Increased pastoralist livestock mobility is associated with large-scale rangeland restoration and soil carbon sequestration

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Abstract

Semi-arid rangelands cover 40% of the earth's land surface, but their ecosystem services have declined due to, among other factors, increasingly sedentary livestock husbandry by pastoralists. Such degradation might be reversed by adopting frequent, large-scale livestock movements similar to those associated with traditional nomadic pastoralist habits but data to support this hypothesis is lacking. We report here the consequences of implementing a program to increase the mobility of livestock grazing across 2 million ha in northern Kenya from 2014 - 2021, as compared to conditions prior to the program, 2001- 2013. Despite increased human populations and livestock numbers on the study area during the program, nearly 60% of 213 sampled locations experienced the single major grazing events per year intended by the grazing program in at least 6 out of 7 years, with accompanying rainfall-corrected increases in forage biomass and soil carbon sequestration. Locations where mobile grazing was less consistently applied experienced neither of these responses. We demonstrate that increasing mobility in grazing practices within large-scale pastoralist systems can be implemented to improve range condition and soils and consequently sequester substantial CO_2 in soil organic carbon.

Introduction

Rangelands cover approximately 40% of the earth's land surface (Bai and Cotrufo 2022) and have historically served as the basis for livestock husbandry and habitat for the world's large mammal megafauna (Fynn and Bonyongo 2011). For millennia, humans have coexisted with wildlife and engaged the landscape as nomadic pastoralists or agro-pastoralists, and this remained true in many portions of the world until a few decades ago (Mbow et al. 2019). Since the mid-20th Century, the accelerated provision of localized services by government and markets, along with increased population size, have incentivized the abandonment of nomadic lifestyles by pastoralist communities. Settlement by pastoralists has been shown to reduce seasonal movements and access to green vegetation, which result in support of lower livestock densities (Peacock 1987, Ole Seno and Tome 2013).

Decline in livestock mobility and herding from more permanent residences results in repeated use of grazing areas during the plant growing season and significant ecosystem impacts. Such impacts include the consequent loss of perennial forage, increase in bare ground, soil erosion and loss of soil organic layers, disruption of the hydrologic cycle and the loss of surface water sources for people and livestock (Savory and Butterfield 2010, Teague et al. 2011, Ritchie 2020, Ritchie and Penner 2020, Apfelbaum et al. 2022a). These ecological changes induce a cycle of impoverishment in human livelihoods, a persistent and dramatic decline in productivity, and increased competition among livestock and wildlife for forage and water (Young et al. 2005, Western et al. 2009a, Western et al. 2009b). Coupled with increases in human population size and total livestock numbers, the outcome of sedentary, continuous grazing is often a decline in ecosystem services such as soil fertility and carbon sequestration, primary production, water availability, and livestock production per household, as well as biodiversity (Haro et al. 2005, Glew 2012, Ogutu et al. 2016).

Given that sedentary livestock husbandry practices are likely a major contributor to rangeland degradation (Western et al. 2009a, McSherry and Ritchie 2013, Turner et al. 2014, Dlamini et al. 2016, Kinuthia and Wahome 2019, Kim et al. 2023), there is intense interest in identifying alternative management choices that could reverse this trend (Duffy et al. 2022, Li et al. 2023). Semi-arid rangelands such as savannas in East Africa are a focus of these activities because they harbor the last remaining populations of many "charismatic" large mammal species and people engaged in traditional pastoral livelihoods. These savannas also hold the potential to remove and store globally important amounts of greenhouse gases (Dobson et al. 2022, Rui et al. 2022).

Livestock management focused on mimicking the migratory movements of wildlife is a commonly proposed solution to rangeland degradation driven by grazing (Reid and Ellis 1995, Kinuthia and Wahome 2019, Phukubye et al. 2022). The concept is that regular movements of large herbivores allow plants the opportunity to recover or compensate for the offtake of biomass while plant resources, namely water, are still available. Sufficiently intense biomass removals by herbivores maintain plants in an exponential growth phase over the growing season, thereby increasing total carbon assimilation and primary production over the growing season. Known by various labels as "holistic," "rotational," "short-duration high density," "adaptive multi-paddock," etc., mobile grazing may be a tool of restoration and sustainable land use (Menke 1992, Chaplot et al. 2016, Provencher et al. 2023), but this outcome is controversial and has not been demonstrated at the scale at which pastoralist grazing systems operate (Robinson et al. 2017). Due to a lack of persistent, landscape-level evaluation, the debate persists as to whether more mobile livestock herds produce the hypothesized beneficial effects.

A key barrier to changing practices across entire landscapes is the financial investment needed to support education and training of herders to adopt more mobile grazing practices, improved governance to support implementation, and sufficient monitoring of livestock movements to document these changes. Potential options to incentivize change may exist via "nature-based" financial incentives in the form of shared revenues to communities from tourism and/or from carbon markets. In the latter case, shifts from sedentary to mobile livestock management practices are viewed as additional activities that may sequester carbon and therefore be converted into saleable carbon credits. However, there are virtually no test cases of whether such financial incentives can motivate community-wide re-adoption of more traditional highly mobile grazing practices. Thus, the idea that nature-based finance can stimulate sustainable community use of rangelands remains highly uncertain.

Our study evaluates for the first time, to our knowledge, the success and impact on key range condition metrics from implementation of mobile grazing practices across a traditional pastoralist landscape. Here we report the outcome of a program of planned, mobile grazing in 13 different pastoralist community areas featuring at least six different ethnic groups, across a large (~ 2 million ha) traditionally migratory grazing system in northern Kenya (Fig. 1). We label the objective of the program as “rapid rotational grazing” or RRG, to emphasize the intention for livestock to aggregate and then move frequently. Achievement of this objective is manifested in a single one-month period of livestock visitation and reduction in forage biomass at a location each year. To assess achievement of a single grazing event, we first developed predictive monitoring tools using satellite-based MODIS imagery to assess short-term impacts of livestock on forage biomass as calibrated by direct recording of livestock locations by human observers. We then determined whether communities effectively implemented single annual grazing events within their community conservation lands, designated as “conservancies.” We then measured the consequences of changes in livestock management by comparing key metrics of green vegetation cover (i.e., satellite-based NDVI) and soil carbon as modeled with a simple process-based soil carbon dynamics model SNAP (Ritchie 2014) and as measured by sequential soil carbon surveys initially either in 2012 or 2016 and then again in 2021.

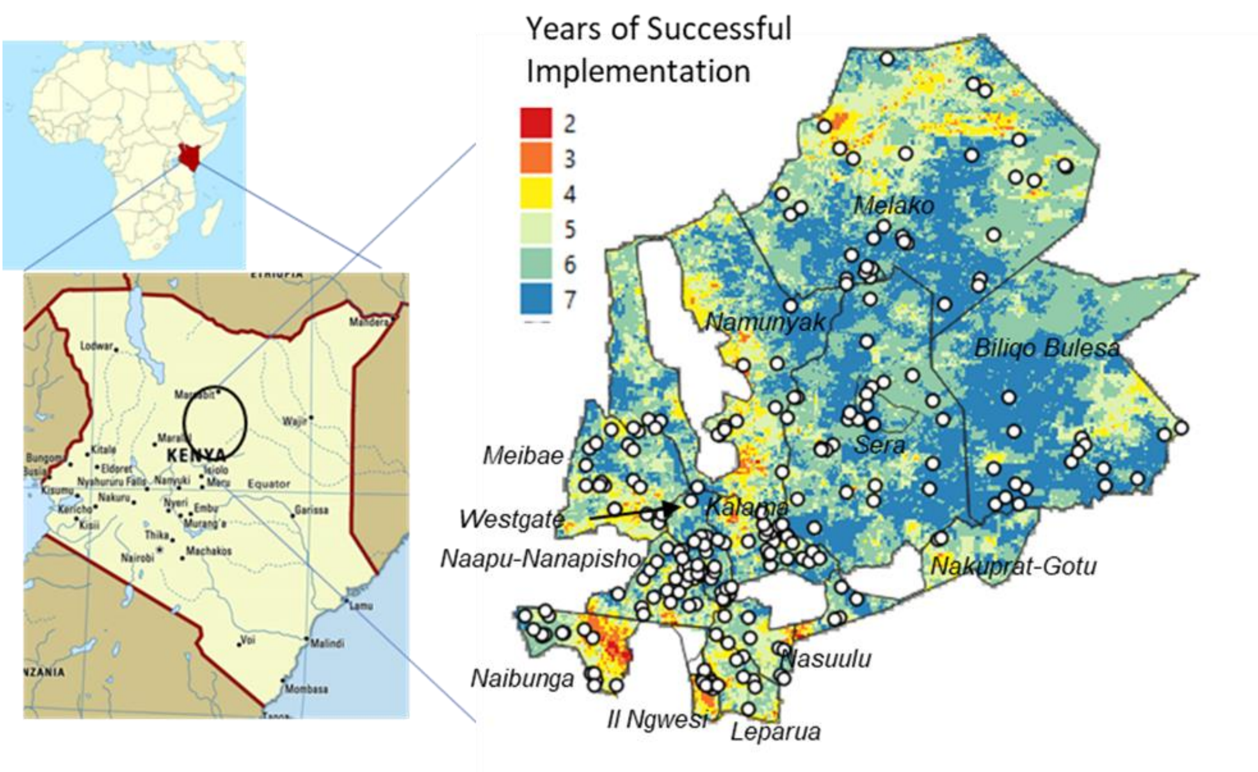

Figure 1. Location of the improved grazing management project in northern Kenya., the 13 communities (conservancies) that participated, the spatial distribution of the annual frequency with which short duration high density grazing was successfully implemented (number of years of successful implementation out of 7 years) and the distribution of 213 soil and vegetation sampling points (white circles).

