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Rehabilitation of community-owned, mixed-use rangelands: lessons from the Ewaso ecosystem in Kenya

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Abstract Globally, 10–20% of arid and semi-arid rangelands have been classified as severely degraded (UNCCD, in Elaboration of an International Convention to Combat Desertification in countries experiencing serious drought and/or desertification, particularly in Africa 1994; MEA, in Ecosystems and human well-being: current state and trends. Island Press, Washington, DC, 2005), and in sub-Saharan Africa specifically, 70% of rangelands are considered moderately to severely degraded (UNCCD 1994). Given that these drylands make up 43% of Africa's land area and support approximately 45% of its population, restoring, maintaining and even increasing their productivity is imperative from both conservation and food security standpoints. In the Laikipia and Samburu counties of Kenya, degradation manifests itself through the increase of bare ground and the replacement of perennial grasses by undesirable plant species, primarily *Acacia reficiens* and *Opuntia stricta*, resulting in reduced forage availability. Further complicating management is the fact that most land in this ecosystem is owned by community conservancies, where the land is managed to support both wildlife and livestock grazing. There has been considerable effort targeted towards using mechanical clearing coupled with reseeding to combat *A. reficiens* spread. Additionally, the use of both traditional and modern mobile cattle enclosures (commonly referred to as *bomas*) has been used to create vegetation patches in areas with increasing bare ground. Here, we look at the challenges faced in implementing these interventions, as well as the successes and opportunities associated with them.

Keywords *Acacia reficiens* · *BOMAS* · Community conservancies · *Opuntia* · Restoration

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Introduction

Globally, drylands are a significant land type, covering more than 40% of the land area and support more than 35% of the global human population (Reynolds et al. 2007; James et al. 2013). Pressures for food production in these marginalized lands has increased as a result of rising human populations and is typified by overgrazing as a result of increased stocking rate or reduced forage availability (D'Odorico et al. 2013). Overgrazing leads to land degradation through vegetation composition shifts and reduced primary productivity (MEA 2005; Asner et al. 2004; Kinyua et al. 2010). Increasing bare ground and reduced community resilience often lead to detrimental positive feedback loops wherein gradual modification of edaphic conditions further affects vegetation productivity (van der Merwe and Kellner 1999; Kinyua et al. 2010; Bestelmeyer et al. 2015). Altered edaphic conditions predispose landscapes to invasion by opportunistic species with adaptations to degraded conditions (Alpert et al. 2000; Pierson et al. 2011), and continued grazing is often higher on remaining local species if they are more palatable than invasive species (Ash et al. 2011), leading to shifts in plant community structure. This degradation cycle often creates the conditions necessary for invasive species to become established and out-compete forage species (Strum et al. 2015).

African rangelands are primarily utilized by nomadic pastoral communities through various types of traditional livestock production systems, in addition to supporting a significant portion of the continent's wildlife (Fratkin 2001; Sankaran et al. 2005; Georgiadis et al. 2007). Specifically, in the greater Ewaso ecosystem, which covers the majority of north and central Kenya, complex historical land tenure systems have resulted in a patchwork of private commercial ranches and community-owned conservancies that are characterized by mixed-use between livestock production and wildlife conservation for tourism (Gadd 2005; Georgiadis et al. 2007). Historical overgrazing combined with highly variable rainfall patterns and drought cycles has resulted in significant spread of bare ground patches across the landscape. These bare ground patches are characterized by physical crusts overlying a compacted argillic horizon that has been exposed due to erosion of the surface horizons (Kimiti et al. 2016).

Exposed argillic horizons have a propensity for forming physical crusts and reducing infiltration rates (van der Merwe and Kellner 1999; Kinyua et al. 2010; Kimiti et al. 2016). Soil water infiltration is further reduced because of reduced grass basal cover and homogenous surface soil conditions, resulting in accelerated run-off and susceptibility to gully formation (Pierson et al. 2011). These gullies in turn affect productivity by altering water flow patterns through modifying the soil microtopography and thus reducing available moisture, as well as affecting the soil fertility of the surrounding landscape (Poesen et al. 2003; Valentin et al. 2005; Stavi et al. 2010; Xu et al. 2016).

Native vegetation loss and soil degradation can create alternative ecological states, which provide conditions for opportunistic native or exotic species to invade disturbed areas and further encroach on the remaining native vegetation (MacDougall and Turkington 2005; Suding et al 2004; Strum et al. 2015). In the Ewaso ecosystem, modifications of soil and vegetation have encouraged the spread of *Acacia reficiens* and *Opuntia stricta* (Ward 2005, Strum et al. 2015). *A. reficiens* is a medium-sized indigenous tree that has become dominant in Samburu County and suppresses regrowth of other herbaceous species (Bester 1999; Ward 2005; Vågen and Winowiecki 2014), while *O. stricta* is an exotic invasive species in the cactus family that has become dominant in Laikipia County and has low forage potential for most domestic and wild ungulate species (Kunyanga et al. 2009, Strum et al. 2015). Dominance of these lands by woody species is likely accelerated by reduced fire frequency which in turn is attributed to a loss of fine fuels due to overgrazing and the increasing proportion of bare ground in the system (Roques et al. 2001; Eldridge et al. 2011, 2012).

