and smallholder livestock farming remains the dominant land use in the region, and 90 % of the land is grazed (Desmet and Marsh 2008). Communal farming is a land use strategy employed on 16 % of agricultural land in South Africa. On communal farms, rural populations practice small-scale farming over large collectively managed areas. User rights and obligations are typically not clearly defined, leading to the over exploitation of the resource base (De Lange 1994). Local natural systems are, therefore, as much a product of human activities as they are that of evolutionary and climatic histories, and local residents are closely reliant on these natural systems (Hoffman and Rohde 2007).

While there is some debate as to whether domestic livestock have resulted in landscape level changes in Succulent Karoo vegetation (Vetter 2009), there is agreement that, at the paddock scale, injudicious grazing management does alter vegetation composition (Milton et al. 1994; O'Connor and Roux 1995) and soil processes (Roux and Opperman 1986). Hence, observed losses in productivity and diversity have been attributed to the overuse of rangelands by a narrow suite of domesticated herbivores (Milton et al. 1994).

Grazing-related degradation in the Succulent Karoo results in a simplification of the shrub layer and changed patterns of primary productivity (Thompson et al. 2009). Severely degraded areas show evidence of serious erosion and trampling, have a low cover and diversity of perennial plants, are dominated by unpalatable shrubs, and contain an abundance of toxic plants (Thompson et al. 2009). Intact areas have a high cover and diversity of perennial plants, little evidence of trampling or erosion, and an abundance of palatable plants (Thompson et al. 2009). Moderately degraded areas are intermediate between intact and severely degraded. In this paper, the authors have used the term 'restoration success' to refer to an effective transition from a severely degraded state to a state more characteristic of intact vegetation.

Ecological restoration to reverse or reduce degradation is promoted as an effective and cost-efficient means for building resilience in local and regional ecosystems (Roberts et al. 2012). Restoration efforts focused on maintaining agricultural productivity; promoting water infiltration through mulching, brush packing, and micro-catchments; stabilising soils through erosion control; and re-introducing lost biodiversity through direct seeding have been ongoing in the broader Karoo for two decades (Milton 2001; Blood 2006; Carrick and Kruger 2007; Carrick et al. 2015). However, restoration success has proved difficult to achieve. In dryland areas, low levels of success are attributed to the unique floral diversity, low and variable rainfall, and complex spatial and biological dynamics (Carrick and Kruger 2007) and driven by stochastic events (Ellis 1994). The autogenic recovery of indigenous perennial vegetation is very slow once disturbed and unlikely to occur within human lifetimes, if at all (Van der Merwe and Van Rooyen 2011). Milton and Dean (2015) provide an example from Sampson (1986) of pre-colonial corrals which remain denuded of vegetation more than 300 years after their abandonment.

Given the potentially significant impacts of degradation on livestock production systems, biodiversity, and livelihoods in the Succulent Karoo, the need for effective restoration and improved management is clear. We identified current best practice for rangeland restoration and re-vegetation in South African winter rainfall drylands and implemented a small participatory restoration trial with communal farmers in Steinkopf in the Nama Khoi local municipality, Namaqualand, South Africa. In order to improve our understanding of the likely return on investment from restoration, we additionally undertook an economic analysis of restoration costs and benefits at the scale of the Nama Khoi local municipality. We present the results of both of these analyses here.

Study site

This study took place in the Nama Khoi local municipality, located in the Namakwa District, Northern Cape, South Africa (Fig. 1). The restoration trial