Results

Local communities organized herder activities (the grazing program) to implement rotations of livestock among 8-12 grazing blocks within their associated conservancies in the wet season and coordinated long-distance (50-200 km) migrations to and through dry season ranges in Sera, Melako, and Biliqo Bulesa conservancies (Fig. 1, **Supplemental Information** Fig.S1). Implementation of the grazing program occurred through community-led governance mechanisms, featuring an elected grazing committee within each community, community meetings to discuss options, and penalties for non-compliance coupled with regional meetings among leaders of the different communities.

Dynamics of rangeland degradation drivers

Three key potential drivers of rangeland condition, and, by proxy, soil carbon, were human population sizes, livestock numbers and rainfall. Annual rainfall averaged across the study area varied considerably among years, with dry (below-normal rainfall) years in 2005, 2007-2009, 2016-2017 and 2021 (Fig. 2A). Annual rainfall amounts varied even more strongly across space, from near 900 mm in the southwest to near 200 mm in the north and northeast (Fig. 2B **Supplemental Materials**, Fig. **S1,**). Over the entire study period rainfall showed no significant trend with year ($R^2 = 0.036$, $N = 20$, $P = 0.425$). However, during the grazing program period 2014-2021, well above average mean annual rainfall occurred during 2018-2020 (Fig. 2A, B). Rainfall became increasingly spatially unpredictable, as rainfall in some drier conservancies (e.g., Biliqo Bulesa) exceeded rainfall in normally wetter ones (Fig. 2B) in both 2016 and 2020, while such switches did not occur during 2001-2013.

Household numbers increased by nearly a factor of 2.5 over the study period 2004–2021 (Fig. 2C). While livestock numbers are often proportional to number of households, high mortality rates, particularly of cattle and donkeys during the drought of 2009-2010, led to declines in cattle numbers following the drought, but cattle numbers have increased after 2013 to pre-2009 levels by 2020 (**Extended data Fig. S2**) In contrast, sheep and goat numbers were not strongly affected by the 2009-2010 drought and have increased steadily during the study period (**Extended data Fig. S2)**. As an estimate of changes in total livestock numbers weighted by the average forage demand for each species, tropical livestock units (TLUs) increased during 2001-2008 but showed sharp drop immediately following the 2009-2010 drought (driven by high cattle mortality) followed by recovery to pre-2009 levels after 2014 (Fig. 2D).

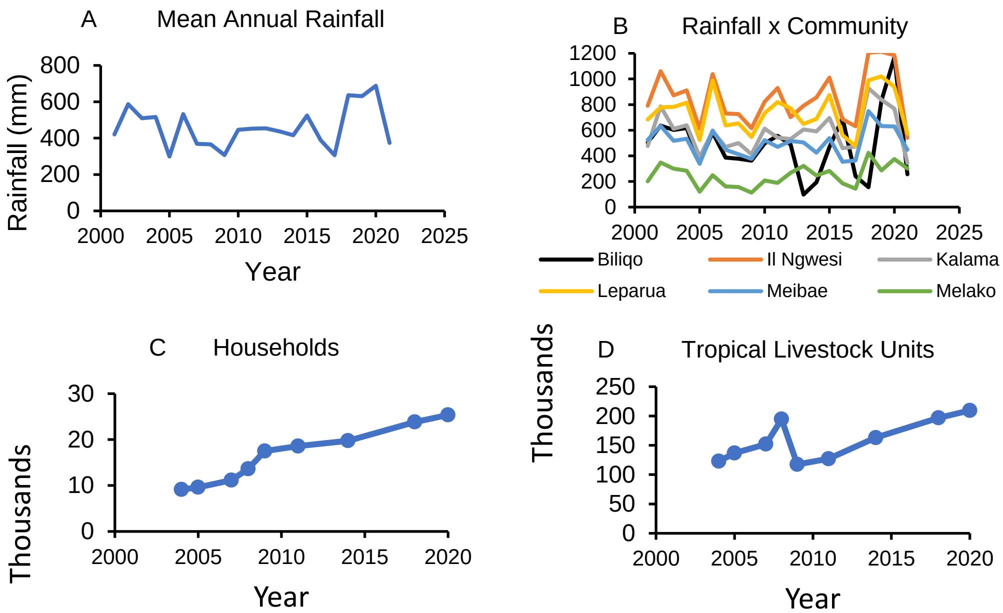

Figure 2. Dynamics in the study area of three key drivers of rangeland condition common in semi-arid climates: A. mean annual rainfall, B. spatial variation in rainfall, C. estimated human households in the study area, and D. estimated tropical livestock units (total numbers of livestock animals weighted by the relative forage consumption for each species).

Grazing management

NDVI was calibrated to actual biomass measures during July and August 2014 (**Extended data** Fig. S4) at both the study area (Kenya) and in Serengeti National Park (Tanzania). Data from Serengeti, a tropical savanna grazed by wild herbivores, were added to increase the range of NDVI and biomass values in assessing the calibration. Across both study areas, NDVI was strongly correlated with green forage biomass (R^2 = 0.77, N = 63, P < 0.0001) but was also strongly correlated within study areas: northern Kenya (R^2 = 0.596, N = 29, P < 0.0001); Serengeti (R^2 = 0.637, N = 34, P < 0.001). The slopes and intercepts of the two separate regressions were nearly identical and not significantly different (**Extended data** Table S1). Thus, NDVI was shown to be a reasonable measure of green forage biomass.

Based on these calibrations, the relative change in NDVI, corrected for weather-influenced changes in NDVI (see **Methods)**, $\Delta NDVI_{rel}$, strongly and significantly corresponded to reported presence or absence of livestock at each of 130 sample point x month combinations in May-July 2014 and January–May 2017 (**Extended data** Fig. S4A). Further, the $\Delta NDVI_{rel}$ value of 0.05 $\pm$

0.015 SE was the threshold at which the probability of livestock presence was > 50%. Thus, for a given sample point *i*, a month *m* with $\Delta NDVI_{rel,i,m} > 0.05$ was considered indicative of a significant livestock grazing event. Effective implementation of grazing corresponded to a single significant grazing event during the year. The occurrence of one or fewer grazing events at a sample point *i* in year *Y* was determined by summing the $\Delta NDVI_{rel,i,m}$ for each month over the year (see **Methods**) to obtain $SUM\Delta NDVI_{rel,i,Y}$. The number of months for which $\Delta NDVI_{rel,i,m} >$ 0.05 was determined for a subset of sample points in 2014 and 2017 and a threshold of $SUM\Delta NDVI_{rel,i,Y} > 0.05$ was associated with failure to successfully implement desired grazing management (two or more grazing events in a year) (**Extended data** Fig. S4B).

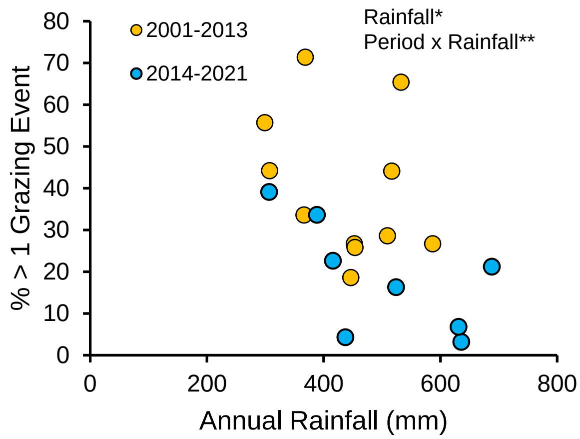

Figure 3. Percent of the study area experiencing > 1 major grazing event/yr as a function of annual rainfall (mean across the study area) for each year during two periods, 2001-2013 prior to the grazing program (yellow circles), and 2014-2021 during the grazing program (blue circles, $R^2 = 0.413$, N = 8, $P = 0.085$). The outcome of the General Linear Model indicated significant negative association of > 1 grazing event with increasing annual rainfall and a significant interaction (Period x Rainfall) between period and rainfall (* $P < 0.05$; ** $P < 0.01$).

Community-led planning generally led to livestock moving among four zones or grazing blocks within each seasonal range over a six-month season. Thus, in any given month, livestock were expected to occupy approximately a quarter of their seasonal range or 1/8 (12.5%) of the landscape. However, multiple factors related to weather, access to water, restricted migration routes, and encroachment of livestock from communities outside the study area drove deviations from grazing plans by herders (**Supplemental Material**). Success in implementing grazing management objectives (i.e., limitation to only one grazing event/yr) was evaluated using the NDVI-based metric of $SUM\Delta NDVI_{rel,i,y}$ for each MODIS pixel in the study area for all years from 2001-2021. Despite such deviations, the grazing program succeeded in reducing the area experiencing 2 or more grazing events each year. From 2002-2013, an estimated marginal mean (after controlling for rainfall) of 38.6% (± 6.4 SE) of the study area experienced more than one grazing event/year and included three years in which more than 65% of the study area experienced such grazing impacts. In contrast, with the implementation of RRG grazing from

2014-2021, only an estimated marginal mean of 18.8 % (± 3.4 SE) of the study area experienced more than one grazing event. The percent of area with > 1 grazing events/yr declined with increasing annual rainfall, and after accounting for rainfall as a covariate, we found a significant interaction (*P* = 0.004) between annual rainfall and period (Fig. 3, Table S3). We also calculated the number of years in which management implementation during 2014-2021 was successful (Fig. 1). Of the 213 sample points, 58.6% experienced 1 or fewer grazing events (success) in 6 or all 7 of years, while 28% of points experienced success in 5 of 7 years, for a total of 86% of the landscape experiencing success in 5 or more years out of 7.

Trends in NDVI

Mean coefficients across 213 sample points of January NDVI versus rainfall in the previous 12 months were significantly greater than zero both during the pre-project period 2001-2013 and after the start of the grazing program 2014-2021 (Fig. 4A). The mean coefficients for rainfall were not significantly different between pre-project and project time periods (*P* =0.43). This indicated that NDVI was equally strongly associated with rainfall during both time periods. In contrast, the mean NDVI trend across years over the 213 sample points, after correcting for the association between NDVI and rainfall, was significantly negative during the pre-project period (2000-2013) while the mean trend during the period of project activity (2014-2021) was highly significantly positive (Fig. 4B).