Control of woody species invasion and reversal of bare ground prevalence have challenged the management of rangeland systems worldwide (Asner et al. 2004; Han et al. 2008; Liao et al. 2008; Pierson et al. 2011; Eldridge et al. 2011). These challenges are exemplified in the Ewaso ecosystem where efforts to reduce woody plant dominance (Kunyanga et al. 2009; Strum et al. 2015; Witt 2015) and restore grassland vegetation (Mugerwa et al. 2009; Kinyua et al. 2010; Porensky and Veblen 2015; Kimiti et al. 2016) have had mixed success. In this paper, we examine the potential factors causing land degradation, the restoration management practices currently used, and the challenges encountered in the Ewaso ecosystem, looking at two counties (Samburu and Laikipia). We then propose new avenues for restoration research and land management in these degraded mixed-use areas.

Historical context

Samburu

Samburu County is part of the greater Ewaso ecosystem in Kenya (Fig. 1). The majority of this ecosystem is mixed-use semi-arid rangeland supporting both cattle and wildlife. Very little literature exists about the historical vegetation patterns of most eastern African rangelands, and even less so about the Samburu area. The few existing literary and oral indigenous sources indicate that vegetation complexes in this area were historically dominated by dryland *Acacia-Commiphora* savannas. For example, Barkham and Rainy (1976) describe the vegetation communities of the then Samburu-Isiolo Game reserve and its surrounding areas, and identify *Acacia tortillis* as the dominant species, accounting for 44% of all tree/shrub cover. Bronner (1990) also describes the lowland savanna areas as being dominated by *A. tortillis* assemblages, but denotes the presence of this species as representing degradation, given that the prior historical vegetation type was likely open grassland savanna. In most of these historical accounts, *A. reficiens* was not identified as a major constituent of the plant community or a particularly problematic encroaching plant, as it was confined only to certain areas of the landscape.

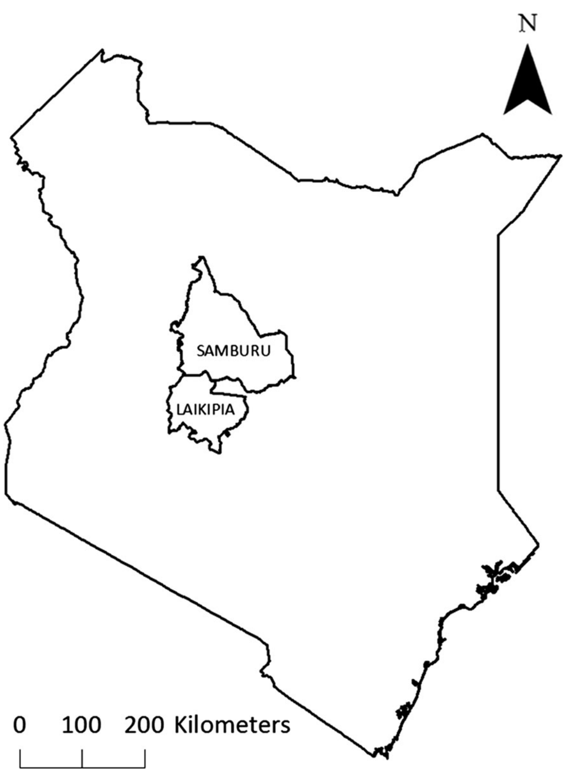

Fig. 1 Map of Kenya showing the location of Laikipia and Samburu counties

Traditionally, resilience of rangeland ecosystems lay in the ability of pastoral communities to move into different areas depending on changing resource availability (Reid et al. 2014). In colonial era Kenya, the British authorities restricted community movements, confining populations into demarcated areas referred to as native reserves (Worthy 1954; Cronk 1989). This was the beginning of a pattern of sedentarization that contributed to the breakdown of traditional coping mechanisms (Galvin 2008, 2009; Reid et al. 2014).

Post-independence Kenya also created an atmosphere in which individual land ownership was increasingly sought after (Okoth-Ogendo 1986; Green 1987). Additionally, migrant communities converted rangelands with higher productivity into agricultural lands, further constricting the available forage base (Ellis and Galvin 1994; Lesorogol 2010). During periods of climatic uncertainty, there has also been a tendency for resource-based inter-community conflict, especially in the less urbanized areas where historically problematic practices like cattle rustling are still part of the cultural identity (Fratkin 2001; Schilling et al. 2012). The resulting conflict and social instability can hamper nomadic movement and increase intensity of use of already scarce land resources (Reid et al. 2014).