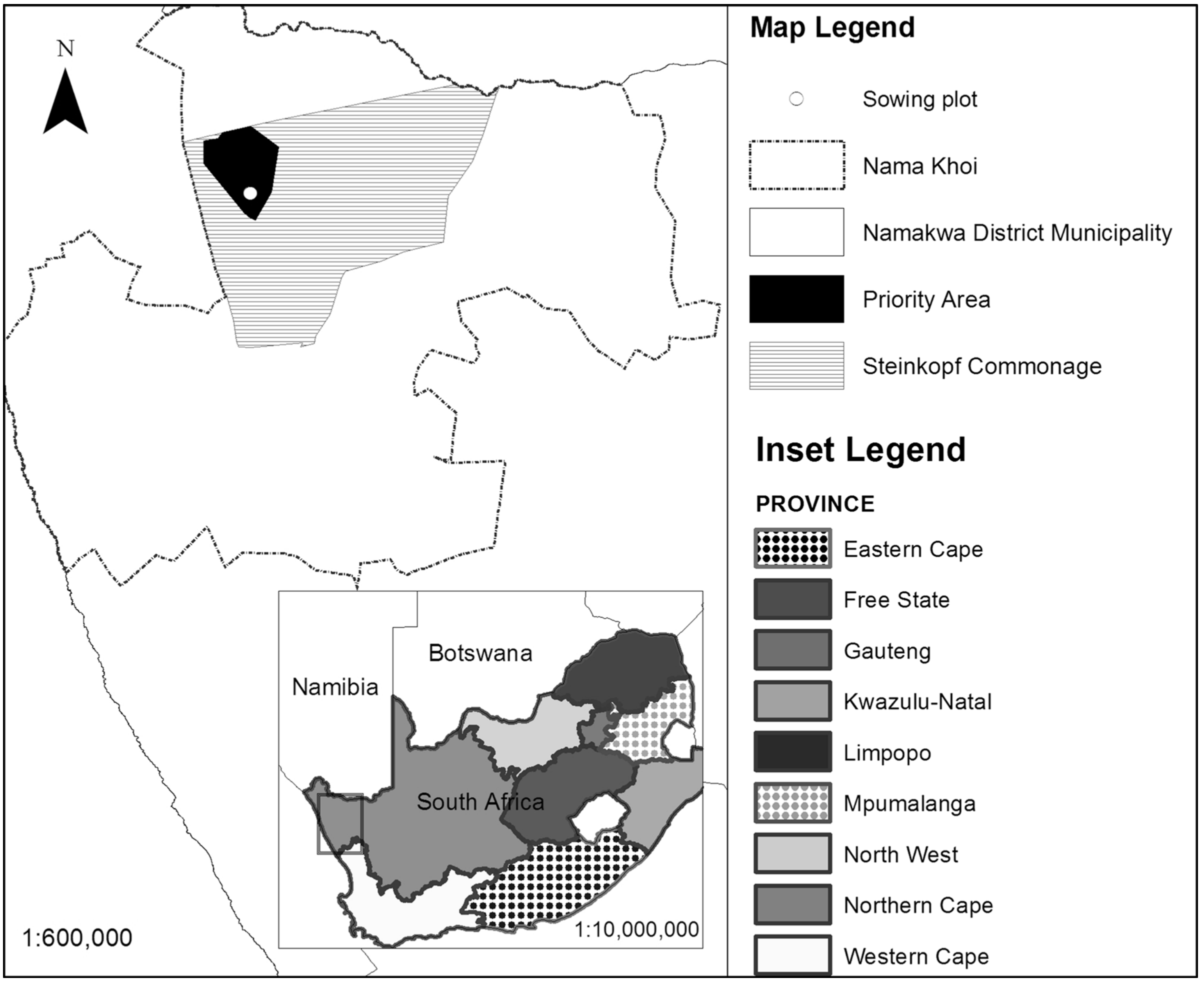

Fig. 1 Geographic location of the study site in Steinkopf, Namakwa District. The study site is shown as a *white circle* on the map. It is located within a 19,650 ha critical biodiversity area, shown in *black*, in the north west of the Steinkopf communal farm, shown by the *grey horizontal lines*

was conducted in a 19,650 hectare (ha) critical biodiversity area in the northwest of Steinkopf (Fig. 1), a 302,125 ha communal farm between 28°59′26.1852″S and 29°12′51.9834″S, and 17°41′27.744″E and 18°13′10.9698″E. In the interests of understanding the feasibility of restoration at scale, the economic analysis of rangeland restoration was undertaken at the larger Nama Khoi local municipality scale. The terrain is mountainous (max 1340 m above sea level) with steep to moderately steep mountain slopes and a relatively flat, undulating plateau (Mucina and Rutherford 2006). Rainfall is low, typically 100–250 mm per annum, 60 % of which falls in winter, between May and September (Desmet 2007). The primary land use is sheep and goat farming, and fodder crops are periodically cultivated.

Methods

Restoration trial

The experiment aimed to test the potential for structurally diverse re-vegetation in disturbed and degraded sites. Restoration treatments were (1) direct seeding, (2) mulching with plant material, (3) mulching with animal manure, (4) micro-catchments, and (5) brush packing with *Galenia africana*. To test the effects, 15 treatment combinations consisting of single treatments or combinations of multiple treatments were replicated eight times each using a randomised complete block design. Eight controls, which consisted of no site treatment, were also included. Farmers voluntarily allocated the land and provided significant input into the design of the treatments, all of which were intended to be low cost options using locally available materials and labour.

Seeds of 16 locally abundant perennial plant species were collected within 5 km of the location of the trial and used for the seeded treatments (Table 1). Species used included highly palatable and less palatable plants and were chosen to represent five different growth forms: small spreading mesembs, upright mesembs, perennial grasses, small shrubs, and legumes. Just before sowing, seeds were cleaned, weighed, and separated into standardised amounts per species for sowing in the seeded treatments. Identical seed mixtures and amounts were sown in each seeded treatment at a rate of approximately 5 kg per ha. For the duration of the trial, grazing was excluded from the site via fencing.

The exclosure plot contained 128 treatment plots (2 m × 2 m each). Within each plot, a discrete 0.5 m × 0.5 m quadrat was allocated for application of the different treatments (128 treatment application quadrats) (Fig. 2).

Each 0.5 m × 0.5 m treatment application quadrat was irrigated with 2 litres of water on the day of seeding, 23 May 2014. Treatment application quadrats were revisited twice for data collection, in Winter 2014 (August) and Summer 2015 (February). Perennial and annual plants within each quadrat were recorded in Winter 2014, but only the targeted perennial species were recorded in Summer 2015. Annuals die back in the summer months and were therefore not present in the plots in Summer 2015. The number of established individual plants per species per quadrat, as well as their corresponding height, width, cumulative stem length, and number of leaf pairs, was recorded. All plant species that had not been seeded as part of the study were removed by hand after they had been identified and measured.

The proportion of annual to perennial plant species in each treatment area was calculated for Winter 2014. A factorial analysis of variance (ANOVA) was conducted to compare the main effects of different treatment combinations on the observed proportion of targeted perennials. For both Winter 2014 and Summer 2015, species richness and species diversity, using the Shannon-Wiener Diversity Index (SWDI—Kent and Coker 1992), were calculated for each seeded treatment. Species density (species per m^2) and frequency, calculated as the percentage of total treatment application quadrats containing at least one individual of a species, was determined within each treatment combination. A one-way ANOVA and post hoc comparison *t* test was used to calculate the effect of species characteristics on density and treatment effect on species richness, SWDI, and plant establishment. Statistical analyses were carried out in the R statistical environment version x64 3.2.4 using the core packages (R Development Core Team 2015).

Some results are only reported at the species level, because multiple species within a genus were collected (specifically *Hermannia disermifilia,*

Table 1 Table listing the locally abundant plant species used for the restoration trial. For each species, the respective plant family and growth form are shown

Species	Family	Growth form
Cheiridopsis denticulata	Mesembryanthemaceae	Small spreading mesemb
Drosanthemum hispidum	Mesembryanthemaceae	Small spreading mesemb
Conicosia elongata	Aizoaceae	Upright mesemb
Fingerhuthia africana	Poaceae	Perennial grass
Ehrharta calycina	Poaceae	Perennial grass
Hermannia spp.	Malvaceae	Small shrub
Hirpicium alienatum	Asteraceae	Small shrub
Manochlamys albicans	Chenopodiaceae	Small shrub
Pentzia incana	Asteraceae	Small shrub
Pteronia spp.	Asteraceae	Small shrub
Tetragonia fruticosa	Aizoaceae	Small shrub
Tripteris spp.	Asteraceae	Small shrub
Eriocephalus africanus	Asteraceae	Small shrub
Lessertia diffusa	Fabaceae	Legume

