ACCESS, LIVELIHOOD-VULNERABILITY, AND LANDSCAPE-LEVEL VEGETATION CHANGE IN LAIKIPIA, KENYA

by

RYAN ROBERT UNKS

(Under the Direction of Elizabeth G. King)

ABSTRACT

Mobility in semi-arid lands is essential for wildlife and herders alike to buffer spatially and temporally variable key resources. On the Laikipia Plateau, recent decreases in pastoralists' seasonal grazing access are related to shifts in land tenure, decreasing porosity of boundary lines, wildlife conservation, and increasing intensity of conflicts in surrounding lands. Interrelated to these changes are the influences of wildlife NGOs in pastoralist governance and as employers. We used an interdisciplinary approach, drawing from ethnographic and landscape ecological methods, to ask how changes in informal and formal institutions have impacted herding ecology over the past 30 years. Using qualitative and quantitative methods we explored how the restricted mobility of pastoralist herding at one group ranch has led to cascading social and livelihood changes, and how today both institutional and biophysical factors create new compounded stressors that are experienced unevenly. Institutional change has led to reserve forage access being available in very few areas, requiring either numerous household assets, or relationships with private land owners to gain access. Shifting norms of cooperative herding, new institutions shaping access, and employment all appear to be exacerbating inequality and stratifying herding strategies, with most relying on small amounts of goats, or illicit grazing, to subsist. We built upon this understanding of reorganization of herding to analyze how novel pressure relates to vegetation changes that are frequently attributed to livestock, in contrast with pastoralists' own accounts. We used GIS methods to estimate pressure and test correlations between metrics of vegetation change. Most changes did not show meaningful correlations to livestock. There was little evidence that the most dramatic changes detected, with 18% of the area experiencing loss of *Euphorbia spp.* canopy, and 37% of the land experiencing shrub encroachment, were related to livestock. Correlations with livestock pressure implied that cattle may have had impacts on grasses during the dry season and contributed to increases in two encroaching understory species. Smalll stock estimates were correlated with some decreases in shrubs, vines, and grasses. We interpret human-environment interactions as embedded in complex social, political, and economic context and make recommendations to inform livelihood and conservation policy.

INDEX WORDS: Kenya, Laikipia, Landscape Ecology, LeUaso, Livelihoods, Mukogodo Division, Plant Community Ecology, Pastoralism, Political Ecology, Remote Sensing, Social Ecological Systems, Vulnerability

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DEDICATION

This dissertation is dedicated to the people of Koija Group Ranch.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

The framing of much of this dissertation has been influenced by taking two different perspectives on human and environment relationships. The first one, drawing from the closely related frameworks of social-ecological systems (SES) and coupled human and natural systems, is rooted in complex systems theory and has origins in biophysical science. The other perspective, critical political ecology, has a lineage tracing back to dialectical thinking and political economy critiques of cultural ecology, with origins in questioning the framing of historical human-environment studies (Blaikie 1985). Considering past paradigms that have oversimplified the dynamics of ecosystems (Holling 1973), or imposed excessively top down logics (Scott 1998), both of these perspectives have provided novel insights into human-environment studies over the past several decades.

These perspectives informed a case-study, split into four chapters, on changes in pastoralist livelihoods and semi-arid rangelands in central Kenya over the past 30 years. In semi-arid lands, concepts such as fixed carrying capacity or maximum sustained yield lose traction when applied to livestock management, due to the high seasonal and spatial variability in rainfall and vegetation (Ellis and Swift 1988, Vetter 2005). This leads to a need for high levels of flexibility in access (Mwangi and Ostrom 2009), as uncertainties are compounded from herders' perspectives when social access to resources is spatially variable (Ash et al. 2002, McPeak 2003). As vegetation resources and ecological heterogeneity are tightly linked to both livelihoods and biodiversity concerns, a number of disciplinary specializations in the natural and social sciences have emerged to try to grapple with the non-equilibrial, multi-scalar, complex interactions that need to be considered in a robust socio-ecological analysis of pastoralism (Behnke et al. 1993, Ellis and Swift 1988, McCabe 2003).

One heuristic that has been proposed as a framework to understand the ability of an ecosystem to experience fluctuating conditions but to return to one stable state rather than shift to another stable state is known as resilience (Holling 1973). This is distinct from a system's resistance, or ability to buffer changing conditions without exhibiting changes (Holling 1996). When a system is experiencing a decrease in resilience, for example due to gradually changing underlying conditions such as nutrient levels or climate, this means that the system is more sensitive to perturbations. Importantly, it is thought that gradual changes in ecosystem conditions may decrease the resilience of the system with little observable difference, but resulting in increased possibility of catastrophic shift (Sheffer et al. 2003). This perspective has origins in ecological science, but has been used as a basis, informed by complex systems theory, for understandings of the complex interaction of social and ecological factors at different temporal and spatial scales (Olsson 2015). Resilience-based human-environment framings fundamentally recognize that multi-equilibrium or non-equilibrium states might be exhibited by ecosystems (Briske et al. 2003) and that it is appropriate that this is reflected in interpretations that affect social systems and vice versa (Berkes and Folke 1998, Turner et al. 2003). The SES perspective, drawing in large part from ecological concepts and complex systems theory, provides a basis for analysis of the dynamics of interaction of social and ecological factors at different temporal and spatial scales (Berkes and Folke 1998, Holling 1973, Walker et al. 2004, Ostrom 2009) as well as the ability to move beyond individual actor models (Nelson et al. 2007).

Social-ecological systems research is especially adept at providing a systematic approach to understanding how livelihoods and ecological process are intertwined and can cascade, causing interrelated shifts between alternate states (Kinzig et al. 2006).

Approaches building on this body of theory have been utilized extensively in rangeland ecology because of the ability to improve upon the shortcomings of previous models and practices applied to the ecology of semi-arid rangelands (Anderies et al. 2004, Bestelmeyer and Briske 2012, Briske et al. 2003, Briske et al. 2005, Walker et al. 1993, Westoby et al. 1989). Drylands have provided a robust empirical study system for testing theory informed by systems models (e.g. Ludwig et al. 2005). Past equilibrium understandings utilized in rangelands tended to emphasize how different human land uses might impact ecological integrity and were focused upon sustaining the maximal use of rangeland resources by applying a fixed stocking rate in hopes of sustaining the productivity of rangelands. However, in semi-arid rangelands with highly variable rainfall, it has been increasingly recognized that non-equilibrium conditions might instead prevail (Ellis and Swift 1988). Non-equilibrium understandings of these rangelands concluded that high variability of rainfall leads to high fluctuations in vegetation as well as herbivores, and under conditions where animals either migrated or died due to droughts, the impacts of herbivores on vegetation was negligible (Ellis and Swift 1988, Briske et al. 2003, Vetter 2005).

Non-equilibrium understandings of the ecological dynamics of semi-arid rangelands allow for explicit consideration of high seasonal and spatial variability in rainfall and vegetation that might be of greater relevance than stocking rates (Ellis and Swift 1988). This in turn links to how livelihood uncertainties are compounded when social access to resources is spatially variable (Ash et al. 2002, Hobbs et al. 2008). While there is a history of polarization surrounding the implications of non-equilibrium understandings of ecology for pastoralist systems (Vetter 2005), the growing consensus is that semi-arid rangelands express traits along a continuum from equilibrium to non-equilibrium (Briske et al. 2003, Boone et al. 2011). Consequently, the pressure, timing, and duration of livestock herbivory, as well as the coefficient of variation of rainfall, can in some instances be of greater relevance for vegetation than stocking rates (Vetter 2005, Boone et al. 2011). The stability of these semi-arid ecosystems has also been well-documented to sometimes exhibit dynamic traits that bear a strong resemblance to "feedback" mechanisms, and to potentially exhibit non-linear, abrupt responses to changes in conditions, making these theoretical understandings more suitable (Scheffer et al. 2001). These perspectives, taken together, allow for greater contextualization and understanding of wider changes and contingency of landscape transitions (Bestelmeyer et al. 2011, Briske et al. 2005).

These more recent understandings of ecosystem dynamics have been incorporated into understandings in the social sciences as well (Scoones 1999, Zimmerer 1994). A large amount of this work has been done under the banner of political ecology, a diverse approach to examining human-environmental interactions that in itself is not a coherent body of theory (Peet and Watts 2004) as much as it is a loosely grouped body of works of critical scholarly inquiry. While more heterogeneous and less theory-driven, it represents a multi-scalar, non-equilibrial, interdisciplinary body of work, with numerous applications in rangeland studies (e.g. Turner 1993, Able and Blaikie 1989). Early political ecology combined the concerns of political economy and cultural ecology, and sought to understand "forms of access and control over resources and their implication for environmental health and sustainable livelihoods" and eschews simple understandings of human-environment relations (Peet and Watts 2004).Historically, there was a strong component of engaging the biophysical (Blaikie 1985, Fairhead and Leach 1996), but more recently the scope of political ecology has broadened greatly, and draws from heterogeneous fields such as post-colonial studies, feminist studies, and a wide range of post-structuralist thought.

Political ecology is of high relevance for pastoralist studies in Kenya due to the well documented influence of a number of narratives about pastoralist property regimes, herding economy, and rangeland ecology on development initiatives. These historically prominent narratives often focused on how pastoralist common property regimes were inadequate to prevent degradation, or how human population and livestock levels were leading to a lack of balance between livelihoods and ecological processes (Able and Blaikie 1989, Fairhead and Leach 1996, Nelson 2012, Turner 1993). Historical and recent Kenyan policies have sometimes focused on the need to reduce livestock densities, but have been linked to an equilibrium understanding that lead to semi-arid rangelands being thought of as stable and having a set carrying-capacity (Anderson 2002, Roba and Oba 2008, Turner 1993). Some have drawn links between equilibrium interpretations of the ecological impacts of livestock and misunderstandings of pastoralist livelihoods (Little 1994, McCabe 2004) that have led to policies that overlook concerns from pastoralists' perspectives. Others have drawn a direct parallel between the poles of the equilibrium and non-equilibrium ecology perspectives, and two contrasting types of policies that are advocated for pastoralist rangelands (McCabe 2004, Moritz 2008). These authors have indicated how the equilibrium perspective in rangeland studies is closely aligned with what has been called a "modernization" paradigm where it is advocated that pastoralists should become settled and adopt agricultural livelihoods, contrasted with how a non-equilibrium perspective is closely aligned with a "mobility" paradigm, and where the importance of seasonal movements is instead emphasized (McCabe 2004, Moritz 2008).

Integrative Framing

Both resilience-based approaches and critical political ecology have attempted to grapple with complex multi scalar interactions in human-environment problems. Despite the potential for constructive engagement (Peterson 2000), as political ecology and resilience perspectives have both "embraced unbounded, open-ended, historically contingent changes in human-environment relations", there has been somewhat limited engagement between the two approaches (Turner 2014, but see Fabinyi et al. 2014, Widgren 2012, Ingalls 2016, Stone-Jovicich 2015). This is not without good reason (Olsson 2015) and a recent review of the synergies of resilience and political ecology perspectives emphasizes how SES approaches usually emphasize certain theories of human behavior that are most compatible with ecological understandings, such as ecological economics and rational choice theory (Turner 2014). While embracing a framework for non-equilibrium landscape heterogeneity, and historical contingency for ecological dynamics, resilience thinking has also come under fire for fostering static, homogeneous conceptions of society and politics (Hatt 2013). Similarly, using the principles of ecology for understanding social organization may result in a lack of ability to explicitly address issues of power (Brown 2014), leading to a lack of understanding of larger economic forces, as well as a tendency to idealize or oversimplify social relationships by using social science approaches that have greater similarity to ecological perspectives (Cote and Nightingale 2012). This is especially conspicuous when considering the reliance on methodological individualism in much of SES research (Ollson 2015), and the alignment of concepts of self-organization with economic concepts that tend to view actors as rationally self-interested economic actors (Walker and Cooper 2011). SES scholars have come under fire for appearing to view society in a functionalist manner, or with a "top-down" view of "society as organization" (Hatt 2013, Walker and Cooper 2011). Other critiques have involved the way the greater ability to buffer vulnerability (adaptive capacity) of wealthier or more powerful individuals over others has been naturalized (Turner 2014, Watts 2011).

While scholars within the SES perspective have perhaps overlooked links to relevant studies in the social sciences, it is perhaps a lost opportunity when critiques of resilience are used to dismiss it wholesale (Turner 2014, Olsson 2015). The aforementioned shortcomings have not been shown to be inherent to perspectives that draw from understandings of resilience, though the potential for misuse, reification, and overreach must be acknowledged. For example, there is nothing inherent to the SES perspective that bounds the frame of inquiry to exclude factors which are inherently political (Turner 2014). As classic political ecology work shows, restrictive bounding has been common in many human-environment studies by ecologists, and is not just limited to the resilience literature (Blaikie 1985, Turner 1993, Watts and Bohle 1993, Fairhead and Leach 1996).

In using resilience for problem-based research, SES can instead be used as a bridging concept between ecological science and other fields with potential for novel, integrative, synthetic research if more pluralistic approaches are taken (Turner 2014, Olsson 2015). While there is a tendency in SES to frame humans homogeneously, without nuance or sufficient social context, or as an exogenous perturbation that pushes stability toward one stable state or another, it is also possible to add nuance to these types of analysis through borrowing from other perspectives. Others have pointed out how systems concepts such as resilience or regime shifts to characterize complexity in ecosystems alongside political ecology can foster explicit consideration of social processes of power and governance at multiple scales, and thereby lead to an expanded understanding of “how variable access shapes material engagement with the biophysical world" (Turner 2014). Classic biophysical political ecology (Blaikie 1985, Turner 1993, and Abel and Blaikie 1989) analyzed processes such as land degradation as inherently contextual and embedded in multiple scales of political and economic factors. This work called into question many equilibrium-based ecological accounts that have attributed degradation in semi-arid lands largely to localized land use, population growth, or overstocking (Abel and Blaikie 1989, Turner 1993), and has embraced biophysical analysis but at the same time has often contested strictly ecological understandings of these processes. These so-called "critical realist" (Bhaskar 1991, Sayer 1993, Forsyth 2003, Blaikie 2012) approaches in political ecology enable complex understandings of resource-related conflicts that attempt to move beyond scarcity-driven explanations (Turner 2004, LeBillon 2001) and can aid in bridging epistemologies (Goldman et al. 2010).

Political ecology can supplement ecological understandings by adding analysis of how differential access changes according to resources, as well as to power and governance, with cascading implications for human-environment relations (Turner 2014). This can add understanding to the historical, political, economic, and discursive context of livelihoods (Scoones 2009, Carr 2013) and the "chains of causation" of environmental problems (Blaikie 1985). Political ecology enables alternate, complex understandings of resource-related conflicts that attempt to move beyond scarcity-driven or proximate explanations (LeBillon 2001). This approach can also enable consideration of historical factors that influence current livelihood outcomes by drawing from social science that has examined the influence of colonial legacies. For example, particularly relevant to pastoralist studies is the tendency for the colonial authority to present ethnic groups as "nations in miniature" with distinct lines between them (Broch-Due 2000). The colonial process in this area affected not only the way that ethnic identify was portrayed, but also limited the potentials of different livelihoods (Cronk 2004, Spear 1993). However, there is often a lack of engagement with this literature from conservation perspectives, and these critical perspectives frequently do not gain robust consideration in conservation planning.

The approach taken in the four chapters that follow comes from understanding these two perspectives side by side. I have attempted, whenever possible, to write in a manner that is mutually intelligible between scholars of these disparate disciplines, to avoid the jargon and reifications possible within both modes of inquiry. My approach does not however, consider these disparate bodies of work as being fully complementary or capable of synthesis. An ability to synthesize indeed would imply there is an epistemological consistency between these perspectives (Nightingale 2016) or a flattening of the differences between the approaches. I instead draw from the two frameworks as partial, and often contradictory, and as a result able to result in exciting synergies, as well as to spark internal dialogues within one's work – thus enabling each to serve as checks and balances from different perspectives. It is my view that so-called "integrative" frameworks -- bringing two disciplinary lenses side by side -- can lead to emergent insights, the internalization of critiques of the other disciplinary perspective, and lead to growth rather than polarization. In writing this dissertation, I found this tension to be a productive one, encouraging creativity in modes of inquiry and methodology alike. I used biophysical analysis at that landscape scale drawing from resilience-based understandings and the well-developed toolset that scholars using the resilience perspective have developed for understanding semi-arid vegetation change. I explicitly linked these ecological changes to livelihood changes, and applied livelihood and institutional perspectives informed by political ecology to consider the role of the international political and economic factors that create structural influences on local practices. This work was driven in part by the hope that such approaches can inject new relevance into research on how to balance livelihoods while avoiding damage to the ecological functions and biodiversity those livelihoods often rely on. More pluralistic approaches are needed that can consider the multiple perspectives and different ways of knowing of different actors (Goldman 2003, Haraway 1988, Nightingale 2016) in this age variously dubbed the Anthropocene (Crutzen 2002), Capitalocene (Moore 2017), or Plantationocene (Tsing 2015).

I chose to focus my work on Koija group ranch in central Kenya after traveling to central Kenya with my advisor, Lizzie King, in 2012, as her field technician and soon to be graduate student. I had chosen to work with Lizzie based upon our mutual interests in restoration ecology, the interaction of ecological change and livelihoods, and different multi-scalar approaches being used in restoration ecology to address complex environmental problems. Visiting Koija, a Maa-speaking pastoralist herding community, led to me wanting to do research which might speak directly to the day to day concerns of the people who live there. Having a background in plant ecology and restoration ecology, I began designing a study with a focus on vegetation, but in bringing together literatures on livelihoods, entitlements, access, and resilience, a perspective emerged that explicitly engaged herding livelihoods as a central factor in my research framing. However, with herding the only currently viable livelihood for the area, and given the tight links of herding to ecological factors, a robust consideration of livelihoods is inherently also a question of ecological processes. These factors led to an interest in more pluralistic policies rather than those driven by goals such as maximum sustained yield, one that was open to multiple understandings of local factors, human agency, and cognizant of the unique outcomes that can occur in different contexts.

My approach began with one rooted in institutional analysis from a SES approach initially (Ostrom 2015), but it became clear that while there were internal problems with management of grazing resources within Koija, external constraints in productivity and migration were key (Appendix A). Drawing from oral histories that emerged in focus group discussions (Appendix A), many elders indicated how Koija has experienced a gradient of animal use with different watering points being used with greater frequency, and that historically there have been areas of restriction during different seasons. However, that increasing frequency and intensity of drought, and a lack sufficiency of reserve forage areas during dry seasons has become apparent resulting in most of the herbaceous vegetation resources being exhausted quickly, confirming a major barrier to sustainability observed by others (Herren 1991, Letai and Lind 2013, Muthiani et al. 2011). Many scholars have emphasized the cascading impacts of the spatial fragmentation of rangelands on social and ecological factors in areas where subsistence pastoralism is the dominant livelihood (Galvin 2009, Hobbs et al. 2008, Homewood et al. 2009). This literature prompted me to maintain a focus on vegetation changes, an understudied element of rangeland fragmentation, and on landscape vegetation changes. Bringing these bodies of literature led to a strong need for first understanding the social dynamics underlying changes in herding practices to then determine what ecological questions should be asked, and at what scale they should be considered to have the most relevance when considering the history of Koija.

Much of the initial inquiry followed an abductive method, where rather than beginning with deductive, hypothesis-testing driven research, my research program began with exploratory techniques, and exploration of the data, generating multiple working hypotheses, e.g. that local land uses are driving vegetation changes, contrasted with drought driving vegetation changes. This process led, ultimately, to critical examination throughout the chapters of the multiple ways that livestock is and livelihoods are interacting with vegetation changes while at the same time examining the drivers of livelihood challenges.

Using an approach with much in common with grounded theory (Charmaz 2007), while using methods of participant observation, focus groups, and interviews, I attempted to collect rich, diverse data on everyday individual and collective experiences. This research was focused on presenting broader trends and then asked higher-order questions. Multiple periods of field work, separated by periods of analysis of elicited texts (interviews, focus groups) and survey data, allowed for iteration of analyses. What emerged from these data was a collective experience of constraints at Koija, that all households experienced and were stressed by, but which resulted in a diversity of responses and outcomes, requiring gathering of a diversity of perspectives to accurately explain the differential outcomes. This led to use of the access (Ribot and Peluso 2003) and environmental entitlements approaches (Leach et al. 1999) to understand the different pathways of access and livelihood outcomes. Building upon these initial understandings, using an ethnographic approach to guide exploration of wider trends in survey data, I tried to also understand what factors have shaped the social processes of sedentarization and transformation of livestock husbandry practices that have occurred historically.

The research process began with an exploratory trip in 2012, and a return trip in 2013 to conduct focus group discussions, collect vegetation data, and design a survey for Naiputari Paul Wachira, the invaluable field technician on this project who conducted all surveys and translated all interviews, among many other tasks to conduct with all households. Then in 2014 we conducted in-depth key informant interviews with elder herders on livelihoods and vegetation changes, completed detailed transect-based vegetation sampling across hilltops near the Ewaso Ng’iro river, as well as vegetation sampling in plots distributed across the landscape that aligned with Landsat satellite pixels. In 2015, we completed additional key informant interviews with elder herders and collected vegetation data. Two distinct periods of change in herding were identified, one approximately near 1984, when the majority of cattle were lost during droughts, and one in 2002 when formalization of group ranches occurred. Based upon these discussions, I then created household surveys to gather data on salient factors that were thought to be closely related to herding practices. The intent of these surveys was to establish a baseline data set of household entitlements that include factors such as sites accessed, past sites they no longer access, the last time a household visited reserve forage sites, a given household's ability to split herds, the type of access (e.g. paid, unpaid, through employer relationship), herding labor, herd size and composition, herd off-take rates, labor sharing, and animal sharing.

This led to four distinct, but interrelated research components represented in Figure 1.1. One was a broad focus on the social and institutional processes that mediate livestock herding as well as the implications of changes in access for livelihoods and vegetation within Koija (Chapter 2). Due to recent changes in exclusion policies, formalization of boundaries, conservancy formation, privatization, and conflict, seasonal grazing in areas that were formerly open for grazing has decreased, leading to a number of livelihood changes (Chapter 2). Chapter 2 uses a SES metaphor of scale mismatch (Cummings 2006), or misalignment between institutions and ecological processes, supplemented by a political ecology exploration of the historical, material, and discursive context of institutional change. This evolved into an analysis of a number of institutional constraints that have arisen out of a specific social reorganization of the landscape that began during the colonial era, and ultimately led to an improved understanding of individualization and social unevenness in the way that institutions and mobility interact. This also led to improved understanding of how such a mismatch between pastoralist livelihoods and ecological process has been produced historically, and has come to be reinforced, contradictorily, in the name of wildlife conservation, which seeks to align institutions and wildlife corridors. Chapter 3 then asks whether, and how, the new landscape of institutions that underlies this scale mismatch shapes processes of access and livelihood vulnerability (Figure 1.1) that today has differential impacts among households at Koija. Access to seasonally strategic grazing is dependent on household factors such as herding labor, access to cash through external employment or livestock sales, household ability to split herds, and also social relations (Figure 1.1, Chapter 3). Some households are excluded from certain types of access and the resulting benefit streams, while access and the vulnerability of herders to drought and vegetation changes are inherently intertwined. The objective of Chapter 3 was to determine how institutional factors are mediating herding practices over time at different spatial scales, and how these changes are impacting the vulnerability of herder livelihoods to drought and ecological changes (Figure 1.1). This approach allowed for a systematic accounting of the ways that individuals access and use resources, and how these are shaped by rules and norms, which in turn impact individuals' livelihoods (Chapter 3, Figure 1.1). This merged fluidly with recent work that is critical of a tendency within studies of vulnerability to mask historical and political factors, which can lead to a focus on the local ability of land users to cope and respond, while leaving out wider constraints and the ultimate causes of vulnerability (Ribot 2010, 2014).

Then, in bridging the social and ecologic components, for an article in preparation (Appendix H), I conducted interviews on ecological change to contrast the ways that pastoralists and conservation actors view recent ecological change. This then informed chapters 4 and 5, which focused on developing ways of testing hypotheses about the complex vegetation changes that have happened on Koija over the past 30 years. Using this, along with the above approaches, allowed for a simultaneous exploration of how changes in access and land use relate to changes in landscape-ecological processes (Figure 1.1), estimated using a hybrid approach drawing from landscape ecology and household survey data. Chapter 4 focused on how, methodologically, to estimate herbivory at the landscape scale in rangeland studies, for use in understanding landscape process (Figure 1.1). Chapter 5 then takes an interdisciplinary approach informed by oral histories, but focuses on how to analyze vegetation change (Figure 1.1) using both remote sensing and plot-based data over the past 30 years with respect to the gradients of herbivory estimated in Chapter 4 to consider how in light of development projects that appear to be couched in assumptions of equilibrium conditions, what are the dominant drivers of changes in vegetation This change in landscape-level process was linked to shifts in vegetation, drawing upon state and transitions informed by non-equilibrium ecology to look at changes within Koija, but also aimed to understand the patterns in changes in vegetation that have occurred outside Koija as well. Finally, drawing upon herder understandings of ecological process (Appendix H), the analysis of vegetation shifts across Koija, and herder strategies, I then tried to account for how these shifts in vegetation have in turn also historically affected livelihoods (Chapter 1).

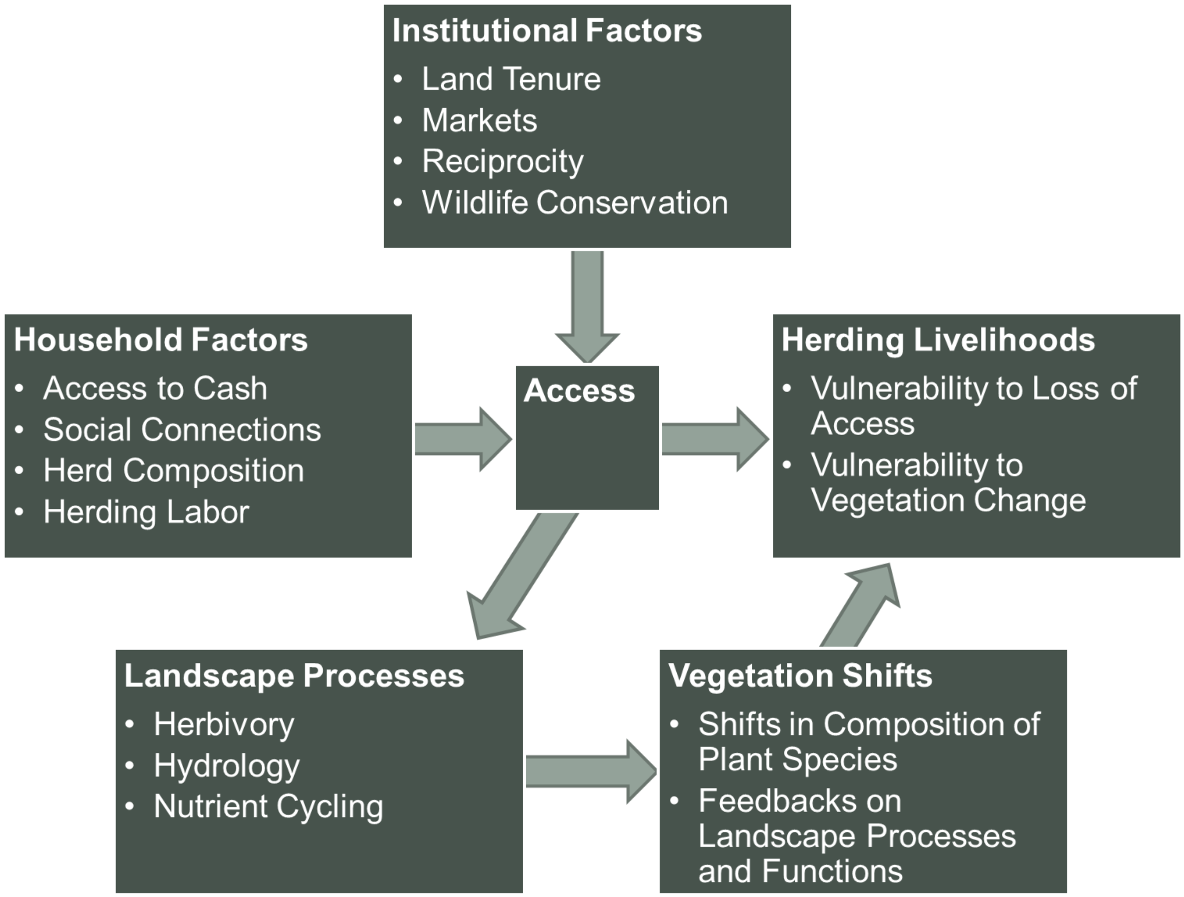

Figure 1.1 Conceptual framework

Strategic Communication

The last time I visited Koija, in May of 2016, I presented my preliminary research results in an open community discussion (*baraza*) at Koija, leaving space for discussion, interruption, and debate. These discussions, which are frequent at Koija, provide an opportunity for elders to discuss and debate different aspects of a given topic in a respectful public setting. This was an incredibly useful exercise in terms of validating the research, because it enabled a community-wide summary of the views that had been expressed individually. I therefore encouraged elders to communicate to me if they disagreed with the conclusions of the research, and to debate it amongst themselves. I also inquired whether there was anything they would like to see come of this research.

This coincided with an internship I was at the time completing with the Nature Conservancy on Loisaba Conservancy in Laikipia. While at Loisaba, which is a large, 56,000-acre ranch directly across the river from Koija, with a long history of relations with Koija, I participated in regular outreach discussions with Loisaba and TNC staff, presenting my research on numerous occasions, including at nearby Mpala Research Center where a number of wildlife-conservation organizations and researchers regularly meet in Laikipia. This finally culminated in me facilitating a discussion between Loisaba staff and Koija, focused on community relations and grazing access. In these conversations, I put myself in a role where I provided my results from my research on Koija residents' livelihoods, preliminary findings of vegetation changes, and sustainability challenges of the region to try to stimulate dialogue. I did this in an open-ended fashion, in an attempt to open up discussion about historical access and livelihood needs at a time when different future policies were being discussed. This was done explicitly to attempt to encourage greater mutual understanding between conservation actors and pastoralists, and to question entrenched narratives and assumptions in an open, honest, and respectful dialogue. At the end of this internship, I prepared a research brief that provided a summary of my research findings from Chapters 2 and 3 of my dissertation to a number of conservation and development NGOs throughout Laikipia (Appendix I) and solicited feedback and attempted to engage in critical dialogue. I additionally will provide a summary of the research findings from Chapters 4 and 5 of my dissertation to these same organizations.

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CHAPTER 2

UNEVENNESS IN SCALE MISMATCHES: INTERRELATED SOCIO-ECOLOGICAL CHANGES IN PASTORALIST LIVESTOCK HUSBANDRY IN LAIKIPIA, KENYA[1]

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Abstract

This paper focuses on how political, economic, and biophysical factors shape rules and norms that can create constraints on livelihoods. These can affect how land use and ecological processes align at different scales, leading to context-specific impacts on livelihoods and ecological outcomes. We used a mixed-methods, multi-scale analysis of livelihoods and institutions to track historical changes in a herding community in central Kenya. We asked how pastoralist livelihoods have adapted within constraints formed during the colonial era, the independence era, and more recently due to wildlife conservation, and related livelihood evolution to patterns of loss of forage access, changes in formal and informal herding institutions, relational dimensions of wildlife conservation, and changing ecological conditions. Drawing from social-ecological systems theory and critical social science perspectives, we detail how political and economic factors have interacted historically with herding ecology to shape outcomes that are linked to global factors, but are unique to the Laikipia context. We discuss how efforts by wildlife conservation actors to align social factors and ecological processes in favor of a specific vision of landscape sustainability has been intertwined in a historically-rooted mismatch between dry season pastoralist livestock mobility and seasonal variability in vegetation. We add to the literature on interpretations of scale mismatch by adding understandings of multiple dimensions of unevenness in how institutional landscapes are produced. Using disparate methods of inquiry side by side, this work shows how in analysis of socio-ecological interactions, a lack of nuanced consideration of social and political factors can, perhaps unintentionally, exacerbate misalignment between livelihoods and ecological process.

Introduction

Pastoralism is the current primary livelihood of four million people living in arid or semi-arid lands in Kenya (Kirkbride and Grahn 2008) that do not support agriculture due to lack of adequate rainfall (Niamir-Fuller 1999). Customary pastoralism involves flexible institutions for securing access to seasonally variable common pool water and pasture resources, and coordination across levels of social organization in times of environmental stress (Blewett 1995, Mwangi and Ostrom 2009). However, pastoralism is a less well-suited livelihood for drylands when seasonal grazing movements cannot occur (Fratkin 2001). Fragmentation of the connectivity of herding ranges can lead to a decreased efficacy of pastoralism as a subsistence practice (Galvin et al. 2008, Hobbs et al. 2008). This fragmentation has been interpreted as a mismatch between the scale of variation in ecological processes and the scale of seasonal movement required to sustain livelihoods (Du Toit et al. 2010).

Building upon an historical literature review combined with quantitative and qualitative ethnographic field work, we analyze the ways that local livestock husbandry practices have evolved in relation to internal and external constraints in one community in Laikipia County, Kenya. As privatized models of wildlife conservation gain prominence, and non-governmental actors increasingly assume a role more typical of the state, we investigated the sequence of changes that have occurred in common property grazing institutions due to colonial, and post-independence government, and more recent non-governmental interventions. We evaluated how these institutional changes have impacted the alignment and interaction of herding livelihoods with variation in vegetation, how these might incentivize different practices, and how herders have adapted their livelihoods to these changes. We built upon understandings of constraints to livelihoods (Barrett et al. 2005, Liao et al. 2015, Speranza et al. 2014) and drivers of socio-ecological change in problem-based research. Drawing on multi-scalar understandings of livelihoods as situated within specific political and economic contexts (Carr 2013, Ribot 2010, Scoones 2009) we identify and describe how recent policies have reinforced a historical scale mismatch (Cash et al. 2006, Cummings 2006) between the institutional landscape and herding livelihoods. By incorporating insights from recent understandings that consider the role of biophysical and non-human factors in social processes (Li 2014, Mitchell 2002, Robbins 2007, Tsing 2005), and increasingly decentralized, market-oriented conservation (Igoe and Brockington 2007, Fletcher 2012), we add an understanding of how herding ecology and institutional factors have interacted in complex ways in Laikipia to shape outcomes.

In hopes of improving understanding of the interaction of social and ecological factors in a context where wildlife conservation and pastoralist herding overlap, and where international non-governmental actors play an increasing role in governance, the approach we used considers livelihood evolution in relation to material, historical, and discursive elements that underlie the way that ecological processes are aligned and that institutional landscapes are produced. Historical contingencies in Laikipia have led to sweeping changes where large private ranches, supplemented by ecotourism profits and conservation NGO support, have the ability to practice strategic, low-intensity grazing with networks of contiguous ranches that are largely unfenced (Georgiadis et al. 2007, Western et al. 2009). Due to increasing support from international wildlife conservation organizations, the requirements of wildlife conservation and low-impact commercial cattle ranching have been increasingly well-aligned with landscape ecological processes within Laikipia. At the same time, however, pastoralist herding ecology has been impacted by fragmentation of dry-season cattle ranges, with a high density of land use now occurring in areas where populations were historically concentrated on areas of lower, more variable rainfall during the colonial era. Historical changes in institutional and biophysical factors have favored a type of pastoralist herding that is increasingly individualized among households and spatially concentrated. Biophysical constraints, such as inability to sustain large herds of goats within one homestead reinforce how livestock husbandry practices have become individualized at the household scale. We argue that the contrasting institutional requirements between private ranches and pastoralists in Laikipia are inherently intertwined, where marginalization of pastoralist requirements has been actively reinforced by wildlife conservation partnerships that utilize group ranches without seasonal grazing access plans as a core element. However, some pastoralists benefit from conservation while at the same time pastoralist livestock husbandry is increasingly unequal and individualized, leading to many pastoralists seeing the benefits as beneficial. We argue that these inter-related social and ecological elements are key factors in holistic understandings of contemporary conservation outcomes, that are well understood using biophysical and critical social science framings side by side.

Social-ecological Systems and Critical Perspectives

The concepts of carrying capacity and stocking rates lose traction when strictly applied to livestock management in semi-arid lands that are highly seasonally and spatially variable in rainfall and vegetation (Ellis and Swift 1988, Vetter 2005). These uncertainties in availability of forage are compounded from herders' perspectives when social access to resources is spatially variable (Ash et al. 2002, McPeak and Barrett 2001). While there is a history of polarization in the debate of the implications of non-equilibrium understandings of ecology for policies that impact pastoralist livelihoods, the growing consensus is that semi-arid rangelands express traits along a continuum from equilibrium to non-equilibrium. Consequently, pressure, timing, and duration of livestock herbivory, as well as the coefficient of variation of rainfall, are perhaps of greater relevance than average stocking rates in explaining vegetation dynamics (Vetter 2005, Boone et al. 2011).

A number of disciplinary specializations in the natural and social sciences have emerged to attempt to grapple with non-equilibrial, multi-scalar, complex interactions in socio-ecological analyses of pastoralism (Hobbs et al. 2008, McCabe 2003), and the social sciences more generally (Scoones 1999, Zimmerer 1994). Social-ecological systems theory is one perspective that has emerged for use in understanding social factors in relation to an understanding of ecological interactions along a hierarchy of ecological levels, building from systems theory (Allen and Star 1982) and understandings of ecological scale (Levin 1992). However, a number of scholars critically examined how using the principles of ecology for understanding social organization can mask the influences of power and culture (Cote and Nightingale 2012, Walker and Cooper 2011, Brown 2014, Hatt 2013), emphasize certain theories of human behavior over others (Turner 2014), neglect understandings of larger economic forces, and idealize or oversimplify social relationships (Davidson 2010, Duit et al. 2010, Cretney 2014, Béné et al. 2012, Welsh 2014, Walsh-Dilley et al. 2013). While tending to neglect these other areas, social-ecological systems approaches can guide rigorous inquiry into complex feedbacks between social and ecological factors (Walker et al. 2004). In what follows, we reemphasize the utility of the social-ecological systems frameworks as a bridging concept (Olsson et al. 2015, Turner 2014) with potential for novel, integrative, synthetic research when combined with robust critical social science perspectives such as political ecology (Able and Blaikie 1989, Blaikie 1985, Blaikie and Brookfield 1987, Turner 1993), leading to a more robust understanding of how social and ecological factors inter-penetrate (Levins and Lewontin 1985) and how human-non-human relations can shape outcomes (Li 2014, Mitchell 2002, Tsing 2005). We treat these as two partial perspectives, with their own disciplinary biases, that when used integratively can produce novel insight into complex interactions between people and environments that neither approach could achieve by itself.

Scale mismatch

Scale mismatch, a concept rooted in the social-ecological systems perspective, builds upon hierarchy theory and spatial ecology to improve conservation outcomes by detecting when the scale of management does not match an ecological process of interest in conservation planning (Cummings 2006, Guerrero et al. 2013). This understanding can include ecological process at spatial, temporal, and functional scales interacting with different institutional scales, hierarchical governance levels, or knowledge levels. Consideration of multiple scales can lead to more effective assessments of how social and ecological processes align (Allen and Holling 2010), and understandings of scale mismatch may produce insight for policy considerations (Folke et al. 2005), when patterns of land use, institutions, and ecological processes occur at different spatial scales.

The approach we used explicitly considers the rules and norms of resource use at different scales as well as the relationship between these social processes and ecological processes (Zimmerer 1994, Scoones 1999, Cumming et al. 2006). The literature on new institutionalism attempts to understand how rules and norms create expectations of how others will act, and thus impact the outcomes of transactions between individuals (North 1990, Ostrom 2015, Agrawal 2010, Lesorogol 2008). Some examples of this approach have used institutional analysis to assess the resilience of social systems in direct relation to environmental stability and food security by examining the diversity of resources a society depends upon, as well as that society's institutional diversity (Ostrom 2015, Adger 2000, Ostrom 2009). Institutions also shape human-environment interactions at different spatial scales (Leach et al. 1999, Kepe and Scoones 1999, Turner 2014) that can influence biophysical processes (Cumming et al. 2006), shape landscape process, and in turn interact with landscape vegetation structure.

Analysis of scale mismatch provides a robust way for identifying misalignment between social and ecological processes, especially when considering how the underlying, driving factors of how some human/environment relations can become marginalized by complex historical, political, and economic contexts (Lebel et al. 2005). For example, in past considerations of scale mismatch, politics have been engaged to some extent (Cash et al. 2006), including how different actors emphasize scales that benefit their own interests and patterns of unevenness (Lebel et al. 2005), and how focus on too few dimensions can create or exacerbate new scale mismatches (Guerrero et al. 2013). While these perspectives are useful for understanding conspicuous dynamics of exclusion and conflict they are perhaps less attuned to more "intimate" aspects of power and governance (e.g. Agrawal 2005, Carr 2013). Critical perspectives have a long history of deepening understandings of unevenness and the factors underlying how one understanding of scale may be privileged and reinforced (Butt 2014, Fairhead and Leach 1996, Goldman 2003, Kull 2004, Peet and Watts 2004). For example, Ahlborg and Nightingale (2012) emphasize how these concerns can often be intertwined with the concerns of powerful actors and on how scale is socially framed by knowledge and narratives and is inherently political and the ways that certain narratives can be excluded (Leach et al. 2010). Use of critical perspectives also can help to avoid how systems metaphors and the process of "scaling-up" might potentially obscure heterogeneity, leading to a specific dominant discourse that can shape interpretation of human-environment relations (Tsing 2012). In this paper, we combine analysis of scale mismatch with specific ways that historical, material, and discursive factors interact to align social and ecological processes in the interest of specific local and international agendas. Understanding the historical ways these mismatches are produced are not just important for understanding how landscapes are contested, but can lead to more nuanced understandings of the historically contingent ways that institutional constraints form and translate into outcomes on livelihoods.

In what follows, we explicitly consider how institutions that mediate the interaction of livelihoods and ecological process can become misaligned in what in social-ecological systems is referred to as a scale mismatch. We show how changes in institutions have been constituted by and intertwined with complex historical, material, and discursive factors. We examine the interplay of local-level institutions with restrictions in land use that occurred during the colonial and post-colonial Kenyan governments, as well as shifts in institutions due to interactions with non-state actors in more recent times. Historically, we focus on the institutions that served to secure access to dry-season and drought forage, to reinforce reciprocity, and to create a "safety net" of cattle distribution and assistance. We then show how historical and ongoing decreases in shared livestock husbandry practices have reinforced present outcomes in the stratification of different types of livestock husbandry. We then explore how in more recent years, a national priority on wildlife conservation has led to specific changes in land-use policies in Laikipia county where much of the role of the state has been delegated to non-governmental actors, and sweeping governance changes have occurred. Landowners of private ranches have sought to secure boundaries to reaffirm their property rights, to promote wildlife conservation and ecotourism in areas of livestock production. At the same time conflict between pastoralist groups outside of Laikipia and wildlife conservancy formation have contributed to geographic constraints on pastoralist cattle herding. We detail how recent NGO practices have overlooked the ways in which historical interventions have shaped current pastoralist livestock husbandry practices and how wildlife-governance changes have further impacted livestock husbandry.

East African Pastoralist Institutions

Customary Maasai land tenure is a common property regime (Ostrom 1990) where individual families own livestock, but pasture and water resources belong to the group as a whole. Customary Maasai governance has a "nested" structure where councils of elders at various spatial levels of organization are responsible for governance decisions, enforcement, dispute resolutions, and sanctions (Mwangi and Ostrom 2009). The rules and norms that elders enforce allow for seasonal access and coordinated response to highly variable vegetation across the landscape to ensure adequate resources and avoid degradation of important grazing areas (Mwangi and Ostrom 2009). Customary pastoralist institutions are considered to reflect uncertain and inherently risky dryland environments (Blewett 1995), as seen in complex networks of reciprocity and "risk pooling" (McCabe 1990, Bollig 1998, Aktipis et al. 2011).

The household is the smallest unit of Maasai social organization, and while autonomous, is frequently connected with other households in a joint herding, food sharing, and residential *nkang*, or grouping of several households (plural *nkangitie*) (Grandin 1991, Spencer 1993). The next higher level of organization is the *elatia* (neighborhood or settlement) where labor is pooled and coordinated for grazing of individually-owned herds, and which have a council of elders responsible for localized grazing regulation (Mwangi and Ostrom 2009). The *elatia* is then located within an *enkutoto* or locality with a council of elders that coordinates local water resources and grazing, and who also settle disputes and ensure that proper management techniques are used to ensure rangeland productivity (Grandin 1991, Mwangi and Ostrom 2009). Each locality contains both dry and wet season grazing, and access to these areas is a right of all residents (Mwangi and Ostrom 2009). Localities then typically have access to grazing within the next largest level of organization, the *oloshon*, or section, which forms the largest livestock grazing unit (Grandin 1991).