Severe droughts in 1984, 1995, 2005, 2006, and 2009 depleted both water and vegetation resources, leading to increased pressure on existing scarce vegetation near persistent water sources (Boruru et al. 2011). When these areas eventually received rainfall, this would have resulted in heavy surface run-off and flash flooding due to reduced vegetation cover, and soil degradation. This in turn would have led to lower infiltration rates, increased run-off velocities and possibly stripping of the existing seed bank, in addition to creation of deep rills and gullies on the landscape (Jones and Esler 2004; Mati 2006).

Climatic stochasticity, alteration of human movements, fluctuation in livestock and wildlife populations, changes in land uses, and increases in land degradation have resulted in shifts of vegetation community compositions across the Samburu study area. There has been a perceived reduction in both cover and diversity of perennial grass species, as well as a concurrent increase in woody vegetation (Schultka and Cornelius 1997; Williams 2002). For example, *A. reficiens* has thrived, taking over a significant amount of grassland, and seemingly outcompeting other more desirable woody vegetation (Schultka and Cornelius 1997; Fig. 2).

Laikipia

Laikipia County (Fig. 1) generally receives higher rainfall than Samburu due to the South–North decreasing rainfall gradient between mounts Kenya and Marsabit and higher elevations (Franz et al. 2010; Georgiadis et al. 2007). Higher rainfall, coupled with many areas of fine-textured soils and cooler temperatures, leads to higher available water holding capacity (Franz et al. 2010) and consequently greater vegetation productivity. Before 1900, accounts from British explorers to Laikipia reported mostly open grassland plains with sporadic woody vegetation cover. Interviews with local land owners and analyses of aerial photos from 1961 and 1962 combined with ground surveys from 1997 indicated an apparent increase in shrub cover from near zero to close to 30% within half a century (Augustine and McNaughton 2004).

Historically, Laikipia was settled by the nomadic Laikipiak and Purko-Kisongo peoples beginning at least in the 1800s, whose numbers were greatly reduced by disease and warfare by the 1900s (Young et al. 1995). Between 1914 and 1915, the British engineered a treaty with these peoples to move them off the land and open the areas up for European settlement (Herren 1987; Young et al. 1995). Most of Laikipia was then divided into commercial private ranches owned by Europeans, a land tenure system which persists today, with most ranches still being privately owned by descendants of the original European settlers (Georgiadis 2010). The conversion of these open pastoral and wildlife plains into

Fig. 2 Photo of an area on Kalama conservancy encroached by *Acacia reficiens*. Note lack of herbaceous understory and general prevalence of bare ground

commercial ranches likely facilitated changing vegetation communities primarily through restricted grazing patterns, suppression of wildfires, and restriction of herbivore movements through fencing (Boughton et al. 2016; Ford et al. 2016; Russell and Ward 2016). Additionally, several indigenous community group ranches characterized by shared ownership exist and are often more degraded than private ranches due to higher stocking rates of domestic animals and continuous grazing (Livingstone 1991; Shackleton 1993; Georgiadis et al. 2007). However, the formation of centrally governed, mixed-use conservancies on some of these properties has possibly provided an avenue for better grazing management and habitat restoration (Okello et al. 2009; Sundaresan and Riginos 2010; Measham and Lumbasi 2013).

Laikipia has experienced increasing drought frequency and greater intensity of storm events similar to Samburu (Huho et al. 2010; Boruru et al. 2011; Ayeri et al. 2012). The influx of large browsers, specifically elephants leaving Samburu due to decreasing food security, has possibly influenced vegetation composition (Thouless 1995). Experiments with animal exclosures on this landscape have shown that large browsers concentrate on palatable broad leaved plant species that lack effective defensive structures like long thorns, leading to increased dominance by narrow leaved thorny *Acacia*s, primarily *Acacia etbaica* (Augustine and McNaughton 2004).

Increase in woody species dominance and reduction in relatively continuous grass cover, coupled with increasingly variable rainfall frequency and intensity, leads to a system that is less diverse, more sensitive to disturbance events, and less likely to recover from disturbance (Eldridge et al. 2011; Ratajczak et al. 2012; Alofs and Fowler 2013). In addition, the reduced forage base and increasing inter-community armed conflict elsewhere in the Ewaso ecosystem has led to Laikipia ranches experiencing an influx of pastoral livestock herds from less productive areas; primarily Samburu, Turkana, and Pokot (Bond 2014). This increase in grazing pressure during drought periods - when the ecosystem is already vulnerable- has possibly led to catastrophic transitions, resulting in novel stable states characterized by large patches of bare ground (Rietkerk et al. 2004) that are vulnerable to colonization by opportunistic invasive species like *O. stricta* (Strum et al. 2015; Fig. 3).