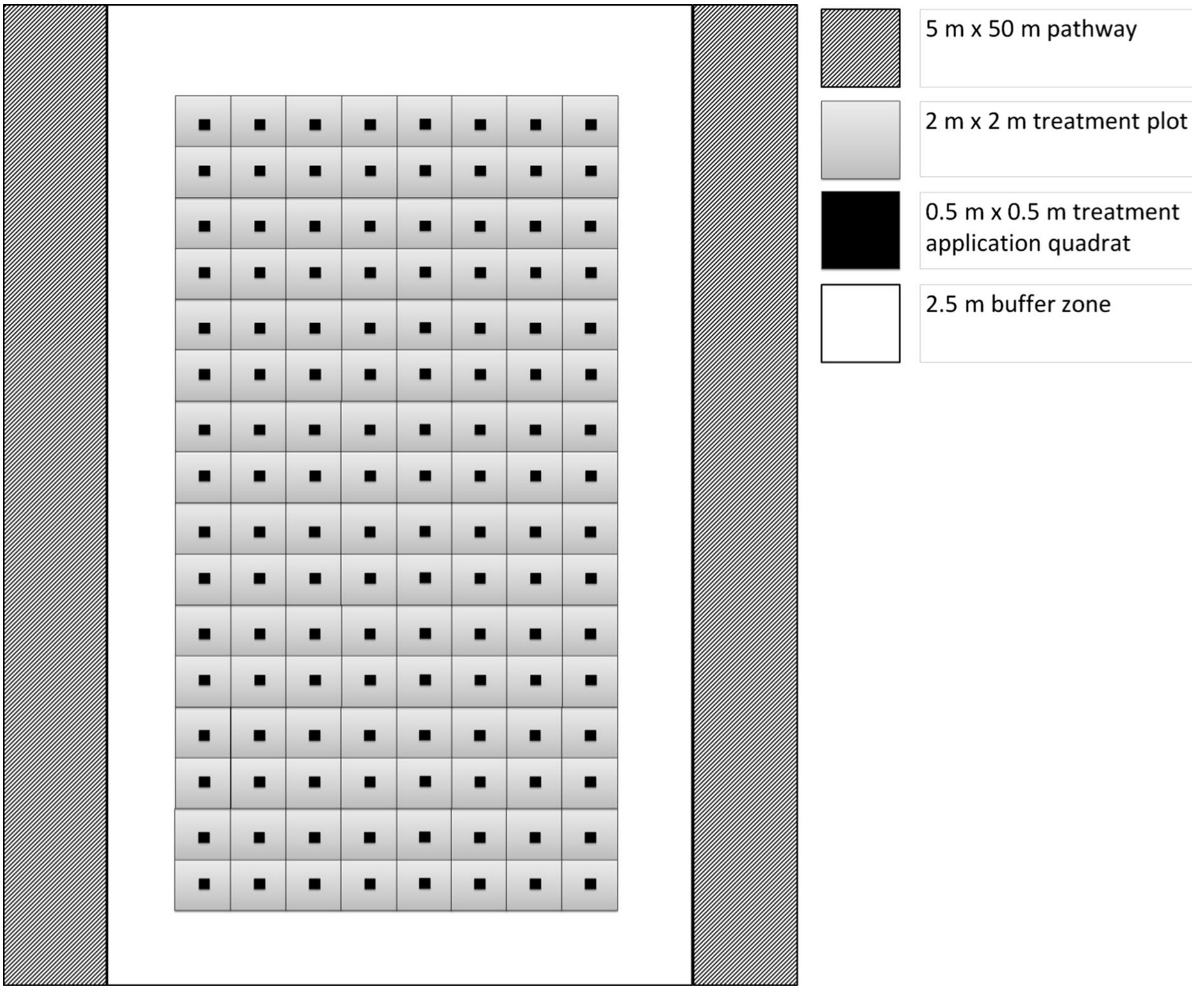

Fig. 2 Schematic showing the layout of the restoration trial comprising 128 treatment blocks in 16 rows of eight, each containing a single 0.5 m × 0.5 m treatment quadrat. A randomised complete block design was used to replicate 15 different treatment combinations eight times each. Eight controls were also implemented

Hermannia trifurca, Pteronia divaricata, Pteronia incana, Tripteris oppositifolia, and *Tripteris sinuata*) but it was not consistently possible to distinguish between species within each genus at the emergent seedling stage.

Economic analysis

We used a scenario-based approach to compare the costs of two alternative approaches to maintaining livestock production on rapidly degrading rangelands: a reactive intervention whereby farmers purchase increasing amounts of supplementary fodder in order to maintain livestock numbers as rangeland productivity declines versus a proactive intervention where degraded areas are actively restored in order to sustain natural rangeland productivity. Active restoration in this analysis involves soil preparation, direct seeding, and mulching/brush packing, similar techniques to those used in the restoration trial.

Determining current levels of degradation in the study area

We estimated, based on literature, that roughly 25 % of the rangelands in Nama Khoi are severely degraded, requiring active restoration intervention (Hoffman and Ashwell 2001; Thompson et al. 2009). Assuming that the remaining 75 % of the rangeland is relatively intact, or at least in a condition where biotic or abiotic thresholds have not been crossed and natural processes of succession are still likely to be able to establish a functioning, stable, and biodiverse ecosystem through secondary succession and judicious land management alone (Hulvey et al. 2013; Carrick et al. 2015), we focused our analysis on the costs of actively restoring 25 % of Nama Khoi land area, or 449,700 ha (NDM 2012).

To estimate the economic value of annual livestock production for the return on investment calculations, we first identified grazing capacity classes for the degraded 25 % land (ranging from 40 to 120 ha per head of cattle) in the municipality[1] and calculated both the number of small livestock that could be supported by each grazing capacity class using the national recommended stocking rates[2] and the kilograms of dry matter produced by each grazing capacity class (Klug et al. 1999; Herling et al. 2009). Meat production potential was set at 1 kg meat per 10 kg dry matter (Cupido and Samuels, Agricultural Research Council, pers comms 2013) and summed for the study area. The monetary value of meat production per annum was then calculated by multiplying meat produced by the US$ per kg mutton price in South Africa.[3] This monetary value, based on maximum production in pristine rangeland, was then adjusted downwards by 50 % to account for reduced productivity in degraded areas based on estimates of degradation impacts on rangeland productivity in southwestern South Africa (Dean and McDonald 1994; Milton and Dean 1995).