In addition to nested councils of elders, a number of "horizontal" relationships exist at different spatial scales. Two types of mutual assistance, individual and clan-based, are core features of social organization (Grandin 1991, Potkanski 1999). Individual assistance occurs between patrilineal and affinal family members as well as between close friends or individuals that share age-set bonds, and can consist of food or livestock that is expected to be repaid (Potkanski 1999). Age-set bonds are formed among *ilmurran* (unmarried males highly trained in cattle herding), a role that all males serve from the time of circumcision until marriage (Grandin 1991). These same individual channels are used to disperse cattle geographically through loans amongst family and friends to minimize risk in case the cattle in the care of one family is lost (Potkanski 1999). Individuals with few cattle are also able to seek aid or employment through family or friends following drought or other events of cattle loss (Blewett 1995). The combined social bonds all serve a crucial means for individuals to secure seasonal grazing access across different spatial scales.

Historical Literature Review

We conducted a brief literature review on the history of pastoralist herding in Laikipia as necessary context for the reader to understand recent institutional changes as they relate to historical contingencies and the evolution of livelihoods. This focused on:

I. *Historical Marginalization*

II. *Changes in the Spatial Scale of Herding, and Historical Evolution of Livelihoods*

III. *Changes in the Authority Structure and Relations with Conservation Actors*

Historical Marginalization

Prior to the imposition of British rule, the overall regional economy in East Africa at this time was thought to be integrated and dynamic, with exchanges and fluid movements between livelihoods of pastoralists, farmers, and hunter-gatherers. This allowed pastoralist societies, such as the Maasai, survival and long-term sustainability, particularly during droughts, warfare, or periods of livestock disease (Spear 1993, Waller 1993). On the Laikpia Plateau of north central Kenya, warfare between the Laikipiak Maasai and the Purko-Kisongo Maasai sections around 1870 ended in the Laikipiak being defeated and forced north from the Laikipia Plateau, while some married into other surrounding hunter-gatherer groups (Herren 1987). By 1890, a rinderpest epidemic, introduced by European cattle, is thought to have decimated herds throughout the region, with some Purko-Kisongo Maasai losing up to 95% of their herds within a matter of days (Herren 1987). In 1904 the British were able to force the weakened Purko-Kisongo Maasai to sign a treaty confining them to two “reserves” that were a fraction of their territory at the time. A second treaty in 1911 established that the Maasai would all inhabit a single reserve at Kajiado in southern Kenya to enable the inclusion of the Laikipia Plateau as part of the “White Highlands,” a vast area stretching from northwestern Kenya to Mt. Kenya that was to be set aside for European farming and commercial ranching (Herren 1987). In 1914 the Maasai were deported to Kajiado from the Laikipia Plateau (Hughes 2006) to areas that were less desirable for European commercial ranching due to aridity. Some avoided capture and intermarried with Kikuyu farmers, Samburu pastoralists, or one of five other hunter-gatherer groups in the area (Herren 1987, Cronk 2004). In 1934 the Kenya Land Commission (also known as the Carter Commission) concluded that these hunter-gatherer groups, despite having disparate lineages were all “Dorobo”, a British term derived from the pejorative Maasai word *il-*

torrobo used to refer to hunter-gatherers (Cronk 2004). As a result, the "Dorobo Reserve" (Mukogodo Division from here on) was demarcated in 1936 on extremely arid lands of the Laikipia Plateau that were considered undesirable for European ranching and farming (Herren 1987). Today, the five lineages that are descendants of the previously mentioned groups and that inhabit this area are all Maa-speaking pastoralists, several groups of which also keep bees (Herren 1987, Cronk 2004) and supplement their diets with hunting.

Changes in the Spatial Scale of Herding, and Historical Evolution of Livelihoods

A process of increasing confinement of Mukogodo Division occurred by the early 1950's. Some of the borders of the reserve were fenced off and livestock disease quarantine areas became enforced, largely stemming seasonal grazing movements into other areas as well (Herren 1991, Letai and Lind 2013). Internal pressures on land mounted as large numbers of both Samburu and Maasai who were increasingly forced from European controlled ranching areas moved into Mukogodo Division (Herren 1987). This led to an increase of individual households selling their livestock to buy grains, which coupled with low rates of animal productivity, led to a decreasing regeneration of herds (Herren 1991). Sale of animals was largely under the terms of the African Livestock Marketing Organization (also Livestock Management Division), which required a license for trades, discouraging individual exchanges across customary horizontal lines, and led to decreased investments in traditional safety nets and a monetized economy by the early 1960's (Herren 1991). During the drought of 1964 animal numbers became so low that it was impossible for pastoralists to subsist upon livestock alone, and for all but the wealthiest households it became a necessity to sell animals to buy grains, with most transitioning to heavily maize-based diets at this time, and many forced to migrate in search of paid labor (Herren 1991). Additionally, families began to invest more heavily in small stock (sheep and goats), as they are more drought-tolerant than cattle, have higher rates of reproduction, and are more easily sold (Herren 1991). Increases in livestock wealth made through individual's herd reproduction capacity were offset by a need for market off-take (Herren 1991). The customary system of mutual loans was only thought to be significant for the wealthiest families at this point, and the poorest families had decreased their investment in traditional security networks where animals were exchanged to distribute wealth in a security net (Herren 1991).

The East Africa Royal Commission (also referred to as the Dow Commission) of 1952 deemed the common property regime of the Maasai to be the root cause of land degradation in rangelands, and recommended that subdivision and private property rights should be the goals of policy (Mwangi and Ostrom 2009). The Swynnerton Plan of 1955 mandated that pastoralist reserves destock below a set carrying capacity, that access to markets be assured, that a permanent water source be developed, and that owners should manage grazing to a controlled and productive level (Grandin 1991). This policy was based in part upon assumptions that a "cattle complex" that led pastoralists to irrationally accumulate cattle, leading to negative impacts on vegetation and soil erosion (Blewett 1995). Compared to the southern Maasai reserves, little government presence other than minimal infrastructure, promotion of livestock sales, a historical cattle tax and forced vaccinations was noted at Mukogodo Division at the time, while grazing controls and stocking levels were thought to have been "half-heartedly" enforced (Herren 1987).

Following Kenyan independence in 1963, many of the large ranches of the White Highlands that did not remain property of Europeans were consolidated by land buying companies under the presidency of Jomo Kenyatta (Letai and Lind 2013). Other pieces of land were demarcated for landless Kikuyu, the ethnic majority in Kenya, but most were never settled and were instead used by the owners as collateral for loans. Herders at Mukogodo Division began to utilize these lands as well as other open-access government lands for grazing (Letai and Lind 2013). Encounters with East Coast Fever and Contagious Caprine Pleuro-Pneumonia during droughts in 1981 and 1984, respectively, led to further cattle losses of ~60% each event (Herren 1991). Combined with collapse of markets and grain supplies, this led to famine and impoverishment of all pastoralists at Mukogodo Division, forcing many into migratory labor and the remaining population became increasingly stratified and more incorporated into the market economy, with the wealthier producers owning the vast majority of the total livestock (Herren 1991).

The Land Adjudication Act of 1968, though post-independence, followed directly from the conclusions of the Swynnerton Plan (Grandin 1991), which was backed by the World Bank, USAID, UNDP, and FAO, and advocated for group ranches, or subdivisions within pastoralist reserves. This act was supported as a means of moving pastoralists away from subsistence practices and toward commercial beef production (Grandin 1991). Subdivisions were intended to create tenure security and encourage investments in land to increase carrying capacity of the land, prevent degradation, reduce stocking rates, and to provide collateral for loans (Grandin 1991, Mwangi 2007), based upon the logic of the time that individual land tenure would bring these changes (Hardin 1968, Campbell 1993). Group ranches were frequently not delineated with respect to seasonal water and grazing access (Coldham 1982). Thus, while group ranches could potentially provide some benefits by excluding other groups, the boundaries often cross seasonal migration lines - resulting in decreased ability to access reserve grazing in other areas (Coldham 1982, Halderman 1972).

Changes in Authority Structure and Relations with Conservation Actors

The Land Group Representatives Act of 1968 set out a system of internal governance for group ranches. This took the form of elected officials at a single level with no semblance to the Maasai elder councils (Coldham 1982), and was based on the assumption that rangelands lacked management (Kibugi 2008). The customary authority of elders at higher levels was replaced by elected committee members, who in turn are expected to enforce wider government rules, such as those on grazing, creating a novel hierarchical structure that is at odds with customary authority (Kibugi 2008, Ostrom and Mwangi 2009, Rutten 1992). Committee members were intended to guide range management, commercial practices, land use, and animal husbandry, and prepare land development plans (Kibugi 2008).

Changes due to delineation of group ranches did not occur until recently in Laikipia compared to other areas in Kenya (Kaye-Zweibel 2011). While boundaries were officially delineated in the mid-1970's and ultimately resulted in the current 13 subdivisions at Mukogodo Division, these subdivisions were not formally recognized in the affairs of pastoralists, and group ranch committees were not recognized until the late 1990's or early 2000's. Today, Mukogodo Division accounts for 7.45% of Laikipia County, and consists of 13 groups ranches with several small tracts of privately-owned land within it (Letai 2011). To the south and west is largely privately-held lands based on 99 year leases. 48 of these large private ranches comprise 40% of Laikipia County (Letai 2011). A transition in use of these private ranches to wildlife conservation came about following approximately 75 years of these lands being utilized primarily for beef production, after this industry became largely unviable due to collapse of the export market to the Middle East and the end of the Kenya Meat Commission (Heath 2001). The main income generating activities on former commercial cattle ranches shifted at this time, and came to include ecotourism, horticulture, and livestock breeding (Letai 2011). Ecotourism currently contributes nearly 1 billion dollars annually to Kenyan GDP (Homewood et al. 2008) and has been a large driver of the increasing prominence wildlife conservation, research, and ecotourism on private ranches in Laikipia. A number of these ranches have recently become recognized as "community conservation organizations", utilizing changes in the 2010 constitution that enable a consortium to manage the land in perpetuity.

Following a series of tensions in the early 2000's between private ranches and pastoralist communities, where pastoralists made claims to ancestral Maasai lands and occupied private ranches (Kantai 2007), conservation and development models came to prominence in several group ranches in Laikipia (Kaye-Zweibel 2011). This resulted in a number of "partnerships" (Kaye-Zweibel 2011, NAREDA 2004, Lamers 2014) between group ranches, private ranches, and a consortium of NGOs and ecotourism enterprises. These conservation partnerships were based upon United States Agency for International Development (USAID) models, and led to title deeds to group ranches being obtained, group ranch boundaries within Mukogodo Division being formally recognized, adoption of formal group ranch governance structure in accord with national law (Kibugi 2008), and growing authority of conservation actors within the internal management of group ranch affairs (Zaye-Zweibel 2011, German et al. 2016). For several group ranches, the African Wildlife Foundation (AWF) played a pivotal role in securing the title deed from the Ministry of Lands and in providing legal support in drafting group ranch constitutions (NAREDA 2004). At this time AWF also led group ranches to adopt internal land use zoning into designated housing areas, grazing areas, and conservation areas intended to exclude livestock (NAREDA 2004, Sumba et al. 2007). From a wildlife conservation perspective, one motivator for these partnerships was to establish areas of designated wildlife habitat as well as create a corridor for large mammals to move between connected conservancies. Known motivations for neighboring private ranches to have entered into partnerships include wildlife conservation, and efforts to accrue funds for medical, educational, and infrastructural development, but also involves an additional concern of leverage against future land claims or grazing access demands (Sumba et al. 2007, Letai and Lind 2013). These projects were established on the premises that, "wildlife would generate real income both from non-consumptive wildlife utilization, that the opportunity cost to pastoralists would be lower than the benefits (income) from wildlife, that the benefits will be shared fairly and equitably, and that ecosystem balance will be sustainably maintained" (Muthiani et al. 2011).

Numerous different aspects of these conservation trusts have been examined in terms of the direct livelihood impacts, and the politics that has sometimes led to instability in these partnerships (see Fennessy 2009, Muthiani et al. 2011, Ramser 2007, Sumba et al. 2007, Kaye-Zweibiel 2011, Lamers 2014). However, it is not well understood how livestock husbandry has been impacted as a result of these partnerships and the resulting changes that have occurred. Changes in governance, management, and norms due to new arrangements with conservation actors have translated into indirect impacts in herding. Formalization of group ranches has resulted in decreased cross-boundary movements (Letai and Lind 2013), but at the same time, since conservation trusts were established, private ranches have provided group ranch residents with greater access to regular paid grazing on private lands (Kibet et al. 2016). Additionally, while income from established conservation enterprises have provided low amounts of direct livelihood benefits to households (Sumba et al. 2007), employment on private ranches is frequent, and numerous changes in relations between Mukogodo residents and conservation actors have occurred (Kaye-Zweibiel 2011).

Building upon this historical understanding of the national and local changes in herding institutions within Mukogodo Division, we asked how these changes have impacted livelihoods, and how the institutional factors and ecological factors underpinning pastoralist herding are aligned. In what follows we complete a detailed empirical analysis of changes in livestock husbandry and livelihoods in one group ranch to examine how recent changes in livestock husbandry practices relate to the following themes in Laikipia:

1. *General Changes in Herding Rules and Norms within Group Ranches*
2. *Recent Changes in External Access*
3. *Individualization of Livestock Husbandry, and Employment Relations with Conservation Actors*
4. *Interaction of Herding Practices and Ecological Changes*

Study Site

Koija group ranch is approximately 7605 ha, and the majority of the people who reside at Koija group ranch trace their lineage to the LeUaso group, with some stating historical ties to Maasai, Samburu, and Laikipiak Maasai groups. Frequent references in casual conversation are made to recent ancestors who primarily hunted, gathered, and kept bees for a living. Today, while being primarily pastoralists, many people continue to keep bees. Koija residents live in *nkangitie* (*nkang*, singular) or residential compounds containing one or several households, today usually all of patrilineal descent. Koija is located at an elevation of 1700 meters, with a mean annual precipitation of approximately 450mm per year. The coefficient of variation of rainfall is close to 40%; it experiences substantially higher variability and lower annual rainfall compared to the majority of Laikipia County (Franz et al. 2010). The landscape vegetation is highly heterogeneous with patches of alternating *Acacia spp*. mixed with grasses, shrubs, and succulents, and other areas that are vertisol savanna dominated by perennial grasses. There are dense areas where *Acacia mellifera* and *Acacia reficiens* shrubs have recently encroached.

Koija is an example of a group ranch within Mukogodo Division that in 2001 entered into a relationship with AWF and Loisaba Wilderness, a ~22,600-hectare ecotourism and cattle ranch. As a result, AWF and Loisaba led Koija to obtain a title deed for their land, which provided the basis for legally binding contracts between Koija and partnership organizations. In the most prominent partnership endeavor, USAID provided a loan for construction of an ecolodge on Koija that was intended to produce employment and direct income to Koija, with Loisaba managing the lodge (Sumba et al. 2007). Koija's portion of the profits from the lodge were deposited in an account managed by a board of trustees and allocated toward health, education, and infrastructure expenses, on the conditions that Koija would maintain a designated conservation area (Figure 2.1, Muthiani et al. 2011), and agree to an AWF land use designation plan (Figure 2.1, Sumba et al. 2007) that included no homesteads located in the area near to the Ewaso Ng'iro River, which forms the western boundary of Koija (Figure 2.1). While the lodge is no longer functional today, the group ranch governance structure remains in place and the formal boundaries of Koija remain recognized. Additionally, a number of Koija residents are employed on Loisaba, and Loisaba sometimes provides a limited amount of grazing to Koija residents, where animals are selected from across Koija as part of a quota (see German et al. 2017 for discussion of this), as well as to their employees.

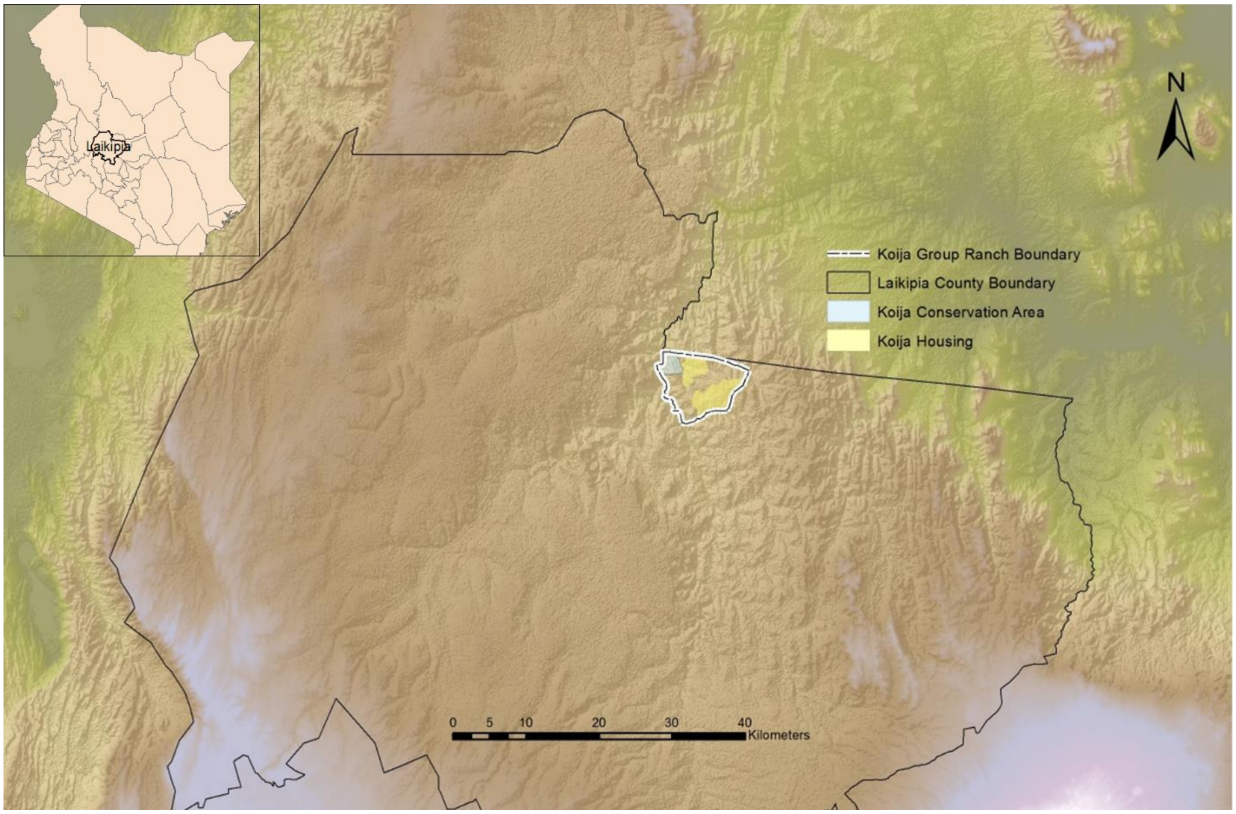

Figure 2.1. Map of study site.

Methods

We analyzed historical changes in access, herding institutions, herding ecology, and relations with neighboring private ranches in terms of the constraints on herding livelihoods experienced by pastoralists at Koija group ranch, and how their livelihoods have shifted to adapt to these constraints, using field work conducted from 2012-2016. We conducted a detailed empirical analysis of changes in livestock husbandry and livelihoods at Koija, and explored recent changes in livestock husbandry practices and the underlying drivers. Beginning in 2013, we used focus-group discussions with elder herders to determine salient ecological, institutional, and livelihood changes that have occurred over recent history (1980-2015). We then conducted surveys with an elder at each *nkang* who is involved with herding decisions (male or female, average age estimated at ~48.2 yrs.). Co-author Naiputari transcribed and translated all interviews and surveys from Maa to English. We used a systematic sampling approach, attempting to sample every *nkang* (n=225 out 245 *nkangitie* total). Two brief follow-up surveys were done at each *nkang* in 2014 and 2015. We were unable to arrange to speak with 18 families to complete follow-up surveys, and between 2013 and 2014, and four *nkangitie* relocated to areas outside of Koija, so follow-up surveys were not completed. Data collected included information on livestock wealth, income, livestock husbandry practices, seasonal herding locations, and views of conservation trust benefits. Current livestock numbers were verified using a systematic count of the entire group ranch, as well as a comparison to recent counts done by the Koija grazing committee. Historical estimates from 2002 were compared to group ranch counts conducted by AWF. In calculating tropical livestock units (TLU) we followed Zaal and Dietz (1999), with an equivalence of 10 small stock, 1.42 head of cattle, or 1 camel to 1 TLU. Average adult male equivalents (AAME) were calculated following (Nestel 1986). Finally, 20 in-depth key informant interviews were done with senior elders about ecological changes and herding practices over the previous 30 years. With 3 elders as key informants, we compiled a history of former large, multi-family *nkangitie* locations and all nuclear families that lived within them at the time, and verified locations of these sites and their sizes based upon current locations and size of glades (lawn-like grassy areas that form on former homestead sites, Young et al. 1995) as well as through informal conversations. Nine elders participated in a forage preference ranking exercise. Qualitative data were coded and analyzed using NVIVO software (Version 11).

Results

1. General Changes in Internal Herding Rules and Norms

Life histories from focus-group discussions indicate that the majority of elders we spoke with had lived most of their lives at homesteads based within the present boundaries of Koija group ranch. The areas of seasonal restriction and use within Koija were said to remain consistent, except during periods of extreme drought conditions, when some people moved homesteads near to the river. These restrictions change depending on availability of forage in hilltop glades and water sources near homesteads. When those resources are deemed adequate, then restrictions are placed on all animals using watering points and grazing areas near the Ewaso Ng'iro river. While this practice itself is thought to be a customary system of seasonal restriction based upon elder consensus, in place as long as all elders could remember, today this decision-making is formalized at the group ranch level, through the elected grazing committee. This restriction was last in place for the duration of a rainy season in mid-2013. However, during dry periods in the past, these restrictions would be lifted, and then when reserve grazing was exhausted, the elders would coordinate travel outside the group ranch, typically with the *ilmurran* leading cattle to neighboring areas that had experienced sufficient rainfalls to support grass growth (Figure 2.2). Therefore, during the long dry season in February and March the *ilmurran* would typically be on *porr/lale* (migration) with the livestock. If the April rains failed, this would lead to a prolonged drought (*Olamei*) requiring continued movements tracking rains, and if the October rains failed they would remain on *porr* in November as well.

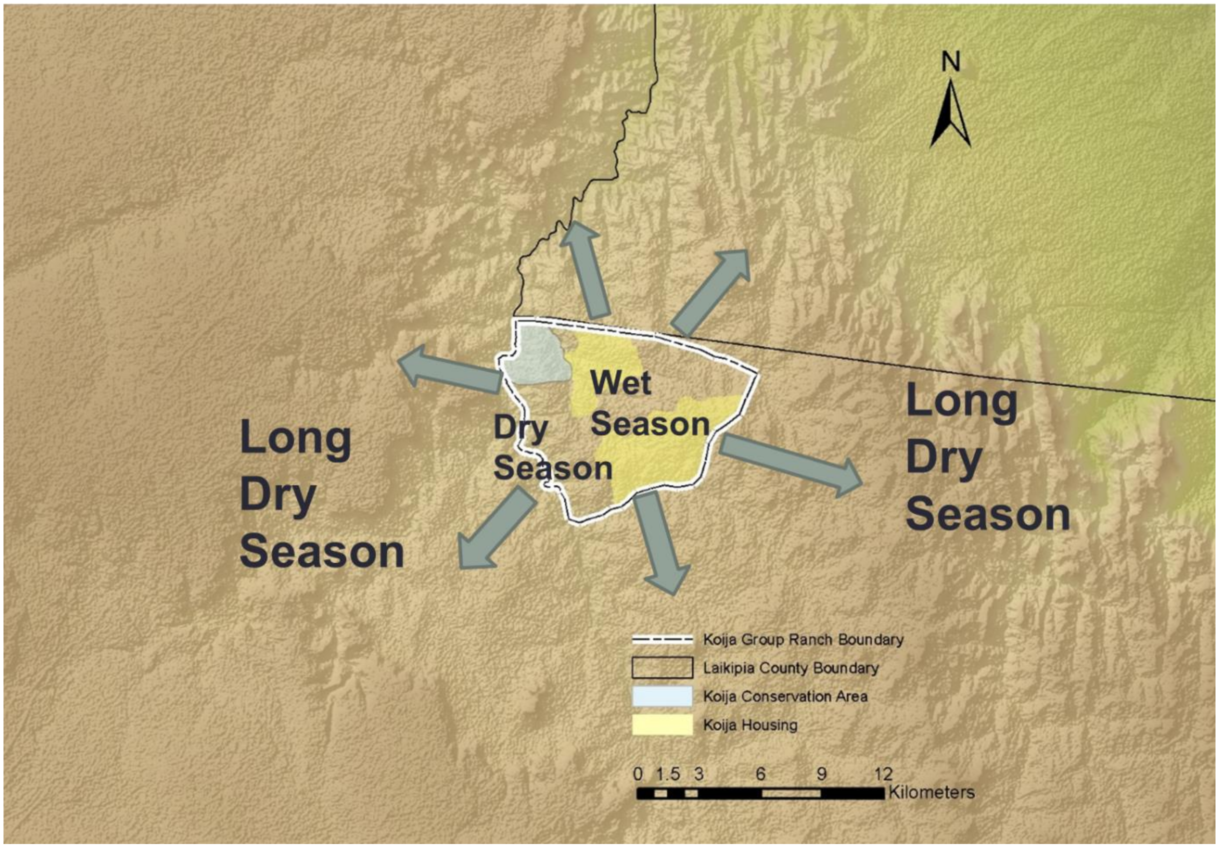

Figure 2.2. Schematic of past seasonal grazing areas within and outside of Koija group ranch

2. Recent Changes in External Access

Key-informants and focus group participants indicated that livelihoods and growth of herds were primarily limited by drought and disease, with an emphasis on access to forage resources outside of the group ranch, during drought. A marked shift from the past was indicated by a number of places where access no longer occurs. This in turn was indicated to lead to poor forage regeneration within the group ranch. This led to an indication that management within group ranches rather than an inability to regulate forage use within the group ranch was forming a main constraint. Forage access outside of Koija group ranch has decreased sharply over the past 30 years (Figure 2.3). Ninety *nkangitie* indicated that they had accessed neighboring private ranches located to the west until the early 1980’s (Figure 2.3, Table 2.1) when these ranches began excluding pastoralist access under threat of punishment with jailtime and/or directly paid fines for illegal grazing. Before this time, most herders stated in interviews that there was a recognition of informal seasonal access within areas that are currently privately owned, including one vast private ranch to the west of Koija, or at least pastoralists were not actively excluded from this area. These changes effectively eliminated Koija residents’ access as a whole to areas to the west of Koija (Figure 2.3, Table 2.1), where place names linked to Koija residents suggest longstanding past access.

A second wave of exclusion from other seasonal grazing access areas occurred during the 1990’s when conflicts to the north and east of Mukogodo Division, in present Isiolo county, limited access to these areas (Figure 2.3, Table 2.1). In the late 1990’s, additional conflicts in these areas further decreased access. In the early 2000s, as mentioned in the introduction, a wave of formalization of tenure (title deed acquisition followed by formalization, conservation trusts, and exclusion) swept throughout Mukogodo Division, decreasing most access to the immediately surrounding areas to the east (Table 2.1). At the same time, areas to the immediate north, while not becoming strictly formalized, began to exclude Koija residents, as indicated in Table 2.1 and Figure 2.3. These changes involve areas which consist of former government holding grounds, titled to National Youth Services and Livestock Marketing Division. These areas served as de facto open access in the past following disuse as a holding ground, as often seen in government-held lands, but were then said to have become occupied by Samburu herders in the late 1980's. Creation of conservancies in a number of these areas is pending, apparently in collaboration with conservation NGOs and private ranches. 67 out of 225 *nkangitie* reported that they do not currently leave Koija to access forage resources.

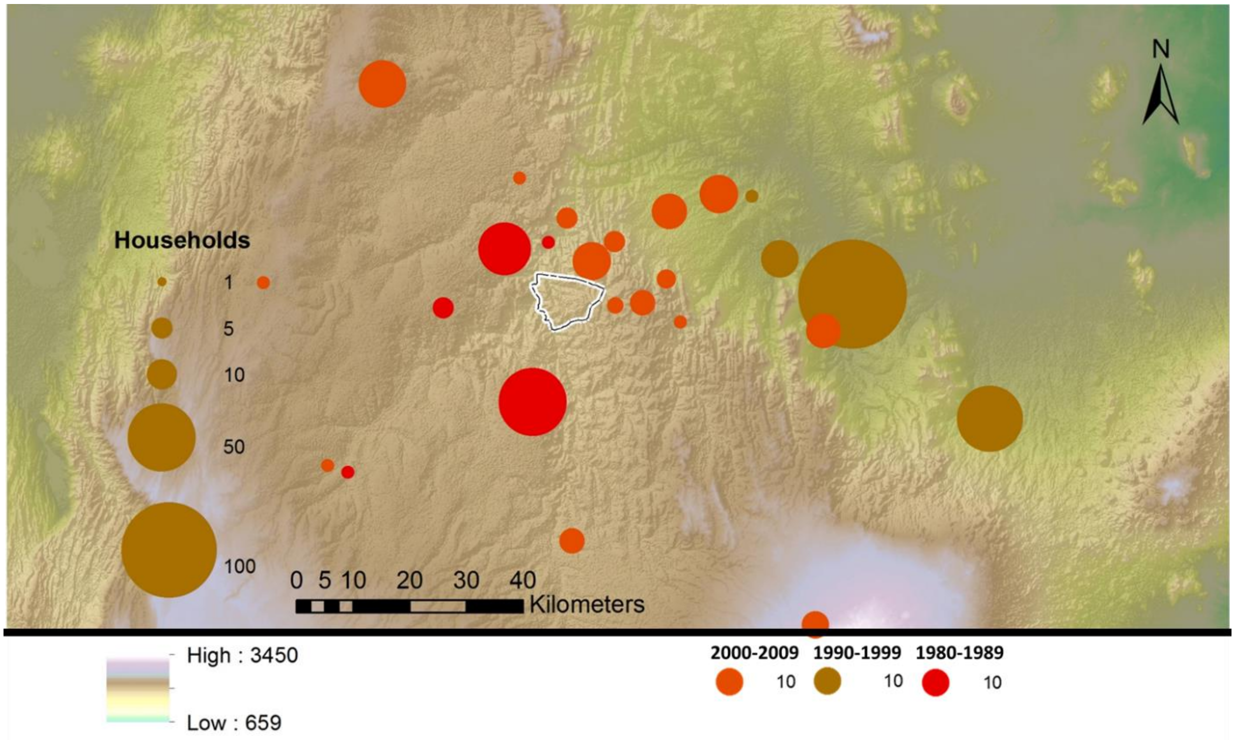

Figure 2.3 – Locations of areas formerly accessed outside of Koija. Size of dot indicates number of households that reported a loss of access to these sites during the time period referenced, color indicates time period that last access occurred. These were located according to place names and the number of *nkangitie* that use or formerly used these places was recorded.

Table 2.1. Reasons and timing of loss of grazing access outside of Koija (mentioned by elders from n=225 nkangitie).

Stated Reason	Time Period	Number of *nkangitie* Reporting
Exclusion from Private Ranches	1980 -1985	90
Conflicts within Former Seasonal Access Area	1980 -1985 1990 - 1995 1995 - 2000 2000 - 2005	4 145 48 34
Group Ranch Formalization or Conservancy Formation	1990 - 1995 1995 - 2000 2000 - 2005	3 20 59
Avoidance of "Crowding" in Former Seasonal Access Area	1995 - 2000 2000 - 2005	4 19
Closed Migration Routes	2000 - 2005	9

Today, livestock leaving Koija are taken to a small number of places. Paid grazing on neighboring private ranches began in the early 2000's and was said to offer benefits to those that have cattle by guaranteeing their survival and health in times of drought, and by providing herding labor. However, there were also negative side effects discussed: lactating cows are not located at the homestead for a source of day to day milk, the care given to animals may not meet an owner's standards, and many *nkangitie* lack access to cash to pay for the paid grazing. In 2013, at a time that was not considered drought, but when forage was considered inadequate for cattle on Koija, paid grazing was made available on private ranches. At that time, 1284 head of cattle were located on private ranches, through either paid grazing or employment arrangements, which was 85% of the total cattle taken outside Koija. While the quota for paid grazing on

Loisaba at this time was thought to be 480 head of cattle, 408 were present on nearby Mpala ranch, and the remaining 396 of Koija's cattle are thought to have also been at Loisaba through access granted due to employment relationships. Only 219, or 14.57% of the cattle that were located outside of Koija, were located in low densities at areas with unpaid access that also had a historical precedence of being accessed by pastoralists. low densities at twelve unpaid, areas with informal access arrangements. At the same time 3994, or 87.82% of sheep that were located outside Koija, were in just 3 areas where access is not paid for and where there is a historical precedent of pastoralist use, while most of the remainder were located in very low densities at 16 different sites.

3. Changes in Livestock Husbandry, Individualization, and Employment Relations with Conservation Actors

We analyzed the descent of different *nkangitie*, and which families lived together in *nkangitie* in 1984 as compared to today. Using a combination of oral histories and the presence of perennial grass-dominated glades (See Young et al. 1995 for a detailed description) indicating a former *nkang*, we determined there would have been 16 *nkangitie* within Koija at that time, with a much larger number of people living within each *nkangitie* (Table 2.2) and a distribution as shown in Figure 2.4. We then estimated the number of livestock at Koija using an average of the numbers of livestock that each elder who was a member of a nuclear family estimated were owned by their family at that time. The word *entare* (goats and sheep together) was used for this portion of the survey, as individuals could typically not recall the exact numbers of individual goats and sheep prior to 2002. Since 2004, 32 nuclear families have moved to nearby locations, many on other nearby group ranches, perhaps leading to an underestimate of historical numbers. At the same time 19 families are known to have relocated into Koija since 2004 and this increase was included in the 2016 count. The numbers of goats and cattle have increased slightly since 2002, camels have increased slightly, from 0 to 299, and sheep have increased (Table 2.2). This is thought to be largely due to recovery over time following a severe drought in 2000, and changes in herding strategy and forage access, with 52.34% of sheep located year-round on sites located outside of Koija (discussion of the ecological condition of these sites to follow). In the early 1980s, prior to a major drought in 1984, we estimated that the total livestock on Koija was 5357 head of cattle, and 2692 *entare*, for a total of 4041.74 TLUs (Table 2.2). Compared to estimates of total livestock on Koija today, there has been about a 35% decrease in cattle since the 1980's, with a simultaneous approximately tenfold increase in sheep and goats (Table 2.2).

TLUs overall have increased over the past 30 years, with 5623.52 TLUs total considering livestock holdings located off of Koija (Table 2.2). Using population density estimates (Herren 1989), we calculated there were approximately 1316 people living on Koija in the late 1980s compared to 2761 according to our current estimate, imply a doubling of population with less overall livestock per person. The average in 1980 for this region was estimated at about 3.07 TLUs per person, with an average value of 2.04 TLUs per person today. A GINI coefficient calculated for matched pairs of 201 families (221 complete surveys, excluding 20 families that arrived after 2002) for 2002 and 2016, showed that between 2002 and 2016 there has been a marked increase in inequality in livestock holdings except camels (Table 2.3), with the sheep and goat holdings contributing the highest and second highest, respectively, to the overall increase in inequality of TLUs. Considering livestock holdings today: 12 out of 225 *nkangitie* had no livestock, and 155 out of 225 *nkangitie* have no cattle, 70 *nkangitie* had less than 0.70 TLUs/AAME (the equivalent of less than one cow per person), and just 37 *nkangitie* had above 4 TLUs per AAME.

Table 2.2. Historical changes in livestock numbers estimated from surveys.

	1980	2002	2016
Cattle	5357	2644	3530
Sheep and Goats	2692	21329	28386
TLU	4041.74	4063.87	5623.52
# *nkangitie*	16	221	243
Average # persons per *nkangitie*	82.25	n/a	11.36

Table 2.3. Comparison of inequality amongst *nkangitie* (GINI coefficient values) between 2002 and 2016 (subset of 221 *nkangitie)*.

	2002	2016
CATTLE	0.573	0.613
GOATS	0.423	0.506
SHEEP	0.559	0.669
CAMELS	0.895	0.863
TLU	0.537	0.606

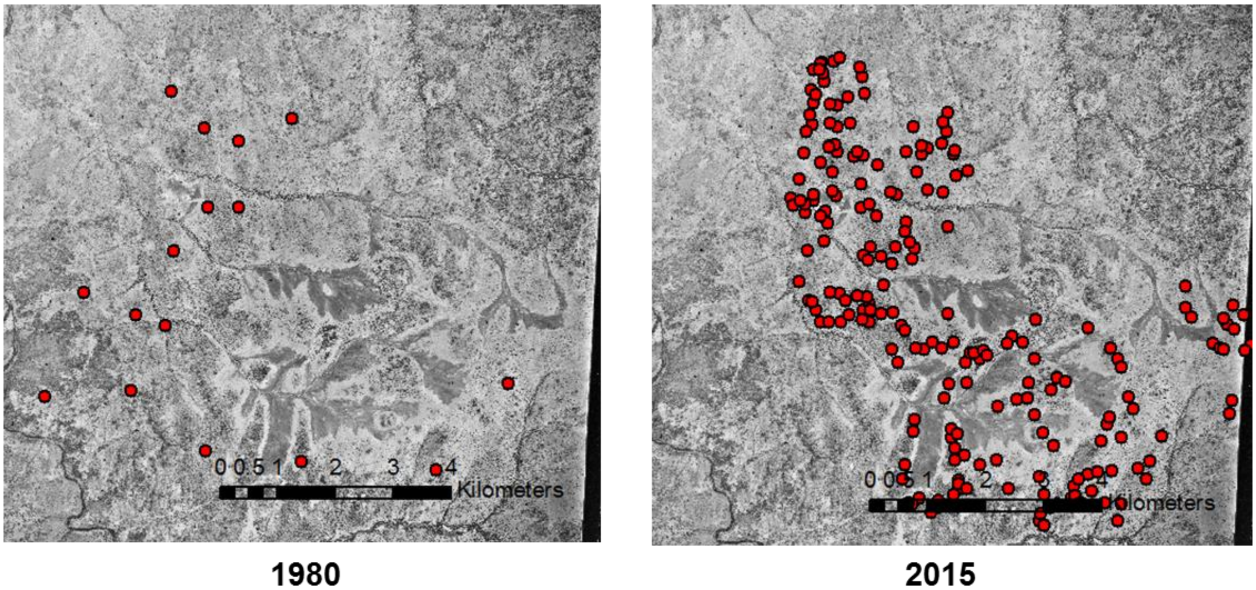

Figure 2.4. Spatial distributions of *nkangitie* in 1980 and 2015 across Koija

In addition to those detailed by Herren (1991), a number of recent droughts were frequently referred to as drivers of large decreases in cattle (1997, 1999-2001). In more recent droughts (2009), disease was said to be an additional factor, causing death of small stock as well. In explaining historical increases in small stock, it was stated that goats in particular have been increasingly favored due to fast reproduction, ability to recover rapidly following loss of animals, ease of sale, the high reliance on grain based diet, drought resistance, ease of slaughter, and rapid use of meat, in general agreement with previous studies (e.g. Herren 1991, Hauck 2013). However, the role of small stock as a source of income necessary to support cattle keeping, was often also emphasized, where small stock are frequently sold to access cash that is needed to support cattle. Some indicated that drought is leading them to only keep camels and goats today, while others emphasized the importance and interdependence of the different species for maintaining herds, with small stock enabling people to buy and maintain cattle.

Other more nuanced and complex aspects of herding ecology emerged in focus groups (distilled summary in Appendix A) and key-informant interviews (Appendices E-G). Human-non-human relationships are different from commercial producers in a number of ways, including cosmological relationships, identity, and status. In interviews, it was also emphasized that the higher sale value and higher quality milk for cattle compared to small stock gives cattle preference in terms of utility, despite their sensitivity to drought. Additionally, it was emphasized that a single goat can easily be slaughtered or sold to feed a family with a smaller impact to a herd, while slaughtering a cow is usually only done during ceremonies. Most interviewed said that cattle are only sold after great deliberation, but small stock are sold readily. It was indicated that cattle are difficult to sell during droughts when they have become thin, while goats can be sold any time, though at a greatly reduced price during drought.

Historically in this region, nuclear families of patrilineal descent have decreased the practice of living together in one large *nkang*. It was stated in both focus-group discussions and key-informant interviews that the practice of many *nkangitie* migrating together has greatly decreased recently, and today it usually occurs between close relatives, friends, and immediate neighbors only, while in the past everyone's cattle were the responsibility of the whole community. Labor sharing was also said to have decreased on a day to day basis, while 25 families currently hire herders from another family at Koija. One reason stated for a lack of sharing labor was children being in school, and that people who did not have children in school now expected payment for their children to help others. Further, some emphasized a change in the overall view of children in the community, stating that raising children is no longer viewed as the responsibility of the community as a whole, and that individual families' values now take precedence. While considered much less frequent in comparison to the past, participants generally agreed that animals may still be shared if one family owns multiple cattle and another is in need. Some continue to share animals when one family has more lactating cows, one house will loan another house one to provide milk, and then give them a calf when they return to collect the cow. In the past this debt was repaid, but today reciprocation seems to have decreased.

In informal discussions, herders also attribute changes to the material basis of small-stock, mentioning the difficulty of herding them in large groups, and the difference in accumulation of manure within the livestock enclosures, where goat and sheep dung must be regularly removed and piled outside but cow manure is not. It has been indicated informally by several elder herders that large herds of entare are unmanageable. Comparing *nkangitie* with multiple herds, 26 out of 40 *nkangitie* herded their small stock together, while 14 *nkangitie* did not. The *nkangitie* that herded small stock collectively had significantly smaller mean total herd sizes (mean total herd size of 145.85 *entare*) than the *nkangitie* in which each nuclear family herded their small stock separately (mean total herd size of 238.79 entare, $t_{26.41} = -1.72$, $p=0.048$). This is in contrast with *nkangitie* with multiple herds of cattle, where they nearly always (37/39) herded their cattle together, regardless of herd size.

Respondents frequently emphasized the benefits of employment in determining *nkang* livestock husbandry success, by providing additional financial resources to offset sales, to afford medicine, and pay for other costs of keeping livestock (detailed mechanistic analysis of how this translates into individual benefits found in Chapter 3). We considered whether and how employment may be impacting herding institutions directly by exacerbating individualization of livestock husbandry practice. We based this upon repeated mentions of the advantage that is given to individual *nkangitie* that are able to supplement their livestock through outside income.

Employment on private ranches also typically allows grazing access for employees, which constitutes the bulk of secure, non-illicit pastoralist access to these private lands, as reported above. We examined the increases in cattle between 2002 and 2016, and found that those with full time employment had significantly greater increases in cattle owned (paired t-test, $t_{53.63} = 1.75$, $p = 0.043$), as well as for sheep ($t_{66.71} = 2.02$, $p = 0.024$) but this relationship was not significant for total TLUs or goats ($t_{57.03} = 1.641$, $p = 0.948$, $t_{61.01} = 1.08$, $p = 0.857$, respectively).

Thirty-five *nkangitie* reported having members with current full-time jobs on conservation ranches, and four reported having one person that had other full-time employment, in 2015. Fifteen reported a member with some type of part-time local employment (frequently on conservation ranches). Sixty-six reported having a member who worked for conservation ranches or for Koija's ecolodge at some time in the past, and nineteen of these reported that they had lost their job between surveys, from 2014 to 2015, most of which were due to a fire that burned down the main lodge on Loisaba in 2014. Six reported other past full-time employment not on conservation ranches. Sixteen reported members having a current job within Koija (including as infrastructure guard, herder, teacher, dispensary employee, etc.), and twenty reported past employment with Koija (including management committee, guard, teacher), but were considered separately as these jobs are typically low-paying, at half or less the rates formerly paid at Koija's ecolodge, or on Loisaba.

Direct benefits of employment on conservation ranches through wages do not provide a complete understanding of the potential benefits of employment, and this emerged in ethnographic interviews in aspects that proved difficult to quantify. These benefits were stated to include Koija residents that work for conservation ranches gaining benefits from ranches in free grazing, those that work as guards gaining direct cash from bribes from illicit grazers, and both indirect and direct benefits gained in exchange for informing on group ranch members who participate in illicit grazing and wildlife poaching to their employers.

4. Interaction of Herding Practices and Ecological Changes

Elder herders in focus group discussions, interviews, and informal conversations indicated a number of changes in vegetation. They frequently indicated especially dramatic increases in two dominant tree species within the past 30 years, *ilmunichoi* (*Acacia mellifera)* and *nchurai* (*Acacia reficiens)*. *Acacia mellifera* was nearly uniformly mentioned as the primary species responsible for supporting individual *nkangitie*' goat production, while *Acacia reficiens* was often said to be of little nutritional values and to cause decreases in grass in areas they have become established. Additionally, one introduced encroaching species *ilmatundai* (*Opuntia stricta*) has increased rapidly recently (Strum et al. 2015) and was reported to create many problems for herders, especially when animals are unattended. Other now dominant species that were regularly mentioned to have increased within the last 30 years are *laraiti (Cissus rotundifolia)*, *ndulele (Solanum incanum), lokiteng (Ipomoea kituensis), and ldupai sero (Sansevieria volkensii*, see King et al. (2012) for a detailed description of this species), all of which were repeatedly said to have low forage value.