Types of degradation

Degradation of rangeland resources can include loss of soil and vegetation resources, as well as spread of exotic or unwanted plant species (Snyman 2003; James et al. 2013). Mixed-use rangelands in northern Kenya are facing these concerns, and efforts to reverse their degradation are ongoing. In this section, we shall describe how degradation in the Ewaso ecosystem has manifested as bare ground spread and gully formation, spread of *Acacia reficiens*, and spread of *Opuntia stricta*.

Increasing bare ground cover and gully formation

Degradation of land resources in eastern and southern African rangelands commonly manifests as spread of bare ground (e.g., land devoid of perennial herbaceous plants and typified by reduced infiltration and increased surface run-off), which can lead to erosion of soil substrate, nutrients, and minerals (van der Merwe and Kellner 1999; Kinyua et al. 2010). Many areas are characterized by deep gullies, while others have a loss in A horizon, which exposes the clay-rich argillic B horizon and can be susceptible to induration (van der Merwe and Kellner 1999; Beukes and Cowling 2003; Kimiti et al. 2016; Mukai 2016).

If the primary drivers of degradation are mitigated before a threshold is crossed, such as the loss of surface soil, some landscapes can recover to previous levels of plant productivity and diversity (Verdoodt et al. 2010; Park et al. 2013; Mureithi et al. 2014). In some instances, however, abiotic changes occur that cannot be reversed simply through rest and autogenic recovery (Bestelmeyer et al. 2015). Restoration efforts in these areas must initially focus on repairing damage to the edaphic characteristics of the landscape, which often requires mechanical intervention through various practices, such as ripping compacted soils, constructing soil erosion barriers, and ameliorating surface soils through input of organic matter (Sankaran and Anderson 2009; Kinyua et al. 2010).

Acacia reficens encroachment

In large parts of Samburu, increasing disturbances and subsequent alterations in edaphic conditions (e.g., loss of topsoil and nutrient resources), combined with increasingly irregular weather conditions, have

Fig. 3 Photo of area on Mpala conservancy heavily colonized by *Opuntia stricta* (in the foreground) including in the inter canopy spaces between *Acacia xanthophloeia* and *Acacia tortillis* stands (background)

worked in concert to create a conducive environment for the spread of the undesirable plant species *A. reficiens* (Alpert et al. 2000). This indigenous tree has historically existed in the less productive sections of the landscape in low proportions, with local community members observing a spread in cover over the last two to three decades (Schultka and Cornelius 1997; Lelukae, pers. comm.).

A. reficiens spread is associated with a lack of persistent herbaceous understory growth (Kimiti pers. observ.). The exact mechanism of this exclusion is not well understood, but studies of invasive *Acacia* species in other ecosystems have pinpointed microhabitat modification, soil property changes and to a lesser extent, allelopathy (Marchante et al. 2008; Lorenzo et al. 2016). Because *A. reficiens* forms a monoculture, large areas that seem healthy from a superficial 'vegetation cover versus bare ground' perspective are in effect still degraded since there is no herbaceous cover under the trees to provide forage to grazers and minimize run-off velocity associated with soil erosion.

By colonizing these large tracts of community land, *A. reficiens* has also increased degradation in the rest of the landscape by increasing pressure on existing forage resources for wildlife and livestock through overgrazing remaining grassland areas. This leads to a destructive positive feedback loop that in turn creates more substrate for *A. reficiens* to occupy (Keane and Crawley 2002; Mitchell et al. 2006).

Opuntia stricta encroachment

In many degraded areas, altered disturbance regimes present an opportunity for plants that are superior competitors to become more prevalent and persist in alternative stable states (Strum et al. 2015). In the Laikipia section of the ecosystem, this is visible in the spread of *Opuntia stricta*, an exotic species introduced by the British settlers during the colonial era in the middle of the twentieth century (Strum et al. 2015). *O. stricta* can reproduce both sexually and vegetatively, giving it an advantage over many indigenous plant species (Mandujano et al. 1997; Padrón et al. 2011).

The Laikipia Plateau hosts extremely high native biodiversity (Kinnaird and O'Brien 2012), including numerous potential seed dispersers and pollinators that can aid reproduction and spread of *O. stricta* across the landscape (Padrón et al. 2011). Central and northern regions of Laikipia are extremely water limited for much of the year (Georgiadis et al. 2007; Kinnaird and O'Brien 2012), and the adaptation of *O. stricta* to arid environments allows it to survive and produce nutritious fruit through the dry season (Strum et al. 2015), making it particularly attractive to native wildlife. Olive baboons (*Papio anubis*) are found to be especially important dispersers of *O. stricta* seeds (Foxcroft and Rejmánek 2007; Strum et al. 2015).

The result is that *O. stricta* has spread across substantial portions of the Laikipia ecosystem. This has had a detrimental effect on the livestock production systems that are the major land use in this area. Camels and cattle periodically experience eye punctures while consuming *O. stricta* (Littlewood, pers. comm.), and some Maasai communities are losing up to one-third of their goats annually to mortalities from ingestion (Imerinyi, pers. comm.). Raw *O. stricta* paddles also contain oxalates, which act as laxatives and block nutrient absorption when consumed in large quantities (Stintzing et al. 2005).