Two degradation response scenarios

The starting point for both scenarios is that 25 % of the total area requires restoration intervention. Costs and benefits were estimated over a 50-year period.

Reactive scenario In this scenario, no action is taken to address degradation. Current land uses, including overstocking, continue and no management or restoration action is undertaken. Consequently, we assume that degradation worsens over the 50-year time period from severe (50 % of intact grazing capacity) to very severe (25 % of intact grazing capacity), and apply a simple linear model of gradually declining livestock production by 5 % per decade over 50 years. The number of livestock that can be maintained on land in the respective degradation categories was calculated following Dean and McDonald (1994) and Milton and Dean (1995).

Livestock numbers are maintained on degraded areas over time by purchasing supplementary fodder. The costs assessed included purchase and distribution of fodder according to the annual rangeland productivity deficit. The purchasing cost was calculated following Le Maitre et al. (2009) and Van der Merwe and Smith (1991), with fodder provision defined as 0.25 kg maize and 1.75 kg lucerne (*Medicago sativa*) per adult sheep or goat per day at Kaap Agri 2013 rates of US$260 per metric ton and US$300 per metric ton, respectively. Trucking costs were based on the cost to farmer to distribute the fodder locally, as transport to the area is already included in the price. Calculations for this transportation cost considered *inter alia* the number of collections per week, travel distance, and running costs of vehicles. These values were based on Rotary FY2014-2015 standard global per km rates for personal vehicles.[4] For direct comparison with the restoration scenario (see below), we have taken supplementary feeding only to 70 % of potential productivity on intact rangeland.

Supplementary feeding to maintain livestock production in extremely degraded areas is a strategy that is currently used in some locations. Sterkspruit, one of South Africa's most severely degraded areas (Hoffman and Ashwell 2001), maintains twentieth century livestock numbers by increasingly providing feed inputs (Vetter 2003). Vetter (2009) argues that other 'communal rangeland systems have shown similar persistence despite predictions of imminent collapse'. However, it is unclear how long this strategy will remain viable either in the present as markets fluctuate, or particularly under changing climate conditions in future (Tadross et al. 2011).

[1] Data provided by the Department of Agriculture, Forestry, and Fisheries.

[2] Data provided by the Department of Agriculture, Forestry, and Fisheries.

[3] For the current study, this was taken as US$2.57 per kg mutton, as reported by the Red Meat Producers Organisation ABSA Weekly prices for 22 May 2015. http://www.rpo.co.za/InformationCentre/ABSA/WeeklyPrices.aspx. Accessed 29 May 2015.

[4] Accessed online at https://www.rotary.org/en/document/7321 in May 2015.

Proactive scenario All degraded rangeland is restored to full production potential in the first decade of the 50-year analysis period. The practicalities and costs of rangeland rehabilitation projects were sourced from a review of the literature (Milton-Dean 2010; Merrit and Dixon 2011; Coetzee and Stroebel 2011; Blignaut et al. 2013), the authors' restoration experiences, and consultation with experts implementing restoration in the Succulent Karoo (Table 2). Active restoration activities in these areas include soil stabilising and re-seeding degraded areas to re-establish forage species and increase plant species diversity. Costs are assumed to increase with inflation by 7 % per year.

Following Herling et al. (2009) and Milton-Dean (2010), the opportunity cost of excluding livestock from restored areas during the initial 5-year period of seedling establishment was included. Livestock are then gradually re-introduced, starting at 50 % grazing capacity, increasing by 5 % a year, and capped at 70 % of the rangeland's grazing capacity in the tenth year and beyond to promote long-term sustainability (Vorster 1982; Hobson 1984; Le Roux et al. 1984; Du Toit 1996; Beukes et al. 2002). Rangeland productivity is assumed restored by year ten. Livestock is re-introduced to the model after 5 years due to social and economic pressures to maintain production, even though plant establishment in real terms is likely to take longer than this (Foster et al. 2012).

Least-cost and cost-benefit analyses

The economic efficiency of both scenarios was determined and compared by conducting a least-cost analysis and calculating net present values (NPV) and benefit-cost ratios (BCR) for a cost-effectiveness analysis. NPV was interpreted as favourable when positive and unfavourable when negative. BCR was regarded as the return on investment for every dollar spent, with >1 considered as cost-effective and <1 not cost-effective.

The choice of an appropriate discount rate is central to any economic analysis because this can have a large impact on the final outcome. However, there is little consensus amongst researchers on the appropriate rate, particularly regarding ecological restoration. We performed a sensitivity analysis using three different discount rates over the 50-year time frame to explore the implications for NPV and BCR. These are an 8 % discount rate, as recommended by Mullins et al. (2007) for the South African context, a 3 % discount rate, as suggested by TEEB (2010) for ecosystem services and other public goods, and a 1.3 % discount rate as advocated by Stern (2006) for giving greater

Table 2 Table summarising the costs for restoring 25 % of Nama Khoi land area (449,700 ha) over 10 years at a rate of 44,970 ha per year

Restoration Activity	Description	Total cost (US$)
Collect seed	Labour (US$10 per day, 1 seed picker collects 4 kg per day)[a,b]	6,213,256.65
	Buy seeds—alternative shown for comparative purposes only and not included in totals (US$107 for 4 kg)[c]	66,276,808.67
Digging micro-catchments and sowing seeds	Labour (US$6.5 per day, 12 labour days per ha)[a,d]	49,706,053.19
Clearing invasive plants	Labour (US$6.5 per day, 12 labour days per ha)[b,d]	49,706,053.19
Mulching, brush packing, and fertliser	Labour (US$6.5 per day, 12 labour days per ha)[c,d,e,f]	49,706,053.19
	Fertiliser (US$21 per ha in year 1 increasing with inflation by 7 % per year)[b]	13,109,971.53
Operations	Equipment and transport costs[g]	10,376,138.60
Total restoration cost		178,817,526.33

Cost is estimated from actual restoration costs in South Africa's semi-arid shrublands. An estimated 1,798,800 kg of seed are required for restoring 449,700 ha and calculated as approximately 4 kg per ha (Merrit and Dixon 2011)

[a] Milton-Dean 2010, [b] personal communications with Becker and Carrick (Nurture Restore Innovate), Cupido and Samuels (Agricultural Research Council), and Dreyer (South African National Parks) 2013, [c] Mugido 2011, [d] based on standard average public works daily wage 2015, [e] Pauw 2011, [f] De Abreu 2011, [g] based on current restoration project costs in Nama Khoi

weight to the ethical considerations of future generations.

