RESEARCH ARTICLE

Evaluation of Restoration Success to Inform Future Restoration Efforts in *Acacia reficiens* Invaded Rangelands in Northern Kenya

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ABSTRACT

Arid and semi-arid rangelands support a significant portion of the world's human population, as well as its biodiversity. These landscapes are threatened by degradation, through loss of vegetation, increasing spread of invasive or undesirable species, or both. Efforts to halt or reverse degradation exist, but lack of monitoring and reporting of restoration outcomes hampers efforts to replicate and upscale effective practices to other areas. This paper demonstrates how monitoring can inform future efforts through retrospective analysis of restoration projects on *Acacia reficiens* invaded rangelands in northern Kenya. *A. reficiens* has encroached into productive rangeland undermining both livestock production and endangered wildlife species conservation. Using a mobile phone application, LandPKS, we assessed 22 plots across 13 restoration sites in Westgate and Kalama conservancies in Northern Kenya that had been cleared of *A. reficiens* and reseeded with *Cenchrus ciliaris*. We found that these restoration treatments led to increases of more than 25% in overall ground cover, 34% in perennial grass cover, and 60% in standing herbaceous biomass. We therefore suggest that manual clearing of *A. reficiens*, when carried out in the late dry season and combined with both reseeding and prudent pre- and post-treatment seed and soil conservation practices, has the potential to provide an efficient and cost-effective solution to help reverse habitat losses. Our use of mobile phone applications allowed rapid assessment of restoration outcomes, and the resulting data are already being used to help design restoration projects on rangelands in northern Kenya.

Keywords: *Cenchrus ciliaris*, LandPKS, mobile apps, reseeding, tree clearing

Restoration Recap

- Manual clearing of *Acacia reficiens* stands in northern Kenya is demonstrably effective, and positive results persisted at least three to five years following treatment.
- Reseeding with *Cenchrus ciliaris* is a sustainable method of spurring perennial grass regeneration, and newly established stands can provide seed for future reseeding projects.
- The combination of clearing *A. reficens* stands, reseeding with *C. ciliaris* and construction of low-cost, low-tech shallow trenches filled with residual crown material may be a key to restoring the productivity of rangelands currently colonized by this species, provided it is combined with grazing management.
- It is imperative that large scale restoration projects set up treatment and control plots to monitor and evaluate relative restoration success. Easy to use mobile phone applications exist that could facilitate rapid restoration assessment.

Nearly 40% of the global land surface is characterized as dryland, which supports more than a third of the earth's population (MEA 2005, Reynolds et al. 2007). Some studies estimate that 10–20% of global drylands are degraded, with 12 million new hectares a year characterized by losses in productivity (MEA 2005, Brauch and Spring 2009, James et al. 2013). In Africa, arid and semi-arid rangelands cover approximately 43% of the land mass and support close to half of its population. Most of these rangelands are considered degraded, with approximately 65–70% of rangelands in Sub-Saharan Africa classified as being moderately to severely degraded, with significant reductions in vegetation cover, increases in undesirable species, or both (Oldeman et al. 1991, Tamene and Le 2015).

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Several initiatives have been implemented to halt or reverse these losses, from large scale intergovernmental efforts like the African Resilient Landscapes Initiative (ARLI), to smaller scale projects on individual conservancies, parks, and reserves (Clewell and Aronson 2006, Suding 2011, World Bank 2015). The scarcity of resources available to managers of degraded African rangelands discourages experimental testing of restoration methods and subsequent monitoring and reporting of restoration outcomes (Ruiz-Jaen and Aide 2005). This unfortunately leads to the uniform and uncritical application of restoration methods, despite general knowledge that the success of different restoration strategies depends on topography, soils, climate, and the type of degradation.