Current restoration efforts

Control of bare ground spread and gully formation

In Laikipia, most attempts to combat and reverse bare ground have mainly been experimental or at plot scale, with several pilot projects looking at different interventions. The use of tractors for ripping up hard physical crusts has shown promise, but presents inherent cost restrictions, as most community-owned properties, do not have the resources to buy or rent heavy machinery, unless with the aid of outside donors. Some research has also been done using small (2 m length) soil erosion barriers spread across the landscape to modify the micro-topography of the landscape, act as resource traps, and initiate vegetation patch formation following the example of Ludwig and Tongway (1996). The success of the pilot phase of this project led to it being upscaled on 10 hectares on the Mpala Conservancy in Laikipia (Kimiti et al. 2016). In Tiemamut and Koija community conservancies, grazing exclusion plots were set up in the year 2000 through an African Wildlife Foundation restoration program. After 10 years of exclusion, ungrazed areas showed significant reductions in bare ground, and increases in grass cover and biomass relative to continuously grazed zones (Mureithi et al. 2014).

The use of mobile cattle enclosures, commonly referred to as 'bomas' demonstrate the ability to create vegetated patches in degraded areas (Porensky and Veblen 2015). Bomas modify the soil surface texture and add nutrients into the soil, creating fertile patches that are frequently colonized by species of the *Cynodon* genus (Porensky and Veblen 2015). Research in Laikipia has shown the importance of these bomas as islands of productivity that attract wildlife and can persist for centuries (Young et al. 1995; Porensky and Veblen 2012; Veblen 2012). In some parts of Westgate and Kalama community conservancies, old bomas are some of the most vegetated areas in a matrix of bare ground (Kimiti, pers. observ.).

Continuous, long-term grazing may be detrimental to palatable grass species recovery (Verdoodt et al. 2010; Park et al. 2013); thus, bomas need to be moved often to minimize impact on existing vegetation and localized animal impacts associated with overgrazing like soil compaction and trampling of existing vegetation (Veblen and Porensky 2011). This presents a time and labor cost implication which would likely limit their efficiency. However, their effectiveness in reestablishing vegetation patches throughout the landscape, although limited in singular spatial extent, should not be discounted (Porensky and Veblen 2015).

Check-dams are structures that capture sediment in gullies as water moves through, trapping soil, seed, and vegetative material that promotes plant establishment (Xiang-zhou et al. 2004). As the gullies fill up with vegetation and sediment, they become increasingly efficient at slowing down water movement and halting erosion, essentially 'healing' themselves (Valentin et al. 2005; Xiang-zhou et al. 2004). Modified, small-scale run-off mitigation and debris catchment efforts using branches have been successful in some systems (Ludwig and Tongway 1996). However, in places where the surrounding landscape is heavily eroded, high run-off velocities often destroy these barriers and reduce their effectiveness. For example, in the Westgate community conservancy, efforts to arrest and fill in gullies using sand bags and debris that act as modified check-dams have often failed, with several of these barriers collapsing after every heavy rainfall event (Kimiti, pers. observ.). In this case, interventions aimed at increasing ground cover in the surrounding areas will play a key role in the future success of gully control efforts (Valentin et al. 2005).

Control of *Acacia reficiens*

Small-scale attempts at clearing *A. reficiens* from Samburu rangelands have been undertaken since the extent of the problem became evident in the early 2000s (Kimiti, pers. observ.). However, concerted and directed efforts began with a clearing pilot project initiated by the Grevy's Zebra Trust (GZT 2016) on the Westgate Community Conservancy in 2009. Approximately 238 hectares of land was cleared of *A. reficiens* and reseeded with *Cenchrus ciliaris*, a perennial bunched grass that has been successful in reseeding trials in other parts of Kenya (Mureithi et al. 2015; Lugusa et al. 2016). *C. ciliaris* has been used for reseeding on East African rangelands since the 1960s and is one of the three most common species for reseeding in Kenya along with *Eragrostis superba* and *Enteropogon macrostachyus* (Pratt 1963; King and Stanton 2008; Mureithi et al. 2014; Mganga et al. 2015).

The most effective strategy for removing A. reficiens from the landscape is tree clearing at the height of the dry season to avoid coppicing, and subsequently scouring the hard ground and broadcasting seed right before the beginning of the rainy season (Kimiti et al. forthcoming). Additionally, the cleared *A. reficiens* crown material was placed on the reseeded ground to protect the seed and help trap soil material eroding from upslope bare ground areas. Cleared tree material was also placed in gullies to help slow and ultimately reverse erosion.