Results

Restoration trial

Across all treatment combinations, 99.4 % of individual plants counted emerged in the seeded treatments in Winter 2014. Site treatment and seeding had a significant effect ($P < 0.001$ and $P < 0.0001$, respectively) on the emergence and establishment of target plant species, while the interaction between the two was not significant ($P = 0.34$).

In the absence of site treatment and direct seeding, the majority of plant species that emerged after rain were ephemerals. For example, 98 % of individual plants counted in control plots in Winter 2014 were annuals (Fig. 3). Unseeded treatments failed to establish target perennial species despite the exclusion of grazing, and were excluded from further analysis.

Species frequency in the seeded treatments in Winter 2014 was dominated by *Cheiridopsis denticulata,* a locally abundant small succulent shrub, and *Ehrharta calycina*, a highly palatable grass, both of which appeared in 92 % of all seeded treatment quadrats. These species were also recorded in significantly higher densities than any of the other target species during the Winter 2014 monitoring (Table 3). *C. denticulata* had an average of 463.44 individuals per m^2 ($P < 0.0001$) and *E. calycina* had an average of

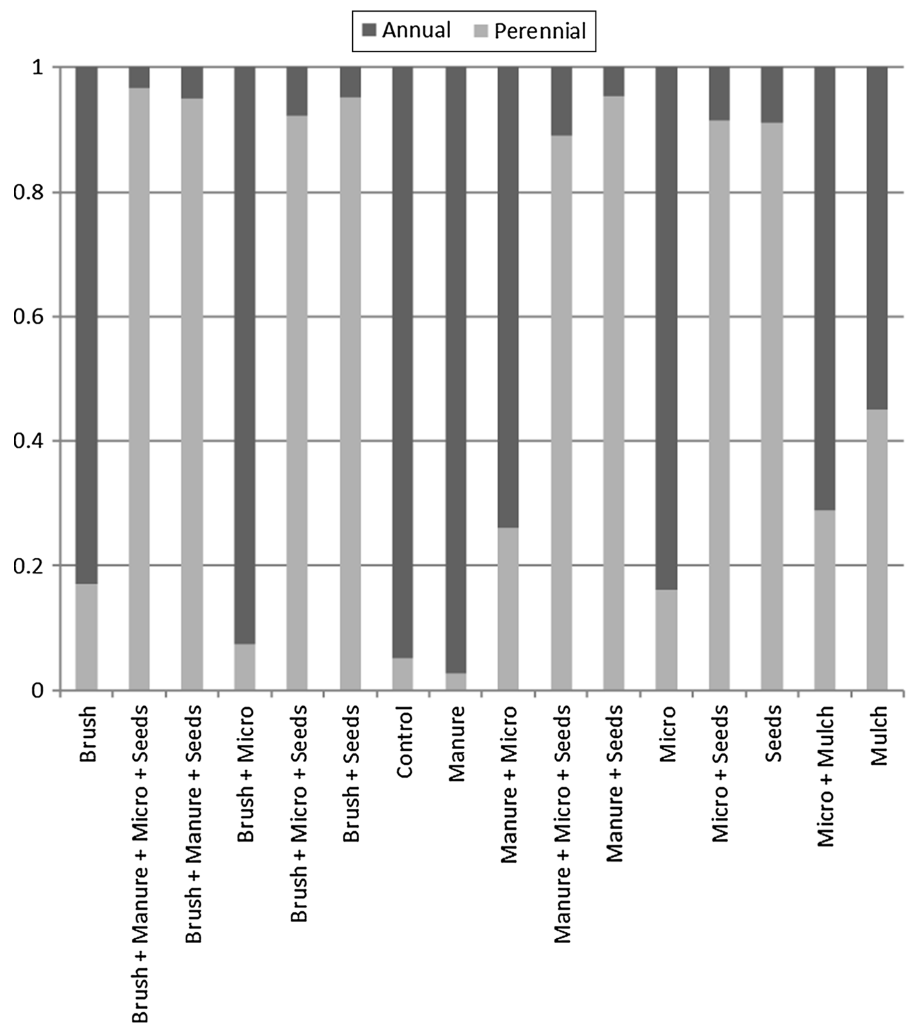

Fig. 3 Proportion of annual to perennial species for Winter 2014 in all treatments. Annual plant species are shown in *dark grey* and perennial plant species are shown in *light grey*. Perennial species dominate in seeded treatments while annual species dominate in unseeded treatments

Table 3 Table presenting a summary of results for the average frequency (%) and density (m^2) of target species in the seeded treatment combinations for Winter 2014 and Summer 2015. Densities of *Cheiridopsis denticulata* are significantly higher than those of other target species. Survival through the dry season was low for all target species

Species	Growth form	Winter 2014		Summer 2015	
		Frequency	Density	Frequency	Density
Cheiridopsis denticulata	Small spreading mesemb	0.92	463.44***	0.59	23.69***
Drosanthemum hispidum	Small spreading mesemb	0.75	25.00	0.30	3.44
Conicosia elongata	Upright mesemb	0.83	15.63	–	–
Ehrharta calycina	Perennial grass	0.92	116.44**	–	–
Fingerhuthia africana	Perennial grass	0.13	39.06	–	–
Eriocephalus africanus	Small shrub	0.91	49.81	0.22	3.13
Hermannia spp.	Small shrub	0.77	4.88	–	–
Hirpicium alienatum	Small shrub	0.91	8.19	0.09	0.56
Manochlamys albicans	Small shrub	0.38	18.19	–	–
Pentzia spp.	Small shrub	0.66	2.63	0.05	0.25
Pteronia spp.	Small shrub	0.63	12.50	0.05	0.63
Tetragonia fruticosa	Small shrub	0.00	35.00	0.03	0.13
Tripteris spp.	Small shrub	0.89	0.00	–	–

* Indicates significant differences in species performance at the following significance levels: 0.0001 '***', 0.001 '**', and 0.05 '*'

116.44 individuals per m^2 ($P < 0.001$). Species such as *Hirpicium alienatum* appeared at high frequencies but low densities, while others, such as *Fingerhuthia africana* occurred at relatively low frequencies, but in high densities where they did occur.