Two canopy species, *pushiruti* (*Euphorbia tiriucalli)* and *bobongi* (*Euphorbia magnicapsula)* were both frequently stated to have decreased recently, with large negative impacts on honey production. A number of other plant species were reported to have decreased, including *loigwaroi*, *denja(Kleinia spp.), loilei* (*Sarcostemma viminale*), *lkimanjoi (Hibiscus greenwayi)*, *loishimi* (*Commiphora sp*.), resulting in large negative impacts on honey production as well as losses of important forage species for goats. Declining milk yields of goats and cattle, widely observed across *nkangitie*, were attributed to changes in vegetation.

Today cattle are reported to rarely ever be kept at homesteads within Koija, due to the lack of rains, but also because as soon as the rains return, the grasses are immediately eaten by small stock and the cattle that return home soon after the rains. Some emphasized that it would be better to allow this grass to regrow, but that usually during these times illicit access to areas is being relied upon for grazing, so herders return home to avoid using these areas under threat of fine. This state of poor vegetation regeneration is now also said to be exacerbated by herders from northern areas moving into Koija immediately following rains, and it was emphasized that when recent attempts have been made to restrict within reserve grazing areas or the designated conservation areas, that tensions have arisen as a result. While it was frequently emphasized in interviews that one of the main benefits of group ranch formalization is to be able to restrict herders from northern areas, many emphasized that this also resulted in a breakdown of reciprocal relationships, where herders from Koija can no longer expect to access areas to the north as a result.

Recent landscape vegetation changes are also thought to contribute to increases in small stock. It was stated that goats are typically taken to lush areas along seasonal streams near homesteads and dense areas where *Acacia mellifera* has proliferated recently, and that these browsing resources are available in all except the most extreme drought. It was emphasized that goats are able to climb trees and to go into dense brush that cattle cannot access. Goats are said to rarely leave Koija, but since sheep and cattle require grass, they usually move during drought. The recent proliferation of *Acacia mellifera*, sometimes seen as a sign of degradation in rangelands studies (but see D'odorico et al. 2012 for a detailed discussion), is said by some to reduce grass cover in areas where it establishes, but is also thought to be beneficial for goat keeping. Overall, changes in vegetation are thought to be supporting goats, but are also thought to be making cattle keeping more difficult because goats are ever-present and impact the grass immediately after it begins growing.

Sheep increases detailed above are likely related to recent access changes, especially considering the vegetation of areas off Koija where access has recently increased. One of these areas has just recently begun to have returned access, following conflicts that had occurred there between other pastoralist groups over the past two decades. This area, as well as the two other external locations that can be accessed informally by anyone from Koija, and that support large amounts of sheep but do not support cattle year-round. This is due to differences in forage type (e.g. high amounts of *Pennisetum sp*., which cattle can only eat following wet seasons) and because of the inadequate amounts of reliable water sources required to regularly support cattle. The large increases occurring in sheep herds in the preceding 14 years may reflect a response to this access.

The above was supported by a simple ranking exercise that compared the preferred use by the three dominant domesticated herbivores (cattle, sheep, goats) of preferred forage type that were clearly dominated by one of two types of grasses or *Acacia spp*. (Table 2.4). As stated before, areas with vertisol soils dominated by *Pennisetum mezianum* are not suitable during the dry season, so had a lower overall ranking as forage for cattle, and especially for goats. However, these areas were ranked higher as a forage source for sheep as compared to cattle. This was corroborated by interviews, where it was commonly stated that the cattle at Koija prefer the red soils over vertisols, and are healthier when eating *nkamurai (Digitaria milanjiana*) (Table 2.4*),* a species found on red soils. In this ranking excercise, individuals indciated that goats had a clear preference for areas with forbs and lianas, *shushei (Barleria spp.),* as well as *girgiri (Acacia brevispica)* or il*munichoi*, but also *ltepes (Acacia tortilis)*, all of which produce seeds that the animals favor (*ldalam and sagaram*), and even the dried leaves (*seu*) that fall. The small number of camels within Koija are able to survive on this vegetation to goats though they prefer *Acacia tortillis*.

Table 2.4. Average rank of three vegetation types (1-10, descending by preference) assigned by herders. (These values were assigned according to forage preference of locations dominated by different vegetation types. This subset of types shown to demonstrate the preferences of different forage by different livestock. The other 7 locations had mixed vegetation, with varying degrees of bare ground, and were excluded from the table for simplicity. Exercise was done with n=9 herders).

	Cattle	Sheep	Goats
Pennisetum mezianum dominant	5.4	4.4	7.9
Digitaria milanjiana dominant	3.1	4.0	3.9
High forb, liana, and *Acacia spp.* density	6.3	5.4	2.4

Discussion

It has been previously observed that drought, decreasing mobility, and increased offtake of animals for sale in order to buy grains have led to an increased reliance on small stock along with decreased reciprocity and increasing stratification by livestock wealth within Mukogodo division (Herren 1991). Building on work that has explored changes in access in the Laikipia landscape (Letai and Lind 2013), we confirmed that these patterns coincided with large shifts in herd composition at Koija Group Ranch. More recent changes in the boundaries of group ranches, their internal governance structure, and rangeland governance rationale have continued as well, influencing livestock husbandry within Koija. A loss of access to surrounding areas has continued over the past 35 years, making individuals increasingly dependent on very few areas of access outside of Koija today. Secure access occurs predominantly through grazing quotas and employment on private ranches and a small number of areas which all have a right to and are accessed informally, while the majority remained reliant on illicit forage access during drought. This has resulted in a large amount of the population being in a constant state of precarity for grazing access, while livestock remains the dominant livelihood. We found that local grazing institutions were mainly constrained by limited external access to forage resources and subsequent lack of internal regeneration of resources, rather than an internal inability to regulate forage resource use. Our analysis indicates that overall livestock amounts held by people living within Koija has increased, but actually decreased in the number of livestock per person over the past 30 years (Table 2.2). Four TLUs per person is thought to be sufficient for health by East African standards using this conversion method when households are subsisting on livestock products (Zaal and Dietz 1999), while the average at Koija is 2.04 TLUs per person today. However, historical livestock to person ratios seem to be largely decoupled, as diet is predominantly based upon a high calorie, low nutrient maize diet today.

During this time rainfall has also become more variable within Mukogodo Division (Franz et al. 2010, Huho et al. 2009), and there have been decreases in multiple plant species that were formerly important for supporting goats, and recent increases in one species (*ilmunichoi*) that provides some reliable forage for goats. Within the current institutional context of access and employment, where maintaining herds of cattle and sheep is dependent on mobility, we showed increases in herds to be significantly related to external employment. Increasing herds of goats represents a historical adaptation to the limitation of dry season grass access, and mirrors adaptation to drought and vegetation seen by other pastoralists (Liao et al. 2016, Opiyo 2015, Österle 2008, Silanikove 2000). While reliance on small stock frequently represents a recovery stragety following loss of herds, or a strategy of less wealthy herders, at Koija it reflects an adaptation to the constraints that have emerged in ongoing changes in access and climate. Additionally, increases in sheep appear to be an adaptation related to recent changes in access, as the few sites where reliable access occurs have ecological conditions that are preferable for supporting sheep. These adaptations of herders at Koija to external constraints have occurred concurrently with a concentration of herbivore pressure at the local scale within Koija. This has large implications for landscape ecological process where the differential impacts of small stock and the localized concentration of all livestock is likely impeding the regrowth of grass forage within Koija during rainy seasons.

In considering the alignment of herding institutions and ecological processes, we interpret the trends that we observed at Koija as a product of a scale mismatch between institutions and ecological factors. The cascading impacts of the fragmentation of pastoralist mobility on social and ecological factors is well documented (Boone et al. 2011, Burnsilver et al. 2008, Galvin 2008, Galvin 2009, Hobbs et al. 2008, Reid et al. 2008, Thornton et al. 2006, Mwangi 2007). When seasonal grazing movements cannot occur and vegetation has less chance to recover seasonally, there can be decreased nutritional returns (Fratkin 2001, McPeak 2003) and degradation can occur due to a lack of movement to reserve grazing sites (Mwangi and Ostrom 2009, Weber and Horst 2011).

Historically the scale of mobility required to buffer variability of rainfall and to access adequate vegetation was fragmented by social processes of reorganizing the institutional Laikipia landscape to favor commercial cattle production by settlers, emphasizing private land ownership and the right of large ranches to exclude others. This study has documented the cascading results within pastoralist areas, seen in decreases in labor sharing arrangements, animal gifting, herd risk pooling, and overall reciprocity (Herren 1991) that have set the stage for ongoing changes. More recently, institutional policies that have been advocated by private ranches in partnership with international conservation NGOs have been motivated by goals of enhancing wildlife conservation efforts in Laikipia, securing tenure and borders of private ranches, and enhancing pastoralist livelihoods. However, in utilizing collective title and group ranch structure as a foundation of partnerships, our literature review revealed a refocusing of authority structure, further changes in boundaries, and additional changes in relations between pastoralist groups and private landholders. This has directly led to an increase in the influence of wildlife conservation actors in group ranch governance (Kaye-Zweibel 2011, German et al. 2016), changes in access to forage resources in other group ranches, and an individualized system of cattle access.

The shape of recent changes also reflects global patterns that have been broadly termed "neoliberal", where authority has increasingly become decentralized, especially with non-state actors assuming the role of the state, emphasizing property lines and landscape security, while market-based incentives are at the same time being proposed to restructure both governance and livelihoods. The extension of NGOs roles of governance into the sphere of pastoralist affairs (DePuy 2011, Kaye-Zweibel 2011) has similarities to delegation (Ribot 2002) where NGOs have assumed a role traditionally played by the government in providing infrastructure, health, and education services (Little 2014).

Through market-based projects aimed to improve livelihoods, partnerships between NGOs, private conservation ranches, and pastoralist group ranches have taken a unique, context-specific shape, but also have assumed similarities to global trends. Firstly, the designation of fixed wildlife conservation areas and construction of ecotourism lodges necessitated a land title as the basis of agreements (Kaye-Zweibel 2011). The lack of provisions for external grazing in these agreements, however, implies a “modernization” paradigm of pastoralist development policies that encourages pastoralists to become fully sedentized (McCabe 2004, Moritz 2008) and to move away from livestock husbandry practices that require mobility. These interventions have proceeded based upon the assumptions that group ranches could be autonomous, self-contained livelihood units nested in a hierarchy intended to manage this landscape (NAREDA 2004) when both historical customary social relations as well as the ecological basis of herding have extended beyond these nested boundaries of group ranch governance levels.

Resulting institutional shifts have supported the goals of increased wildlife connectivity, created more legible property lines, and securitized the landscape. At the same time, these priorities have resulted in a lack of consideration of the dominant pastoralist livelihood concerns in the ability to retain mobility in response to fluctuations in rainfall, because these requirements extend beyond the boundaries of group ranches. Conservancy formation thus far has focused on exclusion of outsiders, setting aside areas as conservation areas, and regulating management practices all within group ranch confines, leading to incompatibility with pastoralist institutions and flexibility of access that has formed the basis of historical pastoralism in semi-arid lands. This has created significant stressors on internal resources that have led to an inability of customary management to regulate internal forage resources at Koija. Our results, as well as historical accounts (Anderson 2002) imply that any system of internal management within group ranch boundaries will likely be insufficient to maintain cattle productivity without also providing for adequate grazing outside of Koija during dry periods and times when regrowth of vegetation is most important. This appears to be rooted in a conception of pastoralist livelihoods that is aligned with equilibrium-rooted discourses observed in numerous other case studies of development projects in pastoralist areas (Anderson 2002, Blaikie and Abel 1989, McCabe 2004, Turner 1993, Waller 2012). However, other approaches have been advocated in the past for Laikipia, where flexible tenure with guaranteed seasonal access could prevent ecological degradation as well as decrease poverty by coordinating emergency grazing access during droughts (Heath 2001), or more robustly, through legal recognition of grazing rights, as factors that would also perhaps help deescalate present tensions that have links to historical land losses (Lengoiboni et al. 2010).

Secondly, institutional changes in Laikipia occurred under the assumption that conservation projects would enhance livelihoods across pastoralist group ranches, and have proceeded with a market-based logic to advocate livelihood shifts, a trait that is common among more recent conservation projects (Fletcher 2012, Holmes and Cavanaugh 2016, Igoe and Brockington 2007, Little 2014). We found that effective livelihood diversification, a central goal of conservation partnerships, was largely limited to those that have jobs on private ranches, and who in turn are increasing their own cattle holdings. These efforts, at this point, have not provided a viable alternative to the dominant livelihood basis for herders at Koija to decrease their sensitivity to drought events, and cannot be expected to in the region (Little 2014). The continuation of current policies relies upon narratives about poor management leading to the marginal state of livelihoods today, and in many ways parallel more indirect (Holmes and Cavanaugh 2016) ways of shifting livelihoods by providing market incentives to shift away from pastoralism. In reading the NGO grey literature documents that trusts are based upon, there is a central conviction that Koija's livestock system is inherently unsustainable, and that recent shifts in vegetation are due primarily to issues of land management, livestock stocking rates, and population (Alexovitch et al. 2012, Fennessy 2009, Lent et al. 2002, NAREDA 2004, Sumba et al. 2007). While it should be emphasized that the stocking rates on many group ranches are indeed high by "best use" standards for maximizing beef production (Kaye-Zweibel 2011), our findings indicate that the common argument -- that population growth is leading to increases in stocking rates that are in turn driving ecological changes -- provides only a partial explanation of livelihood pressures, and tends to overlook the need for mobility as well as the complex interpenetration of factors that underpin the changes in landscape processes and livelihoods from herder's perspectives.

Thirdly, these interventions have been spearheaded in the name of improving livelihoods, this system of promise has incentivized wildlife conservation in communities on the short term, but have worked primarily through individualized benefits. Further, our results imply that the collective interests of pastoralists have become increasingly limited as a result of conservation partnerships. Rather than seeking collective interests, Koija residents have begun forming increasingly individualized relationships and close alliances with neighboring private ranches that benefit their extended households, even informing on other Koija residents in exchange for these benefits.

Over the time since conservation partnerships have emerged, there has been increasing inequality and individualization of herding practices at the *enkang* level on Koija, with herders who are employed on conservation ranches being more likely to have increases in their herds, confirming patterns of elite capture frequently seen in conservation and development projects

(Lemos and Agrawal 2006). Finally, these institutional changes have closely interacted with biophysical factors, most importantly the type of forage that is available in the subset of former areas Koija residents now have access to. This has largely favored goats (relying on resources within Koija) and sheep (relying on resources in few areas outside of Koija), as well as the biophysical difference in herding flocks and maintenance of livestock pens of small stock compared to cattle. These factors likely act together to incentivize livestock husbandry that is individualized at the household level.

Finally, recent economic and cultural shifts due to wildlife conservation, and a pattern of increasing inequality between *nkangitie* have likely influenced patterns of reciprocity where individuals are able to secure access or other benefits for their *nkangitie,* at the same time others are excluded from this access in areas that were historically open to all. This is seen most conspicuously in employment on private ranches. Those with outside employment had higher rates of cattle and sheep increases, and those employed on conservation ranches account for the largest percentage of the pastoralist cattle located on those ranches, indicating that the most robust benefits conferred by wildlife conservation are through employment and through the personal relationships that accompany these positions.

We argue that the process of increasing individualization is likely related to a complex of historical and ongoing factors involving changes in landscape access, livestock markets, the material and ecological aspects of keeping livestock, and changes in relations and employment due to wildlife conservation. However, it should be noted that the drivers of the individualization of households could also be further related to a number of factors not considered in our study, such as attitudes fostered by public education in the region, the dominant economic ideals of wider Kenya society (Lesorogol 2008), changes in structures of authority and norms due to group ranch governance (Kibugi 2008), as well as market interactions that incentivize specific types of behavior (Herren 1991).

Our analysis indicates that a suite of external constraints has truncated the ecological scale that herding livelihoods in Laikipia are dependent upon. These constraints have historical roots beginning in the colonial era, that extended into the post-independence era, and have been further solidified through regional conflicts and privatized wildlife conservation. These changes have interacted with and been constrained by biophysical factors as well, while a number of internal institutional norms have shifted as herders have adapted their livelihoods to constraints on the spatial scale of mobility. We interpret these phenomena as a scale mismatch between land use and governance institutions and the resources being utilized and managed (Cumming et al. 2006). The consequence is this study system is there is an increasingly marginal system of livestock husbandry with large negative consequences for vegetation and soils due to decreased mobility and changes in the type of herbivore pressure. However, at the same time, narratives that cast pastoralists as irresponsible land stewards tend to mask the underlying, ultimate causes of livelihood barriers, and can lead to an over-emphasis on proximate causes (e.g. stocking rates). This indicates a need for more nuanced ethnographic approaches that detail how ecological discourses are deployed to support certain types of interventions and scale-making projects in pastoralist lands.

A central goal of conservation research in Laikipia is to determine ways of aligning scale and landscape process to foster sustainable landscapes and livestock-based livelihoods (Sundarasen and Riginos 2010, Kinnaird and O’brien 2012, Georgiadis et al. 2007). Proponents of landscape interventions in Laikipia are very conscious of the closely-related issues of landscape sustainability and livelihoods, as well as the need to garner support and “buy-in” for conservation in local communities (Sundarasen and Riginos 2010, Kibet et al. 2016). Novel conservation governance has largely refocused customary economic bonds onto relationships with ecotourism operators (Kaye-Zweibel 2011) with conservationists' stated goals to "provide alternative income to diversify rural livelihoods" through ecotourism, as well as allocating proceeds from ecotourism to infrastructure, education, and health projects (Sumba et al. 2007).

Conclusions

Contributing to the literature on identifying scale mismatch, using an integrative approach (Scoones 2009), we identified ways that livelihoods are contingent upon structural constraints, the adaptive agency of herders, and biophysical factors. Hierarchical approaches, focused on ecological levels and scale, alongside the levels of social organization, have difficulty accounting for ethnographic accounts of factors such as power imbalances and difficult to quantify livelihood concerns, that some have addressed through focusing on different narratives and knowledges (Leach et al. 2010, Ahlborg and Nightingale 2012). While previous analyses of scale mismatch have incorporated overtly contested politics (Lebel 2005), or the different interests (Cash et al. 2006), different knowledges (Ahlborg and Nightingale 2012), or the imposition of top-down solutions (e.g. Scott 1998), this research shows, that attempts to harmonize social and ecological processes at the landscape can actually be intertwined in deepening inequality and extending power in unintended, non-intuitive ways (e.g. Ferguson 1990, Nadasdy 2005). The present patterns of land use and livelihoods at Koija are not merely the product of apolitical management decisions that misaligned process, but the result of a history of dispossession, loss of access, intervention in pastoralist land use practices, and an economic ideology that has neglected the ecology of pastoralist livelihoods. This unevenness works across levels, and we emphasize the importance of understanding these not just as caveats to governance, but as inherent to governance projects with high relevance for post-colonial conservation settings (Nadasdy 2005, Kull 2004, Goldman 2011, Neumann 2002). Our work suggests scale mismatch to be present not only in current lack of alignment between ecological and social processes, but to be actively reinforced in uneven ways by attempts to align specific ecological and social processes to benefit certain actors in the privatized conservation landscape of Laikipia. Explicit analysis of the historical, material, and discursive practices that have shaped scale mismatch can produce novel-insight as to the ways that scale mismatches are constructed and enforced in social-ecological systems analysis.

Current approaches to wildlife conservation in pastoralist rangelands in Laikipia emphasize large mammal habitat connectivity, social networks that foster maintenance of pro-wildlife habitats, and transformation of livelihoods. Alternative future approaches with a robust landscape conservation vision that is focused on pastoralist well-being, habitat-connectivity, and sustainability as interdependent goals, might instead focus on the highly constrained local management system that is unable to adequately adjust to ecological variability and prevent localized degradation. In closing, using a critical mode of inquiry alongside approaches that are compatible with natural resource management analysis, it is our hope that we can show an example of a bridging between social-ecological systems frameworks and political ecology, where relational, intimate aspects of governance become intelligible to scientists and conservationists trained in social-ecological systems approaches. Using the approach of exploring the alignment of institutional and ecological process while also using a critical approach, allowed for a fluid, indeterminate analysis of how biophysical changes and human-animal relationships shape social elements, leading to an improved understanding of landscape ecological factors and livelihoods alike. It also allowed for understanding of the origins of a scale mismatch in historical, political, economic, discursive, and biophysical factors. While these factors are often underemphasized in conservation planning, they are necessary elements of a nuanced understanding of unevenness and relational aspects that arise in the contexts conservation and governance interventions occur under.

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CHAPTER 3

CONSTRAINTS AND MULTIPLE STRESSORS: STRATIFIED LIVELIHOOD VULNERABILITY IN A MAA-SPEAKING PASTORALIST COMMUNITY OF CENTRAL KENYA[2]

[2] Unks, R.R., E.G. King, D.R. Nelson, N.P. Wachira, and, L.A. German. To be submitted to *Global Environmental Change*

Abstract

The focus of this study is on how changes in formal and informal institutions have differential impacts within communities in terms of vulnerability of livelihoods to drought, and uneven processes that shape adaptation to new conditions. Previously identified institutional and biophysical constraints to herding livelihoods have shaped herding practices over time in one pastoralist community in Laikipia, Kenya. Here we analyze how drought and market interactions differentially impact herders who are operating within those constraints. Using an integrative approach that combines analyses of adaptive capacity, entitlements, and access, we detail the interactions between household assets, different types of resource access, and social relations. Our analyses investigated the roles of these factors in shaping both livelihood strategy and responses to drought events. We asked how changing access and individualization of herding practices may act as constraints that are experienced differentially, as drought and livestock markets together act as combined stressors on herder livelihoods. We found that the distribution of entitlement sets that enable the ability to cope with these stressors were related to the uneven vulnerability of herding livelihoods to these stressors. Herders with higher livestock wealth are more able to access secure cattle grazing on private lands, and to access more distance areas with herds of sheep and cattle. However, those with lower wealth rely disproportionately on illicit, precarious access to external grazing resources. Higher wealth families experienced disproportionately lower losses of cattle to drought, and likely have lowered sensitivity to market influences. In this context, there have been two long-term extremes of outcomes, one where wealthier actors possess entitlement sets that allow for continued cattle and sheep herding through reduced exposure and sensitivity to multiple stressors, while less wealthy actors rely primarily on keeping herds of goats or no longer keep livestock. These results have implications

for recent market-oriented interventions and governance reforms that are intended to restructure people-environment relations in East African wildlife conservation settings. As climate change is expected to have large impacts on livelihoods in the region, considering the drivers of current vulnerabilities in detailed context across households within communities is of high importance.

Introduction

Previous analyses in Laikipia, Kenya have documented how livestock husbandry practices have changed over time, leading to stratification in livelihoods, with large differences in herd sizes and market relations (Herren 1987, Herren 1991). In our own work, it has been shown that privatization, conflict, formalization of group ranches, and conservation trusts have all contributed to changes in the access of Maa-speaking pastoralists to forage resources in one community in Laikipia, Kenya, leading to restructuring of livestock husbandry over the past thirty years (Chapter 2). Livestock husbandry has become increasingly individualized, as are the benefits of wildlife conservation, and the landscape of access to forage resources. Due to these factors, we expected the structure of the specific ways that people gain access to forage resources, and are able to utilize forage resources to enhance their livelihoods and cope with stressors such as drought, to have changed over time for communities as a whole. In what follows, we use the approach of considering entitlements, access, and adaptive capacity to analyze the structure of herding practices in a Maa-speaking pastoralist community in Laikipia, Kenya to try to understand the expected impacts on local livelihoods that occur in an area where privatized wildlife conservation has become the main economic focus of policies (Western et al. 2009). Drawing from previous studies of livelihood vulnerability, and our understandings of recent institutional changes (Chapter 2) we asked how types of access are distributed among households, what is required to achieve access, and how these relate to livelihood outcomes and differential vulnerability to multiple stressors today. Additionally, we asked if differential vulnerability has been reinforced by historical institutional changes in the privatized wildlife conservation landscape of Laikipia. We relate the structure of access to herders' response capacity and ability to sustain their livelihoods, and relate the structure of vulnerability to recent changes in governance, interventions, and policies.

A long line of scholarship has explored the context-specific factors that mediate complex human-environment interactions as they shape livelihoods, and the ability to cope with and adapt to variable and/or changing climates. Much of this work can be traced back to early approaches in hazards, livelihoods, and political ecology (Adger 2006). Historically, hazards research traced causality directly to the biophysical hazard, while entitlements and livelihood approaches traced causality to political-economic and social factors that interact with the biophysical hazard (Ribot 2014). However, there has increasingly been a large amount of interdisciplinary borrowing and overlap between these approaches to vulnerability analysis (Turner et al. 2003, Adger 2006), building upon work that emphasized the importance of considering numerous dimensions of social context in addition to biophysical hazards (Watts and Bohle 1993, Blaikie et al. 1994, Pelling 1999). More recently, "integrative" vulnerability analyses (Ribot 2010) such as Turner et al.'s framework (2003) have provided a synthesis, merging perspectives of entitlements (Leach et al. 1999, Sen 1981, Sen 1984), livelihoods (Scoones 2009) and political ecology (Blaikie and Brookfield 1987) with biophysical analysis of hazards. These approaches have strong abilities to incorporate institutional and political economic factors explicitly (Eakin and Leurs 2006), and see vulnerability as "embedded in complex social relations and processes" (Nelson and Finan 2009), with understandings of an uneven ability of different people within a society to respond to hazards (Turner et al. 2003, Ribot 2010).

The impacts of climate change are expected to be experienced unevenly in many contexts (Bassett and Fogelman 2013, Marino and Ribot 2012), illustrating the importance of approaches that are attentive to those who are most sensitive or exposed to disturbance and those who are most likely to bear the brunt of climate change impacts. Further, many authors have emphasized the importance of multiple stressors (Adger 2006, O'Brien et al. 2004, Turner et al. 2003, Räsänen et al. 2016, McDowell 2012), in particular how global economic factors interact with climate change to structure vulnerability (Eakin 2005, O'Brien and Lienchenko 2000). Some analyses stress the deep interdependence between development, ecological variability, and vulnerability (Adger 2006, Nelson and Finan 2009), placing vital importance on decisions at multiple scales (O'Brien et al. 2007) and understanding the implications for the livelihoods of those most vulnerable to stressors (Ribot 2010).

The ability to cope and adapt to hazards is often structured by the interaction of institutional, environmental, and household factors at multiple scales, leading to a complex biophysical and social basis of differential abilities to respond (O'Brien 2007). Eakin (2005) and Agrawal (2010) show how multi-scalar institutional changes can impact the contextual vulnerability of rural livelihoods. There is thus a need for understanding how people experience multiple interacting stressors and the impacts on their livelihoods, necessitating multiple metrics of livelihood outcomes (Eakin and Leurs 2006). In many contexts there is a need for detail about specific household factors, and how those interact with institutions and shape the ways that people respond to changing environments. There is usually no single variable with a strong correlation between livelihoods and stressors that can be measured, so there is a need for consideration of cultural, political-economic, and biophysical factors, captured through quantitative empirical accounts, but also ethnographic accounts of relative and perceptual dimensions of vulnerability (Eakin and Leurs 2006). Such an approach can lead to understandings of causality that indicate a wider range of policy options, rather than policies that merely address proximate factors (Ribot 2010).

In attempting to bridge understandings of the interaction of ecological and social factors from a social-ecological systems perspective, the concept of adaptive capacity (Folke et al. 2002, Gallopin 2006, Nelson et al. 2007, Smit and Wandel 2006) is frequently used to conceptualize and analyze the underlying social conditions that shape adaptation to environmental change. For example, adaptive capacity in semi-arid pastoralist systems has also been used to explore how systems of exchange and reciprocity have allowed for flexibility in movement as a buffering strategy against spatial and temporal variability in environmental resources (Leslie and McCabe 2013). These analyses from a social-ecological systems perspective also link closely to multi-scalar understandings of how social factors create shifts in human/environmental relations at the landscape scale (Cumming et al. 2006). However, the idea of adaptive capacity has faced criticism (Cote and Nightingale 2012) in its tendency to conceal the drivers of changes, leading to an attribution of vulnerabilities to a localized, proximate lack of response rather than to the ultimate underlying factors (Bassett and Fogelman 2013, Ribot 2014). In focusing on the internal characteristics of capacity rather than external factors, the political economic and social factors that have historically shaped vulnerability can be masked. Ribot (2014) argued that crises are often rooted in historical and social factors, challenging researchers to go beyond simply understanding the capacity to adapt and cope with vulnerability.

To understand different pastoralist herding livelihood pathways and how vulnerability is structured at multiple scales, we began with an analysis of institutions, i.e. the “rules of the game”, or constraints that shape human interactions (North 1990, Ostrom 2015). The institutional dimensions of access can then be addressed systematically using an entitlements approach to understanding access to important resources (Leach et al. 1999, Sen 1981, Sen 1984). Entitlements are ‘the set of alternative commodity bundles that a person can command in a society using the totality of rights and opportunities that he or she faces' (Sen 1984) that mediate their ability to utilize other resources or endowments (Leach et al. 1999). They can be thought of as the gains that people are able to achieve, given their endowments, or assets (Bebbington 1999) and based upon their own production (Adger 2006). In analyzing entitlements, exposure to stresses due to ecological change can be examined as experienced differentially across social strata, and resulting in differential vulnerability that is related to institutional and economic factors (Adger 2006). Leach et al. (1999) emphasize that intertwined ecological and social outcomes differ at different scales and how institutions can mediate these processes. Other have explored how in pastoralist settings, changing institutions can necessitate a new set of entitlements to access resources, and that in turn shape how livelihoods adapt to these novel institutional conditions (Goldman and Riosmena 2013). This approach interjects an improved toolkit for understanding social and political contexts (Leach et al. 1999), adding greater nuance to other formulations of the entitlements approach (Devereux 2001).

Other extensions of entitlements approaches have been put forth to gain greater depth and clarity with respect to property and property rights (Devereux 2001). However, approaches that expand beyond a one-dimensional understanding of property (Leach et al. 1999), and draw from studies of a legal pluralism and a more socially based understanding of access and local systems of legitimation (Ribot and Peluso 2003, Sikor and Lund 2009), greatly expand the understanding of entitlements beyond a purely legalistic understanding of access. Such analyses can be further extended to analyze dimensions of access that go beyond property and rights, in order to understand "bundles of power" (Ribot and Peluso 2003) in the structure of entitlement and access, and to indicate unevenness of vulnerability.

When analyzing adaptive capacity (Folke et al. 2006, Nelson et al. 2007), or the ability of a system to evolve in order to cope with greater environmental variability (Adger 2006), an environmental entitlements approach to map access and benefits (Leach et al. 1999) can be used to identify different constraints and possibilities for change. Simultaneously, using an entitlements approach can lead to an understanding of the mechanisms by which benefit flows are "gained, controlled, and maintained" (Ribot and Peluso 2003), leading to an elaboration on the underlying power relations that structure access, and that also play a role in the ability of a system to change. Thus, an entitlements framework allows for a systematic analysis of the ways that household factors such as assets and endowments interact with institutional factors to structure access to resources, leading to differential livelihood outcomes and vulnerabilities to stressors. It also enables consideration of the differential ability of some to adapt, and the institutional constraints to adaptation that may also drive differential impacts, while simultaneously analyzing the ultimate political, social, and economic drivers of vulnerability, leading to a robust consideration of possible policy responses that address the underlying drivers of vulnerability (Ribot 2010).

We drew from the above understandings of access, entitlements, adaptation, and livelihood vulnerability to study recent changes in pastoralist livelihoods in central Kenya. Pastoralism, or reliance on domesticated livestock for over 50% of household income, is thought to be the most well-suited subsistence activity for much of arid or semi-arid lands that cannot support farming (Ellis and Swift 1988, Homewood et al. 2008, Behnke, Scoones, and Kerven 1993). Mobility is essential to gain access to key resources that are highly spatially and temporally variable across semi-arid landscapes (Ash et al. 2002), and fragmentation of semi-arid lands has limited the response capabilities of herders, leading to decreased efficacy of subsistence by pastoralism alone (Hobbs et al. 2008). When seasonal grazing movements cannot occur, vegetation has less chance to recover seasonally, and there may be decreased nutritional returns (Fratkin 2001). Also, when sedentarization occurs, wealthier herders may be better able to accumulate assets and to strategically diversify (McPeak and Barrett 2001). Pastoralists in semi-arid rangelands are expected to experience disproportionate impacts of climate change (Ericksen et al. 2013, Huho et al. 2009), potentially compounding the impacts of rangeland fragmentation.

In analyzing livelihood vulnerability at Koija, a pastoralist group ranch in Laikipia, Kenya, we treat exposure as the inability to avoid drought conditions, and expect that exposure level will be a function of herding entitlements within a given multi-family household (Figure 3.1). We treat sensitivity as the degree to which drought exposure will impact well-being, which is expected to vary as a function of household wealth or herd size, and the ability to buffer herd losses due to drought, diseases, as well as offtake for sales. Using this framing, some households, for example, may be more able to decrease their exposure to drought through mobility, while other households may be less sensitive to stressors, even though they are unable to mobilize to decrease their exposure (Figure 3.1). These two factors can then interact to shape a given households' overall vulnerability (Figure 3.1).

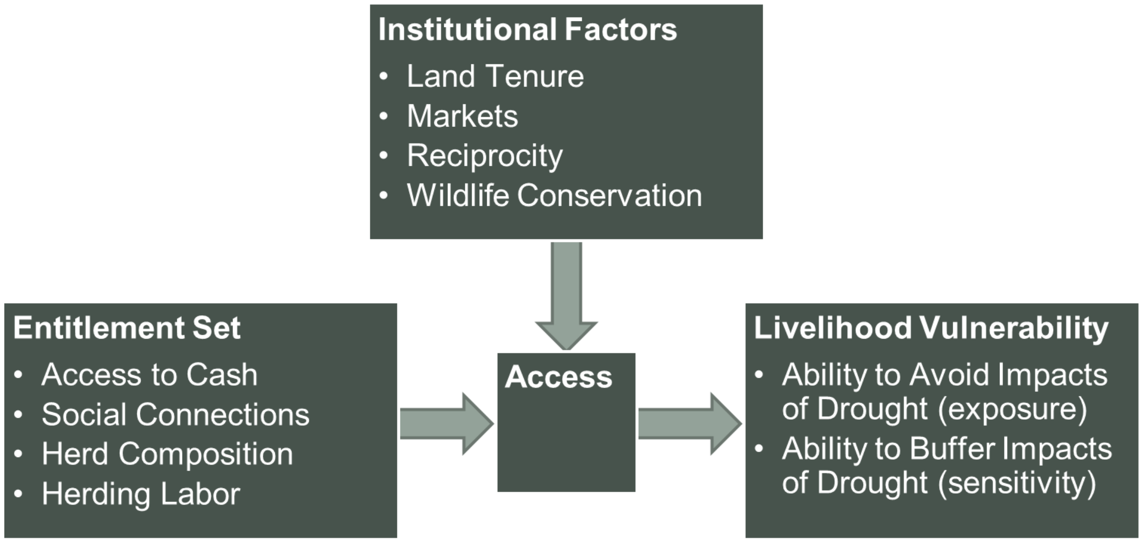

Figure 3.1. Conceptual framework of the interactions between institutions and entitlement sets that shape access, in turn impacting household vulnerability to drought

Building upon a previous analysis of changes in herding institutions, which indicated recent decreases in reciprocity of livestock husbandry at the same time as changes in access to forage and types of animals being kept had occurred (Chapter 2), we asked how types of access are now distributed among households, what underlying factors structure this distribution, and how this new system of access relates to vulnerability to stressors. We then related this structure of vulnerability to historical institutional changes in Laikipia. Our analysis is divided into the following analytical categories that detail the uneven structure of livestock husbandry as experienced through (1) Patterns of household assets that enable reduced exposure to drought, (2) differential access to forage resources, (3) sensitivity to livestock markets, and (4) sensitivity to drought and disease. We systematically consider these categories in terms of household factors that we expect to affect exposure and sensitivity. We explore how those household factors interact with formal and informal constraints to provide detailed analysis of the ways that households access forage resources. We combine a survey-based quantitative analysis of vulnerability, an in-depth ethnographic analysis of herding livelihoods, and statistical analysis of trends informed by ethnographic accounts. We explore how access to forage is structured according to livestock wealth, and how entitlements at the household level interact with employment, livestock markets, and novel informal and formal herding institutions during both drought and non-drought times. Our findings indicate that a novel set of entitlements are necessary for maintaining access to drought forage resources, and that current patterns of entitlement are correlated with bifurcation of livelihood strategies and deepening stratification of wealth within one group ranch.

Historical Changes in Pastoralist Subsistence Practices in Laikipia

Prior to mass removals that occurred under the British colonial authority throughout the early 1900's, Laikipia was populated by both hunter-gatherers and pastoralists (Cronk 2004, Hughes 2006). In 1914 many of the Purko-Kisongo Maasai who occupied Laikipia at the time, weakened following a war between sections, were forcibly relocated along with other pastoralists to reserves in southern Kenya to make way for the creation of the "White Highlands" (Hughes 2006). Subsequent legislation forced the remaining indigenous inhabitants of Laikipia onto the driest, most marginal land (Herren 1987), later known as the Mukogodo Reserve, and today as Mukogodo Division. Following this forced relocation, a variety of complex interactions are thought to have resulted in increased intermarriage between various hunter-gatherers and the remaining pastoralists in this area, and the predominant livelihood is thought to have shifted from hunter-gatherer to primarily pastoralist between 1925 and 1936 (Cronk 2004).

The East Africa Royal Commission (also referred to as the Dow Commission) of 1952 deemed the common property regime of pastoralists to be the root cause of land degradation, and recommended that eventual subdivision should be the goal of policy (Mwangi and Ostrom 2009). This was in large part based upon an understanding of the pastoralist commons as areas with no regulatory institutions that would inevitably become degraded unless privatized (Rutten 1992). Commercial ranching for beef production has consistently been favored by official Kenyan policies (Broch-Due and Anderson 1999), and in 1970, group ranches, or subdivisions within reserves (Group Representatives Land Act Chapter 287), were legislated based upon the rationale of conversion of livelihoods toward commercial beef production, with the intention to reduce livestock and avoid land degradation (Rutten 1992), a move that was internationally supported by USAID (Fratkin 2001). Subdivisions within Mukogodo Division were designated at the time, but group ranches here, as well as throughout Kenya, did not become commercial livestock producers for a variety of reasons (Waller 2012), and the subdivisions were largely ignored (Kaye-Zwiebel 2011). Prior to the early 2000's, government presence mostly involved enforcing the outside borders of Mukogodo Division to constrain cattle movements, encouraging market integration (Herren 1987), and extracting taxes on cattle.

While the number of livestock units per household has decreased dramatically throughout East Africa since the colonial era, Mukogodo Division has been shown to have very low levels of livestock wealth compared to other Kenyan pastoralists, and notably high numbers of small stock (sheep and goats) relative to cattle (Herren 1991). These differences in herd composition are thought to be driven by a combination of loss of grazing access outside of Mukogodo Division, and the subsequent drought events, disease, and excessive offtake as herders sought to purchase grains to survive (Herren 1991). Reciprocal networks of animal exchange, common among pastoralists, are thought to have begun to break down in the mid-1980’s due to extreme stratification of the population following a series of droughts and increased market integration in the early 1980's (Herren1987, Herren 1991). Today there is a continuum of wealth differences between families and between group ranches not just in terms of cattle (Herren 1991), but in ability to subsidize pastoralism and engage in other activities that supplement wealth (Kaye-Zwiebel 2011). Additionally, some households are able to engage in livestock markets under relatively beneficial conditions, while other producers sell animals under less favorable conditions (Hauck 2013). Over the past 30 years, as the landscape has continued to become more fragmented, seasonal grazing access inequality between households has deepened, and a more individualistic approach to livestock husbandry has emerged (Chapter 2). Where herding labor for large multi-family groups was historically provided by the *ilmurran* (unmarried males highly trained in cattle herding) collectively for the entire group, today this herding labor, as well as many other dimensions of livestock husbandry has become increasingly individualized among households (Chapter 2). The reasons driving individualization are complex and include livestock markets, reduced reciprocity in animal exchanges (Herren 1991), and the focus of Kenyan public education (Lesorogol 2008). Decreased sharing of labor between households also is related to continued loss of access to seasonal grazing, the biophysical constraints of herding small stock, rising inequality, and the relations of individuals with conservation actors (Chapter 2).

The collapse of the Kenya Meat Commission at the end of the l980's, coupled with decreased exports of animals to the Middle East, led to a decrease in the viability of commercial cattle-only ranches in Kenya (Heath 2001), driving shifts to a new business model of pro-wildlife conservation ranching and ecotourism in Kenya. Livestock market prices today are mainly determined by private buyers and are highly variable, especially during drought. A global trend toward ecotourism and charismatic large mammal conservation created a new set of incentives for conservation at the national, regional, and global scale (Homewood 2008). These have come to affect Laikipia's group ranches as well. In addition to changes in land use on neighboring private ranches with an increasing orientation toward wildlife conservation, group ranches have formed conservation trusts in partnership with ecotourism operators and conservation-oriented NGOs. These partnerships have resulted in the formal titling and adoption of statutory group ranch governance and management structures in Mukogodo Division (Kaye-Zwiebel 2011). The main international backers for this were The African Wildlife Foundation (AWF) and The United States Agency for International Development (USAID), along with several private conservancies and NGOs based in Laikipia. As part of the formation of these trusts, group ranches set aside wildlife conservation areas that are intended to exclude livestock, while ecotourism enterprises were developed to generate revenues to be channeled toward employment, infrastructure, healthcare, and education services (Sumba et al. 2007). This can be understood as a hybrid governance system, with some state functions devolved by the state to local conservation NGOs (Ribot 2002). NGO governance then interacts with the formally recognized group ranch governance structure, which in turn relies on customary authority for support (Kibugi 2008, Kaye-Zwiebel 2011, German et al. 2016).

Study Site

Koija group ranch is approximately 7605 ha and home to at least 2761 people living in approximately 243 *nkangitie* (*nkang*, singular; a residential compound of an extended household of several nuclear families, usually of patrilineal descent). The majority of people who reside at Koija group ranch trace their lineage to the LeUaso hunter-gatherer group, with some stating historical ties to Maasai, Samburu, and Laikipiak Maasai groups. Frequent references in casual conversation are made to recent ancestors who primarily hunted, gathered, and kept bees for a living. Today, while being primarily pastoralists, many people continue to keep bees. Koija is located on the western edge of Mukogodo Division, bordering the Ewaso Ng'iro river to the west, other pastoralist group ranches to the east and south, and Isiolo county to the north. It is at an elevation of 1700 meters, with a mean annual precipitation of approximately 450mm per year, with a coefficient of variation of annual rainfall of about 40%. The precipitation patterns at Koija show substantially higher variability and lower mean annual rainfall compared to the majority of Laikipia county (Franz et al. 2010). Rainfall is bimodal, with the highest amounts typically occurring in April-May and November. It is characterized as semi-arid, with long, unpredictable periods where rainfall is very low and consequently vegetation is highly variable seasonally. The landscape vegetation matrix is highly heterogeneous with patches of alternating *Acacia* spp. mixed with grasses, herbaceous glades, areas of recent succulent encroachment (King et al. 2012), vertisol soils dominated by perennial grasses, and dense *Acacia mellifera* areas where shrubs have recently encroached.

Methods

Beginning in 2013, we (Unks and co-author Naiputari) completed eight focus-group discussions with elder herders (men and women), two within each of four areas of where *nkangitie* are clustered together at Koija, to determine salient ecological and livelihood changes that have occurred over recent history. Naiputari then completed surveys (sampling all households) with an elder at each *nkangitie* who is involved with herding decisions (male or female, average age estimated at ~48.2 yrs.). Two brief follow-up surveys were done at each *nkang* in 2014 and 2015. We were unable to arrange to speak with 18 *nkangitie* to complete surveys, and between 2013 and 2014, four *nkangitie* relocated to areas outside of Koija, so follow-up surveys were not completed for these *nkangitie*. Data collected included information on livestock and *nkang* wealth, income, herding practices, seasonal herding location place names, livestock sales, response to and impacts of drought, and herding labor to analyze household entitlements in relation to access and sensitivities. Recent livestock numbers were confirmed using a systematic count of the entire group ranch by Naiputari in 2016, as well as a comparison to recent counts done by the Koija grazing committee. Historical estimates were also compared to Group Ranch counts conducted by AWF in 2004. Income for salaried employees and other sources of income were estimated per year based upon Naiputari's knowledge of different positions, approximate pay scales, and market value of different goods, such as honey. Finally, 20 in-depth key informant interviews were done in 2013-2014 with senior elders about herding ecology and livelihood changes over the previous 30 years. Naiputari translated and transcribed all focus-groups, surveys, and interviews from Maa to English. Trends were coded and analyzed using NVIVO software (Version 11). Statistical analyses of survey data (ANOVA, multivariate clustering, t-tests, contingency analysis, correspondence analysis, and analysis of means for proportions, spearman correlations) was completed using JMP Pro software (Version 12).