Following overgrazing and recurrent drought, much of the Ewaso ecosystem in northern Kenya is degraded (Kimiti et al. 2017a). These lands are characterized by large patches of bare ground and the spread of undesirable native species like *Acacia reficiens* and exotic invasive species like *Opuntia stricta*. The spread of *A. reficiens* in this ecosystem is of great concern as it reduces both habitat for endangered wildlife species like the Grevy's zebra, as well as available forage for pastoral communities (Shultka and Cornelius 1997, Kimiti et al. 2017a). *A. reficiens* generally has little to no herbaceous understory and is relatively unpalatable to most livestock and wildlife species. Because the species forms a monoculture, large areas that seem healthy from a superficial perspective are often unproductive due to limited grass or forb production. The lack of herbaceous basal plant cover also leads to increased runoff velocity, which invariably leads to increased soil erosion (Thurow et al. 1988). This has placed significant pressure on the pastoralist communities that rely on this landscape for grazing and browse resources for livestock. They often have to travel great distances in search of forage, or concentrate on the few remaining productive areas (Keane and Crawley 2002, Mitchell et al. 2006). It also negatively impacts wildlife, dealing a blow to important community based eco-tourism efforts, and therefore placing pastoralist livelihoods in further peril.

In the mid-2000s, community members in affected landscapes in Samburu county began searching for options for halting *A. reficiens* encroachment and possibly reclaiming former grazing land on encroached lands (Alex Lekalaile, pers. comm.). Beginning in 2009, the Northern Rangelands Trust (NRT), with support from the Grevy's Zebra Trust (GZT), set up pilot restoration projects. These projects were aimed at reducing density and cover of *A. reficiens* through manual clearing with machetes and increasing forage available to livestock and wildlife in these mixed-use community conservancies through reseeding with the native perennial *Cenchrus ciliaris*. NRT is a regional organization that provides management support for wildlife conservancies in north-central Kenya. Based on anecdotal information, these projects are considered successful, but lack of quantitative information precludes objective evaluation and the ability to implement informed adaptive management. Even the limited datasets that do exist do not include sufficient soil and climate information to define the conditions under which these approaches are likely to be successful.

The definition of "restoration success" has always been a point of contention, with many considering the use of the term "success" misleading. Typically, people measure conditions, structure, processes, ecosystem development, similarity to reference sites, and potential for self-sustainability (Zedler 2007). An alternative approach to restoration success evaluation incorporates ecological process status relative to a reference state corresponding to each unique ecological site. An ecological site is defined as terrain with specific soil and physical characteristics that differs from other kinds of land in its ability to produce distinctive vegetation assemblages, and in its ability to respond similarly to management actions and natural disturbances (USDA-NRCS 1997). A necessary component in the incorporation of these ecological site descriptions into practical short-term management is the development and use of state-and-transition models (STMs; Herrick et al. 2006). STMs are representations of two or more alternative ecological states (each with unique vegetation assemblages) possible on a given ecological site under specific physio-climatic properties and disturbance regimes (Westoby et al. 1989, Stringham 2003, Briske et al. 2008). Although this system works well in the United States, the information necessary to define ecological sites and develop STMs rarely exist in developing nations (Clewell and Rieger 1997). Basic Ecological Site Descriptions (ESDs) and STMs for select African savannas do exist (Dougill et al. 1999, Joubert et al. 2008), but there is no easily accessible documentation of reference states that could be used as a reference for restoration for most African rangelands.

Several rapid assessment options have been explored that would further encourage natural resource managers to collect information on restoration effectiveness without requiring detailed state and transition models or ecological site descriptions. A widely used approach includes establishing untreated control sites, in addition to treatment sites, in a Before-After, Control-Impact (BACI) sampling framework (Smith 2014). This approach involves selecting response variables of interest and measuring them both on treated and untreated plots before any action is taken. These responses are then continuously measured following restoration or management action, and comparisons are made both between and within treatment and untreated areas, spatially and temporally. Successful application of such an approach would provide a low cost, easily repeatable option for restoration practitioners to assess restoration results. However, to be effective, it requires establishing paired plots where the conditions (soils, topography, climate, and current vegetation) that are likely to affect response to restoration are as similar as possible.