In Summer 2015, species frequency was greatest amongst the small spreading mesembs, *C. denticulata* (59 %) and *Drosanthemum hispidum* (30 %). These two succulent species also exhibited the highest densities in the treatment quadrats after the dry season, at 23.69 per m^2 and 3.44 per m^2, respectively. While densities of *C. denticulata* are significantly different from those of the other remaining species recorded in Summer 2015 ($P < 0.0001$), densities of *D. hispidum* are not ($P = 0.37$). No grass species were recorded in treatment quadrats in Summer 2015.

Only 4 % of individual plants survived from the dry season to Summer 2015. Species richness and diversity, as well as plant establishment in terms of absolute numbers of individual plants recorded, were all lower in Summer 2015 than in Winter 2014 (Table 4).

Species richness was consistently high in Winter 2014, with most target species noted in all the treatment combinations. The SWDI values were low across all treatment combinations, highlighting the dominance of a small number of species (notably *C. denticulata*), and differed significantly across treatments ($P = 0.0353$). Treatment combinations which contained manure and brush in addition to seeds performed marginally better in terms of species diversity and evenness (SWDI) in both Winter 2014 and Summer 2015. Plant establishment was dominated by *C. denticulata*, and this did not differ significantly across treatments. Overall, the differences between treatment combinations once unseeded treatments were removed and *C. denticulata* dominance was accounted for were negligible.

Established plants were still small after two growing seasons, with only 9.7 % of individuals exceeding 20 mm in height. All small spreading mesembs were below 2.5 mm, and 85 % of upright mesembs below 2.5 mm. With 87 % of shrubs between 15 and 50 mm, these plants would still be highly vulnerable to grazing impacts and other disturbance. By Summer 2015, after the dry season, plant establishment amongst the more palatable of the seeded species—the grasses, soft shrubs, and legumes—was extremely poor, with negligible levels of survival across species and treatments.

Table 4 Table presenting a summary of results for the maximum species richness achieved per treatment (as number of different target species recorded), average Shannon Weiner Diversity index values per treatment, and total number of individual plants established per treatment, considering only the seeded treatment combinations for Winter 2014 and Summer 2015. The effects of treatments additional to seeding was negligible

	Species richness		Shannon weiner diversity index		Plant establishment	
	Winter 2014	Summer 2015	Winter 2014	Summer 2015	Winter 2014	Summer 2015
Brush + manure + micro + seeds	12	6	1.90	1.61	950	13
Brush + manure + seeds	12	4	1.81	1.14	1106	21
Brush + micro + seeds	12	6	1.34	0.62	2283	187
Brush + seeds	12	6	1.1*	1.05	2544	93
Manure + micro + seeds	12	2	1.89	0.61	750	10
Manure + seeds	11	3	1.59	0.79	1192	30
Micro + seeds	12	5	1.25*	0.74	1880	101
Seeds	11	3	1.35	0.71	1947	54

* Indicates significant differences across treatments at the following significance levels: 0.0001 '***', 0.001 '**', and 0.05 '*'

Economic analysis

On degraded land in Nama Khoi, we estimate approximate annual revenue of US$5.3 million from meat production. We project this will decline to around $2.6 million per annum by 2050. Through restoration and better management, further degradation could be avoided and production increased to approximately US$7.5 million after 10 years. Supplementary feeding in the reactive scenario achieves the same levels of production immediately.

Least-cost analysis

In Nama Khoi, the proactive scenario is more expensive than the reactive scenario, at all discount rates (Fig. 3). At the 1.3 % discount rate, the two options are similar—with restoration discounted to ca. $156 million and supplementary feeding discounted to ca. $141 million. At the 8 % discount rate, restoration costs appear to be more than double supplementary feeding, at ca. $108 million as opposed to ca. $41 million, respectively (Fig. 4).

Cost-effectiveness analysis

The reactive scenario showed positive NPV as well as favourable cost-benefit ratios (>1) at all discount rates (Table 4). The proactive scenario proved less cost-effective: NPV were positive and cost-benefit ratios >1 only at the two lower discount rates. At the standard South African discount rate of 8 %, restoration was not cost-effective. Importantly, the proactive scenario was more sensitive to the discount rate applied than the reactive scenario, while higher discount rates favoured the reactive scenario. Restoration showed the lowest BCR of the two scenarios (0.51 at a discount rate of 8 %). The highest BCR was supplementary feeding at the 8 % discount rate (2.38). Overall, supplementary feeding appears to be less expensive and more cost-efficient than restoration (Table 5).

Discussion

Dryland systems worldwide face significant and ongoing degradation. While existing initiatives promote the restoration of these systems, poor plant establishment in restored areas has been observed for a diversity of indigenous dryland species (Esler and Phillips 1994; Wiegand et al. 2000; Milton 2001; Hanke et al. 2011; De Malach et al. 2014).