To enable statistical analysis of the herding characteristics of *nkangitie*, we analytically grouped *nkangitie* according to livestock wealth characteristics using hierarchical clustering in JMP Pro (Version 12). This enabled grouping of *nkangitie*, based upon both composition and size of herds. We considered 214 out of 244 of Koija's *nkangitie*, excluding those with no livestock or for which we had incomplete surveys. We separated *nkangitie* into clusters of similarity based upon livestock numbers from the 2016 counts. Using a two-stage hierarchical clustering approach (Ward's method, standardized data), with numbers of cattle, goats, and sheep per average adult male equivalents (Nestel 1986) within each *nkang* as input variables. We achieved maximum separation between groups with 5 clusters. By analyzing the graphical representation of clusters, as well as the average numbers of each livestock species in each cluster, we then determined that three clusters all represented higher overall wealth, with all over 5.8 TLUs/AAME, an amount considered above that required for subsistence purposes (Potkansky 1999), and these three groups were merged. This resulted in three final clusters with relatively equal numbers to enable statistical comparison, which we refer to as low, mid, and high wealth levels in the results. Average livestock holdings within these three resulting categories are shown in Table 3.1.

For each household asset analyzed, we tested whether the proportion of households with access to that asset differed between the livestock wealth levels, using chi-square contingency tests. We tested whether continuous variables such as household income differed between livestock wealth categories using one-way ANOVA. All analyses were performed with JMP Pro software (Version 12).

Table 3.1. Three clusters of households representing livestock wealth levels, and their average livestock holdings divided by active adult male equivalents (AAME). Means for clusters are shown with standard error in parentheses.

Cluster	Livestock wealth level	N	Cattle/AAME	Goats/AAME	Sheep/AAME	Camels/AAME
1	high	50	3.25(0.25)	15.45(1.17)	13.96(2.75)	0.34(0.10)
2	medium	82	1.19(0.08)	7.37(0.35)	2.97(0.25)	0.04(0.02)
3	low	82	0.24(0.3)	2.51(0.17)	0.67(0.12)	0.02(0.01)

Results

The results are organized into four analytical categories that track the structure of exposure and sensitivity to livelihoods observed at Koija, and conceptually link to our approach of integrative analysis of vulnerability. These include:

1. Patterns of household assets that enable reduced exposure to drought
2. Analysis of seasonal forage access (exposure to drought)
3. Sales and herd offtake (sensitivity to livestock markets)
4. Household sensitivity to drought, and the interactions of drought with livestock diseases (sensitivity to drought)

1. Patterns of household assets that enable reduced exposure to drought

In this section, we analyze the underlying factors that facilitate the different types of access to grazing resources, and examined how access was associated with each livestock wealth level. These factors include: outside income required to pay for herding costs, household assets that allow for certain types of access, household labor that supports increased livestock mobility, and social relations that can secure access. In interviews, respondents most frequently emphasized that an additional house and cattle enclosure (boma) located in a grazing area outside of Koija, and the herding labor to be able to travel and/or split *nkangitie* -- were closely related to use of informally accessed areas outside of Koija. When asked why some herders may avoid these areas, it was commonly said that in order to exploit these areas, in addition to having a house and boma there, additional medicines and the ability to split *nkang* herding labor are required. Another factor that was sometimes emphasized was the aid of a motorbike to carry necessary items such as building materials, food, and water, and also to purchase medicine, salt, and grains at reduced rates in nearby cities. Motorbikes also aid in transporting family members who are unable to walk the long distance to informally accessed areas, as well as to transport newborn and juvenile animals that lack the required mobility, and to scout for forage during droughts.

When we analyzed these reported patterns quantitatively, we found that high wealth *nkangitie* were significantly more likely to have an additional house and livestock enclosure outside of Koija compared to medium wealth *nkangitie*, and both were more likely to have an additional house and livestock enclosure compared to low wealth *nkangitie* (Table 3.2). Also, high wealth *nkangitie* were more likely to own motorbikes compared to medium wealth *nkangitie*, and both high and medium wealth *nkangitie* were more likely to own a motorbike compared to low wealth *nkangitie* (Table 3.2).

The ability to move to informally accessed areas was reported to largely depend on the *nkang* labor required to build and maintain a house and cattle enclosure, as well as the herding labor required. High wealth *nkangitie* were more likely than medium wealth *nkangitie* to combine their herds when migrating, and both were more likely to combine when migrating compared to low wealth *nkangitie* (Table 3.2). Herders in the high wealth category were also more likely to hire *ilmurran* for herding labor compared to medium wealth *nkangitie*, with both being more likely than low wealth *nkangitie* to hire *ilmurran* (Table 3.2). Finally, high and medium wealth *nkangitie* were more likely to express confidence in herders than low wealth *nkangitie* (Table 3.2). These patterns demonstrated that access to all the key enabling assets for external grazing mentioned in interviews, were positively associated with household livestock wealth.

Table 3.2. Percentages of households in each livestock wealth category that reported access to key herding assets, as reported in 2014 surveys (*indicates category significantly differs from all others in chi square contingency analyses)

Livestock wealth Category	Additional House and Livestock Enclosure	Owned Motorcycle	Combines Herds When Migrating	Hire *ilmurran*	Confidence in herder
High (n)	34.00* (50)	58.00* (50)	36.73* (49)	24.00* (50)	97.96 (49)
Medium (n)	10.98* (82)	28.57* (77)	25.61* (82)	3.70* (81)	95.12 (82)
Low (n)	1.22* (82)	8.00* (75)	13.41* (82)	0.00* (82)	81.58* (76)
Pearson χ^2 (df)	30.578 (2)	37.035 (2)	9.624 (2)	29.555 (2)	12.602 (2)
Probability	<0.0001	<0.0001	0.0081	<0.0001	0.0018

We then assessed the costs of herding and the role of different types of access to cash in facilitating herding success. While there was no significant difference between the three livestock wealth categories in the prevalence of outside employment (Pearson χ^2=0.087, p=0.9572), high livestock wealth households had significantly higher levels of income due to jobs located outside of Koija, typically on neighboring private ranches (ANOVA, $F_{2, 211}$=4.1379, p=0.0173). It was frequently emphasized in interviews that outside income, usually from work on a conservation-oriented ranch, allows individuals to offset the costs of herding, and in previous work we found that there was a greater likelihood of cattle and sheep herd increases between 2002 and 2016 for *nkangitie* with a member that was employed (Chapter 1).

Costs incurred on a day to day basis that were reported were subsequently coded into categories, reported here as ranked by number of times mentioned by *nkangitie* within different livestock wealth categories, and then also reported according to the number of times that factor was volunteered first, as a proxy for salience (Table 3.3). Analyzing these by wealth cluster indicated that *nkangitie* in the low livestock wealth category were less likely to indicate the costs of all categories except medicine and salt (Table 3.3), primarily due to the lower amounts of livestock that they keep. Medium wealth *nkangitie* were less likely to indicate the costs of paid grazing, the costs of herder payments, and the costs of paying for food for herders compared to high wealth herders, but were at the same time more likely to indicate the costs of illicit grazing (Table 3.3), a factor that was frequently emphasized in interviews.

Table 3.3. Herding costs (N=214) according to if they were ever mentioned, or mentioned first, and contingency analysis if ever mentioned

	Ever Mentioned	Mentioned First	High Wealth Percent	Med Wealth Percent	Low Wealth Percent	Pearson χ^2	Prob>ChiSq
Medicine	211	136	98	98.78	92.68	4.79	0.091
Salt	198	5	90	95.12	87.80	2.82	0.245
Paid Grazing	101	46	86.00*	53.66*	17.07*	61.44	<0.001
Cost of Illicit Grazing	58	14	28.00*	36.59*	17.07*	7.93	0.019
Herder Payments	23	10	28.00*	7.32*	3.66*	20.82	<0.001
Food for Herders	10	0	16*	2.44*	0*	19.34	<0.001

An additional factors determining herding success in general was said to be the ability to allocate cash to purchase food, and herding success thus depending strongly on outside income, the ability to sell animals at good prices at strategic times, and to a lesser extent, the ability to sell honey to supplement income. Another theme mentioned was that people with larger herds do better during droughts, both in terms of their ability to recover, even after losing a large amount of animals, and also in their ability to sell animals to cover other costs. Herd size was frequently said to be important to buffer the necessary offtakes seen in sales, as well as deaths of animals due to drought or predation, particularly if there is no outside income. Further, these impacts of sales on herd size are thought to be compounded during droughts when prices for animals are low.

In summary, while interviewees often emphasized that herding outcomes are a result of luck or skill of herders, individuals with greater abilities to maneuver access in nearby areas with reserve grazing for cattle and sheep, are also often able to allocate *nkang* resources in ways that facilitate splitting of *nkangitie* to multiple locations and mobility, to allocate funds in different ways for livestock care, and to manage herds in times of drought. Households with higher livestock wealth had greater access to many of the assets that provide such abilities, revealing complex interdependencies between herd size, income, and strategy that impact exposure to drought.

2. Analysis of Seasonal Forage Access

In this section, we report the structure of seasonal forage access outside of Koija and analyze how access, as a key element of successfully sustaining sheep and cattle herds, and thus a pinch-point of livelihood vulnerabilities during drought, is distributed among the three livestock wealth categories of *nkangitie*. Within the common areas of Koija, the ability to exploit pastures is open to all members at the group ranch scale. Goats typically remain in closer proximity to *nkangitie* than cattle and sheep, except when traveling to water, as they can subsist on nearby vegetation. Cattle rely on grasses and remain mobile, moving throughout the group ranch and beyond when possible, while sheep require an intermediate level of mobility, surviving to some extent on perennial grass resources that cattle cannot. To access forage outside of Koija, four main pathways exist. One of these is through areas that have been open to all Koija residents over recent decades and are typically either formerly government lands or lands that were granted to absentee landholders in the post-independence era, and that have a recent history of being accessed by pastoralists seasonally. These are referred to as informally accessed sites from here on. Another type of access is through paid grazing arrangements where a set quota of cattle can graze on neighboring privately-owned conservation ranches. The third is access granted through relationships with employers on these same conservation ranches to employees or people with close relations. Both of these permitted grazing arrangements have large beneficial impacts for these animals during drought. The final pathway is through illicit access, where herders access areas that either they or elders understand as their land historically, but for which no legal access rights exist today. In this final case, if the area is formally privately owned, the authority to enforce exclusion is conferred by the state to private landowners, despite this concept of exclusive ownership being locally contested by Koija residents. Forage is rarely made available on privately owned conservation ranches for sheep and goats, so small stock only utilize informal and illicit access pathways.

Goats rarely leave Koija, but sheep were frequently stated by many to be largely dependent on informally accessed areas with different ecological conditions, including areas with vertisol soils and large amounts of *Pennisetum mezianum* (*lgurume*) grass. It was stated in interviews that these areas are only suitable for cattle following consistent rains that make this grass suitable for cattle grazing and create sufficient surface water pools in ephemeral ponds, rock catchments, or seasonal streambeds. Therefore, cattle access is dependent on either permitted grazing on private ranches or in illicit areas for long time periods when no surface watering points are available. All members of Koija are thought to be permitted to access resources in the informally accessed areas, and no one indicated in interviews that anyone from Koija was ever denied access. While it is commonly said that there is no practice of exclusion limiting access at these places, they are often avoided due to density of livestock and risk of disease.

In 2013, when it was not considered drought, but there was reportedly insufficient cattle forage on Koija, 52.9% of cattle owned by Koija residents (1687 of 3189 head) were reported to have remained at homesteads within Koija. Of the 1502 head of cattle reported to be located outside Koija, 86.15% (1294 head), from 142 *nkangitie,* were located on 5 private ranches, and 13.84% (208 head) were located within informally accessed areas. High wealth *nkangitie* had 756 head of cattle from 44 *nkangitie* on private ranches; medium wealth *nkangitie* had 397 head of cattle from 59 *nkangitie* on private ranches, and low wealth *nkangitie* had 141 head of cattle from 39 *nkangitie* on private ranches. Less than 480 of these cattle were thought to have been present through paid grazing, while at least 814 are thought to have been present through personal relationships or through employment. The 10 *nkangitie* with the largest numbers of cattle, who also notably did not mention using illicit areas, had 347 out of their 891 head of cattle on the private ranches, accounting for 38.98% of their own cattle, and making up 26.82% of the overall access permitted on private land at this time. At this same time, it was reported that 98.13% of sheep that were being herded off Koija were accessing forage within informally accessed areas, and 87.82% of sheep that were herded off Koija were in just three of these informally accessed areas.

We analyzed rates of access to external forage areas during severe drought in March-April 2015 as a proxy for the exposure component of vulnerability. At this time, there were no cattle remaining at *nkangitie* on Koija, there was very little herbaceous vegetation remaining, and animals of all livestock breeds were frequently dying of starvation. During this time, recording the number of cattle in specific locations outside of private ranches was difficult as herds were often split and frequently shifted between locations according to forage and water availability. However, patterns of reporting whether or not different types of pathways were used during the drought indicated stratification in the ways that cattle owners were relying on informally accessed, paid access, employed access, and illicit forage access. Higher wealth families were much more likely overall to report they had cattle on private ranches at this time, through paid and employed access (Table 3.4). Use of informally accessed areas was reported primarily by *nkangitie* within the medium and high wealth categories (Table 3.4), with only 11 of the 82 low-wealth *nkangitie* using these areas. At the same time, medium wealth families were most likely to report they were reliant on illicit areas for at least some cattle, which was significantly greater than reported use of illicit areas by either high or low livestock families during this time (Table 3.4). Many emphasized that use of these areas was accompanied by high risks in terms of danger due to buffalo, elephants, lions, and leopards, in addition to potential monetary penalties or jail sentences. However, many indicated this was their only choice at this time to keep their cattle from dying.

Table 3.4. Percentages of *nkangitie* reporting use of different areas for cattle grazing during the 2015 drought (n=154, excluding *nkangitie* with no cattle, df=2, * indicates category significantly differs from all others)

Livestock Wealth Category	Cattle on Private Ranches Grazing	Cattle in Informally Accessed Grazing	Cattle in Illicit Areas
High	72.34*	61.70	85.11*
Medium	50.00	56.06	95.45*
Low	41.46	26.83*	70.73*
Pearson χ^2	9.38	12.35	12.64
p	0.009	0.002	0.002

Similar to cattle access, the medium wealth families reported greater use of illicit areas for sheep during the drought (Table 3.5). All wealth categories differed in accessing grazing resources for sheep in informally accessed areas at this time, as well as in the likelihood of sheep remaining at *nkangitie* on Koija (Table 3.5), with high wealth *nkangitie* being most likely to use informally accessed areas, and low wealth families being most likely to have their sheep at Koija. Considering goats, the lowest wealth *nkangitie* expressed a greater tendency for goats to remain at Koija, as well as being less likely to use informally accessed or illicitly accessed areas (Table 3.6). This was said to be related to lower wealth *nkangitie* with their typically smaller herds of goats being able to survive primarily on recently encroaching *Acacia mellifera* leaves during this time. Among low wealth *nkangitie,* 35.37% indicated their livestock, which were primarily small stock, could survive on Koija, compared to just 4.00% and 7.32% in high and medium wealth *nkangitie,* respectively (Pearson χ^2=30.614, p=<0.001).

Table 3.5. Percentage of high-, medium-, and low-wealth *nkangitie* with sheep using different areas during drought in 2015 (n=167 *nkangitie*, df=2)

	Sheep in Informally AccessedAreas 2015	Sheep in Illicitly Accessed Areas 2015	Sheep Remained at *Nkang* 2015
High	56.00%*	30.00%	14.00%*
Medium	32.43%*	39.19%*	28.38%*
Low	18.60%*	32.56%	48.84%*
Pearson χ^2	19.86	19.86	19.86
	<0.001	<0.001	<0.001

Table 3.6 – Percentage of high-, medium-, and low-wealth *nkangitie* with goats using different areas during drought in 2015 (n=208 nkangitie, df=2).

	Goats in Informally Accessed Areas	Goats in Illicitly Accessed Areas	Goats Remained at *Nkang*
High	14.00%	52.00%	34.00%
Medium	13.41%	43.90%	42.68%
Low	5.26%*	21.05%*	73.68%*
Pearson χ^2	23.76	23.76	23.76
P	<0.001	<0.001	<0.001

To assess potential exposure to continued drought, we asked representatives of *nkangitie* if they had alternative plans for grazing access if the 2014 drought continued into 2015. High wealth *nkangitie* were more likely to indicate they had backup plans compared to medium wealth *nkangitie*, and both were more likely to have a backup plan compared to low wealth *nkangitie* (Pearson $\chi^2(2, N=142) = 14.508$, $p=0.0007$).

In summary, three clear access patterns are apparent for different categories of livestock wealth. The differential use of informally accessed areas for sheep and cattle forage by different wealth classes indicates that herders with higher livestock wealth utilize informally and formal areas to reduce their exposure to drought to a greater extent than medium and lower livestock wealth herders. Notably, the ten *nkangitie* with the highest cattle wealth constituted nearly a quarter of the overall access to private ranches during 2014. Grazing quotas available to a cross-section of Koija constituted a small proportion of required access, and it appears that the ability to secure additional access required cultivation of relations with private ranches through employment or other means. Most of this access was enabled through relationships with conservation ranches, which confer direct grazing privileges to *nkangitie* with employed members and close confidants. It appears that higher wealth herders accessed these pathways more, while medium livestock wealth herders were instead more reliant on illicit areas, and lower livestock wealth herders relied primarily on areas within Koija. Rather than keeping cattle and sheep, lower wealth *nkangitie* that did not navigate the access pathways detailed here were relying nearly exclusively on goats, which are less sensitive to drought than cattle or sheep.

3. Sales and Herd Offtake

In the earlier section on patterns of household assets that enable reduced exposure to drought, we showed how access to cash has numerous impacts on herding, and is used to support cattle herds. Here we consider herd size and market relations to contextualize the balancing act that families must navigate between buying grains as food, trying to avoid offtake, and allocating cash when possible to the costs of herding. These dynamics also affect the ability of *nkangitie* to buffer themselves from, and thus reduce their sensitivity to, negative impacts of droughts.

It was repeatedly emphasized in interviews that employment aids some people in their herding. Alternately, it was emphasized how large herds of small stock could lead to increased sales in order reallocate those funds to maintaining cattle, as cattle require cash access. Through participant observation, we also frequently observed that *nkangitie* without outside income paid for herding costs through animal sales, and it was emphasized by many that employment enables herders to reduce their small stock offtake, which in turn allows their herds to grow.

In 2014, high wealth *nkangitie* sold on average 14.91% (+/-1.26) of their small stock (sheep and goats) and 11.82% (+/-1.67) of their cattle per year, while medium wealth *nkangitie* sold 25.41% (+/-3.31) of their small stock, and 12.25% (+/- 1.44) of their cattle. On the other hand, low wealth *nkangitie* sold 61.36% (+/-11.08) of their small stock and 16.53% (+/-3.76) of their cattle. Also, *nkangitie* with high and medium livestock levels were less likely to sell livestock for food (18.37%, 20.00%, respectively) than low wealth *nkangitie* (66.67%) (Pearson $\chi^2(2, N=201) = 44.71$, p=<0.001).

There were also differences in the specific markets where individuals sold animals, which interviewees emphasized as highly important because of differences in the prices paid and their proximity. The most local market was in Ewaso, the trade center on Koija group ranch. The name of the weekly market is “Soko Mjinga”, which means "fool's market" in Kiswahili, a playful deprecation of the low prices offered compared to the larger markets in two nearby centers, Kimanjo and Oldonyiro, which have government sanctioned livestock auctions every two weeks. Some stressed that especially in more local markets, individuals would commonly buy animals from others who lacked the ability to travel to more distant markets, and then would transport purchased animals to more distant markets where higher sale prices were obtained.

Another salient aspect of livestock sales is that people with larger herds of goats, which reproduce more rapidly, are thought to be able to sell more animals at their convenience when prices are high, while others are forced to sell on a more frequent basis, regardless of current sale prices, to get cash for day to day living expenses. High and medium wealth *nkangitie* were less likely (10.00% and 14.63%, respectively), compared to low wealth (28.05%) to sell at Soko Mjinga (Pearson $\chi^2(2, N=214) = 8.10$, p=0.017). Comparing families with cattle, high wealth *nkangitie* were also more likely to sell cattle at the even more distant and larger market in Rumuruti town (26.00%), when compared to medium wealth *nkangitie* (10.8%), and both high and medium wealth *nkangitie* were more likely to sell there than low wealth *nkangitie* (0.00%, Pearson χ^2 (2, N=172) = 15.68, p=<0.001). No significant differences were found between livestock holding categories and sales rates at Kimanjo or Oldonyiro markets or to nearby ranches. Confirming the greater strain of sales on lower wealth *nkangitie*, low wealth *nkangitie* were more likely (22.67%) compared to high wealth (6.00%) and medium wealth (8.64%) *nkangitie* to say that they were unable to sustain livestock without another source of income to supplement their livelihood (Pearson $\chi^2(2, N=206) = 9.654$, p=0.008). Due to having only one year of sales data and the high fluctuation in prices between years, we were unable to quantitatively analyze the direct impact of offtake on herd dynamics. However, herders often emphasized the strain of selling animals for food and school fees, and the interactions between offtake, food, employment, and the costs of maintaining cattle. Goats were reported as crucial to keeping cattle during times of drought, especially in terms of being able to sell or exchange goats to pay penalties when caught grazing within illicit areas or to pay fees for paid grazing arrangements during these times.

Finally, it was also frequently mentioned that these factors were compounded during drought, as market prices decreased dramatically. In summary, external employment appears to enable some to forego livestock sales that are otherwise necessary for meeting basic needs. Additionally, those with larger herds have lower percentages of offtake in their herds and are more likely to sell under favorable conditions, that likely translate into decreased strains on livelihoods and decreased sensitivity to drought.

4. Sensitivity to Drought and Interactions with Livestock Diseases

Finally, we explored herd losses through drought as a stressor. The drought of 2015 reached its most extreme period in April 2015, with very little forage available, and with rains then ending the drought by the end of the month. A total of 2479 sheep reportedly died during this time. Sheep had higher reported rates of death when herded outside Koija (26.49%) during the drought, compared to animals located within dry season grazing areas within Koija (18.63%), with both differing from the lowest rates of death which were seen for sheep located at the *nkangitie* (7.27%). ($F_{2, 155}$=15.22, p=<0.001). A similar pattern applied to the 2092 goats that died, with higher percentages dying outside Koija in either informally or illicitly accessed areas (14.68%, 11.29%, respectively), compared to 4.30% of those that remained at *nkangitie* on Koija. ($F_{2, 206}$=11.4016, p=<0.001). Based upon key-informant interviews, these differences in rates of death appeared to be related at least in part to the higher exposure to disease for animals outside Koija, but also due to animals being weakened by overall lack of forage consumed during drought. Some characterized these as combined risks experienced when leaving Koija that they simply had to take. On the other hand, others felt that there was less risk involved in staying at Koija rather than leaving with their animals. Supporting the high perceived risk of loss to disease, 92.59% of medium wealth *nkangitie* reported that disease was a dominant barrier to successful herding, compared to 86% and 77.33% of high and low wealth *nkangitie*, respectively (Pearson $\chi^2(2, N=206) = 7.31$, p=0.026). This pattern reflects medium livestock wealth households needing to leave Koija during drought, and being in a state of high precarity of access compared to high wealth families, which is compounded by a decreased ability to afford medicine, and greater overall risk.

Of goats that died during the drought, it was reported that 13.29% died from disease, 28.75% died from drought, and 57.96% died from a combination of drought and disease. Of the sheep that reportedly died during the drought, 22.87% died from drought, 16.44% died from disease, and 60.69% died from a combination of drought and disease. Out of 243 head of cattle that died during the drought, 54.73% were thought to have died from drought, 14.81% died from disease, and 30.45% were thought to have died from a combination of drought and disease.

To evaluate whether animals from smaller herds experienced higher mortality rates, we first had to account for the higher likelihood of sampling error if mortality rates are calculated for small herds individually (for example, if a *nkangitie* only owns 2 head of cattle, the mortality rate can only be 0%, 50%, or 100%). To overcome this, we first ranked all *nkangitie* from highest to lowest cattle holdings, then used a binning procedure to create bins of multiple *nkangitie*, such that each bin contained about the same total number of cattle. The largest bin was made up of the two *nkangitie* with the largest cattle herds, adding up to 340 head total. We then binned the next largest *nkangitie* until the bin contained approximately 340 head total. This procedure was repeated, working down the list of ranked *nkangitie*, resulting in 10 bins, each of which contained a set of *nkangitie* whose total cattle holding were about 340 head. Survey data for each of the *nkangitie* indicated the number of cattle that died in the drought. We could therefore calculate the mortality rate within each bin, and the average cattle herd size for *nkangitie* within that bin. There was a strongly negative correlation between average cattle herd size rank and mortality rate (Spearman ρ = -0.70, p=0.025). A similar approach to analyzing deaths of sheep, resulted in 17 bins that each contained a total of about 600 sheep. We found a non-significant negative correlation (Spearman ρ = -0.43, p=0.082). This same approach when applied to goats (39 bins, bin size of 415 goats) also yielded a non-significant relationship (Spearman ρ = 0.23, p=0.159), implying that goat and sheep mortality rates were not related to herd size, and that goats' sensitivity to drought was independent of the wealth level of their owners.

Overall, the deaths of cattle and sheep appear to have been disproportionately borne by those with the smallest herds, with the bottom 49.26% of *nkangitie* (*N*=148) ranked according to cattle wealth experiencing 154 losses (62.85% of total losses), while the top 51.74% (*N*=26) experienced 91 losses (37.15% of total losses) in cattle. A similar pattern held for sheep, where by rank of size of herd, those with smaller herd size (49.6%, *N*=159) experienced 61.55% of the deaths, while the 20 *nkangitie* that held the other approximate half of the sheep experienced just 38.45% of the deaths. However, the deaths of goats were shared much more equally, with the bottom 49.45% of *nkangitie* (*N* = 174) experiencing 49.67% of the goat deaths, and the 40 largest *nkangitie* holding 50.55% of the goats experiencing 50.33% of the deaths.

Considering only *nkangitie* with cattle (*N* =159), a larger percentage of *nkangitie* with high wealth used cattle dewormer (68.09%) compared to those with medium wealth (45.45%), and both were more likely to use dewormers compared to low (32.61%) wealth *nkangitie* (Pearson $\chi^2(2, N=159) = 12.114$, p=0.0023). No significant differences were found in the use of goat or sheep dewormers. Additional aspects that were emphasized in interviews was the importance of preventative use of medicine, and having cash on hand to be able cover the costs of medicine to treat livestock diseases.

Discussion

The structure of livestock holdings at the *nkang* level in Koija was significantly related to uneven patterns of seasonal forage, in terms of cattle access to private conservation ranches, and the ability to herd sheep and cattle outside of Koija. A suite of *nkangitie* assets and endowments, including the ability to split *nkangitie*, to have transportation, and to supplement labor through combining herds and hiring herders allows for enhanced ability to decrease exposure to drought. In addition to vulnerability being differentially structured, our analysis indicates that control of access is structured in a patron/client manner between conservation actors and herders, primarily revolving around employment relationships. Further, access is closely related to interrelated factors of outside employment, access to cash, and herd sizes, which have complex interactions and ultimately impact *nkang* sensitivity to drought. Coupled with changes in individualization of herding practices, this individualized system of access necessitates those that access the limited forage that is accessed informally or on private lands outside of Koija have the assets required to sustain cattle and sheep herds today. At the same time other herders rely upon marginal subsistence strategies found in the use of illicit areas, or keeping small numbers of goats that can survive on Koija.

For those unable to offset sales with access to outside cash or larger herd size and reproduction, market factors have a disproportionately negative impact, yet herders are forced to sell animals to buy grains to feed their families. Coupled with low market prices, especially during droughts when offtake is necessary for those that jobs are unavailable to, this leads to a positive, reinforcing feedback on marginal livelihoods. Further, the drought event we document here indicated that overall losses are disproportionately borne by families with lower livestock holdings who are in turn less buffered against losses, factors which likely combine to create a threshold of constraints where goat herding remains the only viable strategy for many. The shift to goats as the basis of livelihood, an adaptation to external access constraints and the need for a cash income, can decrease *nkangitie* sensitivity to drought, but at the same time may involve greater sensitivity to market factors. Herders that have enhanced entitlement sets due to employment, access to cash, allocation of herding labor, maintenance of second *nkangitie*, and access to medicine and transportation, seem to simultaneously be able to avoid exposure to drought and to be better able to negotiate market interactions. Our analysis indicates overall a complex interplay between multiple factors, that leads to three heuristics of strategies to cope with drought: those of the employed and those who can otherwise subsidize herding, those who somewhat successfully rely on precarious, illicit access to areas outside of Koija, and those who rely primarily on small herds of goats herded within Koija alone.

Numerous factors currently interact to create multiple stressors at different scales and shape the vulnerability of livelihoods. Synthetic analysis of vulnerability using access and entitlements elucidated how constraints on herding success have morphed over time, and pinpointed the ways they act on *nkangitie*. The current structure of seasonal forage access preferentially allows herders with certain *nkang* assets and endowments to readily access grazing, and we found that *nkangitie* have vastly differential exposures to drought. While animals differ in their sensitivity, the amount of sensitivity at the *nkang* scale then is also a function of herd size and income, both which relate to the ability to buffer herd loses in drought and reduce offtake of animals for sale in livestock markets. Taken together, stratified access to seasonal grazing, market interactions, and exposure to disease when migrating constitute multiple stressors that interact and shape vulnerability. Our approach added a more pluralistic account of rights (Sen 1984, Leach et al. 1999) beyond understandings based upon legal rights of access alone, that is closely compatible with analysis of access (Ribot and Peluso 2003). This approach necessitated analysis of the historical and continued marginalization of pastoralist livelihoods and institutions and the ultimate causes of vulnerability (Ribot 2010). The unequal ability to decrease exposure to drought, coupled with heightened sensitivity to drought and market offtake in *nkangitie* without income or with small herds, creates an uneven structure of vulnerability within Koija. This clarifies how in an analysis of adaptive capacity, the ability to adjust responses to drought and ecological change is socially stratified, and how some *nkangitie* may be differentially favored within the wider institutional context. It also clarifies that the current systemic causes of vulnerability are closely linked not just to local livelihoods at Koija, but to institutional factors that have occurred as a result of historical colonial, post-colonial, and conservation-era institutional changes that have shaped the possibilities for adaptive capacity (Chapter 2). Rather, the uneven outcomes of the ability to adapt to the novel context cannot be explained by *nkang* characteristics alone at Koija, and do not simply indicate a setting where some have adapted and some have not. On the contrary, those that herd goats also have also adapted to changing conditions, however, the entitlements approach that we took makes clear how the adaptive options in communities are uneven, stratified, and closely linked to complex institutional changes that have occurred recently. The ultimate causes of this differential vulnerability and adaptive capacity can be understood with greater nuance when considering changes in wider institutions of access and livestock markets, individualization of herding practices, personal relations with conservation actors, and monetization of access and livelihoods in a context of limited opportunity for waged labor. Our results indicate what amounts to a transformation of the customary norms of access, to one that requires capital and relational access, that when considered in historical context amounts to a privatization of the use of these resources by the wealthiest and most closely associated with wildlife conservation actors. These changes have future implications for pastoralist herding in Laikipia, as climate change is expected to impact the seasonal variation of rainfall in drylands throughout east Africa. Overall, these constraints also restrict the ability of herders to respond to environmental conditions, leading to localized concentration of livestock that potentially lead to runaway impacts on important forage species (Chapter 2).

Conclusion

With vegetation resources being highly seasonally variable in semi-arid rangelands, pastoralist grazing arrangements in Kenya are thought to have been very flexible and fluid in the past (Mwangi and Ostrom 2009). Livelihoods at Mukogodo Division are marked by their ongoing adaptation to fragmentation of their access to resources, and numerous dimensions of historical changes in livelihoods (Herren 1991, Letai and Lind 2013, Huho and Kosonei 2013). Recent changes have shaped the institutional context at multiple scales, these include changes in land use and access on lands with private title, conflict in "abandoned" lands (Letai and Lind 2013), changes in pastoralist collective title in surrounding lands, as well as changes in livestock markets and employment, to informal changes in livestock husbandry institutions and relations with conservation actors. Approximately twelve months of intensive study of pastoralist livestock husbandry between 2012 and 2016 on Koija group ranch led to an improved understanding of the way that herders make decisions about their herding and assets. Herders are repositioning themselves to adapt to changing conditions of forage access and livestock markets in a context where privatized wildlife conservation has also become increasingly prevalent.

Previous analysis emphasized how the institutional changes underlying access and individualized livestock husbandry had led to constraints on livelihoods, and that inequality was being deepened as a result (Chapter 2). In that chapter, we also found that current informally accessed grazing sites cannot adequately support cattle year-round, and that the current quota of paid grazing is insufficient, forcing herders that lack the entitlement required to attain grazing through individual relational means on private ranches, or to graze illicitly in areas that were historically accessed. In the past, however, social strata with lower holdings would likely have sent animals with families and pooled entitlements to access off-site forage resources, but today this has shifted to a landscape of access where the costs and benefits of mobility are borne individually. Building upon this understanding, we analyzed how limited forage availability, coupled with highly individualized access, shapes present herding outcomes. Here we analyzed how inequality is structured among extended families, and shapes differential outcomes in vulnerability in a context of increasing drought and changing vegetation.

Through this analysis of the structure of vulnerability and the underlying drivers, we hope to inform decision making that is imbedded in transnational wildlife conservation networks, local power relations, and social stratification that all play roles in outcomes. Our findings add further nuance to understandings of livelihood changes in pastoralist livestock husbandry within a landscape where many state functions have essentially been delegated to wildlife NGOs (DePuy 2011, Kaye-Zwiebel 2011), catalyzing political, economic, and social changes. This shift in economics and land use has been driven by the global push for large mammal conservation and Laikipia's emphasis of wildlife conservation on private lands (Western 2009), and increasingly is driven by a rush of outsiders buying land in Laikipia (Letai and Lind 2013). In contrast to the intended diversification of pastoralist livelihoods (Sumba et al. 2007, Sundaresan and Riginos

2010) sometimes assumed to accompany the growth of the conservation economy, and while being one of the main creators of jobs in the region likely has only a slight potential to create cascading changes in livelihoods (Little 2014). The number of jobs at Koija have decreased recently (Chapter 2) and the overwhelmingly dominant livelihood remains livestock. Most herders at Koija find themselves in a state of constant precarity, and ultimately, these changes have likely interacted to drive a historical bifurcation of strategies and deepening inequality along the lines of knowledge, wealth, and social connection.

It is common in Laikipia for wildlife conservation actors to interpret current pastoralist livelihood barriers as due to internal constraints such as population growth or lack of group ranch grazing management. However, the synthetic approach to understanding livelihood vulnerability we used indicates that the current capacity of the majority of groups of extended families is to adapt to factors such as drought and changing climate via mobility is limited by factors that have their origins in institutional changes at multiple scales. At a time when a suite of novel market-oriented reforms and governance structures are being advocated by NGOs and sweeping through Kenyan rangelands, building upon the group ranch model of conservancies, it is important to understand how these changes will shape future access, and reinforce or change the trajectory of these past trends. The proposed shape of these projects are based upon limiting the current access of herders to seasonal grazing areas, currently through quotas, and initiating market-based incentives to reduce stocking rates and focus on keeping fewer, larger, more profitable beef cattle. However, such projects must explicitly consider the current stratification of livelihoods and the overwhelming reliance of the majority of the population of group ranches in Mukogodo Division on small stock, if they are not going to further deepen inequality. In areas where mobility and access are so crucial to livelihoods, we expect conservation negotiations to be inherently tied to multiple dimensions of governance, conservation, and range management. It is our hope that the approach we have taken here, considering the politics of access and the ultimate causes of vulnerability, will contribute to improved outcomes and an openness to more inclusive consideration of the interests of overlapping pastoralist livelihoods and wildlife conservation outcomes.

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CHAPTER 4

AN INTERDISCIPLINARY METHODOLOGY FOR ESTIMATION OF LANDSCAPE-LEVEL LIVESTOCK PRESSURE IN PASTORALIST HERDING SYSTEMS USING SURVEY DATA AND LEAST COST CORRIDORS[3]

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Abstract

Pastoralist systems around the world are increasingly experiencing multiple stressors in the form of social, political, economic, and climatic challenges. These challenges are often intertwined with geographic constraints on mobility, which play a role in patterns of increasing sedentarization. The relative ecological impacts of changes in herding practices are frequently dramatically different from historical impacts, and difficult to tease out from other processes at the landscape scale, as they often involve changes in herd composition, timing of seasonal use, and overlapping types of novel pressures. Using an interdisciplinary approach drawing from landscape ecology and from ethnographic accounts, we used survey data and readily accessible GIS software to create estimates of non-migratory livestock pressure within one pastoralist group ranch in Laikipia, Kenya. Using the least cost path algorithm, a resistance surface optimization procedure was performed to maximize prediction of landscape use based upon slope, norms of land use, and proximity to other families' homesteads. The optimization procedure was iterated until it maximized prediction of the preferred livestock watering points of each extended family living within a homestead compared to their actual use of watering points. These predictions were verified by comparison to a subset of homesteads that were withheld from the analysis, and to predictions of preferred watering points based upon distance alone. Summed least-cost corridor maps for landscape use from all homesteads were then created and validated with community-wide herding patterns based upon dung counts and known pathways. Validation indicated that the optimized resistance surfaces led to lowered error in predictions considering individual homesteads, but negligible improvements over predictions based upon distance alone. Despite lack of improved community-wide watering point preference, verification showed that corridors calculated using optimized resistance surfaces were able to explain 21% of the

variation in observed livestock densities for goats. This indicates potential future utility in more nuanced estimation of complex, heterogeneous landscape use. While this analysis provides a partial insight into understanding changes in landscape process, it fills a methodological gap in the current ability to understand interactions at multiple scales in coupled human and natural systems inquiry.

Introduction

Understanding the complex ways that rural land users directly interact with ecosystems is a key question in coupled human and natural systems research (Liu et al. 2007), yet such studies face a suite of methodological challenges. In systems where rural livelihoods are directly dependent on ecosystems, and also simultaneously produce feedbacks that affect ecosystems, interdisciplinary approaches are required to understand the ways interactions between ecosystems and livelihoods are impacted by ongoing changes such as intensifying land use (Turner et al. 2003) or increasing ecological variability as climate changes (Kramer et al. 2017). Pastoralist ecology in semi-arid savannas provides an example of a tightly coupled interaction between ecosystems and livelihoods, with vegetation availability and regeneration being key determinants of pastoralist livelihood sustainability.

The structure and ecological functioning of savanna vegetation is shaped by interactions with topography (Augustine 2003), rainfall (Good and Caylor 2011), largescale disturbances such as fire, and complex biotic interactions including ungulate herbivory (Tarnita et al. 2017, Sankaran et al. 2013). Increasingly, other factors such as climate and atmospheric CO_2 concentration are thought to play roles in vegetation changes such as shrub encroachment (Stevens 2017, Archer 2010, Archer 2017). Recent approaches using state and transition models of coupled human and natural system change can provide guidance in grappling with the complexity of ecosystem changes (Bestelmeyer 2011), linking process to pattern at the landscape scale (Turner 2010). However, the historical contingency of different states and multiple drivers of change in non-equilibrium semi-arid ecosystems pose methodological challenges in understanding how changes in land use may relate to shifts in ecosystem state.

When trying to understand the impacts of changes in domestic herbivore pressure on vegetation states in semi-arid lands, a key part of the challenge is estimating the herbivore pressure itself, especially in areas that lack known or clearly contrasting historical land uses. In what follows, we present a case study where we used an interdisciplinary approach drawing from social science and landscape ecology to estimate species-specific livestock pressure across a collectively-titled pastoralist group ranch in Laikipia, Kenya. We asked whether or not, using a hybrid methodology where community-wide herder descriptions of their daily ranges were used to create estimates of generalized land use in a GIS, we could improve upon current methodologies of estimating herbivore pressure. The approach used combined general daily herding ranges elicited through ethnographic methods, combined with current animal movement analysis methodologies commonly used in landscape ecology. Within the area selected for study, a number of changes in livestock husbandry and vegetation have occurred simultaneously over the past 30 years. We completed this study with the aim of generating high-resolution estimates of grazing pressure, which would enable tests of alternative hypotheses about livestock's contribution to observed changes in vegetation. Establishing a more nuanced understanding of whether, and which, vegetation changes are in fact associated with livestock pressure is particularly important in a context where wildlife conservation is increasingly prominent, and pastoralist herding is commonly viewed by wildlife conservation actors as the primary factor responsible for degradation of vegetation and soils.

Study Rationale and Background

Increasing attention has been given to understanding how sedentarization and fragmentation of cattle herding ranges can lead to concentration of ecological pressure (Groom and Western 2013, Hobbs et al. 2008, Reid et al. 2008). Yet because of the complex interactions that affect vegetation structure, it is difficult to discern the relative contributions of these factors to changes in vegetation. An additional complexity arises because vegetation sensitivity to herbivore pressure changes seasonally, dry season periods can be a pinch-point when herbaceous vegetation is especially susceptible to impacts (Hodgkinson 1995). Furthermore, different livestock species affect vegetation in different ways. As many pastoralist groups undergo sedentarization, they tend to shift herd composition to include more sheep and goats -- smaller, drought-hardy livestock species that do not require mobility. Thus, understanding the effects of fragmentation of herding ranges and sedentarization must also account for the effects of small stock on vegetation, for example the differences of impacts due to differences in mouthparts (Rook et al. 2004), as well as cascading impacts on herbaceous vegetation due to impacts on woody structure (Gabay et al. 2011).

Semi-arid ecosystems often express spatial variation in the structure of vegetation across a landscape as a result of heterogeneous abiotic conditions, in addition to gradients of human use. Landscape structure can often be the result of historically contingent, context-specific responses of different vegetation types to multiple processes including human land use. Within many pastoralist areas, many of which have remained until very recently largely unfenced, there are often relatively few areas where replication can occur across similar vegetation types that have experienced contrasting land use treatments. Rather, there is often a heterogeneous mosaic of complex intergrades of land uses. There are exceptional cases when one can distinguish sharply contrasting areas of historical land use across a landscape with a common vegetation type (e.g., Harrison and Shackleton 1999), but often, such tightly controlled ideal scenarios that mimic experimental conditions are not present (Bestelmeyer et al. 2003).

These combined methodological problems facing analysis of land use change are often compounded in pastoralist studies by the lack of available data on landscape-scale use. The scales of available data on livestock presence (e.g. census data) are often difficult to appropriately analyze for correlation with ecological change, due to their coarse spatial scale or low frequency of measurement (Turner and Hiernaux 2002, Moritz 2010). In many instances, this lack of sufficient data is worked around methodologically through use of piosphere models (Andrew 1986, Graetz and Ludwig 1976, Heshmatti et al. 2002, Lange 1969), which use distance gradients from a concentration point (e.g., a watering point or settlement) to estimate herbivore pressure. While piosphere models have the advantage of being theory-driven and generalizable, and their use has contributed novel understandings of rangeland ecology, they lack the nuance to account for several factors known to influence livestock densities: the differences in land use strategies that occur between families, multiple forms of landscape heterogeneity that affect herding decisions, and higher relative use of areas that are far from concentration points (Turner and Hiernaux 2002). All of these can lead to deviations from the assumed monotonic decreases in grazing pressure with distance from concentration points.

Alternately, the use of livestock GPS collars can be used to gain detailed information on landscape use. Yet because of financial constraints, typically only a small fraction of herds utilizing an area are usually tracked. Thus, GPS analyses of herding ranges may not account for inter-familial diversity in herding strategy, and thus may not provide representations of the community-wide landscape process of livestock grazing (Turner and Hiernaux 2002). GPS collar data by itself is context-specific; in order to generalize beyond the time period and subset of animals that were monitored, additional data and analyses are needed to infer what landscape or decision-making factors may be governing observed movement patterns.

As an alternative to piosphere and animal GPS collar methods, Turner and Hiernaux (2002) suggested using herder accounts of their day-to-day landscape use to this land use. This approach offers four important advantages. The first relates to efficiency and feasibility of data collection. While lacking the precision of GPS studies, in a single account, herders can provide information about their land use in different seasons, and broad surveys of many or all land users can lead to improved estimates of community-wide use of a larger landscape matrix. Second, accounts provide details about individual herder's habits, and can help identify the general strategies employed by individual land users. This information is valuable for building more generalized knowledge regarding the factors -- beyond distance from concentration points -- that may influence herbivore pressure. Thus, in terms of building generalizable or predictive understanding, the method can provide a middle ground between GPS and piosphere approaches. Third, such an approach can also include differences in the composition of herds, and the differences in ranges of these different species on a day-to-day basis. Especially, if the purpose of estimating herbivore pressure is to relate it to vegetation change, these are important nuances to capture. The fourth advantage is that herder accounts facilitate more holistic understanding of pastoralist systems. These accounts can provide an overall representation of landscape use that synthesizes relative diversity of use from herder's perspectives while also linking it to livelihoods and the salient concerns of herders (Turner and Hiernaux 2002). For these reasons, we argue that GIS analyses of survey data, characterizing each herding household's use of the landscape, can not only provide a clearer understanding of landscape-level grazing pressure, but can also provide the basis for a generalizable, yet nuanced model of herding strategies and grazing pressures that is suitable for tests of hypotheses about land use and land cover change. The goal of this study was to develop an aggregate understanding of herbivore pressure across a landscape based on herder accounts, then develop a least cost path model of grazing pressure based on reported land use strategies, which can predict empirically observed land use and herbivore density patterns.