We took advantage of a simple, free mobile phone application to rapidly characterize and record soil, topographic and vegetation properties on existing restoration projects on *A. reficiens* invaded rangelands in northern Kenya (see Kimiti et al. 2020—in this issue). This app allowed us to collect and analyze land potential information on a mobile phone, facilitating data acquisition from global soils and climate databases. Additionally, the app allowed vegetation data collection, collation, and cloud upload, negating need for data summarization and reducing arithmetic errors. We then used the app to collect quantitative data on vegetation variables useful to land managers in this landscape. Since we could not conduct a full BACI analysis, having carried out our assessments post-treatment, we identified treatment plots and matched them to controls which were determined likely to have had similar initial vegetation based on similar biophysical potential (soil, climate, topography) and disturbance regime. We then used these paired plots to conduct a retrospective analysis of vegetation responses. Our objective was to evaluate the effectiveness of this approach for determining the success of the restoration techniques piloted in this project (Nordlind and Östlund 2003).

Methods

Study site

Samburu County is part of the greater Ewaso Ecosystem in North Central Kenya. This ecosystem is dominated by mixed use semi-arid rangeland supporting both cattle and wildlife. Vegetation composition in the area is related to proximity to the Ewaso Ng'iro river. In low lying riverine areas, vegetation is dominated by *Acacia elatior* (Fabaceae) and palms of the *Hyphaene* genus (Arecaceae). In drier areas further from the river, vegetation communities are characterized by *Acacia-Commiphora* semi-arid scrub woodland, primarily *Acacia tortillis, Acacia reficiens,* and *Commiphora africana* (Burceraceae) with patches of *Acacia* wooded grassland. Ground cover is dominated by a mixture of both annual and perennial forbs and grasses, including *Indigofera spp.* (Fabaceae)*, Cynodon spp., Eragrostis spp.,* and *Aristida spp.* (Poaceae) (Wittemyer 2001, Low et al. 2009). Common wildlife species include *Loxodonta africana* (elephant)*, Aepyceros melampus* (impala) and *Equus grevyi* (Grevy's Zebra).

Acacia reficiens is a native multi-stemmed bush or flat-topped tree that grows up to 5 m in height. Stem count and diameter varies greatly, with one study finding trees 5 m in height having an average of three stems per plant, and an average stem diameter of 62 mm (Coughenour et al. 1990). Information about the regeneration characteristics of this tree is scarce, and although most *Acacia* species are dispersed by animals, seeds of this tree are also believed to be wind dispersed. Seed density in the soil seed bank is lower than most other *Acacia* species (Tybirk et al. 1994).

We focused our study on sites that had been cleared and reseeded on both the Westgate community wildlife conservancy (37°21′ E, 0°28′ N) and the Kalama community wildlife conservancy (37°37′ E, 0°41′ N). These sites had been treated under a rangeland rehabilitation and restoration program cooperatively managed by the NRT and GZT, and we assessed sites that had been cleared and reseeded five, three, and two years prior to this study. Since this assessment project was carried out after the restoration effort, we could not conduct a full BACI experiment, and therefore our assessment was by necessity carried out retrospectively. Data were collected in the late rainy season to coincide with the period of maximum cover.

Treatments

Clearing and reseeding sites were originally set up by community members, with decisions about when and where to apply treatments made by each conservancy's grazing committee, which directs land management decisions. The first step in these treatments was the demarcation of a large area that had been colonized by *A. reficiens*. The area was subsequently cleared of the target species using machetes; cutting the tree down at the midway point between the lowest branches and the tree base. This was done during the height of the dry season when the plants were at maximum water stress and therefore less likely to survive. After clearing, the remnant spiny crown material was then spread in the inter-canopy spaces, which at this point were large patches of bare ground.

As the rainy season began, *Cenchrus ciliaris* seeds, obtained initially from the Murray Trust in Baringo, and thereafter locally, were manually broadcast randomly in the treated areas, with a variable seeding rate, but approximating 300 seeds/m^2 or 0.5 g/m^2. On at least one of these properties, an arbitrary target of 50% perennial grass cover was set by local management as an acceptable result of the reseeding projects (Alex Lekalaile, Westgate Conservancy, pers. comm.). Efforts were made to exclude livestock from treatment areas, however germinating grass material was susceptible to wildlife and trespass livestock herbivory. The thorny crown material spread across the ground was therefore intended to not only trap soil sediment and seeding material during water flow events, but also to protect subsequent seedlings from herbivory by wildlife and livestock (Tongway and Ludwig 1996, Kimiti et al. 2017b).