Within a few months, perennial grass and forb species colonized this pilot area, and the cleared stands of *A. reficiens* did not show signs of coppicing or growing back through seedlings (Kimiti, unpublished data). The Westgate community grazing planning committee was then able to begin using the cleared area as a grazing resource once more and set up a program to harvest *C. ciliaris* seeds, either using them to reseed other areas or selling them to other communities that were interested in carrying out reseeding interventions. After the visible impact on grass cover and livestock production from this pilot project, similar operations were undertaken under the aegis of the Northern Rangelands Trust (NRT 2016) rangelands management program in other parts of Westgate conservancy, and on other conservancies in Samburu and Isiolo counties that had identified *A. reficiens* as a problem species.

Control of *Opuntia stricta*

Regions in which political infrastructure, regulatory enforcement, and/or budget availability are unreliable can be especially susceptible to biological invasions, due to the difficulty of coordinating control measures across the entire affected area (Odom et al. 2005). *Opuntia stricta* control is conducted through manual labor and time intensive mechanical and chemical interventions. Application of herbicides and manual uprooting showed initial promise, but herbicides are expensive and their effect on wildlife has not been clearly tested. Additionally, mechanical uprooting is labor and time intensive, and is not realistic for properties whose management cannot afford to purchase or hire the equipment required. The dual reproductive modes of *O. stricta* (del Mandujano et al. 1997; Padrón et al. 2011) have rendered many of these efforts ineffective; their ability to produce clones from severed paddles means that mechanical removal often furthers regeneration and spread of the plant. In addition, *O. stricta* seeds concealed below the soil surface can be viable for up to 20 years, and some *Opuntia* species can maintain viability for up to 15 years, meaning that removal attempts may only be effective in the short-term (Dodd 1940).

Beginning in late 2014 and early 2015, a cochineal insect that acts as a predator to *O. stricta* in its native range, *Dactylopius opuntiae*, was introduced to Laikipia. Although quantitative results about the effectiveness of this control agent have not yet been published, the insect has spread quickly across the landscape and is killing large volumes of *O. stricta* across Laikipia (Littlewood pers. comm.; Imerinyi pers. comm.; Weller pers. comm.). The continued effectiveness of *D. opuntiae* as densities of *O. stricta* decline has yet to be determined. In Kruger National Park, introduction of *D. opuntiae* was effective in thinning existing densities of *O. stricta*, but not in eradicating the plant or stopping its spread across the landscape (Foxcroft et al. 2004).

Monitoring and evaluation of restoration success

Few restoration projects globally collect quantitative data showing the success or failure of their interventions (Hardegree et al. 2011), and Kenya is no exception. These types of assessments are mainly carried out in areas where restoration efforts are being undertaken as part of a scientific experiment or case study (Kinyua et al. 2010; Mureithi et al. 2014; Kimiti et al. 2016). From a livelihoods and management stand point, most community and private land managers rely heavily on rapid, qualitative assessments to make time-sensitive decisions (Pyke et al. 2002). In heavily degraded areas in the Ewaso ecosystem where there is very little perennial vegetation cover, a very low threshold for success is generally used, with any visible and usable increase in grass cover accepted as 'success.'

In the case of invasive species control, most evaluations are also qualitative, with rudimentary presence–absence visual assessments being the most common assessment method in the Ewaso ecosystem. Most clearance and reseeding projects visited did not have any control plots or historical quantitative data to show, and the main monitoring tool (where one existed) was photographic monitoring at plot level. These visual photographic comparisons can be misleading if the images under comparison were collected at different times during the growing cycle or in different seasons (Hall 2001; Kull 2005; Pupo-Correia et al. 2014). However, when done in the correct corresponding seasons, and when properly geo-referenced, they can be a useful rapid assessment tool that land managers in this ecosystem can use to inform their planning choices (Hall 2001; Webb 2010; Western 2010).

Remote sensing as a tool for monitoring restoration success has not been explored extensively in the Ewaso ecosystem, either in the form of satellites or aerial photography. As early as 1990, rangeland management institutions in Kenya were considering the use of satellite products, including NDVI, for monitoring vegetation. These early assessments showed promise in large scale monitoring, but there were doubts about their usefulness at the functional group scale, let alone species level (Ministry of Livestock Development 1991). An assessment of the usability of NDVI values in this landscape showed a high level of error and uncertainty and cast doubt on the feasibility of this method (Ritchie unpublished report, USAID). However, if high-resolution datasets were available for image analysis, this might allow a relatively rapid assessment of vegetation changes after restoration (Wiesmair et al. 2016). Additionally, resolution of legal roadblocks governing the use of unmanned aerial vehicles could provide another useful avenue for photographic monitoring (Breckenridge and Dakins 2011; Schiffman 2014; Mukwazvure and Magadza 2014).

Additionally, when post hoc control plots are set up, it allows a 'space for time' comparison that can be used to show results and provide quantitative data that are useful for future management. However, this requires very careful determination that land potential, as defined by initial biophysical conditions, including soil, topography, and climate, was nearly identical at the time the treatment was initiated, and that weather was similar throughout the treatment period (Herrick et al. 2013).