Least cost path, least cost corridor (Knaapen et al. 1992, Adriaensen et al. 2003), and circuit theory analysis (McRae et al. 2008) all provide powerful ways of analyzing the potential utilization of a landscape by mobile organisms. While perhaps best known for application in designing distribution corridors for wildlife in fragmented landscapes (Sawyer et al. 2011), this approach has been widely utilized within landscape ecology, spatial ecology, wildlife biology (Epps et al. 2007), and even anthropology (Schild 2015) to understand movements across landscapes. The least cost path utilizes resistance surfaces, where each cell in a raster surface is assigned a resistance value (a "cost") that represents the degree to which that cell's characteristics can constrain movement. Common factors used to determine resistance surfaces in animal movement studies are predator density, food availability, steepness of terrain, or barriers such as roads. These approaches provide alternatives to categorical representations of landscape utilization that use a binary designation of areas as simply habitat or non-habitat (Spear et al. 2010, Cushman et al. 2010). The least cost path between two points in a landscape is the path of contiguous cells with the lowest cumulative value of resistance for all the cells crossed. This methodology has been used to successfully predict the movements of organisms across a landscape matrix in a number of contexts (Sawyer et al. 2011), through creation of resistance surfaces that represent factors that constrain movement. The relative weighting of different surfaces, and the assignment of resistance values to a given resistance surface, can be used to represent a continuum of degrees of constraint to emulate factors at work in physical landscapes. Selection of resistance values is often based upon expert knowledge, animal presence data, habitat suitability models (Beier et al. 2009), or gene flow data (Epps et al. 2007). However, the outcomes of these models are sensitive to the underlying assumptions and methodologies used to create the surfaces (Mateo-Sanchez et al. 2015), and the resistance surfaces themselves are infrequently validated (Sawyer et al. 2011). To apply this approach to livestock movement entails: identifying factors that constrain movement for use as resistance surfaces, parameterizing the resistance values within each surface, and parameterizing the relative weightings of each surface.

The objective of this chapter was to assess whether, using descriptions of daily herding ranges and common landscape ecology methods, we could create estimates of generalized land use that improve upon current methodologies of estimating herbivore pressure. In what follows, we present a method for estimating landscape herbivore pressure that uses a least cost algorithm to simulate the corridors used by multiple herding households and their livestock across a community in Central Kenya. Rather than determining resistance surfaces and values a priori, we selected factors to include in resistance layers based on herder accounts of key factors that they reported as influencing their landscape use strategies. Then we used an iterative approach to optimize resistance surfaces for least cost analysis, based upon their ability to predict a known use pattern in the landscape: household selection of different available watering points. We verified the optimization with a subset of watering point data not used for optimization, and validated the herbivory estimates with empirical dung counts that sampled livestock use intensity across the landscape. Thus, in applying the least cost path/corridor technique in a novel approach to supplement rangeland ecological methods, we have sought to overcome several methodological challenges that commonly limit the estimation of grazing pressure across a landscape. The use of multiple data sources for the model development, selection of resistance surface, verification, and validation processes also attempts to address some of the methodological concerns commonly encountered in resistance surface optimization and ecological modeling more generally (Augusiak et al. 2014).

Study Site

The area of study was on Koija Group Ranch, located in Mukogodo Division, in Laikipia County, Kenya (Figure 4.1). The vegetation is a mixture of *Acacia* savanna, grassland, dense shrubland, and succulent vegetation. A number of recent changes are reported to have occurred in vegetation in the area, with multiple encroaching shrub and succulent species. In Laikipia County, 40.3% of the land is accounted for by large-scale private ranches (Letai 2011), many of which trace their origins to leases established in the colonial era. Many of these ranches aim to maintain wildlife populations alongside cattle ranching (Georgiadis et al. 2007). Of the remaining land, 27.21% is smallholder farms, and 7.45% is pastoralist Group Ranches (Letai 2011). The Laikipia landscape, however, represents a heterogeneity of land uses that are not mutually exclusive, and the ecological connectivity of the landscape in terms of wildlife habitat is very high relative to much of Kenya (Western et al. 2009) with a high degree of overlap in wildlife conservation interests and pastoralist livelihoods. There have been numerous recent ecological changes that have potentially impacted vegetation, including evidence that rainfall is becoming more variable (Huho et al. 2009, Franz et al. 2010), decreased frequency of fire (Augustine 2003), recent increases in elephant populations, and decreases in the abundance of other wild herbivores (Litoroh et al. 2010), which are thought to play a significant role in regulating woody cover (Sankaran et al. 2013).

Koija is bordered to the west by the Ewaso Ng'iro river. It is approximately 7605 ha and home to at least 2761 people living together in approximately 243 *nkangitie* (*nkang,* singular, a residential compound containing one or several households, today usually all of patrilineal descent). The majority of the people who reside at Koija group ranch trace their lineage to the LeUaso hunter-gatherer group, with some stating historical ties to Maasai, Samburu, and Laikipiak Maasai groups. Frequent references in casual conversation are made to recent ancestors who primarily hunted, gathered, and kept bees for a living. Today, while being primarily pastoralists, many continue to keep bees. The pastoralists who currently occupy Koija have been historically marginalized in the political process and stripped of historical grazing lands as a result of colonial policies and continuing legacies resulting in increasing sedentarization (Cronk 2004, Herren 1991, Letai and Lind 2013, Chapter 2). Lack of land administration that ensures spatiotemporal grazing rights (Lengoiboni et al. 2010), combined with livestock market factors, have led to a dramatic restructuring of pastoralist herding practices (Herren 1991, Chapter 2). Historically there was thought to be a large amount of movement outside of Koija during drought, but the loss of these movements has led to increased concentration and change in compositions of livestock within Koija. More recently, increasing loss of seasonal grazing land to private ownership, as well as conflict between other pastoralist groups and conservancy formation, has led to additional constraints to movement (Chapter 2). Though these changes have altogether led to large increases in sheep and goats, decreases in cattle, and some adoption of camels, the majority of people within Mukogodo Division remain reliant on livestock (Chapter 2). In our previous research in this area, we have found there to be a scale mismatch (Chapter 2) between the requirements of customary herding practices and the institutional context, of land use from a livelihood perspective. While the impacts of cattle on vegetation have been studied in Laikipia, these studies have occurred on private ranches with historically low livestock density, primarily using small (1 to 100m^2) to meso-scale (1 to 4ha) exclosures (e.g. Young et al. 1997). The majority of these studies have occurred on sites with different soils, historically different vegetation, higher and less variable rainfall, and much lower densities of small stock and humans than at Koija.

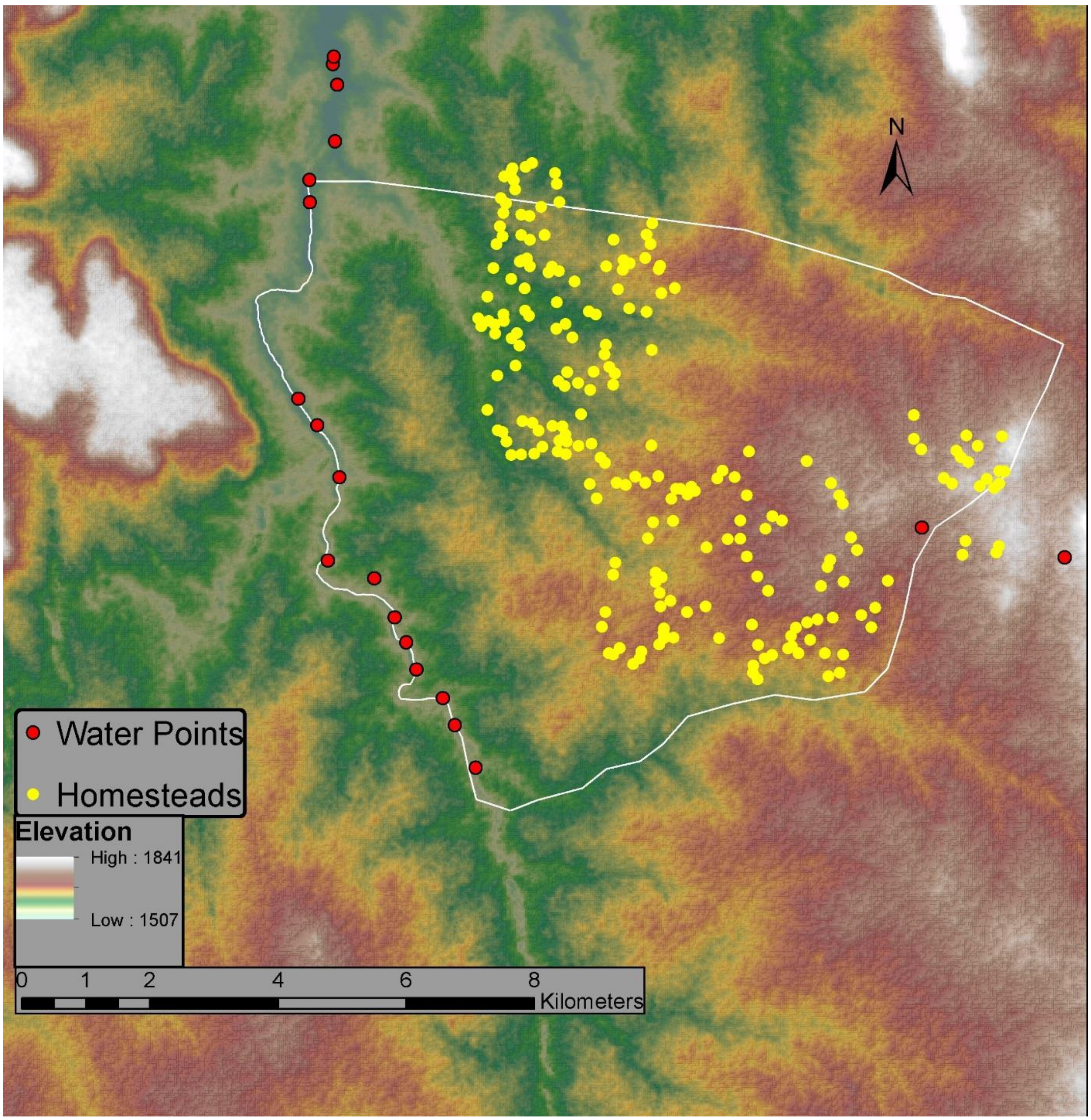

Figure 4.1 - Map of Koija Group Ranch

Methods

Ethnographic elicitation of land use and herding strategies

Participant observation, focus groups, and key-informant interviews were used to gain an in-depth orientation to herding practices and strategies used by community members. In 2013 and 2014 Unks accompanied two herders on several of their 9-11 hour daily grazing routes from homesteads to the river points, and for several months Unks and Naiputari regularly walked herding areas on a daily basis, conducting participant observation of widespread herding practices. These observations indicated some tendencies and decision points that reflected landscape use preferences and constraints on movement, informing our consideration of what parts of the landscape people tended to avoid. In 2013, Unks and Naiputari conducted eight focus group discussions, two within each of four areas of *nkangitie* that are clustered together at Koija. We then conducted 20 key-informant interviews with elder herders responsible for herding decisions in their *nkangitie*. These focus-groups discussions and interviews took place in Maa and followed a semi-structured format, with questions focused on constraints to herding and herding decisions. Focus group discussions and interviews were translated to English during the conversation, and interviews were then transcribed from Maa to English by Naiputari.

In previous studies we showed how these interviews revealed recent intensification of land use within Koija's boundaries due to a lack of access to outside areas (Chapter 2). During focus group discussions and interviews watering points emerged as key points on the landscape that determined the direction of travel and the midpoint of daily foraging routes. These watering points are particularly important during the dry season as there is not water present in catchments near homesteads, so herders are forced to travel to the Ewaso Ng'iro river, one large catchment that holds water during the dry season, or nearby boreholes for water. The majority of families take their cattle to water every day, while small stock tend to travel every other day. On days that herders did not travel to the watering points, they indicated that they tended to travel in one specific direction, but would utilize specific patches of forage more frequently than others within their daily range, for example often spending extensive time near seasonal streams, where forage tends to be ample. Using a systematic sampling of all *nkangitie* across Koija Group Ranch, we conducted a survey designed to address salient characteristics of herding that were identified during focus group discussions and interviews and that were thought to vary among households and to represent heterogeneous uses of the landscape within Koija boundaries. The survey (included as Appendix B) was completed by elders that were responsible for herding decisions at 207 out of the 254 total *nkangitie* at Koija Group Ranch. Each elder was asked to identify the watering points and daily ranges that herders utilized for the three most abundant species of livestock (cattle, sheep, and goats). They were first asked to specify watering points and distance and direction travelled on days that watering points were not used, during periods when restrictions are in place upon reserve forage areas and watering points near the Ewaso Ng'iro river. A second set of responses referred to times when no access restrictions are in place within Koija, aside from one wildlife conservation area which is formally restricted in all periods except during extreme drought. For each of these watering points identified, herders were asked to indicate the frequency of use relative to the other watering points for each livestock species. The format of answers took the form of how many days per week or month one point was used relative to other points, and this value was then coded into a percentage of the total time each point was used. Herders also were asked to indicate how frequently their family took each species of livestock to watering points, and inversely, how many days they only took the animals to forage.

Elders were asked to indicate whether they usually accessed the watering point through one pathway, and whether or not the path extended beyond the watering point before returning to their home. Elders were then asked about their typical herding ranges for each species of livestock on days when they did not take the livestock to water. Elders were asked to indicate a place name or general direction and approximate distance that livestock were herded on days not traveling to water. Finally, herders were asked if there were any places that they avoided within Koija, or could no longer access, and why. GPS coordinates were collected for all *nkangitie* homesteads and watering points.

Construction of least cost path model

Based upon these ethnographic data, we determined three elements that were frequently mentioned as factors, or observed to be factors that affected herding strategies and ranges, and that could be emulated using resistance surface layers within a GIS environment. These were: 1) steepness of topography, 2) areas that were regularly avoided by multiple *nkangitie* due to social norms or danger to herders, and 3) a gradient of proximity to other's *nkangitie* homestead locations. Slope and homestead proximity were chosen as variables based upon our assessments of herding norms and experience during participant observation exercises while accompanying herders. We also chose avoided areas as a factor to include based upon informal and formal rules and norms that emerged in focus group discussions and surveys.

Candidate resistance surface layers were created as follows. For slope, we used an Aster digital elevation model (DEM) as input. Using the ArcMap slope toolbox, the DEM was converted to a layer of percent slope at 30.85-meter resolution for the entire scene in ArcMap software. A quantile approach was then used to classify the continuous percentage values into 7 categories of slope (Figure 2). For homestead proximity, we used GPS locations of all *nkangitie* on Koija. Homestead buffer layers were then created using a 600-meter multi-ring buffer around each homestead (n=218) in ArcMap, with the individual buffers set at 100m, 200m, 400m, and 600m to create five classes of values (Figure 1). These buffer layers were then converted to raster layers with a pixel resolution of 30.85 meters.

In delineating the avoided area layer, land uses were designated as a wildlife conservation area, various avoided areas, and “neutral” areas that were not given resistance weights in the optimization procedure (i.e. no reason for avoidance known) (see Figure 4.2). Five areas were said to be regularly avoided because of elephants. Two areas were said to be avoided because bandits frequented these areas. One is designated as an area generally said to not to be used for herding except during drought events, as part of a conservation trust agreement with neighboring ranches and international NGOs (Sumba et al. 2007). Two areas were designated as potentially avoided because they represented other collectively-titled group ranches beyond the boundaries of Koija to the east. These geographic regions were drawn by tracing existing maps, then locating areas said to be avoided according to the specific place names used by residents. Place names (for ridges, hillslopes, rocks, ponds, old homestead sites, etc.) and residents' location descriptions using combinations of place names and directions were specific enough that nearly any point on the Koija landscape can be described with 10-200m accuracy. Thus the locations of avoided areas could be readily approximated and mapped using a combination of a Quickbird image from Nov 2011 and GPS reference points. For instance, areas avoided because of high elephant densities were designated by using heads-up digitizing in ArcMap and tracing a polygon around areas with dense vegetation that had been indicated in surveys by place name. Polygons were then merged and converted to rasters, at 30.85-meter resolution to match the slope raster.

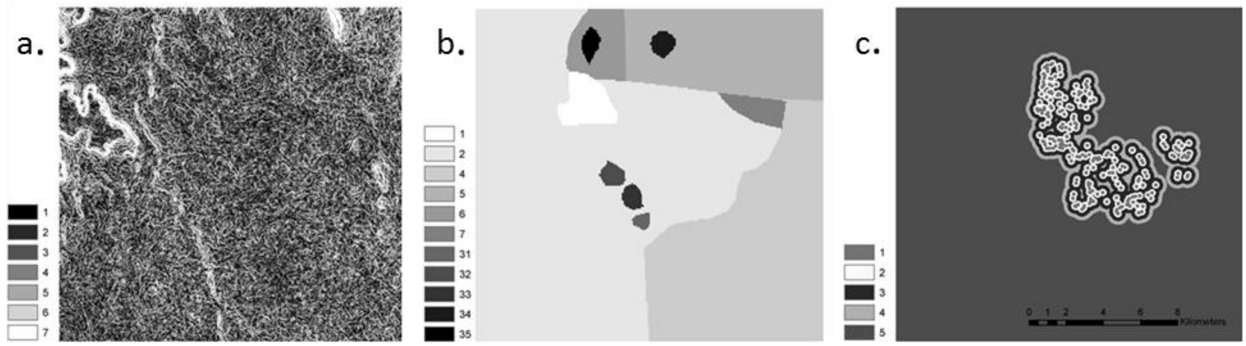

Figure 4.2. Starting inputs for the resistance surface optimization procedure a. slope b. potentially avoided areas c. homestead proximity. The areas within the potentially avoided areas are indicated by 1. Wildlife conservation area, 2. neutral areas 4. Ol donyiro conservancy 5. Nolare conservancy 6. Il motiok and Tiemmamut group ranches 7. area of high banditry, and 31-35 elephant avoidance areas.

All candidate resistance surfaces were created within Arcpy by reclassifying raster surfaces using the reclassify tool to assign resistance values first to the raster values of one individual surface, and then using the weighted sum tool where the relative weight given to each of the raster surfaces input can be designated to create composite surfaces where the importance of any layer relative to the other two could be adjusted (example python script in Appendix L).

Resistance Surface Optimization

A subset of 100 *nkangitie* was randomly selected from the total 205 surveyed *nkangitie* to use for the optimization procedure. To gain an understanding of factors that affected movement across the entire landscape, we chose the time when no restrictions were in place on land use, hereafter referred to as “dry” season watering point data, when herders travel farther distances across the group ranch to water points. Forty-three dry season water points overall were indicated to be used at this time. From focus-groups and participant observation, it was clear that during the dry season most families are forced to travel from their homestead to the river points, one nearby catchment, or a nearby borehole, all of which are farther away than they would typically tend to travel otherwise, and constitute the apogee of their daily grazing orbit. Due to many of these points being used by few families and these points, mostly located in seasonal streams, rarely holding water during the dry season, the overall number was limited to points that were known to always hold water except in extreme drought, and to be used by more than two *nkangitie*. This resulted in a final number of 21 watering points (Figure 4.3) considered in the following optimization procedure, considering the sheep and goats ranges only due to computation time constraints.

For each potential watering point, for each *nkangitie*, and for each candidate resistance surface, a costdistance and backlink raster were created and then the single lowest least cost path was calculated between each homestead location and all 21 water point locations within ArcPy (Figure 4.3, Appendix K).

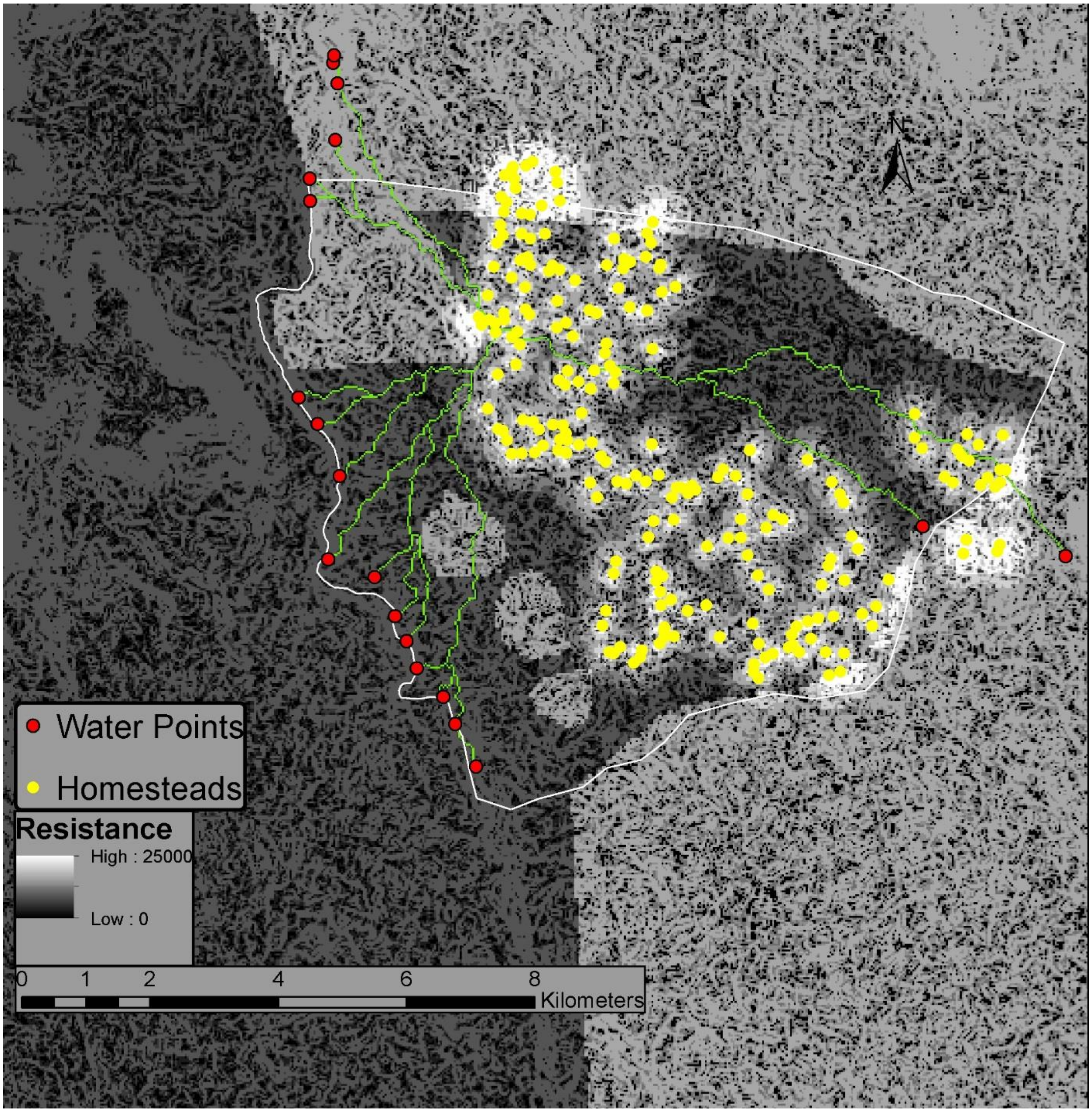

Figure 4.3. Example of least cost paths from one homestead to all possible water points (n=21).

To compare the predicted and observed frequency of watering points used by each homestead, we imported the calculated values of the cost to move to each waterpoint into a spreadsheet (Table 4.1, Columns A and B). A predicted relative rank value was calculated by dividing 1 by the number of possible points (n=21) to determine the rank interval of decrease (Column C). Then for each *nkangitie*, the observed order of watering point preferences -- a subset of less than ten watering points that were stated in surveys to be used by each *nkangitie* -- were arranged according to descending actual frequency of use (Column D), and assigned a relative use rank (Column E, observed relative rank of use). The predicted relative rank, $\hat{\gamma}$, corresponding to each observed watering point (determined using Columns A and C) was transferred to Column F. The absolute value of the difference between observed and predicted relative ranks, $|\widehat{\gamma} - \gamma|$, were calculated for each preferred watering point (Column G).

Table 4.1. Example of the sorting procedure and ranking of the least cost paths from one homestead and one candidate resistance surface, shown for a single homestead (Showing methodology for pairwise comparison of actual and predicted watering points).

A	B	C	D	E	F	G
Predicted Water Point Number	Calculated Cost Value	Predicted Relative Rank of Use	Observed Preference of Water Point	Observed Relative Rank of Use, γ	Lookup Predicted Rank of Use, $\hat{\gamma}$	Absolute Value of Difference $\lvert\hat{\gamma} - \gamma\rvert$
wp20	11787100	0.9524	wp4	0.9524	0.8571	0.0952
wp19	12211500	0.9048	wp20	0.9048	0.9524	0.0476
wp4	12328300	0.8571	wp5	0.8571	0.8095	0.0476
wp5	13474200	0.8095	wp3	0.8095	0.7143	0.0952
wp8	14933200	0.7619	wp7	0.7619	0.6667	0.0952
wp3	15009300	0.7143	wp19	0.7143	0.9048	0.1905
wp7	16060700	0.6667				
wp14	16747200	0.6190				
wp11	17096100	0.5714				
wp6	18089100	0.5238				
wp1	21230500	0.4762				
wp12	21509200	0.4286				
wp17	21840300	0.3810				
wp9	26022500	0.3333				
wp18	29822600	0.2857				
wp10	30565600	0.2381				
wp13	31434400	0.1905				
wp21	32239100	0.1429				
wp16	35678500	0.0952				
wp15	37951500	0.0476				
wp2	47798000	0.0000				

For each candidate raster layer, the above procedure was completed for all homesteads (detailed explanation of weighting procedure follows below). We then took the average of the absolute difference values (Column G) for each homestead to account for the variable number of water points used traveling from different homesteads, and then calculated the Root Mean Square Error (RMSE) of all homesteads for each candidate resistance surface. Where $\hat{\gamma}$ is the predicted relative rank, γ is the actual relative rank of each waterpoint used, n is the number of waterpoints used from each homestead, and N is the total number of homesteads:

$$\text{RMSE} = \sum_{i=1}^{N} \sqrt{\frac{\left(\frac{(|\widehat{\gamma_{1\iota}} - \gamma_{1i}| + |\widehat{\gamma_{2\iota}} - \gamma_{2i}| + \cdots + |\widehat{\gamma_{n\iota}} - \gamma_{ni}|)}{n}\right)^2}{N}}$$

In each iteration of multiple resistance surfaces, we chose the resistance surface with the lowest RMSE as the optimal solution, and adjusted the individual resistance weights of cells within each of the three themes until no adjustments resulted in a lower RMSE. Following this optimization, we then determined the optimal relative weightings for each of the three themes within the composite raster. The overall optimization procedure had a nested structure illustrated in Figure 4, with the following iterated in sequence: 1.) variable resistance values of cells within each resistance theme with equal weights given to the three themes 2.) variable relative weights of each of the three themes of resistance layers in a composite resistance layer, 3.) variable resistance values of cells within each resistance theme. Steps 2 and 3 were then iterated until a single optimum resistance surface was not produced. Below we describe in fuller detail the ways we optimized the resistance value weightings within the avoided areas, slope, and homestead proximity themed rasters.

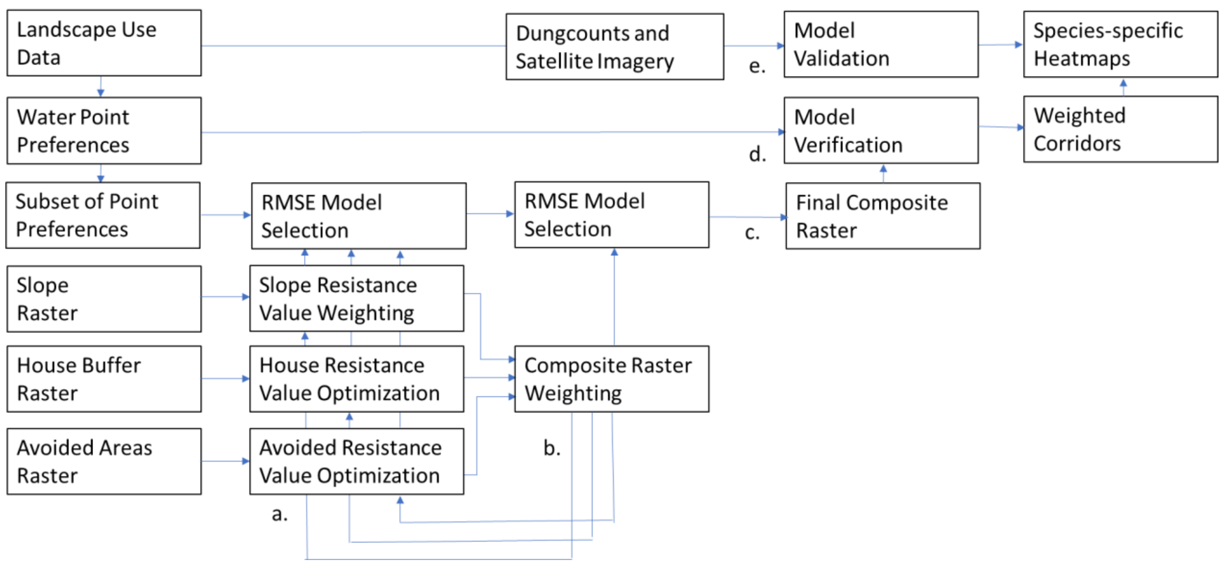

Figure 4.4. Workflow diagram

To find the optimum resistance values for the ten potentially avoided areas we iteratively varied the weights. In the initial candidate raster, cells in all ten different avoided areas were all assigned initial resistance values of 50, and the areas between these locations were designated a value of 0. Forty additional candidate avoided area rasters were created by modulating each avoided area's value by +/- 25% and +/- 50% of the maximum value (100) while holding the other avoided area's values at 50 (see Table 4.2 for example values), then determining which candidate gave the optimal resistance surface (lowest RMSE value), using the RMSE calculation described above. The values used in this layer were then used as the starting value for the next iteration, with alternative candidate values increased or decreased by 12.5% and 6.25% of the maximum value (100), and all other resistance values in the raster retained the previous iteration's values. If one of these new values was then selected in a subsequent iteration, this value was then increased or decreased by 3.125% and 1.5625% of the maximum value (100), and then finally by +/- 0.78125% and +/- 0.390625% (Table 2 shows examples of initial values). An exception was made to the above if the initial value for this class in the optimal resistance value was zero -- in order to avoid negative values in the subsequent iteration of possible values in this case the value was instead increased by multiples of 3.125% up to 12.5% of the maximum value (100) in the next iteration, and then varied by +/- 1.5625% and +/- 0.78125% in the subsequent iteration. This above procedure was repeated until no single candidate raster led to a decrease in RMSE over the previous iteration's optimal raster.

Table 4.2. Portion of the resistance value table for candidate rasters in the first iteration of avoided areas' raster optimization. We show 13 of the 41 candidate rasters in columns, and three of the 10 avoided areas (explained in Figure 4.2) in rows. The first candidate raster had resistance values set to 50 for all avoided areas. Rasters 2-5 modulated the resistance values for Avoided Area 1; Rasters 6-10 modulated resistance values for Avoided Area 2, and Rasters 11-13 modulated resistance values for Avoided Area 3, each time by +/- 50% and +/- 25% of the maximum value (100), while holding values for other avoided areas at 50.

Avoided area	1	2	3	4	5	6	7	8	9	10	11	12	13
Avoided1	50	100	75	25	0	50	50	50	50	50	50	50	50
Avoided2	50	50	50	50	50	100	75	25	0	50	50	50	50
Avoided3	50	50	50	50	50	50	50	50	50	100	75	25	0

For the initial slope raster, the class of highest slope was given a value of 100, and all other slope values were initially set to descending values according to six equal intervals from 0 to 100, with all possibilities for differences in the grade and threshold of slope across the resistance surface iterated (See Table 4.3 for example values). The value contained in the optimum raster was then varied by an increasingly finer increment, varying the slope value up and down by +/- one twelfth for each value, then 1/24th, then 1/48th, and then finally, 1/92nd until no further optimization was possible.

Table 4.3. Subset of resistance values used in the first iteration of slope candidate surface rasters (separate candidate rasters in columns, subset of pixels in rows).

Slope class	1	2	3	4	5	6	7	8	9	10	11
<3.17% Slope	0	0	0	0	0	0	0	0	0	0	0
3.17 - 5.54 % Slope	100	83.33	66.67	50	33.33	16.67	0	0	0	0	0
5.54 - 7.91 % Slope	100	100	83.33	66.67	50	33.33	16.67	0	0	0	0
7.91 - 10.28 %	100	100	100	83.33	66.67	50	33.33	16.67	0	0	0
10.28 - 14.24 %	100	100	100	100	83.33	66.67	50	33.33	16.67	0	0
14.24 - 23.73 %	100	100	100	100	100	83.33	66.67	50	33.33	16.67	0
23.73 - 201.74 %	100	100	100	100	100	100	83.33	66.67	50	33.33	16.67

Homestead buffers were applied initial resistance values of fourths of 100 (0, 25, 50, 75, 100) for all possible variables (limited by the criteria that values had to be progressively increasing with decreased distance from houses). See Table 4.4 for initial example values. In identical fractions to the slope resistance surface above, as new values were selected for each class, the margin of +/- variation in the value assigned was narrowed. For each of the above steps, this process was iterated until the optimal value from the previous round was not improved by subsequent changes in values.

Table 4.4. Subset of resistance values used in the first iteration of homestead buffer surface rasters radiating out from homesteads (separate candidate rasters in columns, buffer maximum distance in rows).

Radius	1	2	3	4	5	6	7	8	9	10	11
100 m	100	100	100	100	75	100	100	100	75	100	100
200 m	100	100	100	75	75	100	100	75	75	100	75
400 m	100	100	75	75	75	100	75	75	75	50	50
600 m	100	75	75	75	75	50	50	50	50	50	50

After optimizing the values within each of the three themed resistance surfaces, we next varied the weights of the different resistance surfaces relative to each other when added together in a composite raster. The initial setting of the composite weights was an equal weighting, as shown in Table 5, candidate composite raster #1. Individual surface's weights relative to each other were then varied with an identical rationale to the avoided areas weighting, varying each surface's contribution to a composite raster by +/- 50% +/- 25% of 100, +/- 12.5%, +/- 6.25% of

100, and then +/- 3.125%, and +/- 1.5625% of 100. See Table 5 for example weights of other candidate resistance surfaces.

Table 4.5. Subset of resistance values used in the first iteration of composite raster weighting.

Themed surface	1	2	3	4	5	6	7	8	9	10	11	12	13
House	50	100	75	25	0	50	50	50	50	50	50	50	50
Slope	50	50	50	50	50	100	75	25	0	50	50	50	50
Avoided	50	50	50	50	50	50	50	50	50	100	75	25	0

Once the weights of the individual rasters to each other were optimized, we returned to optimizing the values of the classes within individual rasters. In this second iteration of these values, all class values were varied by +/- 50%, +/- 25%, 12.5%, 6.25%, 3.125%, and 1.5625% of their values from this first round. This procedure of varying values was sequenced to begin with the raster surface that had the largest relative contribution to the composite layer and ended with the layer with the smallest contribution, with each individual raster layer followed by a series of identical overall percentages of change applied to the composite layer. To allow for the possibility that the optimal value had been underestimated in a previous round, if an increase of 50% led to the optimal outcome in any given round, the potential values of increase were maintained at a level of 50% and 25% for an additional round to allow the ability of this value to increase following changes to other components.

The iterative, multi-tiered optimization is represented in Figure 4.4 as an iterative looping of processes (a) and (b): the weights of pixels within the individual surface layers were varied (shown in Figure 4.4a),then the relative weight of themed resistance surfaces (slope, land use, and homestead buffer surfaces) were varied in the composite rasters (Figure 4.4b). In each subsequent step of (a), varying the resistance values within individual rasters, we worked in order of descending importance assigned to each resistance theme in the optimized composite weightings (b). Subsequent iterations of (a) included the resistance value that minimized RMSE in the next iteration, and modulations of that value by increasingly narrow intervals. The iteration of composite raster weights was omitted if adjustment of raster classes did not produce any changes in the previous step. This sequence was repeated until no further changes occurred and an optimum RMSE value was reached (Figure 4.4c).

Verification

Once the optimal class and composite raster weights were ascertained, the optimal composite resistance surface was then used to run the model with all *nkangitie* that were excluded from the optimization (n=105), and the RMSE value, comparing observed and predicted watering points, was calculated for verification (Figure 4.4d). Total RMSE was also calculated for all homesteads from both the optimization subset and the verification subset (n=205), and compared to the RMSE calculated using the optimization data. In order to compare the result to a “neutral” resistance surface, as a proxy for Euclidian distance alone, the model was also run using a “neutral” resistance surface where all pixel values were identical in the resistance surface, so that there was a cost for distance traveled, but without any spatial variability in resistance otherwise.

Validation

The areas of highest frequency for location of least cost paths were compared to the most commonly observed paths used for herding. To do this, we summed all least cost path output rasters produced using the optimal resistance surface to create a metric of the overlap of predicted use pathways. We visually compared this summed output of predicted pathways to actual pathways apparent on high resolution Quickbird satellite imagery.

The final optimal resistance surface was then used in combination with the detailed dry and wet season ranges depicted in surveys, to create a community-wide "heatmap" of predicted livestock pressure. To spatially represent the specific daily herding ranges that elders described using specified directions and distance, we created lines by using circular buffers with the distance corresponding to the distance of range indicated by elders, and truncating the line to the 45 degrees of cardinal or intercardinal direction that elders had indicated. These lines were then used to represent the end of a corridor of movement from the household origin.

Using the paired homestead origin and destination polygons, as well as the paired homestead origins and known used water points, the corridor tool in ArcMap was used to create use surfaces between each homestead and watering points/forage areas. These were created using both the “neutral” resistance surface and the final optimal dry season resistance surface. For each corridor raster, using map algebra in ArcPy, the minimum value was calculated, and this value was then subtracted from that raster to set the minimum value of all rasters to zero. The maximum value for all rasters was then determined, and each cell in all raster was subtracted from this value to create inverted raster corridors with equivalent maxima. A threshold value including those within the 100th quantile was then set to constrain corridors to simulate an orbit that realistically mimicked those commonly observed and indicated in interviews. A conditional statement was used to set values lowers than this threshold equal to zero (Sample Code in

Appendix M). Watering-point corridors were then combined into composite rasters corresponding to each homestead, where each corridor was weighted by frequency of use the animals were said to be taken to water using the weighted sum tool. Watering points and non-watering point range corridors were then combined using the weighted sum tool and the frequency of days taken to water vs. not taken to water. Finally, a composite raster was then created for the entirety of Koija for each livestock species, summing all composite corridors with an equal weight given to each *nkangitie*.

For comparison to piosphere models, two sets of additional grazing pressure maps were then created for each livestock species. To create these piospheres from all homestead and water-point locations, independently, the Euclidian distance function in ArcMap was used to create a distance raster from each homestead location. A raster depicting the inverse of the distance from homesteads and water-points was created. Water-point raster distances were then weighted by the number of *nkangitie* reporting use of that water-point. All resulting raster surfaces were then summed to estimate the use of these water points considering all *nkangitie*.

We then compared the ability of each of these estimates to predict relative livestock densities as approximated using dung counts. Sampled locations were located along a ~7km ridge that runs parallel to the Ewaso Ng'iro river, and that must be crossed to access water points at the river (Figure 4.5), offering an ideal setting to compare the relative densities of livestock observed versus the modeled estimates. Twenty-eight transects were established in 2013 by creating a grid with nodes spaced 600 meters apart running horizontally to the river, and 300 meters apart perpendicular to the river in ArcMap (Figure 4.5). We then sampled at the location closest to the node that had the highest elevation, did not have signs of a cattle enclosure being located there, had a slope of less than 3%, and was large enough to accommodate transects while satisfying all of these conditions. Dung counts of all mammals were completed by counting the scat of one individual within 1 meter to either side of the tape along three 50 meter transects arranged 20 meters apart in parallel and the center transect aligned with GPS coordinates corresponding to Landsat satellite pixel edges. We then extracted values from the herbivory pressure prediction heatmaps at these locations in ArcMap and calculated the correlation between the number of individuals of livestock present at the time of dung counts and the prediction surface using JMP (Version 12). We used goat dung counts to verify the estimates, as a large percentage of cattle and sheep were not present on Koija at this time.

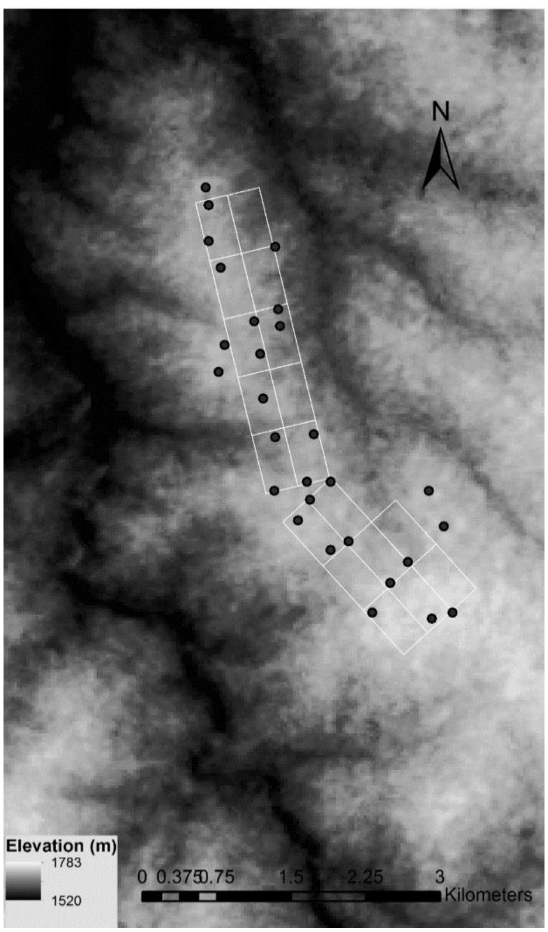

Figure 4.5 – Sampling locations

Results

Resistance Surface Optimization

To implement a least-cost approach to estimating herbivore pressure on the landscape, we created and optimized three themed resistance surfaces, representing constraints to livestock movement due to avoided areas, slope of terrain, and proximity to houses. The optimization was based on minimizing the RMSE of differences between the predicted and observed preferences for watering points used by each household. When compared to the "neutral" resistance model (where all points in the landscape had the same resistance), the independent optimization of each resistance theme (Figures 7-9), led to decreases in the RMSE value by 32.3% due to adjustments in the avoided areas raster, while adjusting the slope raster decreased RMSE by 25.3%, and adjusting the homestead buffer layer decreased RMSE by 1.4% (See Appendix N for the step-by-step results of each of these iterations). Using this baseline determination of individual raster values, the optimal initial weights for the composite raster were then determined (Figure 4.6). Following 3 nested iterations (See Appendix N for detailed results of step-by-step RMSE results within each iteration), the optimization procedure converged upon a single solution. The progression of weights in each step of the procedure for the individual and composite resistance surfaces is shown visually in Figure 4.6. The weights given to each subcomponent throughout this iterative process are shown in Figures 4.8-4.10 and the composite raster weights are summarized in Figure 4.7.