In the Succulent Karoo, ecological restoration focused on re-vegetation and increased structural diversity is challenging with current knowledge and tools. Our field trial demonstrated that restoring a structurally diverse ecosystem will be difficult to

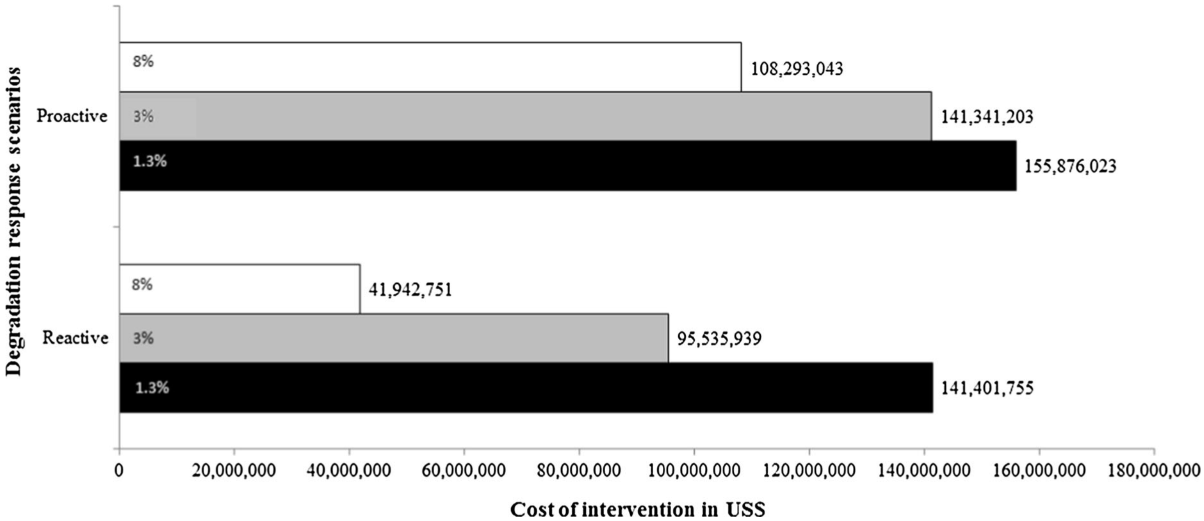

Fig. 4 Costs of proactive (restoration-based) and reactive (supplementary feeding-based) scenarios in Nama Khoi at different discount rates. The 8 % discount rate is shown in *white*, the 3 % discount rate is shown in *grey*, and the 1.3 % discount rate is shown in *black*. The reactive scenario is cheaper than the proactive scenario at all discount rates

Table 5 Table summarising the cost-effectiveness analysis comparing proactive and reactive scenarios for maintaining livestock stocking rates in Nama Khoi at the three discount rates, in terms of net present value and benefit-cost ratio. The proactive, restoration-based scenario has positive net present value and a benefit-cost ratio >1 at the 1.3 and 3 % discount rates, but not at the national standard 8 % discount rate. These values are favourable for the reactive scenario at all discount rates

Discount rate (%)	Nama Khoi (50-year, US$)		
	Scenario	NPV	BCR
1.30	Proactive	$60,459,452	1.39
	Reactive	$139,223,621	2.01
3	Proactive	$4,261,512	1.03
	Reactive	$105,852,029	2.11
8	Proactive	−$49,276,793	0.51
	Reactive	$57,761,256	2.38

achieve, at least in the winter rainfall dryland system investigated here. Plant establishment in the direct seeding treatments was dominated by a single species, the small spreading mesemb, *C. denticulata*. Growth of plants that survived the dry season was extremely slow, leaving them vulnerable to disturbance. Small numbers of targeted perennial seedlings were observed in unseeded treatment quadrats in both Winter 2014 and Summer 2015, suggesting that the natural soil seed bank has been substantially depleted (Simons 2005). Additionally, perennial shrubs that established well in seeded treatments initially after rain struggled to survive the dry season.

The effects of additional treatments were small and the addition of seed is the critical factor for attaining species richness, SWDI, and plant establishment. Overall, low species richness, SWDI values, and levels of plant establishment and dry season survival of individual plants were observed across all treatments. *C. denticulata,* the only species to establish in large numbers, although at very small sizes, is a non-palatable succulent shrub, with limited agricultural value. While its presence may contribute to increased soil stability, this was not tested in the current study. The ability of this species to promote the growth of the more agriculturally productive forage species was also not tested but is questionable given the more likely limiting factors in the environment, which are described below.

Factors that limited successful plant establishment included low rainfall, under-studied seed biology, and losses of top soil and the natural soil seed bank (Simons 2005). In a recent article, Milton and Dean (2015) reflected on 20 years of restoration challenges in South African drylands. They concluded that restoration efforts are often unsuccessful because rainfall is low and unpredictable, falling mainly in the winter when plants are less likely to utilise it and resulting in a very short growing season; perennial species are typically long lived and slow growing, and re-creating the necessary conditions for their establishment is challenging; and degraded areas, denuded of vegetation, are particularly exposed to environmental stressors. Other ecological constraints include 'soil changes, altered hydrology, presence of alien plant species which alter ecosystem processes, changed microclimate or the loss of native seed banks' (Vetter 2009; see also Suding et al. 2004). Some critical processes, such as overcoming seed dormancy and re-creating ideal establishment conditions, are poorly understood. Practical limitations include constraints on sources of seed from both commercial suppliers and wild harvesting.

We have estimated that large areas in Nama Khoi are degraded beyond biotic or abiotic thresholds, requiring active intervention. We have demonstrated that restoration at scale is expensive. Restoration in fragile dryland environments is often risky, time consuming, and labour intensive. Once in process, disturbance is 'difficult and costly to contain or reverse' (Milton and Dean 2015). Where plant establishment is limited by rainfall, the costs of restoration are very likely to greatly exceed the current value of the land and its annual production potential (Blignaut et al. 2013; Tinley and Pringle 2014). Even at the most generous discount rate applied in our analysis, ecological restoration is more expensive than the high costs of replacing lost natural forage with supplementary fodder. While restoration achieves positive NPV and a BCR >1 at both the 3 % and 1.3 % discount rate, it nonetheless appears prohibitively expensive.

Given these results, it may be wise to consider some pre-emptive and/or alternative interventions for the management of degraded areas and the livelihoods they support.

First, improving land management and avoiding further degradation is critical. Once degraded, arid and semi-arid systems cannot easily be restored to their former productivity. Sustainable land management practices, including sustainable grazing practices, must be developed and implemented to ensure that land is used in a way that is both profitable and environmentally sensitive (Milton et al. 2003).