In 2015, the USAID funded and USDA led Land Potential Knowledge System (LandPKS) project began testing the use of simple mobile phone applications in the inventory and assessment of African rangelands (Herrick et al. in review, Landpotential.org). LandInfo, which is a site characterization mobile application (app), was used to describe selected treatment sites, and provided the data for biophysical matching with candidate control plots. The LandCover mobile app uses a modified line intercept method based on Riginos and Herrick (2010) and was used to collect data on various vegetation metrics (Kimiti et al. forthcoming, http://landpotential.org/landpks.html).

Using these mobile apps in areas under the Northern Rangelands Trust umbrella system, post hoc treatment and control plots for large-scale clearing and reseeding projects in the Westgate and Kalama conservancies were identified, matched and assessed. These plots allowed the first quantitative assessment of these restoration interventions, and demonstrated clearly the differences in treated and untreated areas.

For instance, in Kalama conservancy, the visual differences in vegetation cover between treated and untreated sites were visually demonstrable (Fig. 4), but our rapid assessment tools allowed for quick quantification and subsequent generation of rudimentary summary graphs (Fig. 5) and descriptive statistics. Efforts are ongoing to further improve the quantification and reporting of restoration outcomes in the region, especially in areas without long-term presence of academic and scientific research.

Future of rangeland restoration in East Africa

East African savannas are projected to experience more frequent and intense droughts over the next 50 years (Solomon et al. 2007). As such, pressure on land resources will continue to increase, and a combination of best management practices and rehabilitation of degraded land is necessary to mitigate the effects of shrinking grazing resources. Current emerging knowledge about restoration in this landscape needs to be disseminated and tested at larger scales to move the conversation from empirical scientific research into verifiable and assessable management interventions.

In Laikipia, *O. stricta* is expected to continue increasing at the expense of native plants (Snyman et al. 2007). Experimentation and monitoring of *D. opuntiae* as a biological control agent will be necessary to determine its implications for management and effect on native species. Careful disposal of mechanically removed plants, monitoring and removal of seedlings

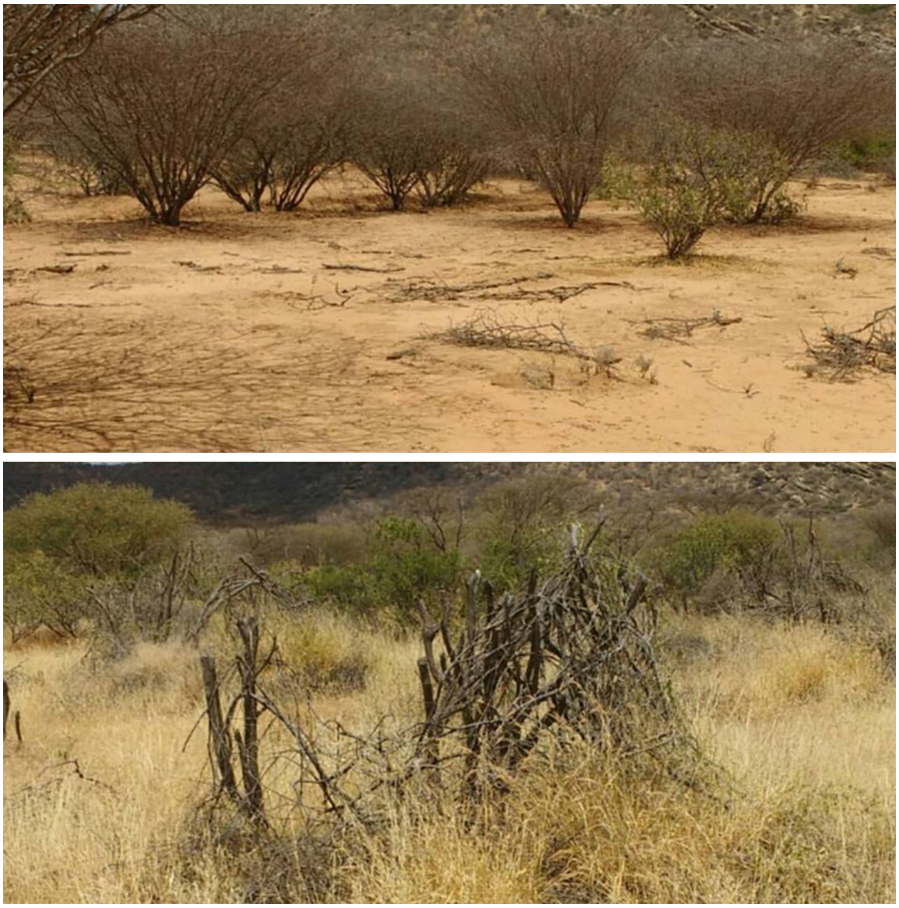

Fig. 4 Photo of plots on Kalama conservancy, Samburu. Both photos taken February 24th 2016. *Photo on the top* was taken in the control zone, where no clearing of *A. reficiens* or reseeding of *Cenchrus ciliaris* had taken place. *Photo on the bottom* was taken on the same day in the treatment zone, where clearing had taken place 2 years before. Most of the grass in the foreground is *C. ciliaris*, while the boundary of uncleared *A. reficiens* is visible in the background