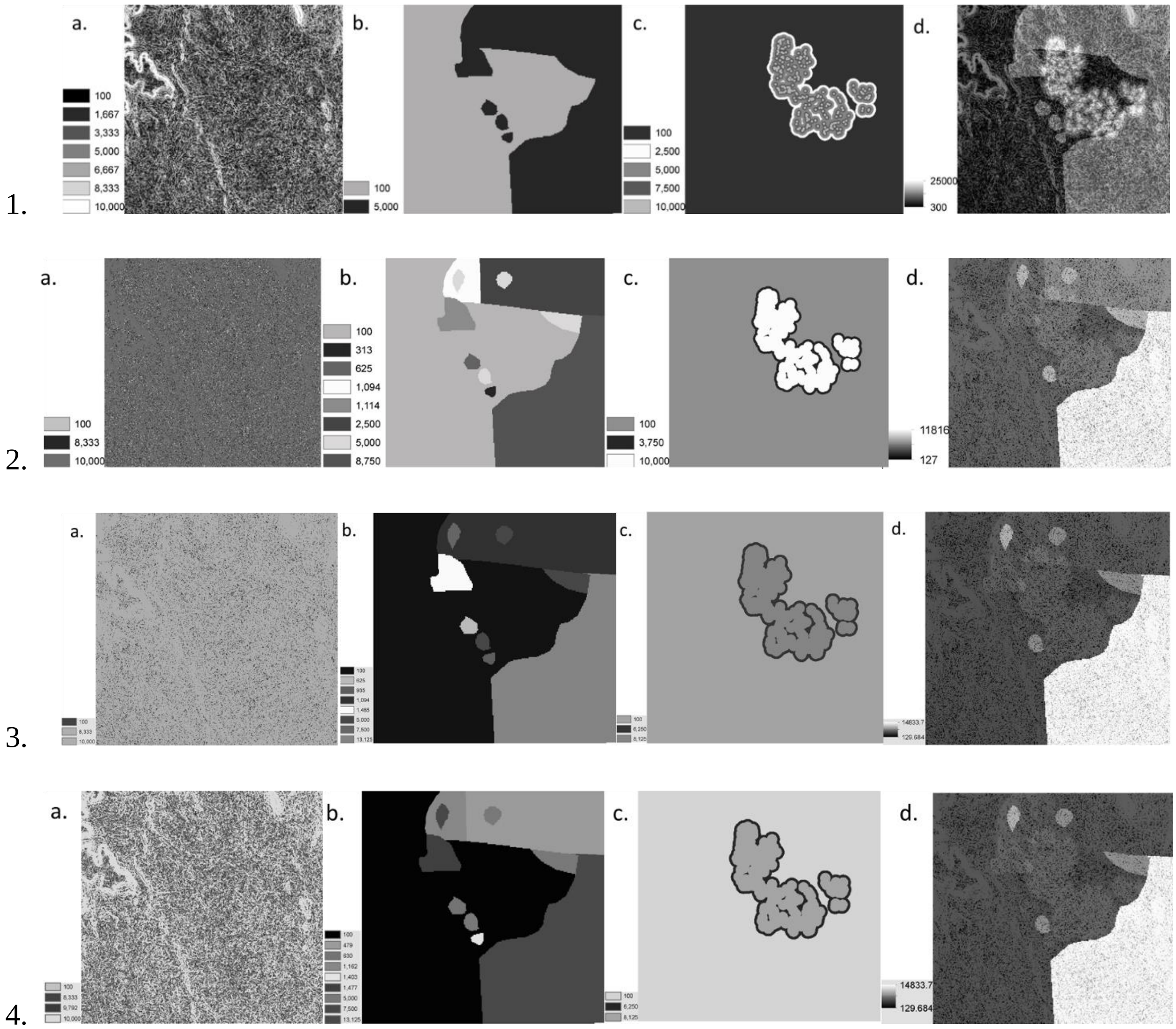

Figure 4.6 – Value settings for the multi-tiered optimization procedure, numbered from first (1) to final (4) iteration of the least cost path model (with resistance surfaces created from a. slope b. potentially avoided areas c. proximity to others' homesteads and d. the weighted composite raster consisting of a-c summed).

Homestead buffers had the smallest percentage of weight in the final resistance raster at just 18.31% (Figure 4.7. The final resistances were high in relative importance within the layer itself however, and for areas where household buffers were present there was a 63% weighting at its maximum range of 4-600 meters from homesteads, and higher equal weights (81% of the original maximum) across the three classes from 400m to 100m distance classes from homesteads (Figure 4.8).

While the overall final weight of the avoided areas raster in the composite raster was highest (Figure 4.6), optimization of weightings within the avoided area surface led to low resistance values for several classes of avoided areas, notably the wildlife conservation area, the neighboring Nolare conservancy located to the north, and one of the areas known for elephant dangers (Figure 4.8).

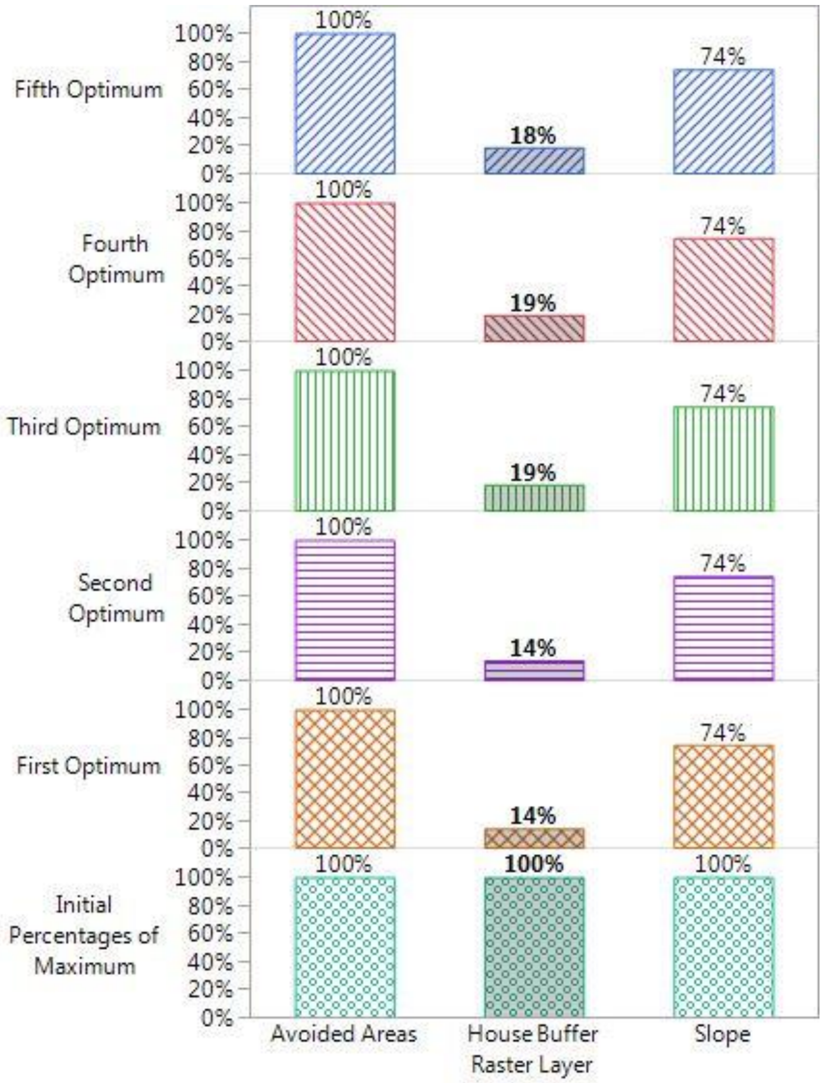

Figure 4.7. – Composite raster optimum relative weightings at the end of each nested iteration of the optimization procedure (re-optimized every time a theme's resistance values changed).

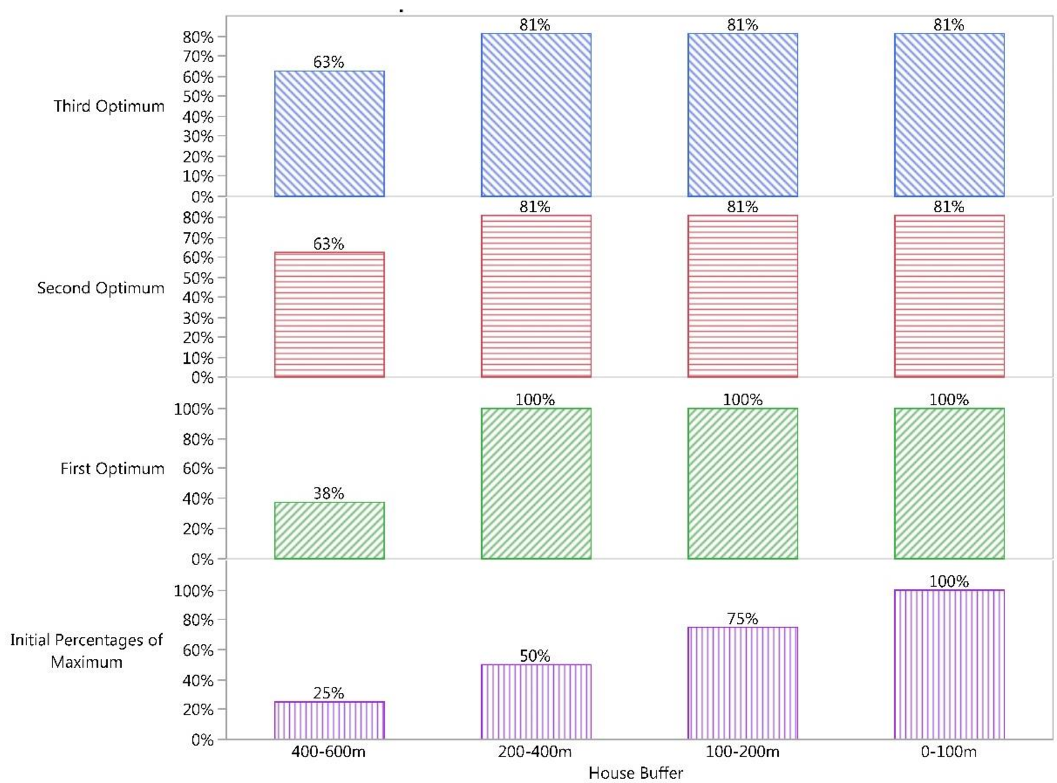

Figure 4.8. Homestead buffer optimum resistance weightings at the end of each of three iterations of the optimization procedure.

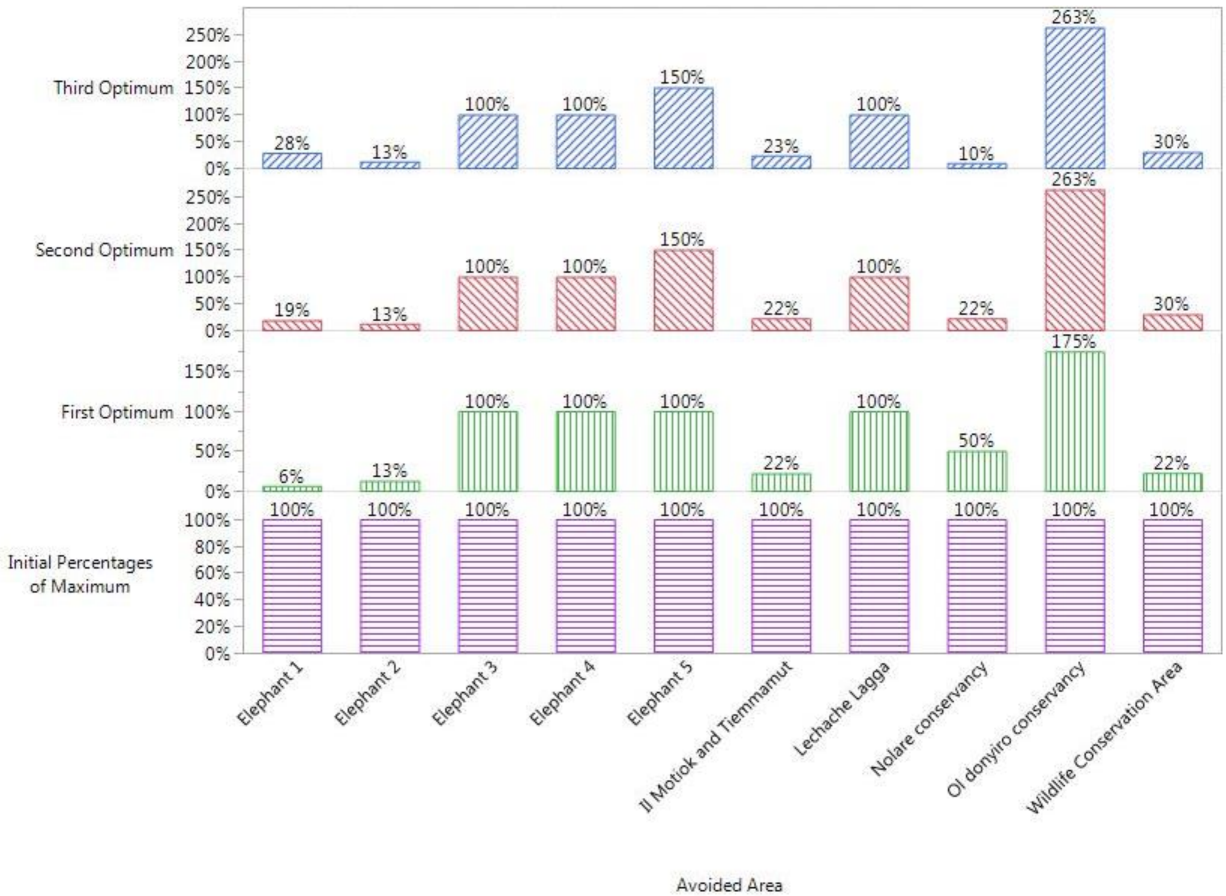

Figure 4.9. Avoided areas optimum resistance weightings at the end of three iterations of the optimization procedure.

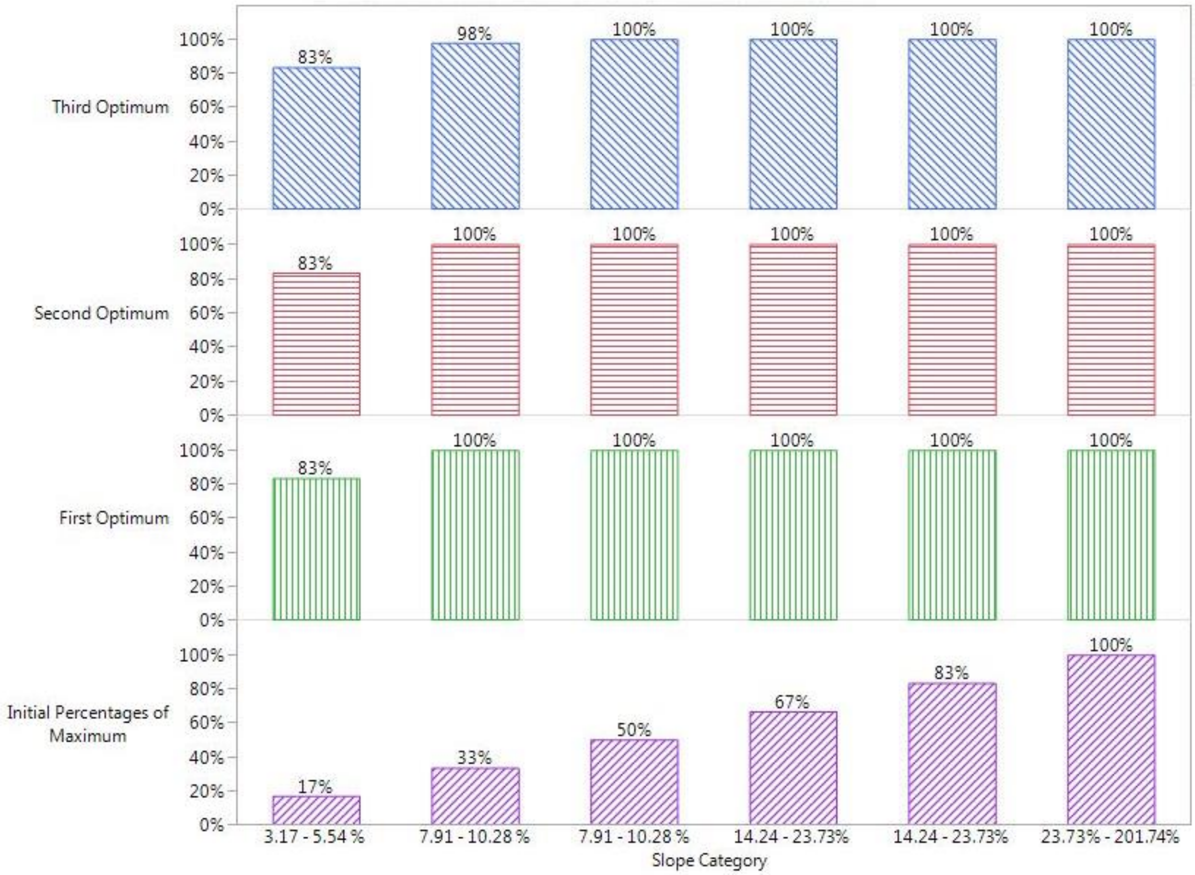

Figure 4.10. Slope optimum resistance weightings at the end of each iteration of its values in three iterations of the optimization procedure.

Verification

The final RMSE value for the optimal resistance surface with a subset of 100 *nkangitie* was 0.100092, compared to the RMSE value of 0.116163 for the “neutral” resistance surface (the proxy for Euclidian distance alone). Calculating the RMSE value for the optimal resistance surface, using only the other 105 *nkangitie* as a verification data set, yielded a value of 0.105283. While this verification value was a higher RMSE value compared to that calculated using the 100 *nkangitie* the resistance surface was optimized for, the value was still lower when compared to the RMSE value calculated for the “neutral” resistance surface (the proxy for Euclidian distance alone) for the 105 *nkangitie* in this verification data set, 0.127504. In the secondary stage of verification, the overall predicted frequency of use of water points with a resistance surface R-squared value of 0.51 (p=0.003) compared to the “neutral” resistance surface value of 0.50 (p=0.003, Figure 4.11), yielding negligible improvement in the ability of the optimal resistance surface to predict frequency of water point use. These two stages of verification indicated that when assessing the model considering water points alone, the procedure of minimizing RMSE led to an improved prediction of specific households’ watering points used, but when considering the overall frequency of community use of specific watering points, there was a negligible difference in predictive ability. Moving forward from this inconclusive validation of improved ability to predict water point use, we proceeded to the validation stage that tested the ability of the different corridors to predict finer-scale land use.

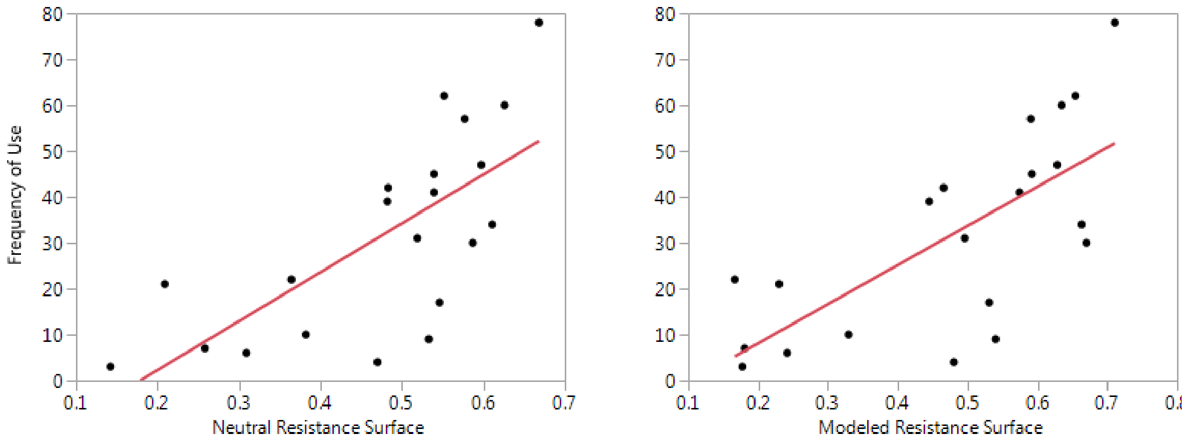

Figure 4.11. Linear fit of "neutral" (distance only) resistance surface and the optimized resistance surface as predictor of community-wide watering point frequency of use.

Validation

In the first stage of validation we used a simple visual comparison of the predicted least cost pathways calculated using both the optimized resistance layer and the distance-only resistance layer to those actually used, as identified in channelized pathways present on high-resolution Quickbird imagery, the least cost paths based upon the optimal resistance surface showed close correspondence to the visible pathways. This gave preliminary confirmation that despite marginal improvement in the community-wide verification stage of ability to predict water point selection, that the optimized resistances might lead to improved abilities to predicting localized land use *between* watering points and homesteads.

In the second stage of validation, we then compared the ability of the summed corridors that were calculated using the optimized resistance surface and those that only considered distance to predict dung counts as a proxy for actual land use. Regression of the small stock corridors calculated for the dry season ranges of goats using the optimized rasters and the field-based dung counts collected in 2014 had an r^2 value of 0.21 (p=0.013, Figure 4.12), while the corridors calculated using distance only yielded a non-significant value (r^2=0.05, NS).

In the third stage of validation we then compared the ability of the summed corridors calculated using the optimized resistance surface to one piosphere based upon watering points, and to another piosphere based upon homestead location. Though the corridors calculated using optimized rasters yielded a modest r^2 value in terms of overall explanation of variance, when compared to the weighted piosphere this indicated a large increase in the ability to predict dung counts (r^2=0.02, p=0.611, Figure 4.13). Further, the corridors based upon optimized resistance surfaces also led to improved prediction compared to the piosphere based upon distance from homesteads (r^2 = 0.11, p=0.087, Figure 4.14).

Summary

In summary, though the validation stage did not indicate whether or not the optimization actually had improved community-wide prediction of watering points compared to distance alone, in the validation stage we found that the community-wide, summed corridors based upon optimized resistance surfaces led to an improved ability to predict dung counts in comparison to all three alternative models: distance alone, a watering point piosphere, and a homestead piosphere. Finally, our ethnographic data revealed that social and ecological factors, such as herding labor, day-to-day decisions based upon variability in forage availability, and differences in landscape concentration and rate of movement due to herd composition also impact landscape use to a high degree, but these were not able to be included in this model as presently configured. Considering these factors and the overall complexity of day-to-day herding decisions and other potential factors affecting landscape use, however, we considered the least-cost corridor algorithm's explanation of 21% of the variance in dung densities to be an indicator of the potential utility for this approach to estimate livestock landscape use. The proportion of variation explained by this extremely simplified model including only slope, avoided areas, and homestead proximity lends a high degree of confidence in the potential of this approach to estimate gradients of livestock use to a higher degree across the landscape in future studies that can incorporate more of these factors.

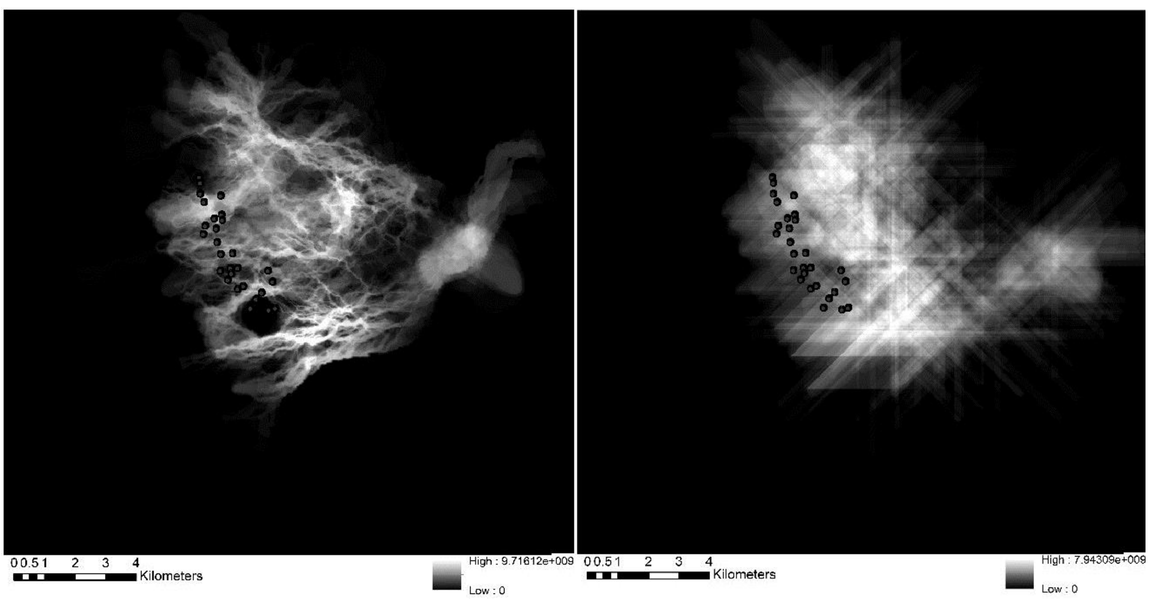

Figure 4.12. Two contrasting estimations of herbivore pressure: 1. Least Cost Corridors on the left 2. Weighted Piosphere on right. Both are unitless measures of predicted livestock use intensity. (Dung count sampling coordinate points from Figure 4.5 overlaid).

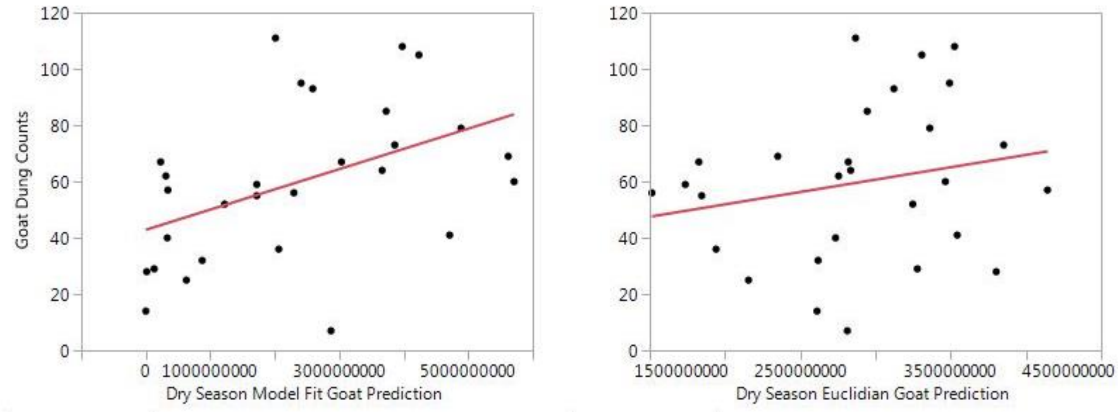

Figure 4.13. Comparison of ability of summed optimized least cost path corridors (left) and "neutral" corridors (based upon distance alone, right) to predict goat densities.

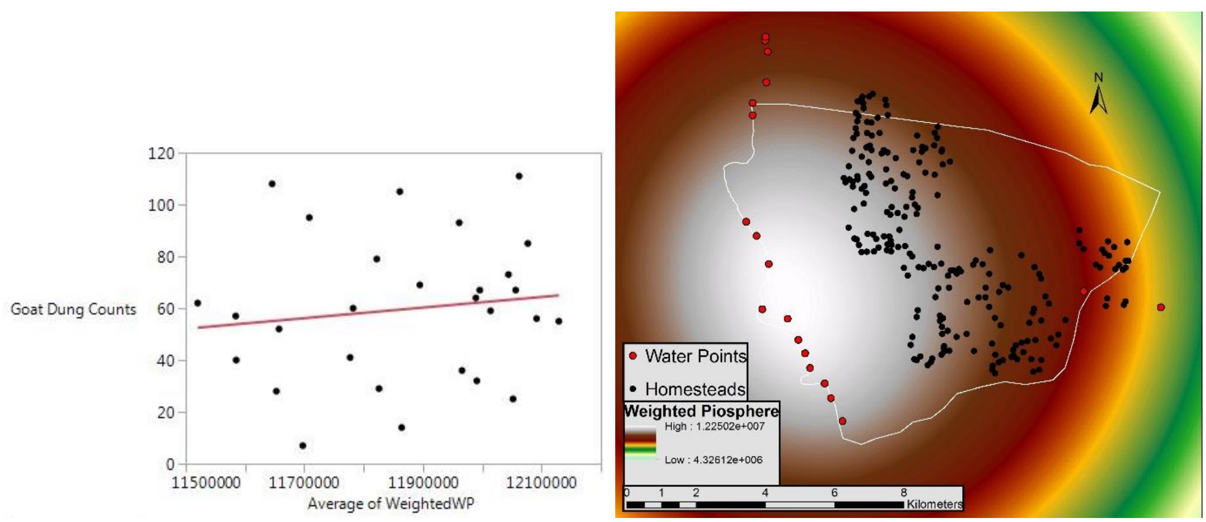

Figure 4.14. Regression of water-point-piosphere predictions of dung counts.

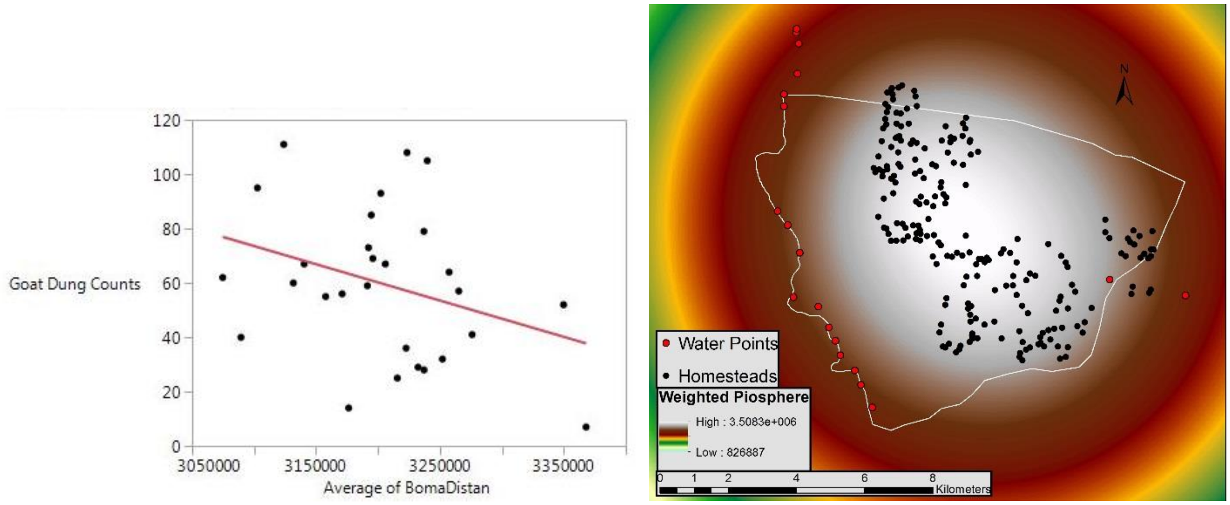

Figure 4.15. Regression of homestead-piosphere predictions of dung counts.

Discussion

Recent limitation of external access to forage resources has led to a greatly simplified system of herding and intensification of land use within Koija. The daily orbits of most herding ranges during the dry season are centered on watering points along the Ewaso Ng'iro river as the apogee of their orbit on a day-to-day basis, and that differential use of these watering points seemed to be related to heterogeneity of land use. Using three variables and data that detailed the preferential use of these watering points across Koija, we tested the ability to improve prediction of community-wide land use. Both the 100 homesteads used to optimize resistance surfaces to use in predictions, and the 105 homesteads used to verify the optimization indicated that these optimized surfaces led to lower RMSE than surfaces that considered distance alone. However, both "neutral" (i.e. calculated based upon distance alone) and optimized resistance surfaces performed similarly for predicting which watering points each *nkang* chose to utilize, explaining about 50% of actual frequencies of *nkangitie* use of different watering points. This indicates, unsurprisingly, that watering point choice is based upon proximity of the watering point to a given homestead, to a high degree. In interviews, water point preferences were frequently explained based on characteristics such as being open-canopied, easy to access (without a step descent or soils that animals may become stuck in), and free of predators such as leopards and jackals. These factors, as well as other unknown qualities of the watering point, or numerous potential unstudied social factors, likely contribute to the unexplained portion of water point selection variance and were not included in our analysis.

While the optimized resistance surfaces produced only a negligible difference in predicting watering point use, the optimized surfaces explained a significant proportion of the variation in land use pressure across the landscape *between* homesteads and watering points – as indicated by empirical dung counts -- while both the "neutral" corridor and the two piosphere models did not. Especially given that this model in no way accounted for disproportionate frequency of use *within* corridors, that no doubt occurs due to a variety of reasons that we are aware of but were unable to include in the model, this result indicates a high amount of potential for development of more sophisticated models using a similar approach. This could potentially lead to development of more detailed estimates of livestock pressure across landscapes, with high potential relevance for use in studies of land cover change, land-use history, and herding institutions. Using relatively simple data elicited about day-to-day herding orbits across a community enabled us to account for a large number of different households with specific land use preferences and variable degrees of mobility and to generalize about livestock pressure across a community. This approach has potential for application in a wide range of scenarios where households have diverse land uses that are not represented in simply understanding the mobility of a few herding families (Turner and Hiernaux 2002, Wario et al. 2016). Our results show one methodology that could be used to feasibly represent this diversity of land uses across communities to gain more generalize understandings of phenomena such as herbivory pressure.

This approach also provided an easily replicable technique for optimizing resistance surfaces used in least cost path, corridor, and circuit analyses in a variety of other contexts, including more traditional use of corridor analysis for wildlife studies. Additionally, applying this type of analysis using an alternate software and methodology (e.g. numerous possibilities in R, Matlab, or Circuitscape), might greatly improve processing speed, allowing for expanded analysis of larger numbers of resistance variables, consideration of larger geographic areas, or more complex configurations that more realistically represent the factors underlying land use. Additionally, there are numerous ways that future studies could explore potential ways to enhance factors we *did* include in our approach, including, for example finer-scale elevation and slope data, and alternative metrics for partitioning slope into classes.

A suite of recent modeling advances have greatly expanded the possibilities beyond the simple least cost path approach we utilized, and could also potentially be incorporated into this analysis, for example, agent-based modeling of choices of watering points, or use of circuit theory as a complementary approach to the least cost path approach (Spear et al. 2010). Software methods such as Circuitscape (Shah and McRae 2008) could be used to facilitate more detailed inverse-modeling of heterogeneity in factors such as differences in the relative use of certain endpoints (e.g. waterpoints in our study), or to model qualities of individual starting points (e.g. livestock composition at homesteads in our study) in circuits. Further, the resistance surface optimization approach that we applied could also be used in conjunction with habitat suitability methods, and finer resolution, GPS-based understandings of movements (e.g. Butt et al. 2009, Butt 2010, Moritz 2010) to resolve finer-scaled utilization of areas, in tandem with an approach like ours to account for wider, generalized patterns across communities. This could add understandings of the fine scale preference and frequency of use within corridors, something that is absent from our corridor-based estimates of use, but known to be important in pastoralist land use (Ellis and Swift 1988).

Though inappropriate for our intended use of the model output in a study of vegetation changes, for other purposes vegetation attributes could also be used as a theme to be considered in creating optimized resistance layers. In our case study vegetation is one factor that herders frequently reported to use in making decisions about land use. While a shortcoming of our study is that the frequency of time spent foraging in areas within corridor boundaries is not accounted for, the approach could be enhanced by adding a vegetation surface and more detailed data on patch utilization, and finer-scale tests of predictions of preferential use of landscapes by herders and their livestock, as well as for wildlife usage alike. This could also potentially allow for additional modeling of fluctuation in time spent in specific areas (e.g. Copollillo 2001). However, the intended use of our herbivory pressure map is to examine correlations with shifts in vegetation composition over time, thus, vegetation attributes would have been multicollinear with the dependent variables in subsequent vegetation analyses (Chapter 5).

Our approach of focusing on day-to-day watering routes was well-suited for the context of highly restricted herding ranges and movement that occur in our study system, however this may not be appropriate in many pastoralist systems. External sociopolitical constraints have progressively eroded pastoralists' access to historically utilized grazing areas outside the group ranch boundaries, resulting in a low degree of flexibility within this herding system and a major shift toward large amounts of small stock that remain on the group ranch year-round, rather than seasonally leaving Koija (Chapter 2 for a detailed examination of these trends). In systems where pastoralists have maintained greater land access to maintain more flexible, complex systems of land use, our approach may prove much more complicated or even inappropriate for larger scales where very frequent changes in herding ranges occur (e.g. see Butt 2009, Liao et al. 2017, Moritz 2010).

While the elements of slope, avoided areas, and homestead proximity did explain a significant portion of the variation in land use, our ethnographic studies revealed numerous other factors that contribute to heterogeneity in herbivore pressure on the landscape, which our approach did not account for. As mentioned above, vegetation characteristics are important in this regard. Herders commonly reported allocating time spent in different areas of the group ranch based upon availability of forage. This is especially true as well in other settings considering cattle, who are typically herded to areas with dense patches of grass and can spend long periods of time in one location (Moritz 2010, Butt 2009). Cattle, when herded by *ilmurran (*unmarried males highly trained in cattle herding), also much more frequently leave the group ranch altogether compared to goats and sheep. Additionally, there are social and ecological factors that likely have large influences on land-use variation such day-to-day decisions and differences in typical ranges, or the social nature of herding, where influence between households over one another's ranges can occur (Turner and Hiernaux 2002), or where youth and adults alike frequently stop to graze their animals in the same places and converse. Incorporating social influences on herding strategies, where strategies are contingent on the activities of other herders, could most appropriately be addressed through agent-based modeling.

One caveat with GIS-based approaches is the abstraction inherent in GIS analysis, which has the potential to mask political factors (Robbins 2003), and has frequently been used to support marginalizing narratives about land use (Robbins 2003, Turner 2003). However, using spatial tools coupled with herder accounts of their daily ranges, can integrate GIS-based analysis while maintaining a degree of relevance to these concerns, while simultaneously acknowledging the partiality of different perspectives and the potential misuse of such models. A challenge in using GIS methods in such contexts, as noted by Robbins (2003) and Turner (2003), is to maintain a nuanced understanding of social complexity, and avoiding that complexity becoming distorted or lost in the GIS matrix. Therefore, building from ethnographic understandings of herding practices (Chapter 2 and 3) and familiarity from long-term fieldwork are helpful in reflexively informing GIS-based studies. Finally, this approach could be expanded, depending upon the intended purpose, to more robustly consider the concerns and lived experience of rural land users through utilization of participatory mapping methods.

Conclusion

This resistance surface optimization exercise showed that adjusting the relative weights of three landscape factors – slope, avoided areas, and proximity to other homes -- led to a more nuanced ability to predict fine-scale animal movements across a herding community, when compared to prediction based upon distance alone. Rather than provide an exact understanding of herding practices, our approach focused on trying to achieve the best possible estimate of livestock pressure at the landscape scale without using methods that are much more difficult to implement, resource intensive, or potentially invasive to research participants (e.g. GPS collars on numerous households' herds). We propose this as a relatively simple methodology drawing from social surveys and widely available GIS tools that can be used to improve understandings of the spatial heterogeneity of livestock pressure in different settings. Incorporating elements from surveys that were commonly stated to influence herding decisions, and that could easily be simulated in a GIS environment, we developed a predictor of livestock use that provides an alternative to methods currently used in rangeland ecology. This exercise represents a potential improvement in understanding landscape pressure over piosphere models while maintaining the generality that is often lacking in GPS-aided measurement of landscape use (Turner and Hiernaux 2002). Future development of similar methods has potential to improve the way that rangeland ecologists understand livestock pressure, enabling more nuanced analyses of vegetation change. By adding several variables that represented some constraints on herder choice of landscape uses, and the least cost path/corridor framework for understanding movements, we were able to greatly improve our understanding of generalized movements across the landscape.

Further, similar resistance surface optimization methods have potential to gain more nuanced and contextual understandings of landscape use for applications in a wide range of settings, from the way that human/livestock assemblages utilize landscapes, to wildlife studies and corridor design. Though a potentially computationally intensive methodology, alternative configurations of variables, or use of open-source software packages such as g-distance for R (van Etten 2017) might provide more efficient means to make such methodologies more widely accessible.

In closing, this case study provides a way of using integrative study to understand changes in landscape ecological process in a setting where livelihoods and vegetation are tightly coupled. Such understanding enables a linkage between understandings of ecological change with our previous analyses of how access and livestock husbandry have changed in recent history (Chapter 2 and 3). Our case study perhaps provides an ideal example of estimating livestock pressure, likely enabled by the extremely simplified land use compared to historical herding extents (see Butt et al. 2009, Moritz 2010 for detailed, GPS understanding of the complexity of these ranges), where decreased porosity and loss of access has concentrated a great deal of land use within a small area. In many other settings, the complexity of nested variables in such a model might need to be expanded greatly to attain generalized pressure maps with a similar degree of predictive power. On this note as well, it should be considered that there were many factors that were knowingly omitted from our model, including a suite of social factors that affect how decisions are made on a daily basis, and the social basis of daily herding activities.

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CHAPTER 5

LANDSCAPE VEGETATION CHANGE IN A MOBILITY-CONSTRAINED SEMI-ARID PASTORALIST COMMONS[4]

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Abstract

In an age of dramatic changes in human-environment relations it is often difficult to distinguish how specific factors such as changes in livelihoods have impacted landscape processes in contexts of complex land-use history. Understanding these changes as embedded in complex, dynamic, path-dependent assemblages requires mixed methodologies and pluralistic understandings of landscapes. In a case-study from a pastoralist group ranch in Laikipia, Kenya, we used an interdisciplinary approach that drew from ethnographic methods to inform analysis of the relationship between landscape process and landscape structure across a pastoralist group ranch. Controlling for historical contingency of vegetation state and using proxies for abiotic context, we evaluated the correlation between recent patterns of the seasonal, species-specific livestock pressure, and changes in vegetation communities. Vegetation methods included remotely-sensed historical vegetation analysis and detailed plot-based analyses of current vegetation. Changes in productivity and species composition in 12 out of 37 community types were shown to be correlated with estimated livestock pressure changes. Yet changes in the other communities showed little to no correlation, or were more correlated with proxies for abiotic landscape factors than livestock pressure estimates. Dry season ranges of cattle were correlated with changes in grasses within some vegetation types, while sheep and goat pressure estimates were correlated with changes in two types of former shrub and, likely, vine vegetation. Canopy losses occurred in 18% of the area considered, while 37% of the area experienced shrub encroachment. We found no correlation between shrub encroachment and historical livestock pressure, but two increasing understory species were correlated with small stock pressure estimates in plant community plots. There were correlations between livestock and changes in areas with former canopy species, but as these canopy losses occurred uniformly across the

landscape, these correlations rather may indicate impacts on exposed understory species. Additional factors such as changes in plant moisture availability, impacts of wild herbivores, and fire suppression should be considered in future analyses of shrub encroachment and canopy loss.

Introduction

A growing body of interdisciplinary work seeks robust understandings of how livelihoods and landscape ecological outcomes are intertwined and embedded in wider land use changes and social factors (Liu et al. 2007). In this study, we explicitly consider whether large-scale changes in landscape processes of pastoralist herding are correlated with vegetation changes, while controlling for several abiotic variables that are known to influence vegetation structure. Pastoralism, or reliance on domesticated livestock for the majority of household income, is the current primary livelihood of four million people in Kenya (Kirkbride and Grahn 2008). As a livelihood, it is thought to be the most well-suited subsistence activity for semi-arid lands that do not support farming (Homewood et al. 2008, Behnke, Scoones, and Kerven 1993). Mobility is essential to gain access to key resources that are highly spatially and temporally variable across the landscape (Hobbs et al. 2008). Therefore, pastoralism is less well-suited as a livelihood when seasonal grazing movements cannot occur (Fratkin 2001). While being tightly coupled to forage availability at multiple spatial scales (Niamir-Fuller 1999, Mwangi and Ostrom 2009), changes in herding processes can have localized impacts on vegetation, leading to cascading changes in ecological function (Ludwig et al. 2005) that can in turn impact herding livelihoods (McPeak 2003). In the case study that follows, we analyzed vegetation change across a community in central Kenya, and considered livestock herding as a historically dominant landscape process interacting with the pattern of plant species composition at the landscape scale (Levin 1992). This work has important implications considering increasing rainfall variability currently being experienced in drylands (Ericksen et al. 2013) at the same time as pastoralist livelihoods are increasingly experiencing livelihood stressors due to rangeland fragmentation and associated factors (see Chapters 1 and 2). While the impacts of fragmentation of herding ranges has been studied extensively from a social perspective, and landscapes in arid and semi-arid lands worldwide are thought to be experiencing negative impacts on vegetation due to sedentarization (Weber and Horst 2011, Groom and Western 2013), the specific impacts of fragmentation and sedentarization on vegetation from an ecological perspective have received less attention.

In what follows, we test whether changes that have occurred in the landscape process of livestock herding in one pastoralist community in central Kenya that have occurred over the past 30 years are correlated to changes in vegetation over the same time period. We built upon an estimation of seasonal livestock pressure to test whether this estimate of recent livestock pressure is related to remotely-sensed changes in vegetation and current patterns of plant composition.