Plot selection

We assessed 22 matched pairs (n = 22) across the 13 restoration sites on the two conservancies. This is because some restoration sites were large enough and heterogenous enough that multiple plots were required. Treatment plots were established by selecting candidate points on a Google

Earth satellite image of the site using stratified random sampling, using visible soil and vegetation differences as strata. For each treatment plot, we identified two candidate control plots in an adjacent untreated area (each less than 100 m away from the identified treatment plot) based on similarity of surface biophysical features including topography, soil mineralogy, and information about vegetation similarity at time of clearing, which was provided by the local conservancy land managers.

Subsequently, each treatment plot and corresponding candidate control plots were then characterized using the LandInfo module of the Land-Potential Knowledge System (LandPKS) mobile phone application (landpotential.org). LandInfo enabled us to collect and manage basic information about the soil profile, including soil depth, texture by feel, and rock fragment percentage at six different layers down the soil profile. Soil texture and rock fragment percentage were recorded at 0–1 cm, 1–10 cm, 10–20 cm, 20–50 cm, 50–70 cm, and 70–90 cm. This soil texture information was then run through a pedo-transfer model, Rosetta (Schaap et al. 2001), to obtain values for Plant Available Water Holding Capacity (PAWHC). This calculation is completed automatically in newer versions of the app. We considered PAWHC as an integrative variable for our soil texture measurements. This information was then combined with information from other global climate and crop growth databases to obtain grass growth potential and soil erosion potential estimates from the comprehensive Agricultural Policy/Environmental eXtender model (APEX; Williams and Izaurralde 2008).

We then used a combination of heuristic matching of surface and subsurface biophysical similarities, PAWHC, estimated productivity, and estimated erosion to select the best matching control plot to each of our treatment plots, resulting in a matched-pairs experimental design.

Vegetation measurements

For each treatment-control pair, we collected data on select vegetation response variables primarily using the LandCover module of the LandPKS mobile app, which is based on a modified line-point intercept (Herrick et al. 2017). To ascertain overall effectiveness of treatment plots at increasing vegetation cover, we collected data on total ground cover, foliar cover, bare ground cover, perennial grass cover, shrub and sub-shrub cover, annual plant cover, and both woody and herbaceous litter cover. To assess differences between treatment and control plots in run-off and wind erosion susceptibility, we collected information about the number of large gaps (> 1 m) between plant bases and plant canopies in each plot. To find out the effectiveness of treatments at reducing the cover of our target woody species, we compared tree cover, as well as the density of *A. reficiens* plants (including saplings) between our treatment and control plots. Additionally, we collected standing biomass data by setting up four 1 m × 1 m quadrats in each plot and clipping, air drying, and weighing all above ground herbaceous forage. This information was collected in the late growing season to capture peak perennial herbaceous growth and avoid pulses of ephemeral plant growth.

Data analysis

The Goodness of Fit function of JMP was used to test if the data were normally distributed. Based on the results of this analyses, parametric statistics were used. To ascertain how similar our treatment plots were to our control plots, we carried out paired t-tests on the slope size (%), PAWHC, and both potential grass productivity and potential soil erosion as predicted by the APEX model. We carried out a Bowker's test of symmetry to assess similarity of soil texture classes between treatment and controls at various depths. Due to varying soil depths across sites, we only analyzed soil texture similarities at four depths (0–1 cm, 1–10 cm, 10–20 cm and 20–50 cm). For all our vegetation response variables, we carried out paired t-tests to compare treatment and controls using the 22 matched pairs.