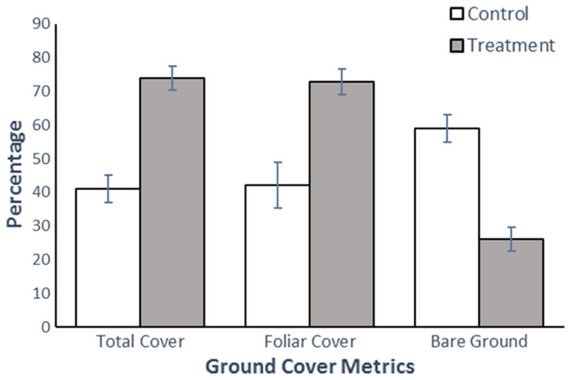

Fig. 5 Graph showing summary findings from data collected using the LandCover mobile application, based on a modified line-point intercept method (Herrick et al., in review), indicating differences in select vegetation metrics between treatment and control plots. Metrics are total ground cover and total foliar cover, which are significantly higher in the treatment areas, and simple bare ground cover, which as expected, is higher in the control areas. Total cover $t = 5.6905$, df = 13, $p < 0.001$. Foliar Cover $t = 3.8377$, df = 13, $p = 0.002$. Bare Ground $t = -5.6905$, df = 13, $p < 0.001$ (Data from Kimiti et al., forthcoming)

from previously infested areas, and efforts to remove fruits from adult plants (limiting dispersal of seeds by animals) should all be employed to mitigate future spread, as well as ecological and economic impacts, of *O. stricta*.

In Samburu, hand clearing of *A. reficiens* stands and reseeding with *C. ciliaris* has proven to be an effective management option (Kimiti et al. forthcoming), though its application is not yet widespread and may be useful in improving other affected areas. A larger number of grasses should be evaluated for inclusion in the reseeding process to increase probability of establishment as well as diversity of perennial grass composition, focusing not only on reducing invasive species, but restoring ecosystem function (Monaco et al. 2016). More research is also needed to understand the specific mechanisms through which *A. reficiens* outcompetes other species, as emphasis shifts from reactionary reclamation to preventative ecological-based invasive plant management. A greater understanding of fire and grazing, and their role in maintaining ecosystem mosaics, is necessary for conservation at a landscape scale (Fuhlendorf et al. 2012).

While a lot of small scale studies have been carried out on fire ecology in the area (Okello and Young 2000; Sensenig et al. 2010; Kimuyu et al. 2014), fire has not been used as a management tool since the 1960s, and most ranchers actively suppress wildfires (Augustine and Mcnaughton 2004; Goheen et al. 2013). Few studies have looked at the cumulative or interactive effects of changing fire regime and herbivore compositions and densities on invasive species encroachment, especially as elephant populations migrate across the system (Thouless 1995; Larson et al. 2013; Kimuyu et al. 2014; Pringle et al. 2015).

Frameworks for suitable rapid and cost-effective assessment and monitoring of restoration projects should be established in order to encourage long-term monitoring by individual land managers (Ruiz-Jaen and Mitchell Aide 2005; Herrick et al. 2006; Klintenberg et al. 2007; Wortley et al. 2013). Each restoration project should be assessed individually based on soils, topography, and climate, as well the type of intervention being carried out (Ruiz-Jaen and Mitchell Aide 2005). The criteria by which each project is deemed a success or failure should also be decided upon in the restoration planning stage to facilitate better evaluation upon completion.

Finally, efforts to reclaim and rehabilitate degraded landscapes in the Ewaso ecosystem and African rangelands in general may be improved through identification and mitigation of causative factors. Grazing system plans and drought preparedness are necessary for effective management of mixed-use properties, especially for community conservancies that rely on both livestock production and wildlife management as a subsistence livelihood system (Kinnaird and O'Brien 2012; Measham and Lumbasi 2013). Maintaining peaceful coexistence among the various land managers will also be imperative, as resource-driven inter-community conflict has historically disrupted management and restoration plans, and exacerbated pressure on natural resources (Berger 2003; Campbell 2009; Schilling et al. 2012). Multiple agencies and land management groups have begun developing, testing, and/or implementing management and restoration interventions to combat land degradation in the Ewaso ecosystem. These efforts should be concerted in order to maximize potential knowledge outputs, improve the quality of recommendations for land managers, and mitigate the complex suite of problems facing this landscape.

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