Background

Drawing from systems ecology (Holling 1973, Noy-Meir 1975, May 1977), understandings of ecosystem dynamics in rangelands have shifted dramatically since the 1980s away from concepts of climax and succession toward ones of considering multiple possible stable states, non-equilibrium dynamics, and non-linear transitions (Ellis and Swift 1988, Westoby et al. 1989, Anderies 2002, Van Langevelde et al. 2003). Semi-arid rangelands can express traits along a continuum from equilibrium to non-equilibrium (Vetter 2005, Boone et al. 2011) and consequently the pressure, timing, and duration of livestock herbivory, as well as the coefficient of variation of rainfall, can be of greater relevance than average stocking rates in determining vegetation conditions (Ellis and Swift 1988, Vetter 2005, Boone et al. 2011). Drylands are marked by infrequent precipitation events with low predictability, and lower precipitation than potential evapotranspiration (D'odorico and Porporato 2006), with semi-arid zones typically occurring in areas of between ~250−600mm of rainfall. Factors such as nutrients, water, herbivory, soils, and fire are known to play strong roles in structuring vegetation in semi-arid lands (Sankaran et al. 2005, Sankaran et al. 2008, D'odorico and Porporato 2006). Above a threshold of 650 mm of mean annual precipitation, savanna systems are thought to be unstable and shift toward being dominated by woody biomass while factors such as herbivory, fire, and soil characteristics greatly impact woody biomass below this upper bound (Sankaran et al. 2005). When between this threshold and the lower bounds of 350 mm (Sankaran et al. 2005), patches of woody canopy with greater soil moisture are expected to be interspersed with areas of drier soil moisture that do not support woody vegetation, while numerous factors are thought to influence the finer structure of this savanna vegetation.

In more recent understandings of shifts in vegetation states in semi-arid rangelands, rather than focusing on stocking rates alone, additional factors are frequently considered such as change in disturbance regimes, herbivore pressure, or how the variability of rainfall can alter feedbacks that are responsible for maintaining vegetation in their current state (Ludwig et al. 2005). Feedbacks can also occur through factors such as decreased soil moisture or decreased organic matter that can also lead to a lack of plant growth, which can in turn lead to cascading impacts on the soils and reduced fertility, reduced biotic activity, erosion, and loss of water infiltration (Ludwig et al. 2005). Shrub encroachment is an example of a phenomenon studied from this perspective of vegetation structure transition due to alteration of feedbacks. While past studies often concluded that herbivory was responsible for shrub encroachment, building upon recent theory, D'odorico et al. (2012) offers a framework for understanding shifting balances between grasses and woody species in rangelands. In their framework, changes such as those in grazing pressure, temperature, rainfall, atmospheric CO_2, nitrogen deposition (Archer 2010, Archer 2017), and the fire regime can alter feedbacks that previously regulated shrub establishment (Stevens et al. 2017). These understandings of vegetation transitions, and the influence of feedbacks at different scales, can lead to insights for interventions that may mediate feedbacks, thereby preventing losses of herbaceous species and increases in bare ground (Tongway and Ludwig 1996). By considering more pluralistic metrics than stocking rates, and including livelihood pressures and recent changes in specific contexts, it is possible to allow for a more nuanced understanding that includes the abiotic context as well historical contingency of landscape transitions (Bestelmeyer et al. 2011).

In East Africa, grazing herbivores and grasses are thought to have been tightly coupled over long periods of time, leading to co-evolution and greater tolerance of grasses to grazing compared to other regions of the world (Milchunas and Lauenroth 1993). However, shifts in vegetation can occur due to changes in landscape processes, such as lack of movement to reserve grazing sites during long dry seasons (Hobbs et al. 2008), where in the past seasonal movements are thought to have prevented degradation (Ellis et al. 2001, Reid et al. 2004, Boone and Hobbs 2004). Further, changes in the timing of forage use, such as the dry-season sensitivity of perennial grasses to grazing damage, are thought to be of high concern when considering seasonal changes in mobility (Ash et al. 2011, Fynn and O'Connor 2000, Hodgkinson 1995).

A number of studies of rangelands have focused on increases in unpalatable species (Richardson et al. 2005) and shrubs (D'odorico et al. 2012) as they relate to the grazing intensity of cattle. On the other hand, cattle are also thought to have beneficial impacts on grass productivity in some cases, where nutrient concentrations have been shown to increase in heavily grazed plots (McNaughton 1997), due to increased uptake of nitrogen and sodium deposited at the surface through decomposition, as well as through urine deposition by grazers that increases soil nitrogen content (Augustine et al. 2003). Such "grazing lawns" occur in areas where livestock pens were located in central Kenya (Young et al. 1995).

Obtaining a nuanced understanding of the potential of livestock to create shifts in vegetation is further complicated when considering small stock, which have increased in many pastoralist communities undergoing sedentarization or increased episodes of drought (Opiyo 2015, Österle 2008). Different livestock species can have distinct impacts on grazed plant communities, and the diets of different domesticated ruminants vary greatly (Harris et al. 2016, Rook et al. 2004). Goats have been shown in some cases to influence plant species diversity and structural heterogeneity through altering the spatial structure of woody plants, which can then lead to cascading impacts on abiotic factors that impact herbaceous vegetation (Gabay et al. 2011). However, the role of changes in herd composition on vegetation changes, and how this potentially interacts with multiple other changing factors in landscape process is difficult to study. The impact of increased numbers of small stock in semi-arid lands is not well studied, but large increases of these species have occurred in herds throughout Kenya (Ogutu 2016).

On the Laikipia Plateau of central Kenya, as in many other East African semi-arid lands, decreasing porosity of boundaries has impeded pastoralists' customary means of securing seasonal grazing access (Herren 1991). This decreasing mobility is due to complex historical and social factors coupled with recent loss of land due to privatization (Herren 1991, Lengoiboni et al. 2010, Letai and Lind 2013), as well as conflict and the recent establishment of wildlife conservancies (Chapter 2). Combined with the influence of livestock markets, need for access to cash, and drought, these changes have led to a dramatic restructuring of pastoralist herding practices (Herren 1991, Chapter 2). Pastoralists and their cattle have been present in Laikipia thousands of years, the above constraints on herding cattle have led to fragmentation of herding ranges and a seasonal concentration of cattle within increasingly smaller ranges, leading to accompanying shifts toward small stock (Herren 1991, Chapter 2). These two factors likely amount to a drastically altered landscape process; that of concentrated pressure due to seasonal changes in cattle mobility, as well as the concentrated, differential impacts of the foraging habits of sheep and goats.

While not specifically shown to be caused by livestock, a, a number of broad changes in vegetation composition and density are thought to have occurred in Laikipia at the same time these herding changes have occurred (Herren 1991, NAREDA 2004, Njenga 2001). A novel landscape dominated by unpalatable plant species has been reported by community members in Laikipia to have arisen relatively recently, with adverse effects on the productivity of livestock (NAREDA 2004). In Laikipia, NGO grey literature frequently calls for alternate rangeland management plans within pastoralist areas frequently cites “overgrazing”, breakdown of common property regimes, or human overpopulation as the justification of a need for novel land use management plans (e.g. see Alexovitch et al. 2012, Fennessy 2009, Lent et al. 2002, NAREDA 2004, Sumba et al. 2007).

We asked how patterns of decreased mobility and a restructuring of herds, which have caused changes in how herbivore pressure is concentrated across the landscape, are related to heterogeneous shifts in vegetation composition in the last three decades. However, large changes have recently occurred in the concentration of cattle, as well as qualitative differences in the impacts of sheep and goats. It is also uncertain to what extent changes in vegetation could be related to the increasing coefficient of variation of rainfall in the area, well documented by Franz et al. (2010). Additional other changes have potentially occurred due to increasing elephant density (Litoroh et al. 2010), loss of browsing wildlife that could regulate shrub cover (Sankaran et al. 2013) and fire suppression (Augustine 2003). There is thus a strong practical need to clarify how restructuring of herding practices has potentially impacted semi-arid vegetation.

Using preliminary findings from ethnographic research on ecological transitions, in which elder herders in focus group discussions and interviews indicated changes in vegetation they had experienced, we were able to gain understanding of some of the salient vegetation and land use changes from life-long Koija residents. This was used in part to explore whether herders had different views of changes from those commonly voiced by conservation actors. Further, linking these approaches to livelihoods, as has been documented elsewhere (Ch. 2), necessitates a qualitative understanding of the livelihood impacts of different vegetation changes, beyond measures of productivity, to make conclusions about the social-ecological impacts of landscape transitions.

A large body of ecological research in Laikipia has been completed on the interactions between wild herbivores and vegetation (e.g. Augustine and McNaughton 2004, Sankaran et al. 2013, Young et al. 1997, Young and Augustine 2007), as well as cattle and vegetation (Riginos and Young 2007, Veblen and Young 2010, Young et al. 2005, Odadi et al. 2007, Mizutani 1999), on privately held conservation ranches in Laikipia. Our research adds an analysis of the specific changes in vegetation that have occurred in relation to our understanding of species-specific livestock herbivore pressure, building upon our analysis of the history of changes in livelihoods and herding ranges in Chapters 2 and 3.

Methods

In the previous chapter (Chapter 4) we developed a methodology to estimate livestock pressure during different seasons using survey data. Here we utilized those gradients of species-specific domestic herbivore pressure to test whether changes in specific types of vegetation over the last 30 years correspond to these gradients. Using mixed methods, in which we combined satellite image classification of vegetation types present between 1987 and 2013 with analysis of both continuous vegetation change metrics and vegetation plots, we tested the varied impacts of herbivory on different types of vegetation. We considered these interactions as context-dependent, contingent upon historical vegetation state, and dependent upon other factors such as soils, slope, and precipitation. We did so to assess whether livestock pressure, just one of the possible drivers, was correlated with changes in plant composition and productivity. Testing hypotheses about these relationships in a highly heterogeneous landscape that has undergone rapid change poses several methodological difficulties. Understanding the underlying drivers of these changes is crucial to inform discussions about the overlapping concerns of wildlife conservation and pastoralist livelihoods. This analysis, while it does not consider all of the potential drivers of vegetation changes and does not test for causation of these changes, contributes to an improved understanding of the specific types of localized impacts that can occur to semi-arid vegetation as a result of livestock pressure from fragmentation of grazing ranges.

State and Transition Framing

Using a state and transition approach (Bestelmeyer et al. 2003, Bestelmeyer et al. 2009, Briske et al. 2003, Briske et al. 2005, Stringham et al. 2003), we outlined a set of possible transitions for the Laikipia context. This was done to attempt to develop a preliminary understanding of recent changes in plant communities in a highly heterogeneous landscape, and to test hypotheses about whether changes in livestock pressure are correlated to landscape vegetation change, to inform discussions about the overlapping concerns of wildlife conservation and pastoralist livelihoods. A state and transition approach enables a systematic framing to bring multiple understandings of how ecosystems respond to drivers of change together to inform management (Bestelmeyer et al. 2017). Using this approach, we structured a methodology for testing hypotheses about broad-scale vegetation transitions based upon our understandings of recent dynamics of individual species. This was informed by numerous sources including: oral histories of vegetation change (Appendix H), one co-author's long-term history of plant ecological work in the local context of Laikipia (King), preliminary examination of current vegetation distributions, and examination of historical remotely sensed imagery.

Study Site

Koija group ranch is located within Mukogodo Division in Laikipia County, Kenya. The majority of the people who reside at Koija group ranch trace their lineage to the LeUaso hunter-gatherer group, with some stating historical ties to Maasai, Samburu, and Laikipiak Maasai groups. References in casual conversation are often made to recent ancestors who primarily hunted, gathered, and kept bees for a living. Today, while being primarily pastoralists, many people continue to keep bees. The study area is located at an elevation of 1700 meters, with a mean annual precipitation of approximately 450mm per year. It is characterized as semi-arid, lands that are typically characterized by long, unpredictable periods where rainfall is very low (Bailey 1979) and have highly heterogeneous in vegetation structure across the landscape due to variability of rainfall. The coefficient of variation of rainfall is close to 40%, and so is substantially higher in variability in this area (as well as lower in overall moisture) compared to the rest of Laikipia county (Franz et al. 2010). This coefficient of variation is near the value where herbivore / vegetation interactions in tropical drylands are considered to be at non-equilibrium, and there is evidence of recent increases in annual variability of rainfall (Huho et al. 2009, Franz et al. 2010). Rainfall is bimodal, with the highest amounts of rainfall typically occurring in April-May and November, with a long dry season occurring in Jan-March. The soils at Koija are a mosaic of luvisols and vertisols. Impala, gazelles, dik-dik, hares, and elephants are common wild herbivores within Koija. Fire is thought to have been largely absent for decades, but to have been an important part of the ecosystem in the past (Augustine 2003). Termites and aardvarks redistribute nutrients and to influence the heterogeneity of the landscape (Pringle et al. 2010, Tarnita et al. 2017). There have been increases in elephant populations through the study period (Litoroh et al. 2010), which are known for their impacts on the densities of trees and vegetation composition (Kimuyu et al. 2014, Western and Maitumo 2004).

Required Methodological Developments

While fine spatial resolution satellite data gives detailed information on the structure of vegetation patches, availability of these data is a limitation when analyzing longer trends in landscape ecological change, and so medium or low spatial resolution must be relied upon. In trying to understand historical changes in vegetation in rangelands, medium resolution Landsat satellite archives provide a powerful resource to understand historical changes in rangeland vegetation due to the archival record and the high compatibility of images from different sensors. However, there are numerous methodological concerns in using medium resolution remote sensing in an area with bimodal rainfall distribution and high spatial and temporal variation of rainfall and vegetation. Here we discuss two of these difficulties encountered in our study, and the approaches we developed to try to overcome these difficulties. The first difficulty relates to classifying complex vegetation types, and the second difficulty relates to understanding change over time.

Firstly, as we lacked sufficient training and ground-truthing data to classify a historical image using supervised classification, we had to develop a method for determining the most appropriate unsupervised classification to use for historical data. There is typically a list of trade-offs that must be made when determining how to produce the most appropriate categorical classification of vegetation and selecting a classification technique (Xie et al. 2008). Supervised classifications, where the analyst first identifies the spectral signatures of known vegetation, can be highly accurate, even largely independent of the classification algorithm used, given precise training data (Li et al. 2014). However, supervised techniques are highly subjective and difficult to replicate in independent studies. There are also rarely sufficient historical data to replicate a fine-scale supervised classification at two time periods within the same study.

Alternately, an unsupervised classification can be used in a situation where training data is limited, or where high replicability of a classification technique is necessary. The k-means classifier is an example of an unsupervised classification algorithm where the spectral signatures are clustered independent of analyst choice of training sites, and is widely used due to its ability to process large datasets. However, despite the many utilities of k-means classifiers, the final classifications produced are known to be highly sensitive to the initial selection of cluster centers (Peña 1999). Determining the optimal unsupervised classification of a highly heterogeneous landscape, where specific types of vegetation need to be distinguished, can be extremely time consuming because numerous classifications must be examined to determine which one represents the landscape. Also, due to the sensitivity of the final classification to centroid selection, and the inability of image analysts to specify centroids in many k-means classifiers, the method can lead to classifications that do not represent landscapes accurately or as intended. These classifications can, for example, downplay important landscape features that might be merged based upon similarity of cluster centroid. Also, spectrally similar classes, which may represent significant differences on the ground, often cannot be reliability differentiated, but may be of specific interest to a researcher. In our preliminary work, we encountered high levels of inaccuracy in clusters generated by the k-means classifier in areas with high amounts of bare ground but very different types of vegetation, so we sought alternatives to k-means classifiers.

To attempt to balance these methodological concerns with the two types of classifiers we developed an approach to reliably distinguish current vegetation classes in this area, and accurately determine past vegetation states for use as a baseline for the analysis. The approach systematically varied numerous parameters in unsupervised classification algorithms, and then compared the numerous resulting classifications to a known reliable supervised classification of recent imagery. Once the best fitting algorithm was selected we then applied this algorithm to past imagery. This led to overall increased ability to determine the most appropriate classifier for this landscape and to maintain replicability between classifications of the scene at different times.

Our second concern, based on known methodological limitations, was that analysis of transitions between categorical classes over time would inadequately capture all of the vegetation dynamics we expected to observe. Categorical post-classification approaches, where classification is performed at two dates and transitions between classes are analyzed, are subject to multiplication of errors that can falsely indicate change (Singh 1989). These post-classification approaches also constrain analysts within a framework of categorical transitions between vegetation states. In such instances, pre-classification techniques such as Principle Components Analysis of images are highly attractive because they allow the user to define the transitions of interest and to determine thresholds of change. Alternatively, continuous approaches to change detection, which often rely on multi-band image differencing or time-series analysis of indices such as NDVI or tasseled cap, can be used to gain understandings of trends in productivity. However, these methods are typically used in areas with homogenous types of vegetation, and can lack the ability to detect historically-contingent, context-specific changes in highly heterogeneous landscapes. The metrics typically utilized in continuous approaches are also highly sensitive to rainfall, and are essentially a metric of productivity that does not capture differences between compositional types of vegetation in heterogeneous landscapes.

The limitations and tradeoffs between different types of remote sensing change detection techniques mirror historical concerns in rangeland ecology methods for understanding vegetation change. Studies that have used biotic indicators, such as vegetation composition alone, as an indicator of ecosystem state have in the past been considered highly problematic when used rigidly in rangelands, due to conflation of biotic change with abiotic change. The focus on biotic indicators is concerned with ecosystem function, but focuses on biodiversity and productivity (Coughenour 1985); however, this emphasis on biotic processes alone may not always be indicative of the underlying processes, such as irreversible shifts in ecosystem behaviors (Shackelford et al. 2013). Metrics such as composition alone may be highly variable and contingent upon rainfall, and may not indicate changes in function or underlying abiotic process (King and Whisenant 2009). Additionally, composition-based surveys in rangelands have in the past frequently drawn upon an idea of ecological stability stemming from a Clementsian understanding of communities (Scoones 1999) where changes from a historical state are assumed to be the result of a disturbance. By examining metrics of both productivity and structure from remotely-sensed imagery, as well as analyzing detailed patterns of plant community composition, we employed an approach that draws from both functional and compositional perspectives.

In sum, the above concerns with metrics as well as classification techniques, combined with limitations in our own historical ground-truth data, led us to use multiple metrics, and a hybrid methodology for change detection. We used a remotely-sensed approach that employed a comparison of two metrics of transition that were selected to capture aspects of both productivity and composition to explore vegetation transitions, and we supplemented it with two composition-based field studies of plant communities using plot sampling and transects to understand fine-scale vegetation compositions. We used a hybrid methodology of categorical and continuous land-cover change approaches (detailed in Figure 5.1), in which we analyzed the continuous metrics of change (NDVI difference, Principle Components Analysis) while controlling for the initial vegetation state in 1987. This allowed us to focus on understanding transitions from past vegetation types as continuous for comparison to ecological gradients rather than interpretation of the drivers of categorical change. This led to an understanding of vegetation transitions as sensitive to the initial starting state of the vegetation, rather than analyzing changes in productivity alone across the landscape. This allowed for an improved understanding of historical contingency, as well as abiotic context in this highly heterogeneous landscape. Further, this hybrid approach also greatly decreased the sensitivity of our change analysis to land cover classification accuracy because of the reduced error multiplication when analyzing transitions. It was our hope that combining consideration of both compositional attibutes and metrics of productivity would be indicative of processes that are sensitive to a number of different states and a continuum of transitions between them (Shackelford et al. 2013). Both remotely sensed change metrics and plot-based data were then analyzed side-by-side in relation to herbivory gradients, while controlling for environmental gradients that are also expected to be correlated with changes that have occurred. The workflow, including all remote sensing steps, is summarized in Figure 5.6.

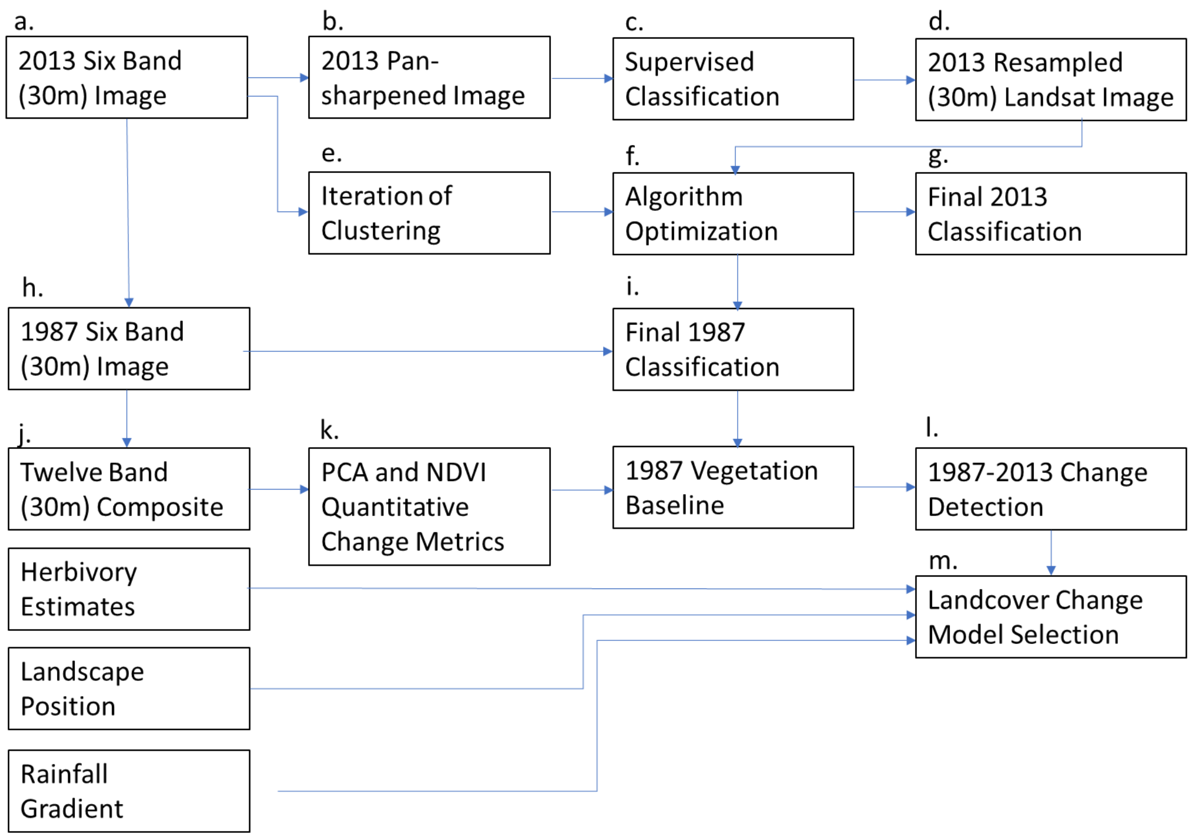

Figure 5.1. Workflow diagram for classification procedure and change detection

Supervised Classification

We began by classifying a pan-sharpened Landsat 8 Image (May 23, 2013, Path 168 / Row 60), using 6 bands: red, green, blue, near-infrared, short-wave infrared 1, and short-wave infrared 2 (Fig 5.1a). Landscapes with complex, heterogenous vegetation assemblages prove difficult to classify with the medium resolution (30m pixel size) data of Landsat imagery because vegetation types are often mixed within pixels. To gain a better understanding of how fine-resolution structure of vegetation is represented within medium, coarser resolution cells, Landsat 8 imagery was converted to a pan-sharpened image using a hybrid wavelet (PCA setting, Figure 5.6b), producing an image that closely matched the spectral qualities of the original image following visual inspection. We used GPS points and knowledge of locations of different types of vegetation to extract spectral values from the pan-sharpened Landsat 8 images and create signatures using Erdas Imagine. Through numerous iterations of classification and preliminary, visual evaluation, we compared classifications to a high-resolution Quickbird image (from November 2011) and our own on the ground knowledge of vegetation to determine whether classes were representative of landscape vegetation assemblages, and were able to capture classes of interest, especially recently encroaching vegetation that proved difficult to distinguish from other types of vegetation. We then supplemented these signatures using a k-means unsupervised classification technique to obtain signatures for landcover types not captured in our ground survey for creation of vegetation signatures. Signatures were then reviewed and edited systematically. If a class erroneously included multiple vegetation class types, it was split into two new classes using our knowledge of landscape vegetation composition, by manually extracting new replacement signatures from known locations. Signature separability was graphically explored within Erdas Imagine signature editor (Appendix O.1), through a separability matrix (Appendix O.2), through graphic examination of the distribution of extracted spectral values for each pixel (Appendix O.3), and using the dendrogram tool within Erdas to further understand similarity. The parallelepiped algorithm in Erdas Imagine was used as the non-parametric setting. We used settings that left areas unclassified if they did not fall within parallelepiped limits. We then extracted signatures from unclassified areas where we knew from on the ground experience what type of vegetation occurred, and classified them accordingly until there were less than 8.89% unclassified pixels. Using an iterative process of classifying the image with no overlap, and then using a fuzzy classification to determine which signatures overlapped most frequently, we adjusted the parallelepiped limits until overlap was minimized. Through ten iterations of this process, images were classified using different mixtures of supervised and unsupervised approaches, adjusting the parallelepiped limits of the signature editor until a signature set was finalized that captured maximum complexity of the landscape.

One type of woody vegetation in luvisol soils was found to be inseparable by this methodology, due to complete overlap of spectral signature and confusion with vertisol soils with perennial grass cover. To work around this problem, a binary mask was created along known edges of vertisol soils, separating them from luvisol soils. Further masking was used to exclude pixels that were mixed with classes not of interest to this study, such as water and buildings, and allowed for greater certainty that pixels were capturing the transitions of interest, rather than mere artifacts of coarse resolution sampling.

Following determining a final supervised classification of the pan-sharpened 2013 Landsat image (Figure 5.1c), it was then resampled back to 30m resolution (Figure 5.1d). New mixed-pixel classes were assigned a unique identifier based upon the top two most frequent classes found within each 30m resampled pixel. All of the resulting classes were then clustered using a hierarchical cluster analysis (Ward's method, JMP Version 11) of the averages of the values for the six bands, to merge the large number of resulting classes (>2000 classes) into 197 new classes. We then used Principle Components Analysis (PCA) on the six original bands of the 30m resolution image to calculate the first three principle components (R Core Team 2014) of pixels falling within Koija, which combined explaining 97.88 % of the variation in the data of the 2013 30m Landsat image. These PCA values were then extracted along with the unique identifier for each mixed-pixel class. Using the scores from the first three components as variables for the x, y, and z axes, the classes were then graphically analyzed for overlap in three-dimensional space of the principle components, with points from classes indicated by differing colors (Appendix O.4). Classes that were dispersed widely over the graph and lacked any distinct clustering in 3d space were removed, and classes that appeared to be redundant or to overlap in 3-dimensional space were merged. This resulted in reduction to 56 classes (Appendix O.5). The centroids (in 6 band spectral space) of these 56 classes were then determined, and analyzed to determine classes that were nearest in spectral space using the FNN package in R (Beygelzimer et al. 2015). Classes with centroids that had a distance value of under 200 were then once again analyzed for overlap in the first three principle components, and any classes that closely overlapped were merged, while classes with highly dispersed patterns that subsumed another class were deleted, leading to 28 final classes (Appendix O.6). The centroids from these 28 classes were then determined, and input as the centroids for a k-means unsupervised classification in R (Maechler et al. 2017).

Unsupervised Classification Fitting

We then varied the parameters of several unsupervised classification approaches to attempt to match this resulting semi-supervised image, beginning once again with the original six input bands from 2013 (Figure 5.1a). Using a customized R script (Appendix P), an iterative, unsupervised clustering was used where the parameters of different classifications were compared to determine the optimal clustering approach (Figure 5.1e). The parameters that we varied were the number of clusters in the classification (2-100), the unsupervised classification approach (k-means, clara, or hierarchical k-means), and the distance measures of the k-means classifier (euclidean, maximum, manhattan, canberra, binary or minkowski). For those that utilized hierarchical k-means clustering, where a hierarchical approach is used first to determine the cluster centers used for the k-means clustering, we varied the method of this hierarchical clustering (ward.D, ward.D2, single, complete, average, mcquitty, median or centroid). Additional iterations were performed where the k-means algorithm (Lloyd or Hartigan-Wong) was varied as well. Finally, to determine which technique most closely matched the semi-supervised image produced at the end of the previous section, the resulting classes were compared for overlap in similarity using the adjusted Rand index (Rand 1971, Hubert and Arabie 1985, Figure 5.1f, Figure 5.2).

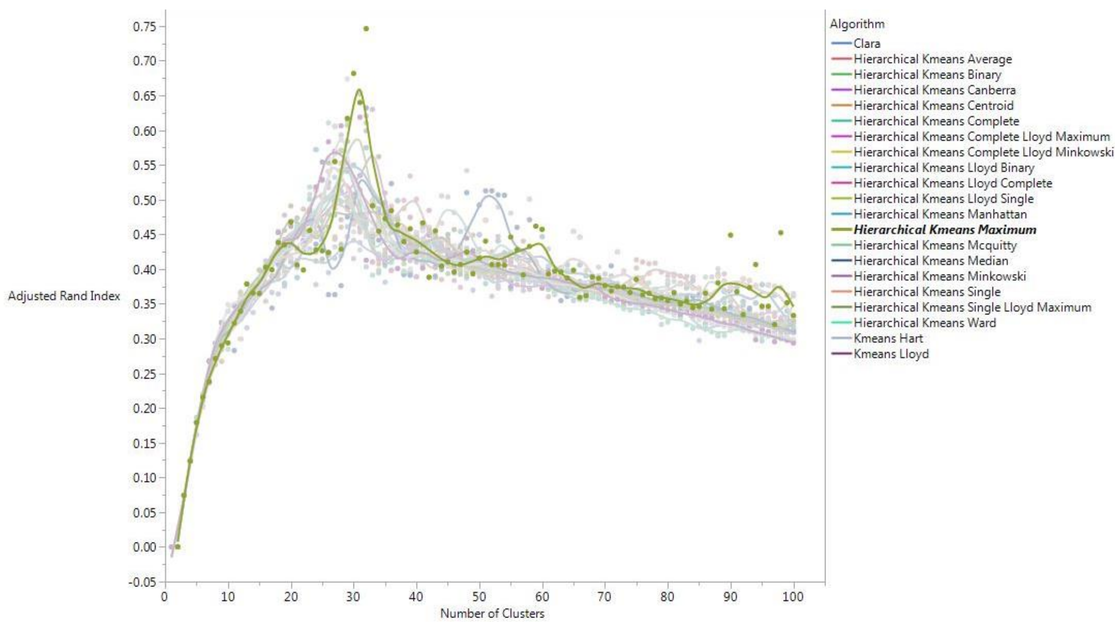

Figure 5.2. Multiple clustering parameters compared to rand index value, with the optimal classification technique highlighted (hierarchical k-means maximum).

Hierarchical K-means with a maximum distance measure produced the best-matching unsupervised image according to the adjusted Rand index (Scrucca et al. 2016) of the 2013 image (Figure 5.1f, Figure 5.2). Once the unsupervised 30m resolution Landsat image from 2013 was validated for similarity to the supervised pan-sharpened classification of the same scene/date and then validated in comparison to field data (5.1g), the identical parameters of the classification algorithm were applied to a 30m resolution Landsat image from 1987 and the same six corresponding bands as for the 2013 image (Path 168/ Row 60, July 3, 1987, Figure 5.1h). As different types of vegetation existed historically, we varied the cluster number in the classification, and the optimal cluster was selected based both upon visual inspection of the classes in comparison to types of vegetation that could be distinguished in 1977 air photos, and the graphical minimization of within-cluster sum of squares within a range of +/- 10 classes from the optimal number of 2013 classes. This was then applied to the Landsat image from 1987 to create a final classified image (Figure 5.1i).

This historical vegetation classification was then compared to preliminary findings from an in-preparation study of oral histories of vegetation changes (See appendices G and H for an abbreviated summary of the methodology and findings). In this study, we first conducted semi-structured interviews with five elders, in which we first determined the Maa vocabulary and taxonomy used for referring to different vegetation assemblage types that occur in different landscape positions (Roba and Oba 2009). We then completed 14 semi-structured interviews at different locations on the landscape selected to represent the different main categories of assemblages that were said to exist. At each of these locations, each elder indicated the changes in vegetation and soils they had experienced in each place since the 1980s. Drawing from the extensive vegetation knowledge of these elders, this information was used to identify the general vegetation types classified within the 1987 vegetation classification, and these vegetation types were also verified using an air photo from 1977.

Finally, the unsupervised classifications were verified for producer's and user's accuracy. We used stratified random sampling of 50 points per class for both 1987 and 2013 classifications. The 1986 unsupervised classification was verified for producer's and consumer's accuracy using comparison to an aerial photograph taken in 1977 (Table 5.1), supplemented with historical knowledge of plant distributions from interviews with herders (Appendix H), where past vegetation types were indicated. Using a Pleiades image from May 2013 (Table 5.2), the same time the Landsat image was captured, we verified the 2013 classification, supplemented by detailed knowledge of plant distributions from field surveys.

Continuous Vegetation Change Metrics

Using Landsat 5 (1987) and Landsat 8 (2013) image scenes that included Koija group ranch we calculated continuous measures of spectral change between the two dates (Figure 5.1j). As vegetation indices in remotely sensed images are highly sensitive to the previous several months of rainfall, Landsat scenes were selected based upon first-hand knowledge of the high amount of rainfall that occurred leading up to the date acquired, or with knowledge of historical records of rainfall. Two of the authors (Unks and Naiputari) were at Koija when the 2013 scene was captured, and by most accounts it was the most rain that had fallen on Koija sequentially in years. This was verified by the cloud cover scene in previous Landsat scenes, and similar cloud cover was verified for scenes predating the scene selected for 1987. Landsat Imagery was obtained as atmospherically corrected, cloud-free, Landsat Surface Reflectance Climate Data Records (CDR) from the United States Geological Survey. NDVI was selected for use as a metric due to its ease of interpretation as change in biomass. Using the raster calculator function in ArcGIS, the NDVI was calculated for 1987 and 2013, and then the 2013 image values were subtracted from the 1987 image for each pixel (Figure 5.1k). This resulted in a minimally negative value if NDVI had greatly increased over this time, and a maximally positive value if NDVI had decreased greatly, as in the case of complete loss of vegetation. A PCA was then used to reduce 12 bands (6 from 2013 Landsat 8 images and 6 from 1987 Landsat 5 images: Red, Green, Blue, Near-infrared, Short-wave Infrared 1, and Short-wave Infrared 2 (Figure 5.1k)). The first three principle components together explained 90.71% of the variation in the 12 combined bands. Principle component 1 (referred hereafter as PC1 variable) was selected as a complementary index of variation to NDVI as it graphically appeared to correspond to changes in plant composition indicated by comparisons of historical air photos and 2013 Pleiades imagery, while PC2 and PC3 both were more closely correlated to NDVI (Appendix R). In sum, we used NDVI as continuous proxy for biomass change, and PC1 as a continuous proxy for vegetation composition change.

Environmental Gradients

Household survey data that indicated preferred watering points, forage points, frequency of use, and number of livestock were used to approximate species-specific herbivore pressure at the landscape scale using a model-fit procedure and the least cost algorithm in ArcGIS (Chapter 4). The least cost corridors produced during this modelling exercise were used to create community-wide composite heat-map raster layers for small stock (sheep and goats, which are typically herded together, figures 5.3 and 5.4) and cattle (Figures 5.5 and 5.6, see Chapter 4 for detailed methodology). These were created for specific times of seasonal restriction in land use, as well as for times when restrictions were lifted and movement to anywhere on Koija was permitted. Times of seasonal restriction corresponded to times when there had been ample rainfall and watering points and forage near homesteads are utilized, allowing restricted areas to regenerate. Times with no land use restrictions corresponded to dry seasons and drought. In the resulting heatmaps, each pixel represents a unitless measure of intensity of use by each type of livestock herd (small stock or cattle) during these different time periods. These rasters were resampled from 30.85m to 30m resolution.

These heatmaps reflect current land use from surveys in 2013 and are thought to be representative of land use since at least 2002. However, to ensure they estimate historical herbivory pressure as well, we examined historical (1987) NDVI images of Koija to try to detect past homestead locations and assess their degree of concordance with current homestead locations. Grazing lawns tend to establish in former homesteads, resulting in isolated contiguous clusters of a few pixels of very high NDVI levels on Landsat images. Our assessment indicated that by 1987, a large number of smaller scattered livestock pens had been used and abandoned, and their locations coincided with current locations of homesteads. In the past, these homesteads were greatly dispersed and had high compositions of cattle kept within them (Chapter 2), and it was confirmed that there were still these dispersed homesteads in 1977, using air photos. Though a number of homesteads were moved in the intervening years, people came and left Koija, and livestock composition fluctuated, this initial analysis showed a general geographic coincidence of homesteads and livestock concentrations in the past and today, and supported the use of our heatmaps as livestock pressure estimates for the time period as a whole as the most accurate estimate that could be created practically, under study constraints.

In creating environmental layers for analysis, we first considered that a subtle gradient of decreasing mean annual precipitation exists across Koija group ranch from west to east, away from the Ewaso Ng'iro river, as shown by Franz et al. (2010). To the east of the settlement areas of Koija, there is also a decrease in frequency of precipitation that has been observed. To account for this spatial pattern of rainfall we created a Euclidian distance raster, with a 30m resolution and where each cell was assigned a value of distance to the Ewaso Ng'iro river. This variable was included not as an explanatory variable of changes, but as a means of controlling for variability that might otherwise be multicollinear with livestock pathways and conflated with the impacts of livestock. We then included raw DEM elevation data, resampled to 30m as a coarse proxy for soils, temperature, and soil moisture. Finally, we included a gradient of slope (>9% slope) that was treated uniformly in heatmap production and so contributed information not present in those maps. This was included based upon observations that the steeper areas of slopes at Koija are frequently bare in comparison to more gently sloping areas nearby, with no clear immediate driver. We included these variables as control for abiotic factors, but also as a way of informing hypotheses for future studies of vegetation change. Multiple regression of both vegetation change indices (NDVI and PC1) were then used as response variables within each 1987 vegetation class (Figure 5.1l), with predictor variables of herbivory, slope, rainfall, and elevation (Figure 5.1m), on 200 stratified random samples.

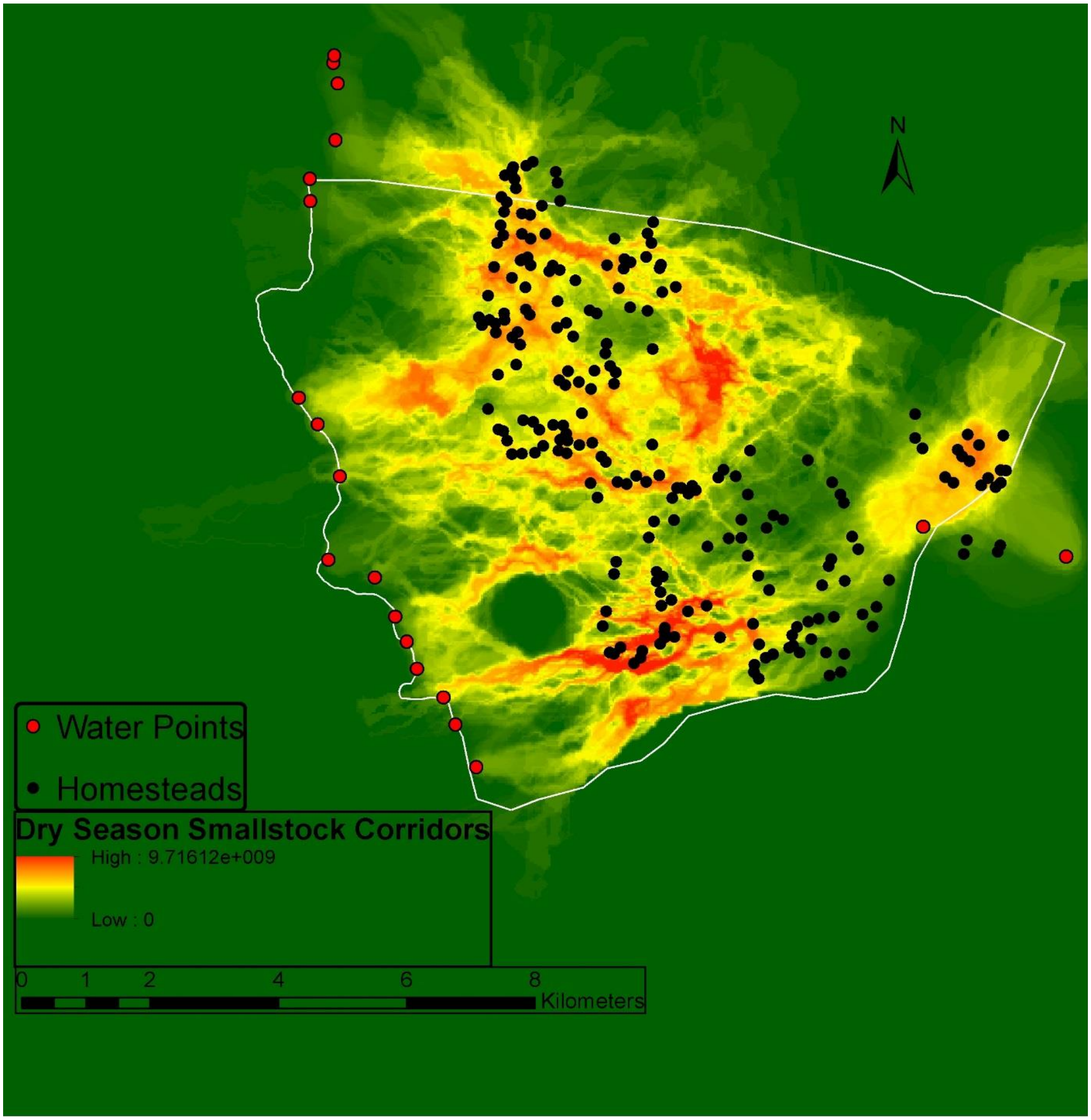

Figure 5.3. Small stock dry season heatmap of estimated pressure (unitless)

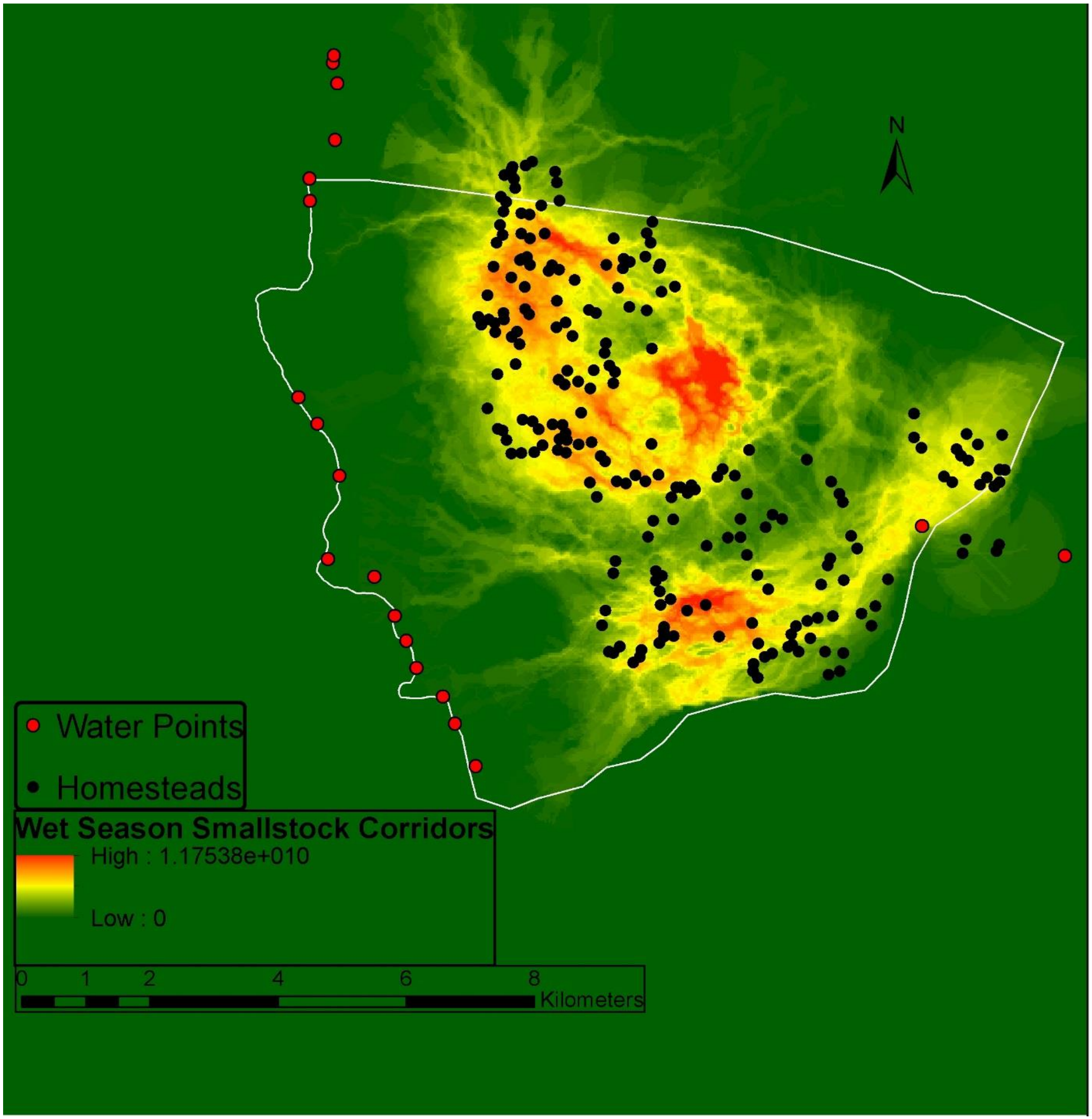

Figure 5.4. Small stock wet season heatmap of estimated pressure (unitless)