To investigate whether there was any influence of site location and selected environmental characteristics on treatment effectiveness, we used t-tests to determine if the treatments were more effective on the Kalama conservancy than the Westgate conservancy. Biomass, cover, and density of *A. reficiens* were compared. One-way analysis of variance was used to determine if year since treatment inception and soil texture (relative proportion of sand, clay, and loam) affected plant cover and density and the magnitudes of differences in plant metrics between treatment and control plots. All analyses were carried out in JMP software. Significance was set at $\alpha = 0.05$.

Results

Plot similarity

There was no difference in the measured slope between treatment ($\bar{x}$ 3.0 ± $S.E._M$ 0.31%) and control (3.4 ± 0.45%) plots (Paired t-test, t = –0.75, df = 21, $p = 0.46$). Estimated potential grass productivity and potential erosion were also not different between treatment and control plots (t = 1.00, df = 21, $p = 0.33$; t = 1.68, df = 21, $p = 0.12$). Treatment and control plots had similar soil texture at the four layers we tested; 0–1 cm (Bowker's test for symmetry, $\chi^2 = 10.33$, 0.78), 1–10 cm ($\chi^2 = 6.67$, $p = 0.76$), 10–20 cm ($\chi^2 = 17$, $p = 0.07$), and 20–50 cm ($\chi^2 = 4.33$, $p = 0.63$). Additionally, there was no difference between treatment (8.37 ± 0.42 cm) and control (8.68 ± 0.44 cm) plots in PAWHC (t = –0.88, df = 21, $p = 0.39$).

Vegetation measurements

Treatment plots had 30% higher total cover (t = –7.74, df = 21, $p < 0.001$), which included 28% higher foliar cover (t = 6.34, df = 21, $p < 0.001$; Figure 1). Perennial grass cover

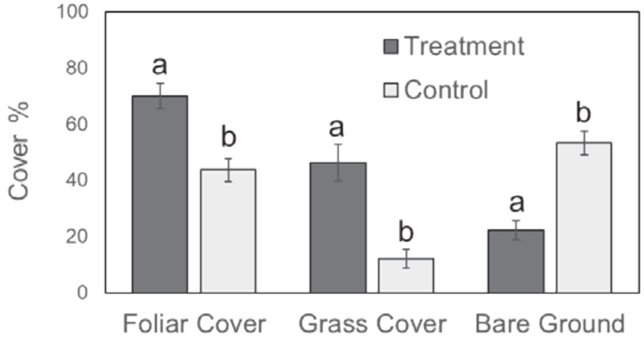

Figure 1. Foliar cover, perennial grass cover, and bare ground cover in treatment and control areas. Error bars show standard errors (n = 22).