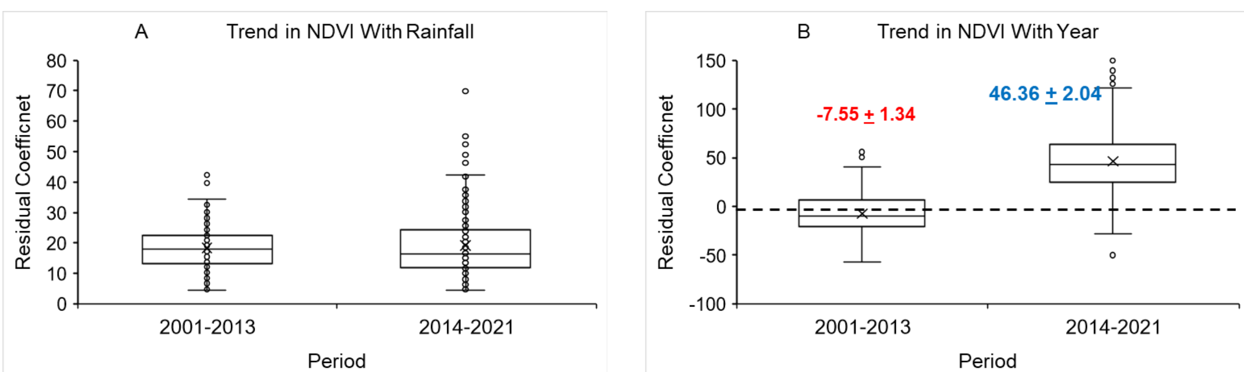

Figure 4. Mean (X) and box and whisker plot distribution of coefficients for January NDVI across 213 sample points explained in a linear model with two variables, rainfall in the previous 12 months (Rainfall) and year. Coefficients are shown for A) response to rainfall and B) change over time (Trend) in NDVI for two time periods: prior to implementation of the grazing program, 2001-2013, and during the period of the grazing program, 2014–2021. Mean (± SE) coefficient of NDVI with year are shown in red for 2001-2013 and in blue for 2014-2021.

The mean coefficient of NDVI with year, after including rainfall in the previous 12 months as an additional variable in the regression, across the 213 sample points was significantly negative prior to the start of grazing management (years 2001-2013) and highly significantly positive over the period 2014-2021 after widespread implementation began (Fig. 1).

Changes in soil carbon

Across the project area, the mean measured ΔSOC across all sampling points was 1.32 ± 0.28 SE Mg/ha to 20 cm depth between initial sampling and 2021 (Fig. 5) and 95% confidence intervals that did not include zero. The SNAP model, validated at the beginning of the grazing program, predicted initial SOC densities (10.52 ± 0.57 SE Mg/ha) with reasonable accuracy (**Extended data** Fig. S5**)**. For the 2014-2021 period of the grazing program, the SNAP model predicted a mean change of 1.26 ± 0.026 SE Mg/ha, a value very close to the measured mean. Model bias (see **Methods**) was -14.6%, indicating moderate underestimation of ΔSOC.

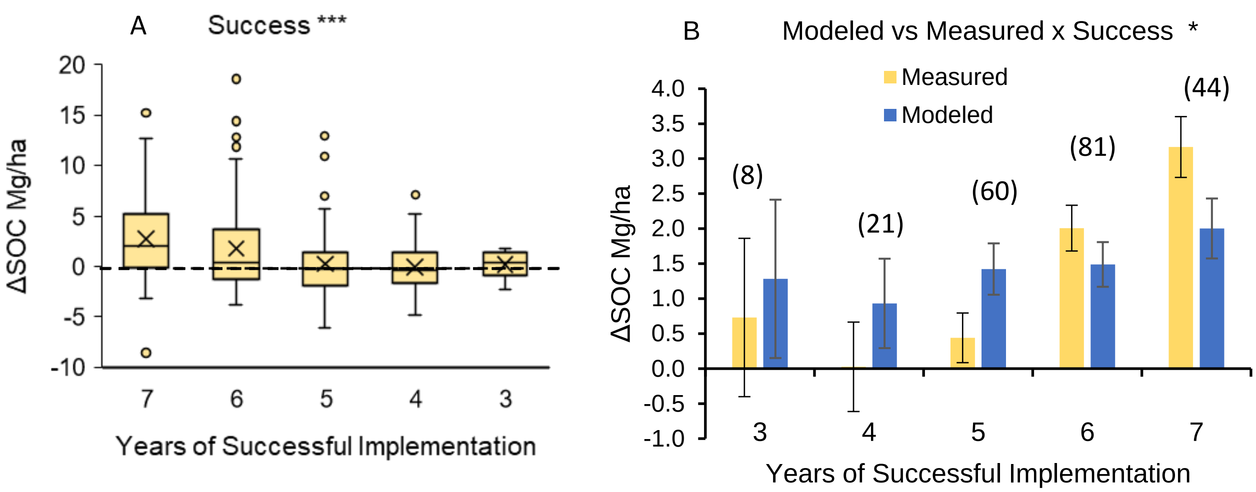

Figure 5. A. Box and whisker plots of measured ΔSOC during the 2014 to 2021 monitoring period at 213 sampling sites across the study area. Horizontal bars inside the box are medians, x is the mean, upper and lower box boundaries are quartiles. B. Matched measured and modeled ΔSOC. Bars are means, error bars are SE. Outcome of generalized linear model analysis is summarized by significant factors (* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$).

A General Linear Model (SPSS 27) of variation in the difference in SOC for 2014 to 2021 (ΔSOC_{MP}) with Source (Modeled vs Measured) and Project Implementation (Successful vs Unsuccessful) as fixed factors with Conservancy as a random factor showed that Modeled and Measured changes in SOC were not significantly different ($P = 0.701$, $\alpha = 0.05$) after including the influence of Conservancy, Year of initial sampling, Initial SOC density, and Success (see **Extended Data** Table S3). In other words, the modeled mean ΔSOC lies well within the 95% confidence interval of measured ΔSOC after accounting for other variables affecting ΔSOC ($F_{1,404} = 0.148$, $P = 0.701$).

Greater positive changes in SOC also appear to occur in areas where successful grazing management is practiced longer; ΔSOC was strongly associated with the number of years of successful grazing plan implementation (Success) ($F_{4,404}$ = 5.68, P < 0.001) but not with the number of years between initial and second sampling (**Extended data,** Table S3). Sites in which target grazing practices were implemented 6 or 7 of years 2014-2020 exhibited mean ΔSOC > 2 metric tons/ha during 2012 or 2016 through 2021, with both means highly significantly different from zero. However, sequestration was substantially less at sites with 5 or fewer years of successful implementation, averaging 0.357 (± 0.496 SE) metric tons/ha, a value not significantly different from zero.

Grazing, vegetation, and soil carbon change.

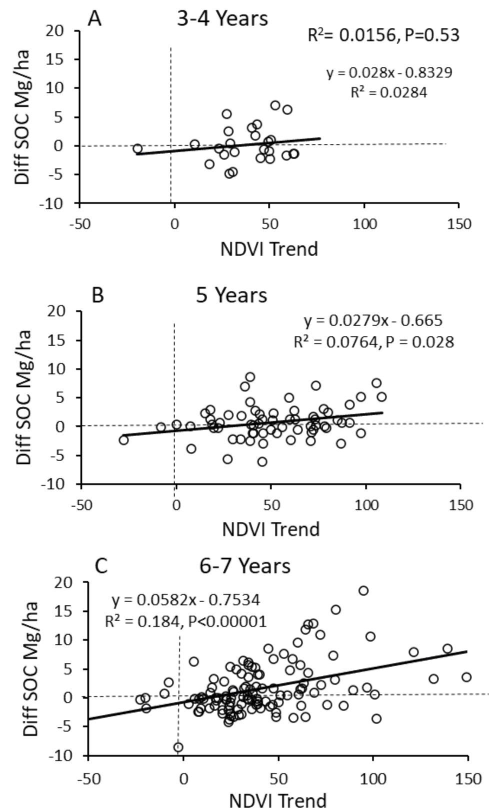

Figure 6. Correlation between the difference in SOC density between initial and second sampling and rainfall-corrected trend in NDVI for sampling locations experiencing different numbers of years (out of 7) of successful grazing management: A. 3-4 years, B. 5 years, and C. 6-7 years.

To assess whether changes in vegetation and soil carbon were associated with each other and with success in grazing management, we evaluated a linear model with ΔSOC as a dependent variable and conservancy as a random factor, with number of years of successful management and NDVI Trend (a coefficient with year after correcting for correlation with rainfall) as independent variables (**Supplemental Material** Table S4). While ΔSOC varied significantly among conservancies, years of successful management was not significant as a main effect (P = 0.539). However, ΔSOC was significantly correlated with NDVI Trend (P < 0.0001) and there was a significant interaction between years of successful management and NDVI trend (P =

0.009). The interaction can be interpreted as a significant relationship between ΔSOC and NDVI Trend occurring only at sampling locations where management success occurred in 6 or 7 out of 7 years (Fig. 6).