Second, accepting some loss as irreversible and identifying alternative land uses for severely degraded areas may be required. Novel land uses for highly degraded areas that are difficult and expensive to restore could include rain-fed cropping with indigenous drought tolerant legumes, or continued light disturbance to promote the growth of seasonal ephemerals. Van der Merwe and Van Rooyen (2011) reported that spring ephemerals dominate abandoned croplands of various ages for almost a decade after the last ploughing. They are then replaced by indigenous pioneer shrubs low in diversity and agricultural value, such as *G. africana.* This transition represents the formation of a novel, but stable, ecosystem (Hobbs et al. 2013). Samuels et al. (2015) have shown that small livestock in Namaqualand depend heavily on seasonal flushes of annual vegetation in disturbed areas. They found that, during the wet season, free-ranging sheep grazed exclusively on annuals in fallow croplands. Although they noted that overdependence on annual vegetation during the wet season could make livestock vulnerable during drought periods when forage production is low, the seasonal availability of nutritious annuals provides a valuable food source and may allow perennial forage resources to rest and regenerate. Where restoration is not possible, cropping or light disturbance may encourage highly productive, albeit seasonal, forage growth, and help avoid both overgrazing in intact areas and the entrenchment of highly stable but unproductive secondary shrublands.

Third, investment in alternative livelihoods, or even relocation of farming communities in extreme cases, may be the best long-term option, as projected climatic changes make farming on the margins ever more challenging (Archer et al. 2008). Rural livelihoods are typically diverse, particularly in poorer communal areas where farming 'serves primarily as a safety-net against unemployment and makes a relatively small contribution towards day-to-day household subsistence' (Vetter 2009). There are often high levels of dependence on off-farm wage labour and government grants. In this context, developing local capacity and skills, for example in eco-tourism (Le Maitre et al. 2009), ecological monitoring and research (Araya et al. 2009), or natural resources management (Turpie et al. 2008; Roberts et al. 2012), may usefully reduce reliance on increasingly marginal agricultural activities.

On the other hand, returning degraded areas to a state of ecological functioning, moderate levels of biodiversity, and acceptable agricultural productivity is desirable. Degraded Succulent Karoo appears not to return to a state comparable with less disturbed sites through rest alone, even if left undisturbed for several decades (Westoby et al. 1989; O'Connor 1991; Milton et al. 1994; Van der Merwe and Van Rooyen 2011). Snyman (2003) has further demonstrated that the exclusion of grazing and the application of management practices in favour of rangeland restoration may not be enough to yield the desired effect on plant cover and density. Some degree of active intervention will be required. The establishment of small spreading mesembs that grow easily could serve to kickstart ecological processes and stabilise soils. Grass growth could be encouraged with irrigation as, although grasses survived poorly through to Summer 2015, they do emerge in large numbers in response to rain.

Economic assessments as a means of estimating the value of an intervention are limited in terms of ascribing value in the context of ecosystem services (TEEB 2010). It is challenging to quantify, in purely monetary terms, the value of a limited resource to the individual or individuals who are directly and wholly dependent upon it (Fourcade 2011). It is difficult to predict long-term future changes to both costs and benefits, as well as unintended consequences of actions, and opportunity costs (Costanza et al. 1997). Some benefits, such as landscape aesthetics or human well-being, are difficult, if not impossible to quantify in monetary terms (Fourcade 2011). Additionally, many critical relationships, such as the potential impact of restoration on ground water recharge, are not well understood and hence benefits (or costs) may be missed.

However, opportunities to develop large-scale restoration through private and public sector investments that look beyond simple economic analyses and seek to address the public good do exist and could be further developed (Carrick et al. 2015). For example, in South Africa, several Expanded Public Works Programmes have been created using tax allocations to maintain, restore, or rehabilitate ecosystems while at the same time creating jobs for, and building marketable skills in, marginalised communities. Government funders of such programmes have indicated that they would prefer to see the labour cost of restoration as a benefit in terms of employment creation. Adjusting restoration targets to focus on achieving interim goals such as soil stabilisation and water infiltration will enable some large-scale rehabilitation to take place through programmes like these while researchers develop more efficient and effective restoration protocols. Taken together, these methodological limitations and development opportunities suggest that decision makers should consider more than the economic efficiency of a project when deciding upon the merits of one or several courses of action.

Responses to degradation in Nama Khoi, and other dryland regions, will depend on available resources, biotic and abiotic constraints, and socio-economic contexts, including societal values, rules, and knowledge (Goddard et al. 2016) and political and economic decisions regards the distribution of resources. Given the long timeframes required for restoration, various decision points throughout the process will be needed to evaluate the best courses of action, depending on how values, rules, and knowledge evolve over time—a framework increasingly referred to as a 'pathways' approach (Wise et al. 2014; Goddard et al. 2016).

Dryland restoration will necessarily form part of the package. While more comprehensive means of assessing return on investment are needed, the effectiveness of the restoration intervention is the critical issue and must be the focus of future research. Although currently costly, researchers around the world are working on improving both the efficiency and effectiveness of dryland restoration (Carrick and Kruger 2007; Madsen et al. 2012; Turner et al. 2013; Erickson 2015; Erickson et al. 2016). Judicious investments in restoration, combined with land use practices that prevent degradation, will be required to sustain land-based livelihoods in this region over the longer term.

Acknowledgments This research was undertaken as part of the International Climate Initiative. All work undertaken was funded by the German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety, which supports this initiative on the basis of a decision adopted by the German Bundestag. The authors thank Nkosinathi Nama for invaluable field support, as well as the farmers of Steinkopf iKosis Ward and the municipal officials of the Nama Khoi Local Municipality for their time, information, and access to land. We thank Dr. Nalini Rao for her early work on the economic analysis, Dr. Heidi Hawkins and Dr. Peter Carrick for their input into the restoration trial study design and analysis, Dr. Todd Erickson for his assistance with the data presentation, our many partner organisations for access to their records, and two anonymous reviewers for their very constructive feedback which has contributed to a greatly improved manuscript. The guest editorial team for this special issue also provided very valuable feedback and guidance.

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