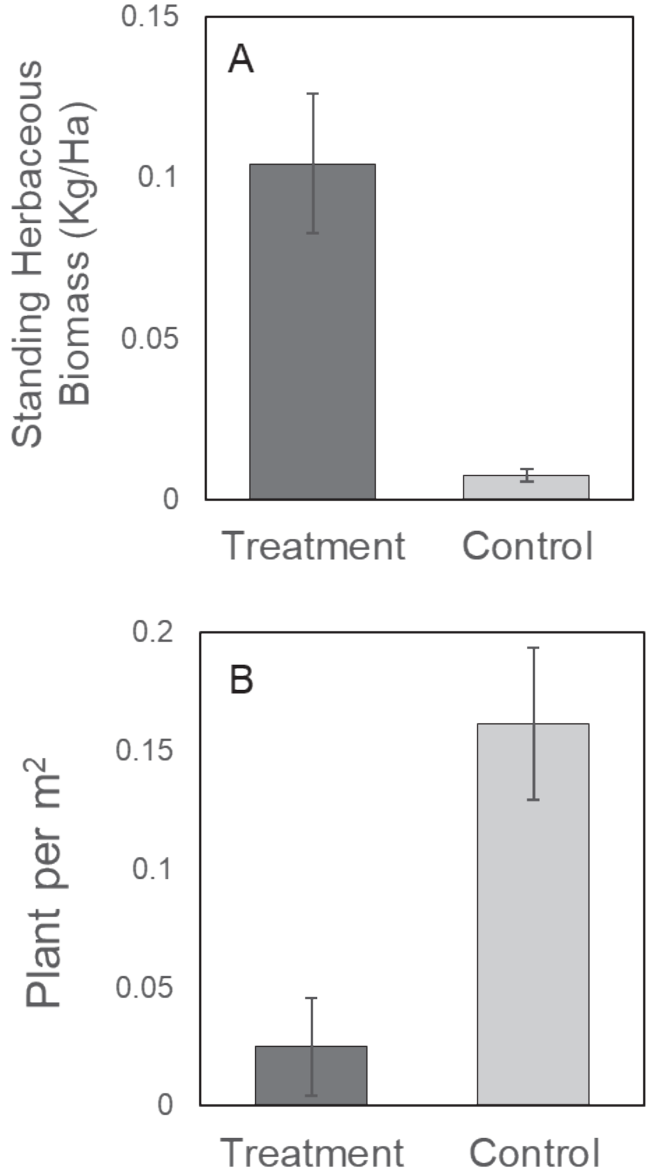

Figure 2. *Acacia reficiens* density A). and standing herbaceous biomass B). in treatment and control areas. Error bars show standard errors (n = 22).

was also nearly 35% higher in the treated areas than in the untreated areas ($t = 7.07$, $df = 21$, $p < 0.0001$; Figure 1).

Cleared areas had 60% more standing herbaceous biomass than control plots, or 100 kg per hectare more forage ($t = 4.39$, $df = 21$, $p < 0.001$; Figure 2). Treated areas had a lower density of *Acacia reficiens* plants ($t = 4.60$, $df = 21$, $p < 0.001$; Figure 2), which amounted to 1364 or 85% fewer plants per hectare. Sites that were cleared had lower tree cover than control areas ($t = 3.70$, $df = 21$, $p = 0.001$), but higher shrub cover ($t = 2.29$, $df = 21$, $p = 0.03$).

Sub-shrub and perennial forb cover was twice as high in the treatment plots as in the control plots ($t = 3.87$, $df = 21$, $p < 0.001$), and treated areas exhibited nearly half as many 1-m gaps between plant bases, an important indicator of potential run-off erosion (Paired t-test, $t = 4.59$, $df = 21$, $p < 0.001$). There was no difference in annual plant cover between treatment and control areas ($t = 0.40$, $df = 21$, $p = 0.69$). As expected, woody litter cover was higher in the treatment areas ($t = 6.08$, $df = 21$, $p < 0.0001$) after clearing, while herbaceous litter was only marginally greater in the treated areas compared to the untreated plots ($t = 1.93$, $df = 21$, $p = 0.07$, Table 1).

Site influences on vegetation responses

Although there were no discernible effects of our environmental variables (site location, slope, and texture at four depths) on the magnitude of differences in total cover, perennial grass cover, or foliar cover between treatment and control sites, there were some influences on other vegetation metrics. Site location influenced biomass, with plots on Kalama conservancy generally having greater biomass responses on treated control plots (Student's t-test, $t = -2.42$, $df = 20$, $p = 0.03$). Additionally, reduction in density of *A. reficiens* plants on treatment plots was higher on sites in Kalama conservancy than those in Westgate conservancy ($t = 2.45$, $df = 20$, $p = 0.02$). Soil texture at 20 to 50 cm influenced percent of basal gaps larger than 1 m, with treatment plots characterized by sandy clay loam textures having nearly 50% fewer large basal gaps than control areas (One-way ANOVA, $f = 3.52$, $df = 3$, $p = 0.04$). Differences in annual plants were also highest in plots with sandy clay textures at this horizon, with treatment areas characterized by sandy clay soils having 24% fewer annual plants than control areas (One-way ANOVA $f = 4.33$, $df = 3$, $p = 0.02$). The year of treatment set up had no influence on any of our vegetation metrics or the magnitudes of differences between treatment and control plots.