Discussion

This study demonstrates several important outcomes relevant to the broader question of whether grazing management can be a large-scale driver of rangeland restoration and carbon sequestration in open pastoralist systems. First, our data suggest that grazing practices organized and governed by communities can achieve community-wide RRG management (Fig. 1, 3). This result is supported by the achievement of a single one-month grazing event per year in a (-at least 6 out of 7 years in nearly 60% of the study area (Fig. 1). Second, after statistically controlling for the influence of rainfall, green forage biomass (as measured with MODIS satellite-based NDVI) increased on average over the 8 years after the start of widespread RRG grazing management in 2014, whereas it declined over the 12 years prior to the start of the program (Fig. 4). Third, the program was associated with an overall significant increase in soil carbon sequestration across the study area, and this increase was accounted for by significant positive ΔSOC at locations that experienced only one grazing event per year (Fig. 5) in 6 or 7 out of 7 years monitored. Fourth, the magnitude of ΔSOC, both within communities and across all sampling points, matched that predicted by a relatively simple process-based soil carbon dynamics model, SNAP (Ritchie 2014, 2020). Finally, these outcomes occurred despite increases in both human population and livestock densities, two commonly identified drivers of degradation in grasslands and semi-arid rangelands (van de Koppel et al. 1997, Stocking 2003, Thompson et al. 2009, Miehe et al. 2010). Thus increases in NDVI and SOC were unlikely to be related to any differences in rainfall patterns between the years of the grazing program (2014-2021) as compared to those prior to the program (2001-2013) period (Figs. 2-4).

Widespread implementation of RRG grazing in this case was achieved despite uncertainties over land tenure, community governance, and cooperation among different ethnic groups, as nearly 60% of the study area experienced successful grazing management (i.e., only one grazing event per year) in at least 6 of the 7 years. The grazing program yielded a large improvement in average impacts during the years of the grazing program (2014-2021) under higher overall livestock densities (Fig. 2D) as compared to the pre-program period 2001-2013 (Fig. 3). The significant interaction of time period and rainfall suggested that, as would be expected, multiple grazing events per year were more likely in drier years, but multiple grazing events were much less likely during the grazing program in wetter years (Fig. 3).

Our evaluation of the success of grazing management was strongly supported by the three calibrations – 1) from green forage biomass to a satellite-based index (**Extended data**, Fig. S3), 2) from changes in NDVI corrected for near-term weather to livestock presence (**Extended data,** Fig. S4A, and 3) from the assessment of temporal grazing patterns from a cumulative calculation of monthly changes in weather-corrected $\Delta NDVI_{rel,i,m}$. (**Extended data,** Figs. S4B). Biomass-

NDVI relationships demonstrate a strong connection between satellite-based measures and actual forage biomass and support the use of satellite imagery in monitoring range conditions and grazing impacts. The strong similarity in slopes and intercepts between northern Kenya and Serengeti study areas (**Extended data,** Fig. S3A) inspires confidence that a reduction in forage biomass from grazing results in a detectable reduction in NDVI.

Our results also suggest reasonable confidence in detecting monthly livestock grazing impacts at each sampling point by the combination of a) $\Delta NDVI_{rel,i,m}$ and b) the number of grazing events in a year by $SUM\Delta NDVI_{rel,i,,y}$. Sensitivity analyses (**Supplemental Information**) of the NDVI thresholds (**Extended data**, Fig. S4) used to determine livestock presence and number of monthly grazing events suggest that the 95% CLs for the logistic regressions include an appropriate range of livestock occupancies (7.5 – 42.5% of the landscape), with the mean threshold of $NDVI_{rel,i,m} \geq 0.05$ corresponding to a livestock occupancy of ~26% that agrees reasonably well with the monthly occupancy expected under RRG grazing of 12.5 – 25%. Given the many potential sources of error in reporting by local observers (e.g., a mismatch in the period of days assessed by community grazing coordinators relative to satellite imagery analysis, the lack of knowledge of livestock distributions in relatively inaccessible areas (Figure 4A)), and the possibility of encroachment by herders from communities outside the project area, the strength of the logistic regression (high R^2 value) indicated a stronger than expected signal of livestock impacts on NDVI.

The improvement in range condition, as judged by green forage biomass for livestock and carbon sequestration at these locations (Figs. 4, 5), occurred despite increases in drivers commonly associated with range degradation: human population size and livestock numbers (Fig. 2C, D). Our results suggest that, as proposed by the originators of the concepts of RRG grazing (Savory and Butterfield 1999, Savory and Butterfield 2010), that the key livestock-related driver of the positive vegetation and carbon changes is the movement of livestock, not reduced livestock density. Classic views of livestock impacts have focused on stocking rates (density) and "carrying capacity," but recent literature reviews suggest that, if livestock are grouped and moved frequently, livestock densities can be stocked at much higher density than expected (Savory and Butterfield 2010, Ritchie 2020, Mosier et al. 2021, Apfelbaum et al. 2022b, Phukubye et al. 2022). Reciprocally, our results demonstrate that, under the right grazing management (Phukubye et al. 2022), reductions in livestock density may not be necessary to achieve rangeland restoration for sustainable development. This may be quite significant for pastoralist systems where wealth and livelihoods are predominantly dependent on livestock and therefore programs that recommend reducing livestock numbers often conflict with the principles of increasing wealth/well-being among pastoralists in the absence of alternative livelihood options.

To our knowledge, our results represent the first large-scale monitoring of the impacts of shifting from continuous seasonal grazing to a plan of RRG grazing (Fig. 3, 4). Importantly, our results document the shift from declining to increasing green forage biomass with year, independent of the influence of rainfall and despite increasing human population density and livestock numbers (Fig. 4). This shift in vegetation trend was accompanied by a first-ever large-scale (> 100,000 ha) demonstration of an increase in soil carbon where RRG grazing was consistently practiced (Fig.

5A), with no increase where it was not. This result aligned well with the predictions of SNAP, a relatively simple process-based model of soil carbon change (Fig. 5B). Given that increases in soil carbon were positively correlated with increased green forage biomass only where RRG management was consistently applied (Fig. 6), these patterns are unlikely to be driven by other factors that might have differed between the pre-project period 2001-2013 and the project period 2014-2021.

Projects focused on implementing RRG grazing management plans as a basis for large scale carbon sequestration projects may be successful at delivering soil carbon credits to voluntary or other carbon markets. Where single grazing events were consistently achieved, mean carbon sequestration over a 6-9 year period averaged 1.68 Mg SOC/ha, equivalent to 6.16 Mg CO_2 removed from the atmosphere. Methane emissions of livestock are approximately 0.0175 Mg CO_2e $ha^{-1}yr^{-1}$, based on ~ 0.175 Mg CO_2e methane emitted annually per TLU. This estimate assumes ~ 1 Mg CO_2e greenhouse gas equivalent/Mg animal (Franz et al. 2010) emitted annually, 0.175 Mg/TLU (Jahnke 1982) and a density of 0.1 TLU/ha (~200,000 TLUs over the approximate 2 million ha study area (Fig. 2)). Thus, net removals to soil carbon during 2014-2021 would be estimated to be 6.14 Mg/ha CO_2e over 8 years. With consistent (6 or 7 out of 7 years) achievement of $\leq$ 1 grazing event/year across 58.6% of the landscape, the grazing program had likely sequestered ~ 7.2 million Mg CO_2e during the 2014-2021 period of the grazing program.

Importantly, this increase can be predicted by soil carbon dynamic process models such as SNAP and its grazing management version SNAPGRAZE (Ritchie 2020), DNDC (Li 2007) or other models formulated to incorporate key details of grazing processes. Methodologies, such as Verra's (http://verr.org) VM0032 or VM0042 that determine carbon credits from new grazing practices allow and rely on models to determine issued credits early in the project, that might otherwise require a financially infeasible 7-10 years of new management activities. Such models may prove accurate enough, if they are conservatively parameterized, to justify such issuances (Ritchie 2014, Khalil et al. 2020, Ritchie 2020). The predicted vs. observed changes in soil carbon by the SNAP model were conservative (i.e., bias < 0%) over the 8 years of the project and well within the 95% CI for observed ΔSOC. Such a means to compensate pastoralists for ecosystem service provision opens the potential for scaling restorative grazing practices across millions of hectares while directly benefiting some of the poorest people in the world (Teka et al. 2019).

Our results support three important ideas for the future of social development and wildlife conservation in pastoralist areas across broad areas of the global tropics and sub-tropics: 1) rangeland restoration under RRG grazing can occur without de-stocking livestock, 2) RRG grazing is an activity that can sequester large volumes of greenhouse gases and support large-scale carbon sequestration projects and the revenue they bring to communities. 3) RRG grazing offers a management choice that can support habitat improvement, coexistence of livestock and wildlife and community-based conservation at a time when traditional protected area models of conservation may constrain indigenous land rights and may not succeed in conserving wildlife populations (Veldhuis et al. 2019). These outcomes offer optimism to those efforts focused on improving the livelihoods and productivity of pastoralists in highly degraded ecosystems.

Materials and Methods

Study Area

The study was conducted in a set of 13 community areas each organized around established wildlife conservancies in northern Kenya and participants in a carbon project validated and verified under the Verra Standard (Fig. 1). These communities herd livestock mostly within an approximately 2 million ha region of savanna woodlands with mean annual precipitation varying from 200 mm in the northeast to near 900 mm in the southwest. The study area features relatively large annual migrations of livestock from more heavily populated higher rainfall areas during the wet season (November – December, April – June) to less populated drier areas during the dry seasons (January – March, July – October) (**Extended Data,** Fig. S1).

Grazing management

Prior to the beginning of the study (2001-2013), herders within each of 13 communities largely conducted unrestricted grazing within each of two different seasonal zones: wet season ranges near their permanent dwellings associated with established wildlife conservancies and dry season ranges in largely unpopulated drier areas that lack permanent settlements (Figs. 1-3). These communities were presented with the idea that the benefits of traditional nomadic grazing practices could be restored by implementation of community-planned mobile grazing management focused on clustering multiple herds, grazing in designated areas, or "blocks" for a planned length of time, and then moving to new pasture and not returning in the same year. We defined this as RRG (rapid rotational) grazing, and the intention was to conduct separate rotations in each seasonal range. Communities participated in the form of employed rangeland coordinators that taught principles, monitored livestock movements and numbers, and supported community-level decision-making and design of grazing plans. Actual grazing plans were designed and approved by grazing committees, a community-led governance mechanism, over the time, timing and density of animal use of different areas. These committees governed both the use of wet and dry season areas in their communities' respective wildlife conservancies.

Community-led planning generally led to livestock moving frequently among grazing blocks within each seasonal range either during the wet or during the dry season. Deviations from the plan (failure of implementation) occurred due to 1) lack of compliance by herders due to incomplete or ineffective communication of grazing plans, 2) prolonged aggregations of livestock near permanent water holes in dry season range, 3) spatially local, monthly gaps in rainfall during the wet season that forced herders to leave zones early and re-use previously visited zones, 4) seasonal migrations by herders across neighboring conservancies, often constricted by the need to avoid inaccessible mountains and protected areas, 5) occasional encroachment of by herders from pastoralist communities outside the study area (such encroachment accounted for less than 2% of total animal-days in the study area).

Effects of grazing management were inferred from a temporal comparison within the study area; conditions from 2001-2013, before RRG grazing was widely implemented, and 2014-2021, a period when RRG grazing was implemented across most of the study area. Making spatial comparisons between the study area and "reference" adjacent rangelands was deemed inappropriate due to the lack of similar climate, comparable history and differentially evolving management during the study period in adjacent rangelands. For example, pastoralist communities in areas adjacent to the study area experienced ~~either~~ mostly different climates e.g., wetter conditions than average for the study area in Laikipia County to the south or in Samburu County to the northwest or drier conditions than the study area average to the north and east (Fig. 2). Instead, inference was based on variation in whether RRG grazing was consistently achieved at different locations within the study area (Fig. 1) after controlling statistically for annual variation among locations in rainfall (Fig. 4, **Extended data** Tables S3, S4) In addition, variation among communities in implementation as well as vegetation and soil carbon response was accounted for statistically by entering Conservancy as a random effect in analyses of changes in forage biomass and soil organic carbon (**Extended data,** Tables S3, S4).

Livestock numbers

Livestock numbers were estimated by multiplying the average number of the different species, cattle, sheep and goats (shoats), donkeys, and camels, per household by the number of households in the study area. Number of people, NP, for each Conservancy at the end of the monitoring period, 2020, was obtained from numbers reported by each Conservancy in the NRT website interactive map (https://www.nrt-kenya.org/interactive-map) and are reported in **Extended Data** Table S4. The number of households NHH in the project area was calculated as NHH = NP / MHHN, where mean household size MHHN was found to be 5.59 (+ 0.09 SE) persons in 1030 households surveyed in four different Conservancies in November and December of 2014 and 2015. The mean number of livestock of different species per household was determined from the same survey, which were chosen randomly from within community areas so as to include households without livestock. These means per household were not significantly different across conservancies. This method was used because human monitors from many community areas were highly inconsistent in reporting actual livestock numbers each month, even in months where all livestock were reported to be present on a conservancy. Changes in livestock numbers were then inferred from changes in the number of households across years reported by community leaders.

Vegetation dynamics

MODIS Normalized Difference Vegetation Index (NDVI) was used as an indicator of green forage biomass in the study area. 16-day composite images of NDVI (MOD13Q1, 250 m resolution) were used to assess biomass of green forage on a bi-weekly basis throughout the project area. NDVI was calibrated (**Extended Data** Fig. S2) by comparing values for individual pixels for a given 16-day range with estimated average aboveground green forage biomass clipped, air-dried at 50°C for 7 days, and weighed on an electric balance with resolution to 0.1, from thirty 0.125 m^2 within the boundaries of that pixel. Biomass estimates for the 16-day period were obtained from 30 pixels chosen to span the range of available biomass in areas grazed by livestock, and thus from a total of 900 quadrats. Calibrations were performed during July 2014 following a late-arriving wet season. These data were combined with NDVI calibrations from Serengeti National Park based on biomass measurements in 27 grazed plots at seven sites spanning a rainfall gradient from 490 – 900 mm/yr (Anderson et al. 2007). NDVI was regressed in a general linear model with green herbaceous biomass from Kenya and Serengeti sites, with separate regressions conducted for each site and for both sites combined, with site as a factor (**Extended Data** Figure S2, Table S1).

Livestock grazing impacts

MODIS NDVI-based monitoring yielded estimates of forage biomass and relative change in biomass (rainfall-corrected). To measure livestock grazing impacts, a change in NDVI (ΔNDVI = earlier NDVI – later NDVI) was calculated, with more positive values indicating stronger reduction in forage biomass (**Extended data**, Fig. S4) between images. ΔNDVI serves as a measure of livestock impact on forage during the 16 days between composite images. Further a *relative* $\Delta NDVI_{rel,i,m}$ for each sample point *i* and month *m* was calculated by subtracting the mean $m\Delta NDVI_{rel,ref}$ for the same period from ungrazed areas (e.g., inaccessible plateaus, fenced wildlife sanctuaries) to correct for expected changes in NDVI due to recent rainfall or drying (see **Supplemental Information** for more details). Livestock presence, as drawn on maps of subsections of the study area by trained, employed local community members, was significantly correlated with relative monthly change in NDVI. The logistic regression suggested that livestock presence could be detected with greater than 95% confidence by relative monthly changes in NDVI that exceeded 0.05.

Calibration of livestock incidence and number of grazing episodes (**see Extended data** Fig. S4) showed that livestock were present with 95% certainty if $\Delta NDVI_{rel,i,m} > 0.05$. For each of 41 sample points spread across five conservancies, with each point in a different grazing “block” within each Conservancy, we assessed livestock presence for 2-12 months, for a total of 130 location x month combinations. In 2014, we used grazing zones in Westgate, southern Namunyak and Biliqo Bulesa Conservancies; in 2017 we used zones in Leparua, Nakuprat, and Biliqo Bulesa conservancies to yield annual evaluations of the number of grazing events for 41 grazing block x year combinations (see examples in Figures 6-8).

Monthly $\Delta NDVI_{rel,i,m}$ exhibited two properties in over 90% of cases: 1) $SUM\Delta NDVI_{rel,i,y}$, the annual sum of $\Delta NDVI_{rel,i,m}$, at a sample point was strongly correlated with the number of grazing events, and 2) two or more grazing events corresponded strongly with $SUM\Delta NDVI_{rel,i,y} > 0.05$ as demonstrated by logistic regression (**Extended Data** Figure S4). The regression suggested a threshold value of $SUM\Delta NDVI < 0.05$ was associated with a probability of successful grazing implementation > 50%, so we used $SUM\Delta NDVI_{rel,i,y} < 0.05$ as the threshold for categorizing a pixel as experiencing successful implementation of rotational grazing. Note that this criterion applied to livestock grazing rotations within conservancies and movements around a larger regional landscape, as occurred during each dry season.

Soil Sampling Design

A total of 226 permanent sampling stations were initially established across the study area in a hierarchical fashion to ensure that as large a range of soil carbon densities as possible be sampled on the project area to test the SNAP soil carbon dynamic model and also to ensure representative sampling across the study area (Fig. 1). To sample soils with high expected SOC density, a total of 7 sampling stations were distributed at random inside each of five "core" zones across the study area. Each core zone was contained within one Conservancy, and was managed as an area that received limited livestock grazing (since 2006) to minimize interference of livestock with tourism operators. Two additional stations were allocated at random to black cotton soils when such soils occurred in these special areas (in 2 conservancies), yielding a total of 39 stations allocated to these special potential strata**.** The remaining 187 sampling stations were apportioned at random throughout the project area using the random point generator in QGIS 2.12.3. Black cotton (higher clay content) soils were underrepresented in this initial apportionment, so an additional 14 stations were assigned at random to black soils resulting in a total of 240 sampling stations. Inaccessibility of sample points either during the initial survey in 2012 or in 2021 led to elimination of 13 sample points, mostly from periodically inaccessible areas of low rainfall in Melako and Biliqo Bulesa Conservancies. Nevertheless both Conservancies were sampled at > 15 sample points each, despite this elimination. Thus, analysis of ΔSOC, its correspondence with predictions from the SNAP model and other potential drivers was performed with a subset of 213 sample points.

Stratification. The study area was initially stratified to sandy loam versus black cotton soils, but this stratification was not used. Modeled ΔSOC in 2013 was found to not differ between soil types and stratification reduced overall uncertainty in the estimate of ΔSOC, so all further analyses did not stratify by soil type. We did, however, include Conservancy as a random factor in assessing model predictions and SOC dynamics to account for uncontrolled potential climate, management and governance biases.

Soil Measurements

We measured soil organic carbon concentration at each permanent station, soils were sampled initially in either 2012 or 2016, and then again in 2021. Two 5 cm diameter cores were taken with an auger to a depth of 20 cm and mixed to form a composite sample, which was then dried and weighed and analysed for total organic carbon using MIR (Mid Infra-Red) spectroscopy at the ICRAF laboratories in Nairobi (Vagen and Winowiecki 2013). A subset of 30 samples was analyzed for total organic carbon by the Walkley-Black method (Moriasi et al. 2007) at Crop Nutrition Services, Nairobi Kenya, to calibrate the MIR method. The MIR predicted the wet chemistry estimate with $R^2 = 0.96$, **Extended Data** Fig. S6).

Bulk density of soils was measured with a separate soil core (5 cm diameter x 20 cm depth) of known volume at each station by pounding a 5 cm pipe to a depth of 20 cm, digging around the pipe sufficiently to insert a metal plate beneath the pipe and then lifting and emptying the pipe into a washtub so as not to lose soil. Rocks and pebbles were removed through a 2 mm mesh sieve and their volume measured by displacement. This volume was subtracted from the core volume to obtain a net soil volume. Soil remaining after rocks and pebbles were removed was sun-dried at 45ºC for 7 days and weighed and divided by net soil volume to obtain bulk density (g/cm^3) (Hao et al. 2007).

Model Parameters

The SNAP soil and vegetation parameters (i.e., soil texture, lignin and cellulose content, historic average grazing intensity) along with interpolated average annual rainfall, were entered into a soil carbon dynamic model called SNAP to predict SOC values to a depth of 20 cm at each of 226 sampling stations in the project area (Fig. 1). The model was not further calibrated. The SNAP model (Ritchie 2014) was used to predict initial soil organic carbon (SOC) stocks based on the estimated history of grazing and other conditions relevant to 2001-2013. Mean, standard errors and ranges of the parameters included in the SNAP model for are presented in **Extended Data,** Tables S6, S7.