Discussion

Mechanical control is often used on areas encroached by woody species like *Mimosa pigra* and *Acacia rigidula*, but terrain limitations on heavy machinery (for bulldozing, grubbing, and chaining) as well as concerns about regrowth from the seedbank and vegetative reproduction has often

Table 1. Means and standard errors ($S.E._M$) for vegetation response variables on treatment and control areas (n = 44, df = 21).

Cover (%)	Treatment	Control	*p* value
Tree cover	6.50 ± 1.98	16.27 ± 1.86	0.001
Shrub cover	5.13 ± 1.57	1.91 ± 0.62	0.03
Perennial forb cover	24.82 ± 4.40	11.59 ± 2.56	< 0.001
Annual plant cover	11.04 ± 1.63	12.18 ± 3.27	0.69
Herb litter	6.36 ± 2.30	1.77 ± 0.59	0.07
Woody litter	17.32 ± 2.04	3.46 ± 0.88	< 0.001
Basal gaps ≥ 1m	28.41 ± 6.42	62.05 ± 6.97	< 0.001

resulted in preference of other methods (Siriworakul and Schultz 1992, DiTomaso 2000, Paynter and Flanagan 2004). The strategy of mechanical clearing and broadcast reseeding was selected due to the relatively straightforward nature of application and ease of access to required tools. We found that manual clearing increased plant cover, density, and biomass of palatable vegetation compared to untreated areas.

One of the problems facing efforts to control invasive or encroaching species worldwide is the risk of regeneration and recolonization after removal has occurred, either through vegetative propagation or seed banks in the soil (Paynter and Flanagan 2004). In our study, *A. reficiens* density (including saplings) inside the treatment plots was lower compared to untreated areas. The relative differences in density of this plant inside and outside treatment areas would indicate that cutting of the crown at the height of the dry season did indeed limit seed production or vegetative propagation from discarded plant material.

Given the lack of observed new vegetative propagation from discarded tree material, and the low rates of reemergence from the seedbank, the pervasive extent of encroachment by this species is surprising (Johansson and Kaarakka 1992, Tybirk et al. 1994). On Kalama conservancy, the only *A. reficiens* saplings we came across were on an abandoned cattle corral after a rainy season, possibly as a result of scarification of seed material after animal ingestion, as well as higher nutrient concentrations due to urine and dung deposits inside the corrals. This would suggest a preference for high nutrient concentration and ideal seed germination conditions for *A. reficiens* to germinate and establish, as opposed to the currently assumed preference for degraded conditions. This perceived lack of continuing recruitment across the landscape might indicate that this species colonizes affected areas following episodic climatic or disturbance events like prolonged droughts and extreme rainfall as well as influx of grazers into an ecosystem (Di Castri 1990, Thibault and Brown 2008, Diez et al. 2012). Further research is required to determine the factors driving initial colonization by this species, as well as the drivers of its subsequent success.

A goal of this intervention was to increase available forage for livestock and wildlife, primarily in the form of perennial grasses. Higher cover and production in the treatment areas as compared to the control areas demonstrates that this objective can be achieved. The 100 kg increase in herbaceous biomass was considered positive by the resident communities given the historical low forage production in the landscape. Seeding with *Cenchrus ciliaris* after clearing contributed to the success of the restoration efforts. On rangelands in the United States and Australia, where it is more commonly known as buffelgrass, *C. ciliaris* is considered a problem invasive species of low forage value (Franks 2002, Clarke et al. 2005, Franklin et al. 2006, Marshall et al. 2012). Ironically, it was initially introduced in these ecosystems mainly to help control erosion and improve pasture due to its high tolerance to drought and capacity to withstand heavy grazing (Miller et al. 2010, Marshall et al. 2012). On its native range on African savannas, however, this species has been used for reseeding projects for decades due to its efficacy in establishing higher cover vegetation patches, thereby improving chances of trapping other seed material from the surrounding landscape and persisting through prolonged dry periods (Ludwig and Tongway 1996, Kimiti et al. 2017b). Together with *Eragrostis superba* and *Enteropogon macrostachyus*, *C. ciliaris* is one of the most commonly used species for reseeding on rangelands in East Africa (Kinyua et al. 2010, Koech et al. 2014, Mureithi et al. 2014, Mganga et al. 2015).