Soil texture of a subset of 30 samples was analysed using a Horiba Model LA 950A2 particle size analyzer at the ICRAF laboratory to find the mean percent sand (*SAND%*) associated with each of the major strata. The mean percent sand from the stratum was then estimated from MIR reflectance data calibrated using the measured textures. The percent sand estimated for each station was used as the *SAND%* parameter in the SNAP model.

Historical grazing intensity (*GI*) at each sample point was measured by an expert whose visual assessment of grazing intensity using a decision tree was trained against estimates obtained from comparing NDVI values at the MODIS pixel containing the sample point against mean NDVI in polygons within areas not grazed by livestock (for details see **Supplemental Information**).

Plant lignin and cellulose content (*MAPLC*) of aboveground vegetation was estimated by collecting samples of the four most common species of the dominant types of herbaceous vegetation: perennial grass, annual grass, legume or forb. Samples for analysis of each type of plant were collected from 58 sampling stations from across five different conservancies (Il Ngwesi, Kalama, Namunyak, Sera, and Westgate). Vegetation was clipped to the ground surface from two 0.125 m^2 quadrats at each station, sun-dried (at 45ºC for 7 days) and weighed. Plant material was ground through a 0.9 mm sieve with a Wiley Mill, and a standard fiber analysis was performed, consisting of sequential digestion in acid detergent (to obtain cellulose) and 12 M sulfuric acid (Jensen et al. 2005) (**Extended data**, Table S6).

Mean Annual Precipitation (*MAP*) was estimated directly reading estimated mean annual rainfall values from WorldClim digital rainfall maps generated by The World Resources Institute taken from satellite data calibrated to local weather stations[1].

Fire frequency was not reported by the local NGO Northern Rangelands Trust on the project area during the study period 2001-2021. Burned area layers of the study area from the Global Fire Atlas (Andela et al. 2019) were inspected to confirm this evaluation, and indeed no burned areas were reported for the study period.

Model Evaluation

The mean predicted SOC stocks at each station, based on presumed historical conditions, was compared against observed SOC at the same station (**Extended Data** Fig. S5, Table S3). Bias was determined by the standard calculation(Moriasi et al. 2007).

$$MBIAS = \frac{\sum_{i=1}^{n}(Y^{obs}{}_{i} - Y^{pred}{}_{i})}{\sum_{i=1}^{n} Y^{obs}{}_{i}} \times 100 \qquad (2)$$

Where, *MBIAS* = Percent bias of carbon model predictions relative to observed data,

n = number of sampling stations tested,

$Y^{obs}{}_{i}$ = observed SOC density at station *i*, and

$Y^{pred}{}_{i}$ = SOC density predicted at station *i*.

Mean predicted SOC values for each Conservancy were then compared with mean observed SOC stocks in those Conservancies. The results suggested that within the NRT Conservancies, the model predicted mean and individual site SOC values with more than 80% accuracy. This analysis was performed and revealed that for each stratum, the 95% CI for the observed baseline SOC overlapped with the 95% CI for the modeled baseline SOC for all 13 Conservancies (**Extended** data, Fig. S5).

Statistical Analyses

Calibrations, model assessments and analyses of drivers of changes in NDVI and soil carbon were analyzed with SPSS 27 (IBM Corp. 2020. IBM SPSS Statistics for Windows (Version 27.0)) using either general linear models or generalized linear models with an identity link function for biomass, NDVI, and SOC density and a logit link function for livestock presence/absence and occurrence of more than 1 significant grazing event per year.

[1] http://datasets.wri.org/dataset/average-annual-rainfall-in-kenya

Extended Data - Figures

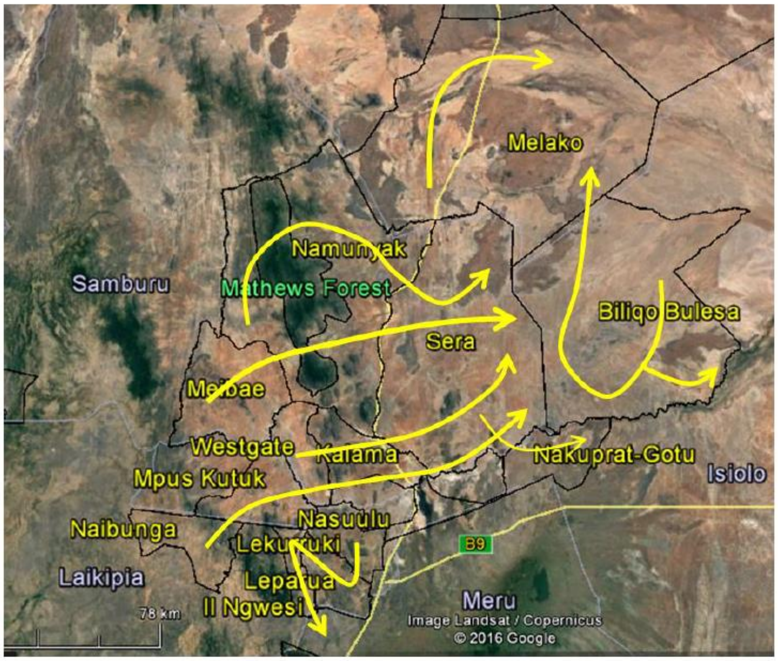

Figure S1. The regional grazing system encompassed by the project zone. Shown are the Conservancy boundaries (black borders) and yellow arrows show major migration routes used in transitioning from wet to dry season range or when rains fail in wet seasons.

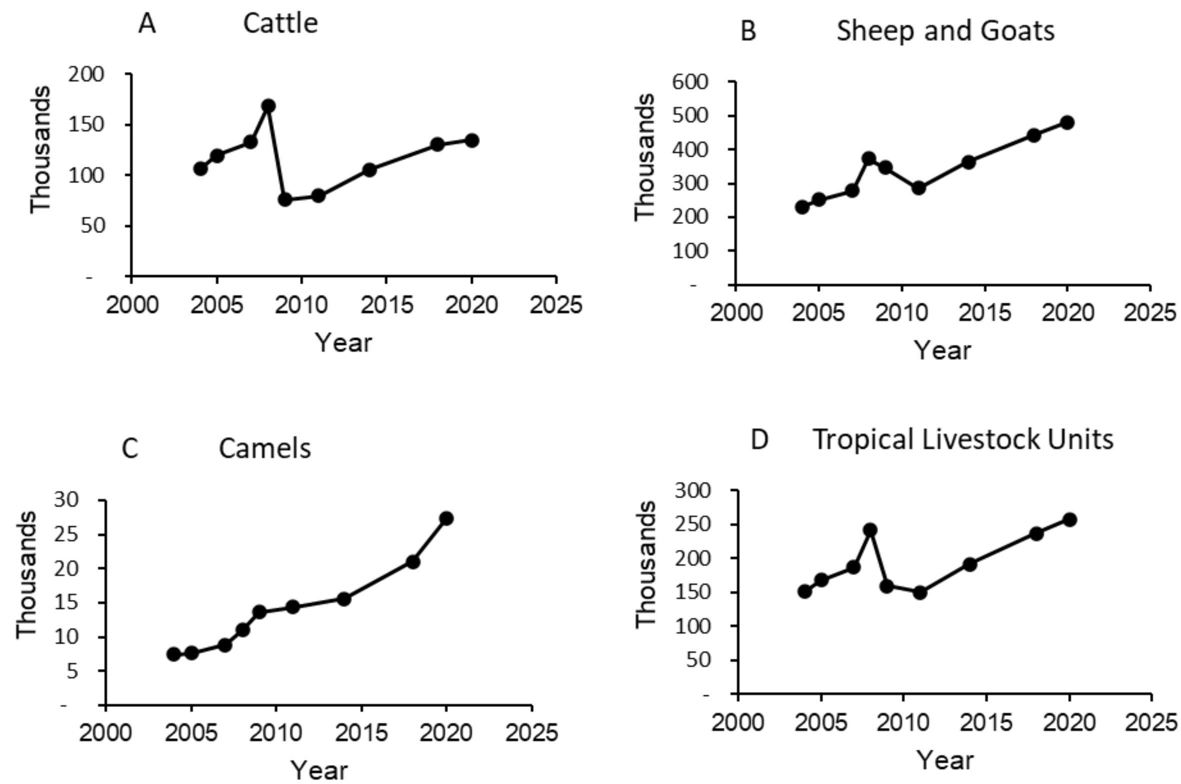

Figure S2. Changes in numbers of the different major species of livestock on the Northern Kenya study area 2003-2020. Note the short duration high density grazing program began in 2014. Sharp declines in donkeys and cattle occurred during the 2008-2010 drought.

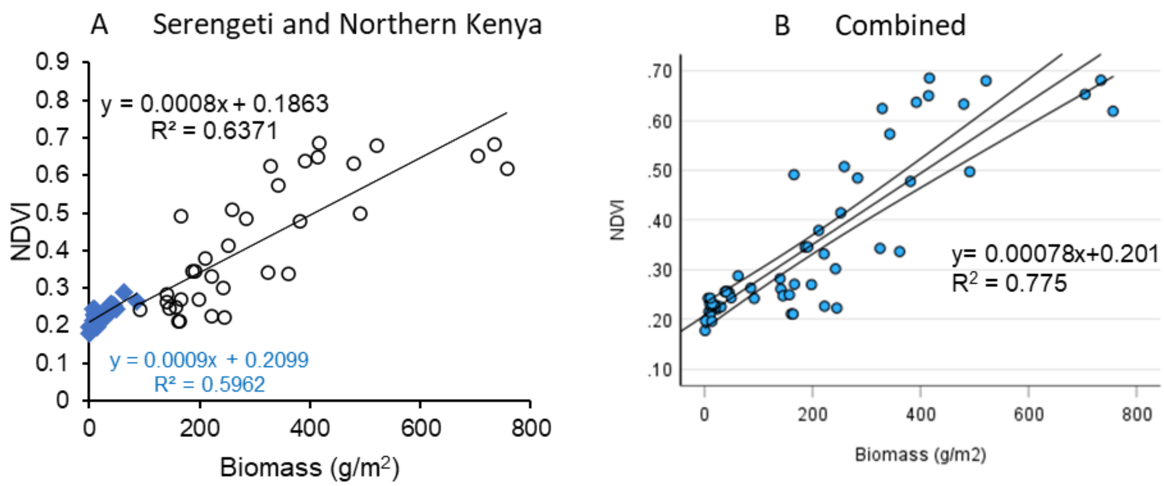

Figure S3. Relationships between NDVI measurements from individual pixels (250 x 250 m) of 16-day composite MODIS NDVI and measured mean forage biomass (shrub leaves and herbaceous plant biomass). A. Shown and regressed separately for the northern Kenya study area (blue) and Serengeti National Park (open). B. Combined data with 95% CL for NDVI values.

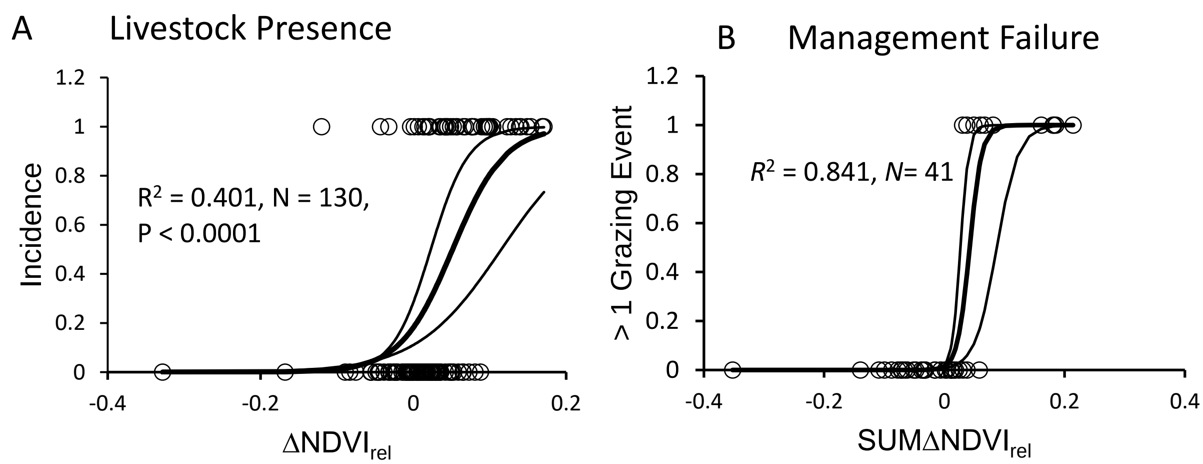

Figure S4. Logistic regression lines (thick curves) with 95% CI (thin curves) in the northern Kenya study area between A. Presence of livestock recorded by observers versus monthly relative changes in NDVI at 130 random sample points, and B. Failure of management (2 or more significant livestock grazing events during a year) versus sum of relative changes in NDVI at 41 sample points over the year (January – December).

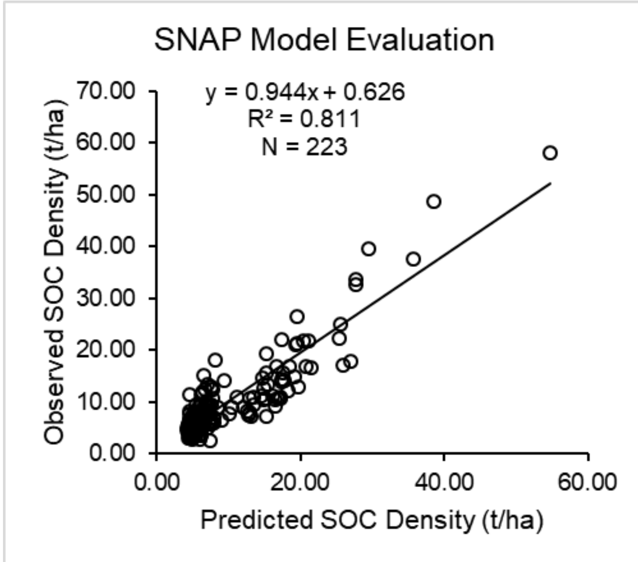

Figure S5. Regressions of observed SOC density at each of the survey sites versus the predicted SOC density from the SNAP model based on rainfall, historical grazing intensity, soil texture, and plant lignin and cellulose at each site.

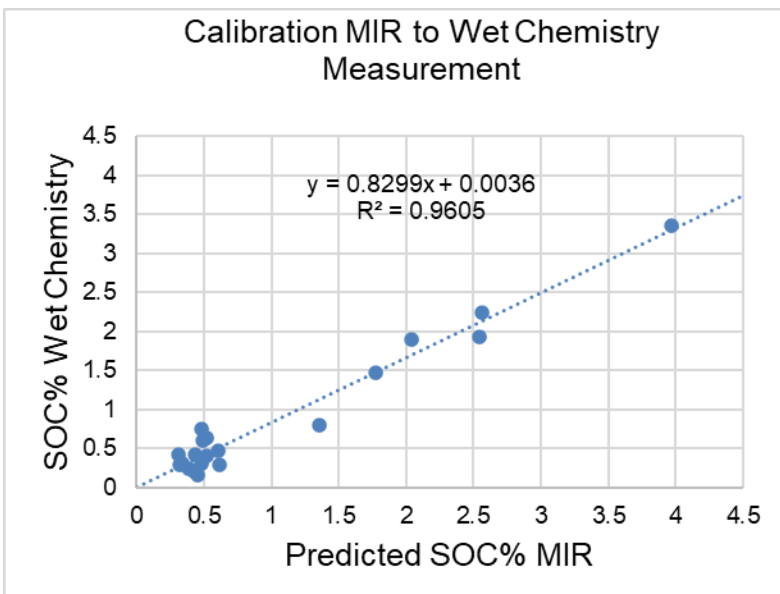

Figure S6. Calibration of Mid-Infrared Spectroscopic method of assessing soil organic carbon content (%) with assessment by the Walkley-Black method.

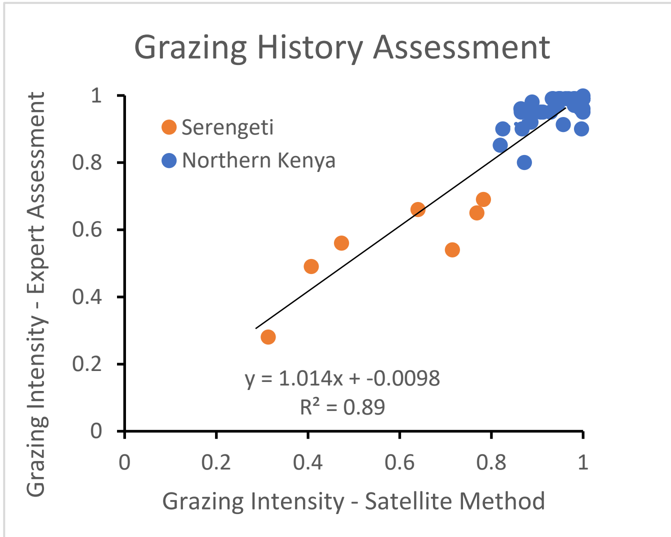

Figure S7. Calibration of the expert assessment method for assessing historical grazing intensity used in this project with outcomes of satellite-based assessment of grazing intensity using NDVI-forage biomass regressions (Fig. 14) to estimate grazing intensity. Data are from the project area (blue points) and from application of the expert assessment in a long-term grazing exclosure experiment in Serengeti National Park (Anderson et al. 2007, Ritchie 2014) to expand the range of grazing intensities.

Extended Data - Tables

Table S1– General Linear Model to for relationship of NDVI to measured green forage biomass.

Dependent Variable: NDVI

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	1.180[a]	3	.393	67.855	<.001
Intercept	.743	1	.743	128.259	<.001
StudyArea	.003	1	.003	.453	.503
Biomass	.028	1	.028	4.816	.032
StudyArea * Biomass	.000	1	.000	.028	.869
Error	.342	59	.006		
Total	8.570	63			
Corrected Total	1.522	62			

a. R Squared = .775 (Adjusted R Squared = .764)

b. Study Area – either northern Kenya or Serengeti National Park (used to increase range of forage biomass values)

Table S2. Generalized Linear Model analysis for the influence of annual rainfall (mm) and period (2001-2013 and 2014-2021) on the % of the study area with > 1 grazing event/yr.

	Wald Chi-Square	df	P
(Intercept)	70.202	1	<.001
Rainfall	3.989	1	.046
Period * Rainfall	8.246	1	.004

Dependent Variable: % Unsuccessful

Model: (Intercept), Rainfall , Period * Rainfall, identity link function, N= 20 years

Table S3. Mixed Model analysis of ΔSOC in the Northern Kenya Rangelands Carbon Project 2014-2021

Source		Type III Sum of Squares	df	Mean Square	*F*	*P*
Intercept	Hypothesis	170.636	1	170.636	14.453	.007
	Error	78.755	6.670	11.807[a]		
Conservancy	Hypothesis	236.500	12	19.708	**2.678**	**.002**
	Error	2973.682	404	7.361[b]		
Year Sampled	Hypothesis	12.019	1	12.019	1.633	.202
	Error	2973.682	404	7.361[b]		
Baseline SOC Density	Hypothesis	.650	1	.650	.088	.766
	Error	2973.682	404	7.361[b]		
Years of Successful Implementation (Success)	Hypothesis	167.249	4	41.812	**5.681**	**.000**
	Error	2973.682	404	7.361[b]		
Modeled vs Measured (Source)	Hypothesis	1.088	1	1.088	.148	.701
	Error	2973.682	404	7.361[b]		
Success x Source	Hypothesis	81.010	4	20.253	**2.751**	**.028**
	Error	2973.682	404	7.361[b]		

a. .222 MS(Conservancy) + .365 MS(YearSampled) + .412 MS(Error)

b. MS(Error)

Table S4. Generalized Linear Model to explain influence of grazing management and NDVI Trend, corrected for rainfall, with year on ΔSOC density during the period 2014-2021 of rapid rotational grazing for the Northern Kenya Grasslands Carbon Project.