Although perennial grass cover was higher in treated than untreated areas, total perennial grass cover was still below the 50% threshold set by management. Treatment plots on both conservancies produced enough grass cover that the communities could harvest seeds at the end of each rainy season that was then used for further reseeding projects on the landscape. Seeding efficiency could potentially be optimized by both regulating seeding rates as well as accounting for micro-topographical disruptions in seed distribution across restoration sites (Tongway and Ludwig 1996). Diversifying native grass species used in reseeding would also be a potential method for increasing species diversity and patch resilience. Distributing spiny crown material in inter-canopy spaces is ostensibly meant to protect seed material, but studies have shown the relative advantage of shallow trenches or berms on certain soils over placement of material on the surface (Tongway and Ludwig 1996, Rango and Havstad 2011, Kimiti et al. 2017b). Monitoring on these sites should continue periodically to assess their long-term ability to spur and sustain increased herbaceous species diversity and achieve restoration goals.

Lack of information about reference conditions and ecological site descriptions affect the ability to evaluate restoration success as well as predict it (Ruiz-Jaen and Aide 2005, Brewer and Menzel 2008). We found that site location, including texture at different depths, affected

Figure 3. Photographic comparison of vegetation differences in a control plot (left) and a treatment plot (right). Plot centers were less than 100m apart. Photos were taken on the same day during the dry season (n = 22).

certain vegetation metrics. Site location and soil texture at the 20–50 cm level can influence vegetation growth and effectiveness of patch-level restoration treatments in this ecosystem. Landscape position, soil texture at different soil profile horizons, slope shape, slope size, and local climate conditions will affect the ability of a site to capture water and nutrients and make them available to plants (Saxton et al. 1986, Lal and Shukla 2004, Herrick et al. 2013). Conceptually, this indicates the need for collecting site characterization data to help explain any differences in treatment success between different locations, and subsequently use of this information will improve our ability to select future areas for restoration based on relative likelihood for successful vegetation establishment. A large body of work describing the influence of local climate, soils, and topographic characteristics on potential productivity of a site is available (Herrick et al. 2013, UNEP 2016). Collecting this information could help land managers better decide where to direct scarce resources for restoration.

Our results demonstrate the effectiveness of manual clearing of *A. reficiens* stands as a restoration method. When paired with reseeding interventions, especially those that optimize seed retention and sediment capture, it has the potential to reverse losses in forage for both wildlife and livestock in affected community owned, mixed-use rangelands. However, for these treatments to have a greater impact on landscape and watershed level restoration, they must be monitored over time, documented, and disseminated to a wider audience to allow upscaling and replication of effective methodologies.

As encouraging as these snapshot results are, there is need to emphasize that any rehabilitation or restoration intervention is unlikely to succeed in the long-term if the underlying factors causing degradation or encroachment are not adequately addressed. In the case of Samburu and Laikipia rangelands, the primary degradation driver has been identified as unsustainable grazing practices leading to overgrazing. Unless grazing management across the larger landscape is restructured to avoid overuse of available forage, then any rehabilitation or restoration programs will only have temporary benefits to the communities implementing them. Further research on impact of *A. reficiens* clearing on browse availability to wildlife and domestic livestock is necessary before "success" can clearly be shown.

The prior lack of quantitative data on the effectiveness of this restoration exercise meant that the results and lessons learned from it could not be confidently shared with a wider audience or to landscapes with similar species encroachment, which limited future restoration efforts. Our use of LandPKS tools to collect both the site description data as well as vegetation metrics allowed us to take advantage of a data sharing portal to display results and provide free access online. Dissemination of these results is facilitating replication of these methods on similarly encroached rangelands across northern Kenya, primarily on community conservation areas (K. Avery, Northern Rangelands Trust; pers. comm.). This type of open access, crowd sourced data storage could be the key to dissemination of both restoration successes and failures, resulting in an increased general awareness of best practices.

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