Variable	Wald Chi-Square	df	P
(Intercept)	.478	1	0.489
Years Successful Grazing	3.111	4	0.539
Conservancy	35.868	12	<0.0001
NDVI Trend	15.710	1	<0.0001
Years Successful Grazing x NDVI Trend	13.425	4	0.009

Note: Years and NDVI Trend were entered as factors and Conservancy as a random effect, Link function = identity, N= 213 sample points

Table S5. Human population data for each community (conservancy) in the project area.

Conservancy	People	Households[1]
Biliqo Bulesa	5,833	1,043
Il Ngwesi	2,137	382
Kalama	9,958	1,781
Leparua	8,590	1,537
Meibae	12,235	2,189
Melako	20,974	3,752
Naibunga	19,106	3,418
Nakuprat-Gotu	6,734	1,205
Namunyak	32,504	5,815
Nasuulu	5,200	930
Oldonyiro	6,758	1,209
Sera	7,214	1,291
Westgate	4,494	804
Total	141,737	25,356

1. Based on estimated 5.59 people per household from baseline household surveys.

Table S6 Lignin and cellulose values (%) for major vegetation functional groups in the project area.

Dominant Plant	N*	Lignin	SE	Cellulose	SE	Lig+Cell	SE
Annual Grass	16	5.70	0.77	17.55	1.90	23.25	1.56
Forb	22	6.81	0.66	15.53	1.35	22.33	1.14
Perennial Grass	9	6.39	1.37	29.29	1.94	35.68	1.42
Woody	11	10.93	1.74	23.30	1.33	34.23	1.69

*Number of sampling stations at which each functional type was dominant and biomass was sampled.

Table S7. Parameters used in the SNAP model of soil carbon dynamics, including mean and SE as well as the range across the permanent sampling stations in the project area.

	Mean*	SE	Smallest	Largest
GI	97.71	0.19659	70	99
MAP	562.61	8.92593	327	1,050
FIRE	0	0	0	0
SAND%	42.77	1.08368	7	74
LIGCELL%**	22.62	0.09628	22	36

* N = 230, **LIGCELL% values entered on the basis of the dominant vegetation functional group: annual grasses, forbs, perennial grasses, shrubs (see Table S6).

Supplemental Material

Study Area

The grazing program and research study was conducted in an approximately 2 million ha region of northern Kenya (Fig. 1), extending from the northern slopes of Mt. Kenya in Laikipia County northward across the Ewaso Nyiro River into portions of Samburu, Isiolo, and Marsabit Counties. The study area features multiple human ethnic groups, including Maasai, Turkana, Samburu, Rendille, Borana, and Somali, all of whom are traditional nomadic or semi-nomadic pastoralists. While population sizes in the region have more than doubled during the past 25 years (Fig. 2C) communities have become more sedentary, with most households occupying permanent dwellings. Nevertheless, the economy remains focused on husbandry of cattle, sheep, goats, donkeys, and to an increasing degree, camels.

The study area spans a large traditional seasonal migration route from higher rainfall areas in the southwest to drier areas in the northeast. (Fig. 2B, **Extended Data** Fig. S1). Herders have many options available depending on different spatial patterns of rainfall and the dependence of each community on cooperation with the others. Dry season grazing areas (endpoints of arrows) typically lie in unsettled areas which often lie between areas dominated by different ethnic groups. Regional grazing plans associated with the grazing program conferred specific recommendations for these movements to minimize inter-ethnic conflicts and competition for forage and water.

Grazing Management

Community-led grazing planning generally led to livestock moving among four zones or grazing blocks within each seasonal range during the season. In any given month livestock therefore occupied approximately a quarter of their seasonal range or 1/8 (12.5%) of the landscape. Deviations from the plan (failure of implementation) occurred due to 1) lack of compliance by herders due to incomplete or ineffective communication of grazing plans, 2) spatially local, monthly gaps in rainfall during the wet season that forced herders to leave zones early and re-use previously visited zones, 3) seasonal migrations by herders across neighboring conservancies, and 4) encroachment by herders from pastoralist communities outside the study area. Some of these factors – water holes, restricted migration routes – are spatially predictable, which was observed from the map of frequency of success (Fig. 1): areas experiencing the least consistent success were found in 1) the southwest where herders from multiple communities moved between several protected areas, such as the Mathews and Mukugodo Forests and Samburu, Buffalo Springs, and Shaba National Reserves on their way to dry season ranges in Sera, Biliqo Bulesa, and Melako Conservancies, and 2) in the north and east in the vicinity of permanent water holes, such as near Kisima Hamsini (translated from Kiswahili as 50 wells) in eastern Sera Conservancy and Kom in northwestern Biliqo Bulesa Conservancy. These deviations led to an average of 18.9%, on average, of the study area experiencing 2 or more grazing events per year with dry years with < 400 mm of rainfall exhibiting higher frequencies of 2 or more grazing events per year (Fig. 3).

Livestock Presence and Impacts

Monthly impacts assessed by $\Delta NDVI_{rel,i,m}$ were then used to determine how many months of the year a given pixel location experienced livestock impacts sufficient for $\Delta NDVI_{rel,i,m} > 0.05$. We calculated the annual sum of monthly relative impacts and found that a threshold of SUMΔNDVI < 0.05 ($5x\ 10^6$) in the raw data) would identify MODIS pixels (250 x 250 m) where rotational grazing practices were successful ($\leq$ 1 grazing event) each year. For a given pixel *i*, NDVI measures are determined for two consecutive months, *m*, *m+1*, in non-reference pixels where success in grazing management is being evaluated and

$$\Delta NDVI_{rel,i,m} = \sum_{m=1}^{12}[(NDVI_{i,m,nonref} - NDVI_{i,m+1,nonref}) - (mNDVI_{m,ref} - mNDVI_{m+1,ref})] \quad (S1)$$

where

$NDVI_{i,m,nonref}$ = NDVI measure from pixel *i* in month *m* in non-reference areas where livestock routinely graze.

$mNDVI_{m,ref}$ = Mean NDVI measure across all pixels in month *m* in reference area polygons

$\Delta NDVI_{rel,i,m}$ = Net relative change in NDVI in non-reference pixel *i* in month *m*.

Reference pixels are those in designated polygons that receive little to no grazing because of management (e.g., wildlife sanctuaries in Il Ngwesi, Westgate, and Sera Conservancies) or topographic inaccessibility (e.g., steep-walled plateaus in Kalama). Monthly changes in NDVI in reference pixels help account for weather related changes in NDVI. These calculations were performed in Google Earth Engine (https://earthengine.google.com).

Estimating Grazing Intensity

The protocol was employed in February and early December 2012:

1. Is grass present? If Yes – Go to 2. If No. Grazing Intensity = 99%
2. Is grass predominantly Perennials? If Yes Go to 4. If No, Go to 3.
3. Grazing Intensity = Percent Bare Ground
4. Is Bare Ground greater than 50%? If Yes, Go to 5. If No, Go to 6.
5. Grazing Intensity = 55% + (1/2)*Percent Bare Ground
6. Grazing Intensity = 1- (Average Height of Grass in the open (no shade) in a 10 meter radius/Maximum height of grass under a Shrub [presumed to be ungrazed due to shrub protection])

Sensitivity of NDVI thresholds

These calibrations were used as the basis for determining thresholds of relative change in NDVI for monthly livestock presence in a MODIS pixel *i* $\Delta NDVI_{rel,i,m}^* \geq 0.05$ that generated our estimates of success in grazing management. The sensitivity was analyzed by calculating the expected $\Delta NDVI_{rel,i,m}$ where the probability of expected livestock presence = 50% for the upper and lower confidence limits of the logistic regression for monthly livestock impact (**Extended data** Fig. S4) and found that $0.02 < \Delta NDVI_{rel,i,m}^* < 0.11$. For the upper confidence interval of the regression, the $\Delta NDVI_{rel,i,m}^* = 0.02$, 52% of $\Delta NDVI_{rel,i,m}$ values across the landscape were above that threshold, suggesting that that livestock occupied 52% of the landscape in any given month. In contrast, for the lower confidence limit of the regression, $\Delta NDVI_{rel,i,m}^* \geq 0.11$, a value suggesting that livestock occurred on only 7.5% of the landscape in any given month. These two upper and lower limits to the thresholds bracket the expected livestock incidence predicted by $\Delta NDVI_{rel,i,m}^* \geq 0.05$ of 26%, which is higher than incidence of livestock expected under successful grazing management:1 out of 8 different grazing zones (12.5 % of landscape), including both wet and dry season ranges, in a given month. However, given deviations from grazing plans that occurred due to weather, restricted migration routes, limited access to water, etc., observed livestock occupancy would have been expected to exceed 12.5%. Thus, while the analysis is fairly sensitive to choice of $\Delta NDVI_{rel,i,m}^*$, expected livestock incidences from planned grazing roughly corroborate our use of the threshold $\Delta NDVI_{rel,i,m}^* \geq 0.05$. These sensitivities transfer to thresholds of $SUM\Delta NDVI_{rel,y}^*$ for the year, where $SUM\Delta NDVI_{rel,y}^* \geq 0.02$ led to the conclusion that only about 42% of the landscape experienced single grazing events in a year, while $SUM\Delta NDVI_{rel,y}^* \geq 0.11$ suggests that single grazing events occurred across 83% of the landscape.

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