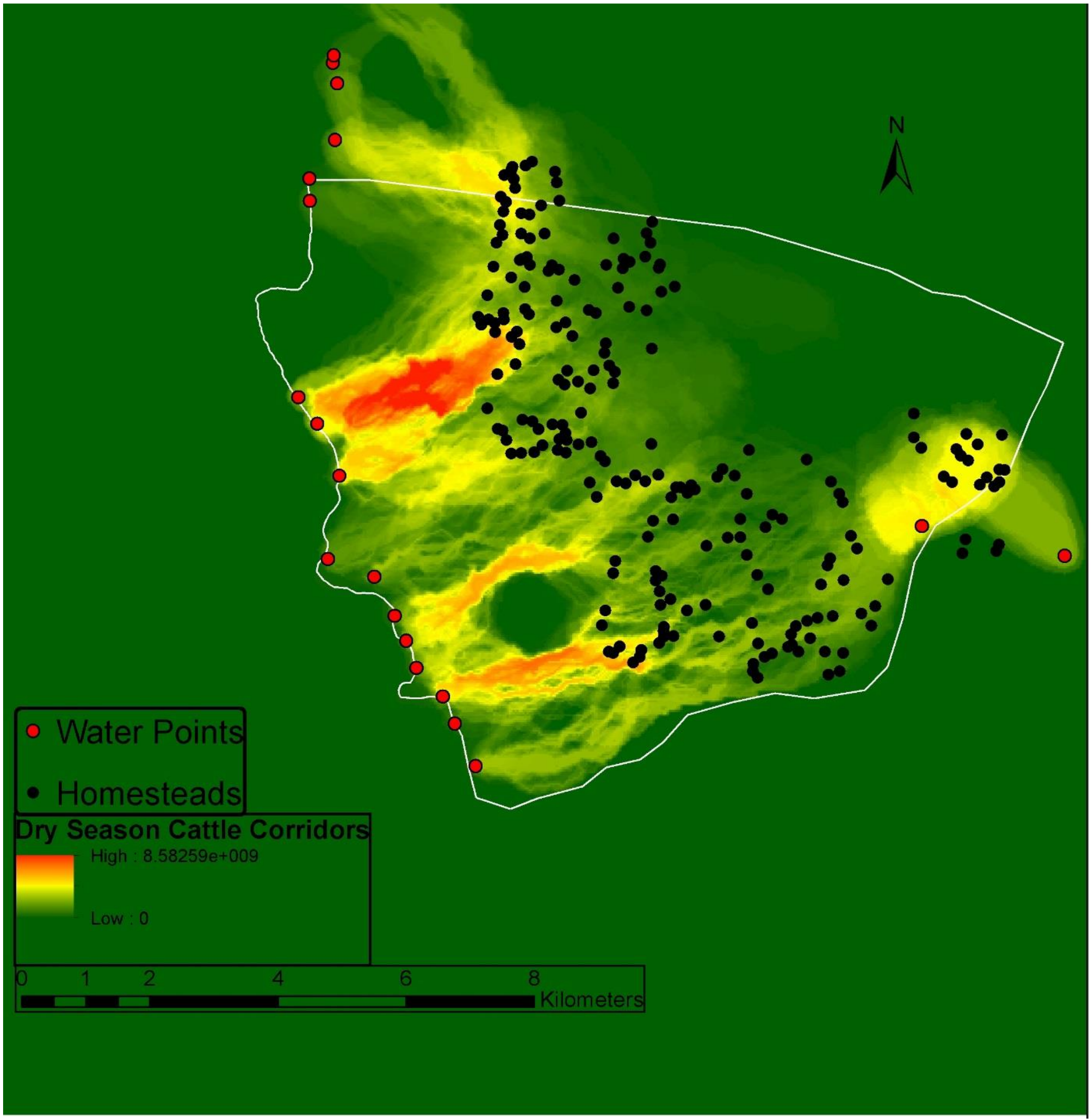

Figure 5.5. Cattle dry season heatmap of estimated pressure (unitless)

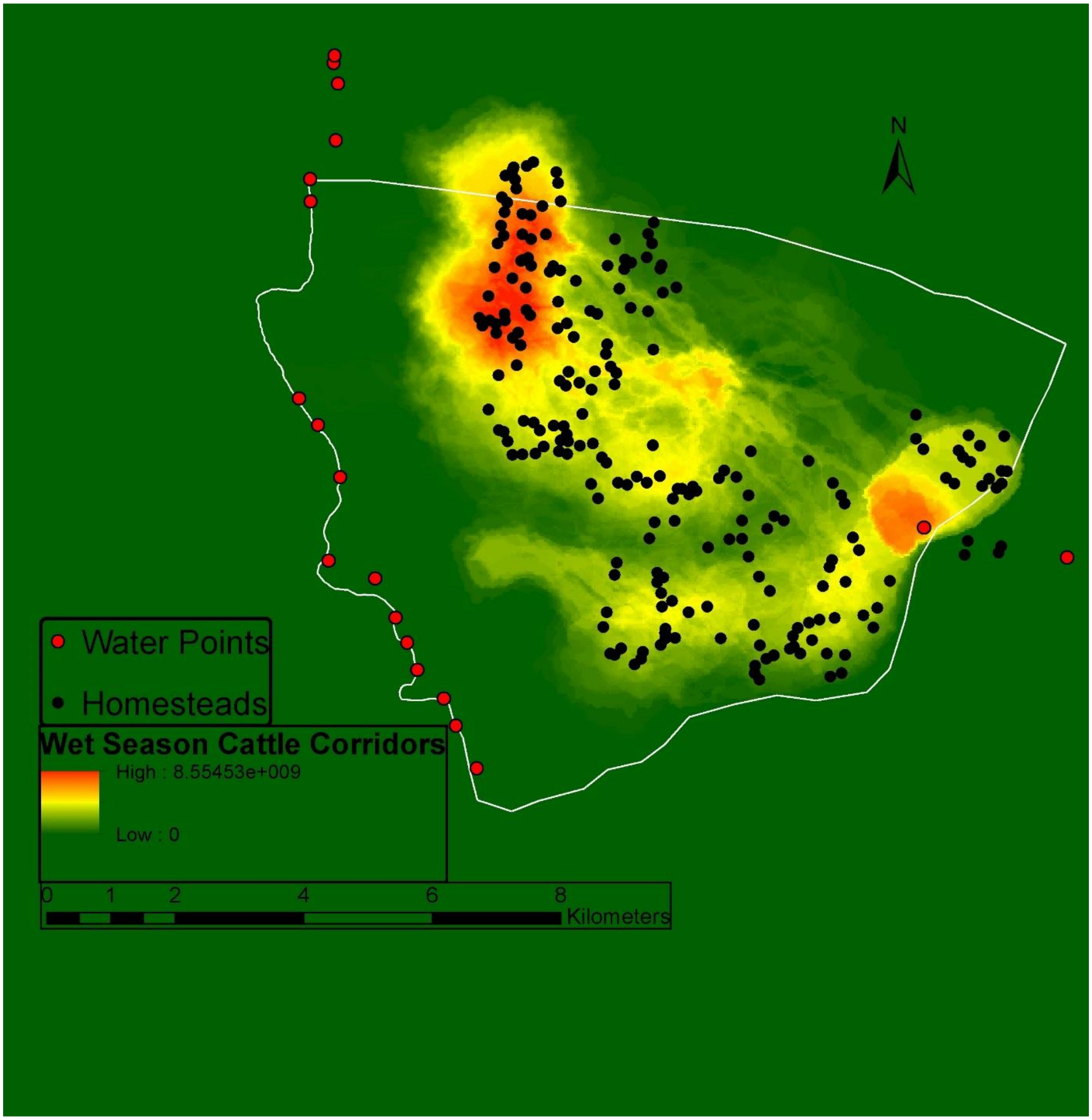

Figure 5.6. Cattle wet season heatmap of estimated pressure (unitless)

Land-cover Change Analysis using Vegetation Indices, Vegetation Classes, Environmental Gradients, and Herbivory Gradient Sampling

We then masked out a number of features, including built roads, permanent structures, fenced areas, and within 50 meters of all homesteads, and excluded all areas outside of Koija from the analysis. For analysis of landcover using the remotely sensed imagery, at each pixel within Koija (Figure 6) we then extracted the value of: the NDVI difference image, the three principal components, the slope layer, elevation data, the rainfall gradient, and the four herbivory heatmap layers, and converted these data to a spatial points data frame in R. A stratified random sample of 200 pixels assigned each 1987 classification cluster was subset in R. This was also repeated for three vegetation types in the 2013 classification which were known areas of dense encroachment by *A. mellifera*, *A. reficiens,* and *S. volkensii*. All independent variables were examined using histograms and tested for normality of distribution using normal quantile plots.

Plant Community and Functional Analysis Sampling Methods

Two plant community datasets, described in the two following sections, were collected with different sampling strategies specifically designed to represent each community type. All environmental gradient and herbivory gradient variables were extracted in an identical manner to the sampling above, but extracted using the GPS coordinates of each sampling plot.

Grazing Lawn Vegetation

This first plant community data set was created through non-random sampling in 2015 and located within *muurua* or grazing lawns (Young et al. 1995) that were thought to be at least 50 years of age based upon oral history. These areas support unique plant communities (Young et al. 1995) and have elevated concentrations of nitrogen, phosphorus, potassium, calcium, and magnesium compared to surrounding areas (van der Waal 2011). Sampling was restricted to very large grazing lawns that correspond to historical locations of very large, multi-family enclosures. We verified with elders that each of these sites had existed as a grazing lawn when they were young. Plots were placed in these conspicuous flat areas with low densities of woody species and high densities of *Digitaria milanjiana* and *Tribulus terrestris*, located on the flattest portions of the tops of hills (Figure 5.7). At each location three points were located that were at least 60m apart, and at each point the four contiguous Landsat (30m) pixels that shared the point as a centroid were sampled for % cover of all vegetation with a presence above 5%. Any points that were indicated to have not been classified as dense perennial grass in 1987 were later excluded from statistical analyses, and the remaining plots at each location were averaged, yielding a final number of 14 locations that were compared. In comparing these data to the herbivory gradient, we only considered the wet season pressure estimates, as the plots were located in a circle around settlements, creating a bias of the gradient away from the sampling points.

Hilltop Vegetation

A second plant community data set was created through systematic sampling along two hilltops that run parallel to the Ewaso Ng'iro river. This region was selected because it lies between the clusters of homesteads and watering points along the river, and the heatmaps generated in Chapter 4 revealed highly heterogeneous intensities of livestock utilization in the area. A grid was created with uniform spacing along the hilltop in ArcMap, with a spacing of 600m x 325m between gridlines. At each vertex of gridlines, using a slope map created using a digital elevation model in ArcMap, we selected the flattest site with multiple contiguous pixels of low slope that was near to the vertex, and that site was visited to assess its suitability for a set of vegetation-sampling transects. If a site was too small to accommodate our transects without sloping downhill, or if the site fell within an area that accommodated a grazing lawn, the next nearest suitable site was selected. At one of the vertices, there was no site that could accommodate transects without overlapping with another site in any direction. At each identified site (n=28, Figure 5.7), we placed three 50-meter tapes on the ground, spaced by 20 meters, oriented north to south, and aligned in the center along the border of the Landsat pixels. We then sampled vegetation along each tape using a modified line-intercept method, whereby for every 1-m line segment, we identified each species intercepted by the line, and estimated the percent of the 1-m line (the number of cm out of 100) that intercepted cover of each species.

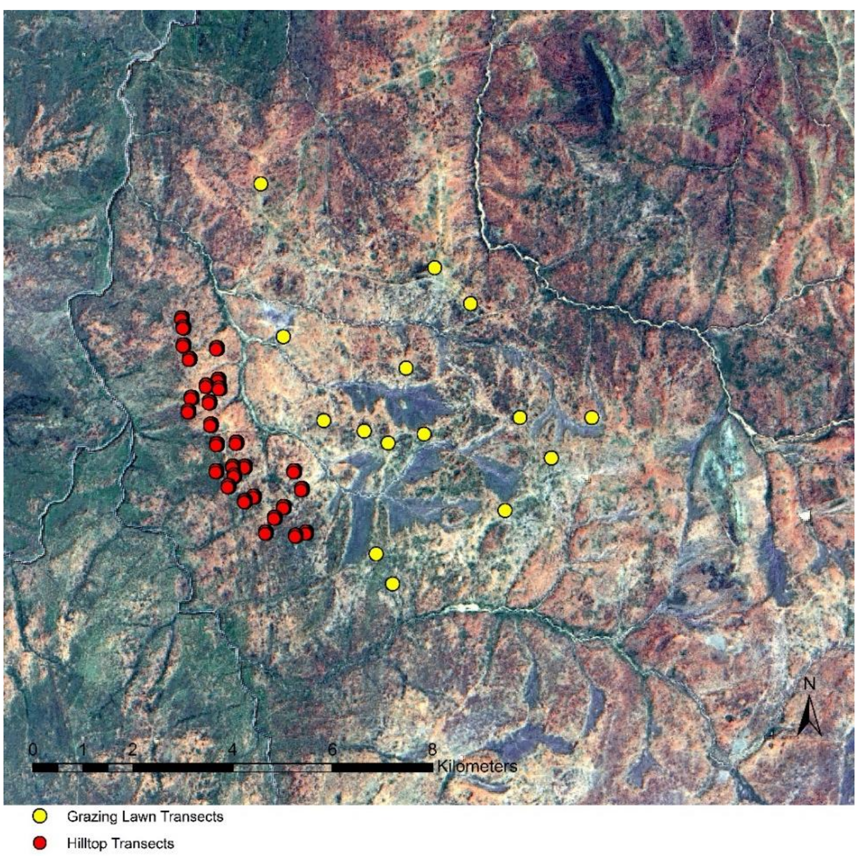

Figure 5.7. Locations of plant community sampling plots

Analysis of Plot Data

We analyzed the two plant community datasets using identical methods that included a multivariate analysis of community composition and individiual species percent cover. Detrended correspondence analysis in PC-ORD (McCune and Mefford 2011) was used to ordinate plot data and to examine the dominant trends in species compositions driving the ordination. We then extracted values of the landscape predictor variables in the model (elevation, rainfall gradient, herbivory estimates) for inclusion as predictor variables in PC-ORD. We ignored correlations that were below r=0.159 between the three axes of the resulting ordination and the gradient variables. Individual species from this dataset that indicated strong correlations between axes that also had strong correlations with predictor variables were noted. We then analyzed these individual species % cover as the dependent variable in multiple regression analysis using the predictor variables of herbivory, elevation, and distance-to-river.

Multiple regression procedure

For the multiple regression of the three sets of dependent variables 1. Landcover change, 2. Glade Plots, and 3. Transects, we used a standardized procedure. Using R (see appendix S for example code) square-root or log transformations were applied if right-skewedness was present in normal quantile plots, and squared transformations were applied if left-skewedness was present. Variables were re-scaled in R prior to analysis to enable comparison of coefficients to determine their relative value and to allow for use in spatial error models. Initial tests were then run as OLS regressions in R without spatial weights using a model selection procedure that considered all possible combinations of independent variables, leading to the model with the lowest AIC score being selected (Mazerolle 2017). The variance inflation factor was then calculated using package car in R (Fox and Weisberg 2011), and if multi-collinearity was indicated by this metric (we used a strict value of >~1) or by the correlation of coefficients matrix, the next best-scoring model in the model selection procedure above was selected. Moran's I was then calculated to determine whether autocorrelation had influenced the model outcome, residual plots were examined, and a Breusch-Pagan test was used to check for heteroskedasticity. If heteroskedasticity was detected, a Box-Cox transformation was performed using the Caret package. If a regression with multiple dependent variables had a significant R-squared, we then calculated the R-squared value while removing one variable at a time, to determine which had the overall largest change in R-squared value. If this change was large relative to other factors, this variable was then considered to have a discernable landscape effect, and was discussed in depth. Finally, if spatial autocorrelation was apparent, LaGrange multiplier diagnostics were then calculated. If LaGrange lag tests alone were significant, a lag autoregressive linear model was applied to adjust the model for autocorrelation in the independent variable alone, while if both lag and error tests were significant, a spatial error model was used. Both of these spatial autoregressive models were calculated using the spdep package in R (Bivand and Piras 2015).

Results

Ethnographic accounts of Landscape Vegetation Change

Ethnographic accounts were rich with descriptions of multiple types of changes in vegetation that had occurred in many areas of Koija, including frequently stated large increases in dominant tree species, especially *Acacia mellifera* (Vahl) Benth. *and Acacia reficiens* Wawra. Now-dominant species that were mentioned to have increased within the last 30 years are *Cissus rotundifolia, Rhamnus staddo, Solanum incanum, Ipomoea kituensis, Opuntia stricta* (Haw.) Haw.*, and Sansevieria volkensii* Gurke. Among the species that were listed that had decreased in abundance were *Euphorbia tirucalli* L., *Euphorbia bussei* (N.E.Br.) S. Carter, *Boscia angustifolia* A. Rich.**,** *Hibiscus greenwayi*, *Euphorbia nubica,* and *Teclea nobilis.* This ethnoecological study (in prep) is summarized as preliminary results in Appendices G and H.

Remotely-sensed Landscape Vegetation Change

The final classifications from 1987 and 2013 are shown in Figure 8, with a 90.11% total accuracy for the 1987 classification, and an 89.58% for the 2013 classification. The accuracy of the individual vegetation classes, assessed for omission and commission errors, is displayed at each date are in Tables 5.1 and 5.2. Our subset areas of analysis included 7180.92 hectares within Koija. The most conspicuous changes in these images were that dense *E. tirucalli* canopy formerly covered at minimum about ~619.74 hectares, or about 8.63% of the study area, and a mix of *E. bussei* and *E. tirucalli* formerly covered at least 678.6 hectares, or about 9.45% of the study site. These species were likely present in other areas as well, but in areas that the two intermingled and were dominant, probably amounted for at least about 18.08% of the land cover in 1987. *Euphorbia tirucalli* is very rare on Koija today, and the areas it formerly dominated today consist of primarily dense perennial grass with sparse shrub cover of *Acacia spp.*, *Euphorbia heterochroma,* and *Croton dichogamus* shrub cover (indicated by green and red bars at the top right of Figure 5.19). Almost all adults of *E. bussei,* have also been lost, except for those located behind fences and protected from elephants, although it is common in a low, shrubby form. Areas that clearly supported medium to dense perennial grass following steady rains in 1987 (Classes 15, 16, 17, 18, 20, 22, 24, 28, 36, and 37) amounted to approximately 1648.98 hectares, or about 22.96% of the area considered. This compared to areas that clearly supported medium to dense perennial grass following weeks of steady rain in 2013 (4, 7, 12, 13, 20, 31) that in total amounted to ~992.25 hectares, or 13.82% of the area considered. However, transitions between these grassy areas were highly heterogenous (Figure 5.9), and did not show clear spatial patterns of change either. Finally, we estimated that 2656.98 hectares within Koija (37.0% of the area) have experienced some amount of *A. mellifera* or *A. reficiens* shrub encroachment between 1987 and 2013 (Including classes 4, 9, 17, 19, 24, 26, 29, 30 from 2013). Many of these transitions are indicated by the blue bars in the bottom right area of Figure 5.19, indicating an increase in NDVI in these areas.

About 78 hectares were bare in 1987, as visible in air photos from 1977, and several elders interviewed indicated these areas had been bare as long as they could remember (Appendix H). About 170 hectares of purely bare area were present in 2013, but many other classes included a greater percentage of bare cover per pixel, and were unable to be quantified accurately. Areas of *Euphorbia spp.* shrubs, the vines they support, and herbaceous vegetation were likely lost from the landscape based upon elder accounts, but as these vegetation types are much more heterogeneous, were difficult to quantify across the landscape, and were instead discussed based upon categories of landscape position in the following analysis.

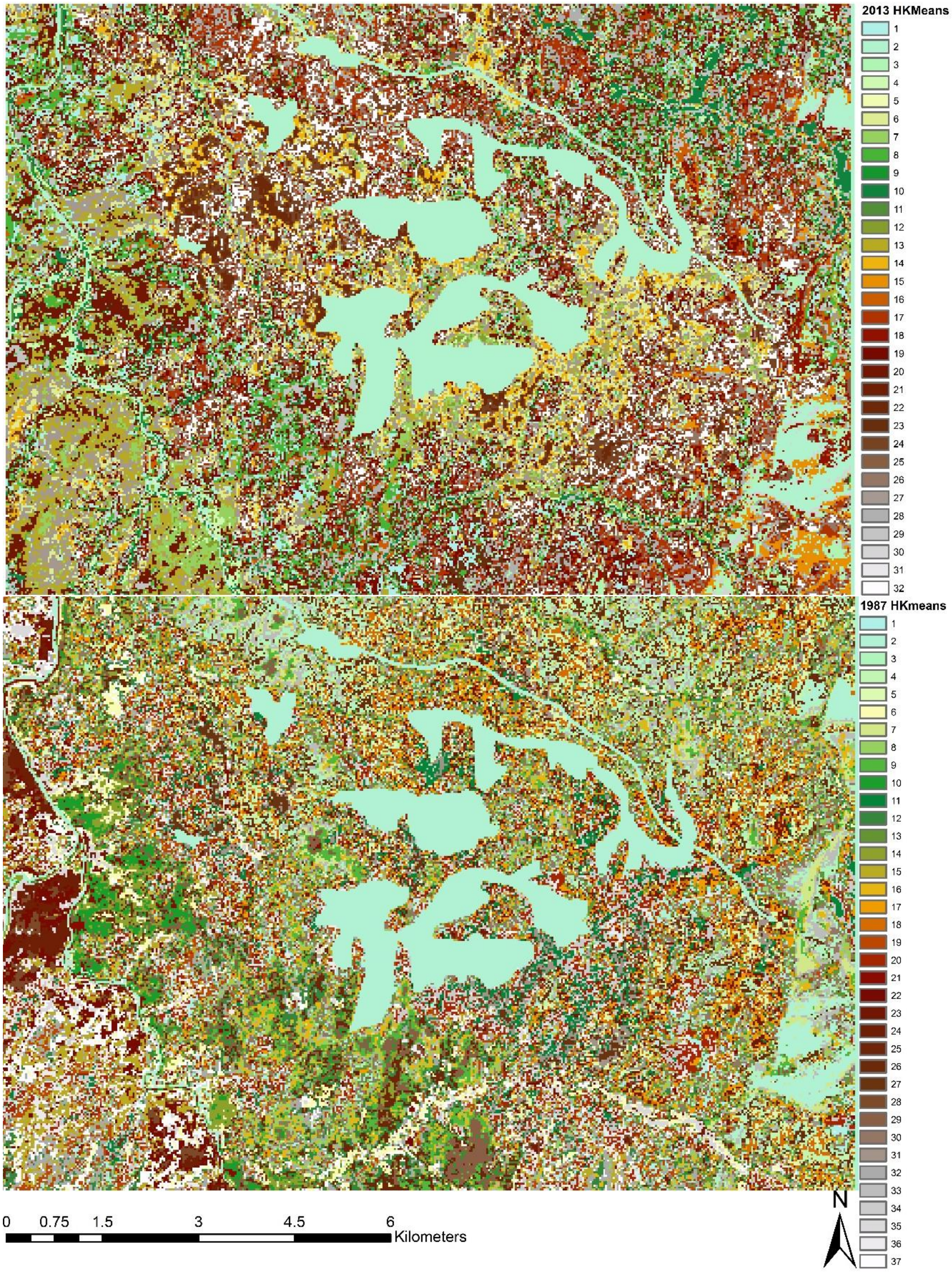

Figure 5.8. Unsupervised hierarchical k-means classifications 2013 (top) and 1987 (bottom). (Colors do not represent reflectance values).

Table 5.1. Accuracy assessment of 1987 hierarchical k-means unsupervised vegetation clusters

Vegetation Type	Class Number	Number Sampled	Total Assigned This Class	Total Correct	Producer's Accuracy (%)	Omission Error (%)	User's Accuracy (%)	Commission Error (%)
Mixed grass and shrub	1	50	48	44	91.67	8.33	88.00	12.00
Low NDVI shrub and grass 1	3	50	48	47	97.92	2.08	94.00	6.00
Low NDVI shrub and grass 2	4	50	49	44	89.80	10.20	88.00	12.00
Low NDVI shrub and grass 3	5	50	50	48	96.00	4.00	96.00	4.00
E. tirucalli forest 1	6	50	38	36	94.74	5.26	72.00	28.00
Very dense shrub and *E. bussei*	7	50	48	46	95.83	4.17	92.00	8.00
Perennial grass and herbaceous patches	8	50	50	46	92.00	8.00	92.00	8.00
Very dense shrub (*E. heterochroma or E. bussei)*	9	50	68	46	67.65	32.35	92.00	8.00
E. tirucalli forest 2	10	50	51	49	96.08	3.92	98.00	2.00
Low NDVI perennial grass	11	50	62	50	80.65	19.35	100.00	0.00
Tree or dense shrub and grass patches 1	12	50	47	45	95.74	4.26	90.00	10.00
Tree or dense shrub and grass patches 2	13	50	58	46	79.31	20.69	92.00	8.00
E. tirucalli forest 3 (Less dense)	14	50	64	47	73.44	26.56	94.00	6.00
Perennial grass and mixed shrub 1	15	50	50	41	82.00	18.00	82.00	18.00
Perennial grass and mixed shrub 2	16	50	52	47	90.38	9.62	94.00	6.00
Medium NDVI grass and forb	17	50	41	38	92.68	7.32	76.00	24.00
Medium NDVI grass and forb 2	18	50	49	47	95.92	4.08	94.00	6.00
Grass/herbaceous and sparse shrub	19	50	42	39	92.86	7.14	78.00	22.00
Medium NDVI grass and forb 3	20	50	57	47	82.46	17.54	94.00	6.00
E. tirucalli forest 4 (Less dense)	21	50	49	49	100.00	0.00	98.00	2.00
High NDVI dense perennial grass 1	22	50	44	44	100.00	0.00	88.00	12.00
Low NDVI shrub and bare	23	50	43	42	97.67	2.33	84.00	16.00
High NDVI dense perennial grass 2	24	50	48	48	100.00	0.00	96.00	4.00
E. heterochroma and mixed shrub	25	50	49	43	87.76	12.24	86.00	14.00
Mixed dense *Euphorbia spp.*	26	50	45	45	100.00	0.00	90.00	10.00
Bare	27	50	49	49	100.00	0.00	98.00	2.00
Dense perennial grass / grazing lawn	28	50	53	50	94.34	5.66	100.00	0.00
E. tirucalli forest 5	29	50	51	50	98.04	1.96	100.00	0.00
Medium NDVI shrub and grass	30	50	52	45	86.54	13.46	90.00	10.00
Grass and sparse *Acacia spp.*	31	50	46	46	100.00	0.00	92.00	8.00
Mixed dense *Euphorbia spp.* 2 (w *E. tirucalli)*	32	50	51	45	88.24	11.76	90.00	10.00
Very dense *E. bussei* or *E. tirucalli* canopy	33	50	48	46	95.83	4.17	92.00	8.00
Medium NDVI grass 3 (with sparse forb or shrub)	34	50	48	46	95.83	4.17	92.00	8.00
Dense mixed Acacia spp. and grass (river or laggas)	35	50	35	34	97.14	2.86	68.00	32.00
High NDVI mixed grass, forb, and shrub 1	36	50	48	45	93.75	6.25	90.00	10.00
High NDVI mixed grass, forb, and shrub 2	37	50	69	42	60.87	39.13	84.00	16.00

Total Accuracy	90.11%

Table 5.2. Accuracy assessment of 2013 hierarchical k-means unsupervised vegetation clusters (Class 2, 5, 9, 11, and 21 were removed by binary mask).

Vegetation Type	Class Number	Number Sampled	Total Assigned This Class	Total Correct	Producer's Accuracy (%)	Omission Error (%)	User's Accuracy (%)	Commission Error (%)
Sparse *Acacia spp.* and grass 1	1	50	49	41	83.67	16.33	82.00	18.00
Dense *Acacia spp.* canopy 1	3	50	51	45	88.24	11.76	90.00	10.00
Dense perennial Grass 2	4	50	46	45	97.83	2.17	90.00	10.00
Low NDVI grass, forb, shrub, and bare 1	6	50	51	42	82.35	17.65	84.00	16.00
Dense perennial grass, forb, and shrub mix 1	7	50	46	46	100.00	0.00	92.00	8.00
High NDVI tree, shrub, grass, and forb mix	8	50	50	44	88.00	12.00	88.00	12.00
Acacia spp. canopy and dense perennial grass	10	50	49	45	91.84	8.16	90.00	10.00
Dense perennial grass	12	50	49	49	100.00	0.00	98.00	2.00
Medium dense grass with sparse shrubs	13	50	53	44	83.02	16.98	88.00	12.00
Shrub, sparse grass, bare 1	14	50	51	46	90.20	9.80	92.00	8.00
Shrubs and bare 1	15	50	46	44	95.65	4.35	88.00	12.00
Dense Acacia spp. canopy, shrub, and grass 1	16	50	54	49	90.74	9.26	98.00	2.00
Shrubs and bare 2	17	50	48	44	91.67	8.33	88.00	12.00
Dense Acacia spp. canopy, shrub, and grass 2	18	50	45	44	97.78	2.22	88.00	12.00
Sparse *Acacia spp.* and grass 2	19	50	47	42	89.36	10.64	84.00	16.00
Dense perennial grass, forb, and shrub mix 2	20	50	92	49	53.26	46.74	98.00	2.00
Bare	22	50	51	48	94.12	5.88	96.00	4.00
Shrubs and bare 2 (often A. reficiens)	23	50	46	45	97.83	2.17	90.00	10.00
Shrub, sparse grass, bare 2 (more bare)	24	50	42	41	97.62	2.38	82.00	18.00
Dense *Acacia spp.* canopy 2	25	50	38	38	100.00	0.00	76.00	24.00
Shrub, sparse grass, bare 3	26	50	59	50	84.75	15.25	100.00	0.00
Low NDVI grass, forb, shrub, and bare 2	27	50	41	40	97.56	2.44	80.00	20.00
Dense Acacia spp. canopy, shrub, and grass 2 (Low NDVI)	28	50	49	47	95.92	4.08	94.00	6.00
Dense *Acacia mellifera* and *Sansevieria volkensii*	29	50	49	48	97.96	2.04	96.00	4.00
Bare / rock	30	18	19	18	94.74	5.26	100.00	0.00
River edge / dense perennial grass	31	47	36	36	100.00	0.00	76.60	23.40
Shrub, grass, forb and bare	32	50	55	48	87.27	12.73	96.00	4.00
			Total Accuracy	89.58%				

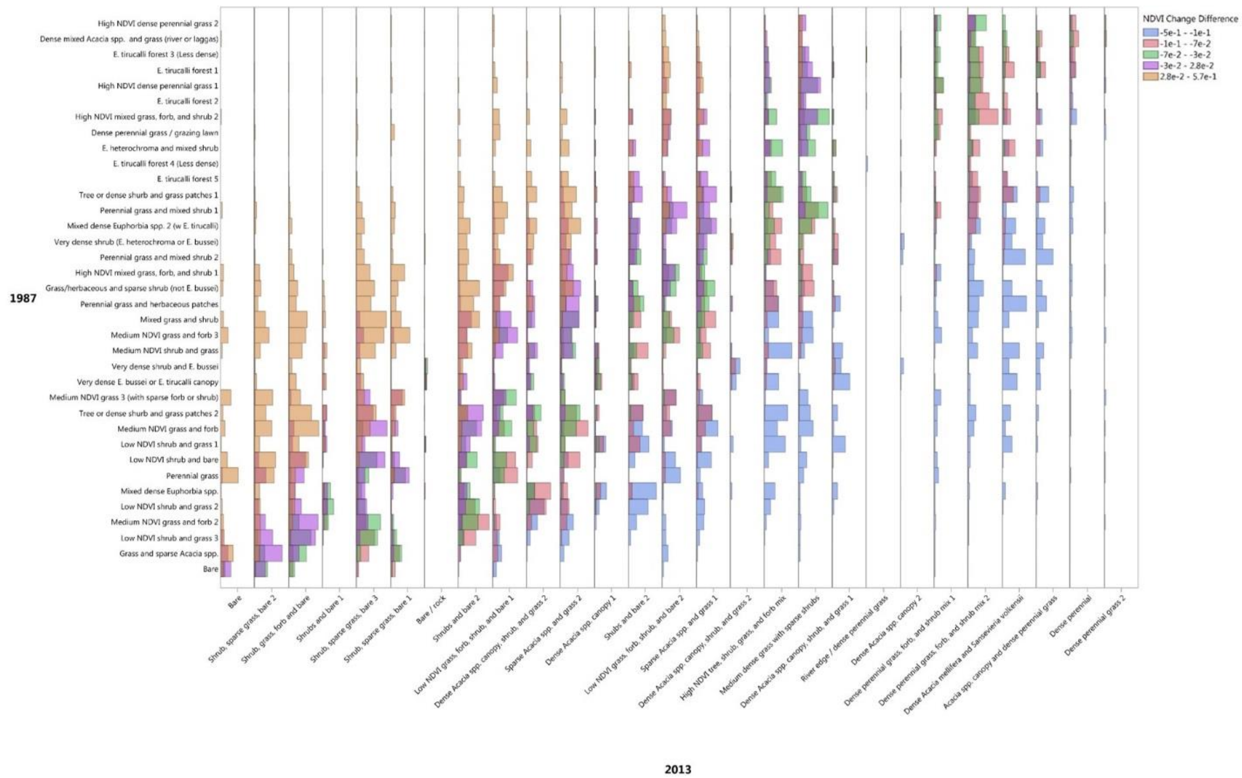

Figure 5.9. Transition matrix between 1987 and 2013 vegetation classes (classes are ordered ascending by the NDVI value at the classification date, bar values indicate the number of transitions between classes that occurred, and change in NDVI value over time is indicated by color-coded quantile -- where negative values indicate increases in NDVI over time, and positive values indicate decreases in NDVI over time).

Integrative analyses of key vegetation transitions

Drawing from elder accounts of vegetation changes, our own observations and vegetation sampling, and analysis of the changes that were observable using remotely-sensed imagery, we identified five different categories of vegetation transitions of interest that we focused on analytically. For each category, we first describe the general plant community, landscape position, species-specific changes indicated both by elders and vegetation classification transitions. Then we report analyses of correlations between remotely sensed change indices (NDVI and PCA of spectral bands), and landscape variables (elevation, slope, distance-to-river, and seasonal livestock pressure), followed by analyses of plot-based vegetation composition with respect to environmental variables. Full regression analysis results for remotely sensed indices are documented in Appendix T. Only summaries of the trends, and significant R-squared values above 0.10 for these tests are reported in the text.

1. Changes in grazing lawns (Muurua)

Areas known as grazing lawns, or (*muurua* in Maa*)*, are often relicts of former cattle pens (Young et al. 1995). Elders reliably reported their locations in 1987, confirmed with 1977 aerial photos and our 1987 vegetation classification (appearing as classes 28, 24, and 20 in 1987, Table 5.1). Some of these areas were said in interviews to have had recent decreases in density of perennial grasses and changes in species composition (Appendix H). Elders indicated that in some of these grazing lawns at Koija there tend to be patterns of loss of grass, increase in bare areas (Figure 5.10), and in some of them, increase in encroaching species such as *A. mellifera* and *S. volkensii*. Considering remotely-sensed changes in areas with historically dense perennial grasses, class 24, referenced areas with dense perennial grass in 1987, and was most correlated with distance to river as a predictor of NDVI (Nagelkerke $r^2 = 0.18$). However, the change in PC1 (Nagelkerke $r^2 = 0.12$) within this class was alternately mostly closely correlated with dry season cattle pressure estimates, suggesting that while the productivity of these areas is very tightly linked to watershed position and soil moisture availability, that cattle impacts also potentially resulted in small changes in this class over time.

We then used detrended correspondence analysis (DCA) to ordinate vegetation composition found in 14 sampling plots in the grazing lawn vegetation, including environmental gradient variables. Wet season cattle pressure estimates had the strongest correlations to axis one of the ordination ($r=0.36$). Wet season small stock was also correlated to axis 3 ($r=-0.26$), while the distance-to-river gradient was most strongly correlated to axis 2 ($r=-0.29$). There were distinct associations with the primary axis that matched our predictions of alternate states of community composition, where increases of *A.mellifera*, *S. volkensii*, and *A. reficiens* were associated with less abundance of *A. tortillis* and *D. milanjiana*, and vice versa. However, none of the individual species that showed a correlation with axis one of $r^2>0.25$ (all listed in Appendix Q), were found to have significant correlations to any predictor variables. Some species did show direct correlations with specific environmental variables in the ordination: *Acacia mellifera* shrub was predicted by elevation ($r^2=0.30$, β = 5.433e-01, SE=2.424e-01, p=0.045), while *D. milanjiana* was predicted by the distance-to-river gradient ($r^2=0.30$, $\beta = 0.55$, SE=0.24, p=0.043). The selected model for *Sansievieria volkensii* density however, had a strong positive correlation ($r^2=0.58$, p= 0.004) with wet season cattle pressure estimates (Figure 5.11a) and elevation, with cattle having the largest contribution to the model ($r^2=0.40$, $\beta = 0.63$, SE=0.22, p=0.015). Finally, proportion of bare ground was strongly correlated with wet season small stock pressure estimates ($r^2=0.51$, $\beta = 0.72$, SE=0.20, p=0.004, Figure 5.11b).

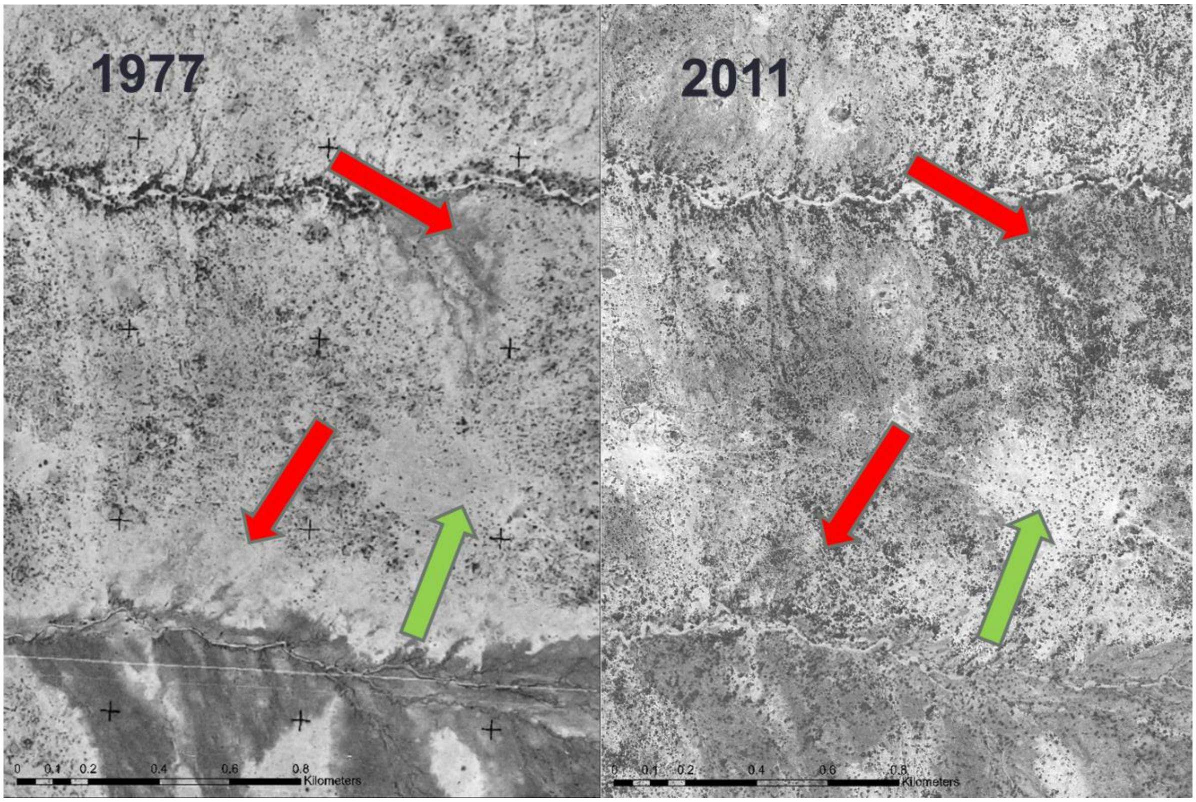

Figure 5.10 Formerly less-densely vegetated hilltop areas where shrub and succulent encroachment by *A. reficiens*, *A. mellifera,* and *S. volkensii* has occurred (red arrows), and a grazing lawn (green arrows) where *Acacia tortillis* trees have established and grass cover has decreased sharply.

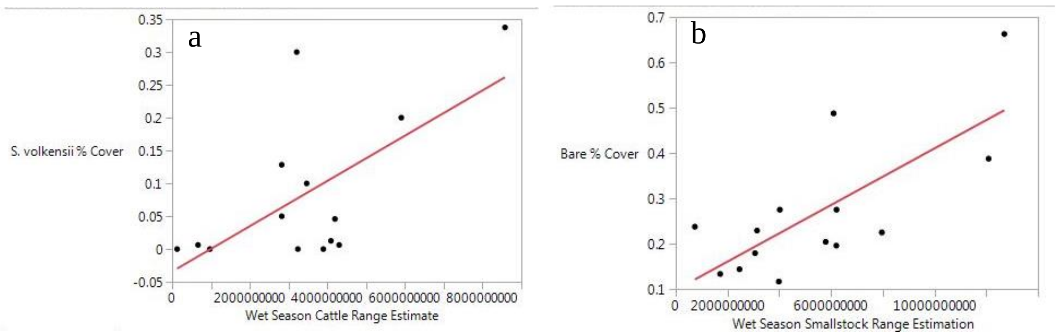

Figure 5.11. (a) *Sansevieria volkensii* plot percent cover as predicted by wet season cattle pressure estimates, and (b) bare ground % cover as predicted by small stock pressure estimates in historic grazing lawn areas.

2. Changes in open-canopied hilltops

Open areas with shrubs and vines or patchy distributions of *Acacia tortilis* (Forssk.) Hayne and herbaceous vegetation were indicated to have recently experienced a gradual decline of abundance in perennial grasses, or to have experienced encroachment of *A. mellifera* shrubs, *Sansevieria volenskii*, and other non-palatable sub-canopy species. Interviewees in this study likewise reported that some areas at Koija that formerly had an open canopy structure and dense perennial grass understories, have now transitioned to dense encroachment of *A. mellifera and S. volkensii*. Elders indicated *Acacia mellifera* has frequently established with it, creating very dense patches of forest with *S. volkensii* understory. While these areas support increased infiltration, and *S. volkensii* are not eaten by livestock, they are generally viewed as unfavorable for herding, with potentially negative livelihood impacts, though they are also associated with higher plant biodiversity (King et al. 2012).

In regression analysis of the remotely-sensed change metrics between 1987 and 2013, Classes 8, 16, and 18, which in 1987 were all shrub/grass vegetation (Table 5.1) and typically occurred on hilltops (Figure 5.12), showed an increase in NDVI since 1987 that was most strongly predicted by the distance-to-river gradient, with R-squared values of 0.21, 0.19, and 0.11 respectively. These classes were fit by models that included other variables in the regressions, but individual regressions indicated that distance-to-river was the most important predictor for all classes (Appendix T). Examining these changes according to the vegetation class transition probabilities, we noted that this increase in NDVI likely reflected an increase in *A. mellifera* for classes 8 and 16 (Table 5.1). Those classes frequently transitioned to class 29 in 2013 (a class dominated by *A. mellifera,* Table 5.2), and the transition was associated with increases in NDVI (Figure 5.9, often coded as blue). PC1 generally showed a similar trend and similar correlations for these same classes, with the exception of class 16 (Table 5.1), where individual regressions including both distance-to-river and wet season cattle pressure estimates had nearly equal contributions in predicting changes in PC1 (Appendix T).

DCA analysis of hilltop vegetation plots yielded weak correlations between vegetation and landscape variables ($r < 0.093$ for distance-to-river, slope, and both cattle and small stock wet season livestock pressure estimates. However, the second axis of the DCA was correlated to dry season cattle pressure estimates ($r^2 = 0.55$), dry season small stock pressure estimates ($r^2=0.399$), and elevation ($r^2=0.550$). We then analyzed individual plant species correlations with the axes, and performed multiple regression on all individual species with r-square values above 0.2, if they were shown to not have a majority of zero values, and could be adjusted for any heteroscedasticity or autocorrelation, if detected in the model (Appendix S).

Considering these individual species correlations, the best fitting model for *Acacia etbaica* included elevation as the only predictor variable ($r^2 = 0.20$, $\beta = 0.45$, SE = 0.08, p = 0.016). Similarly, the best fitting model for *A. tortillis* was nearly significant with elevation included as the sole predictor variable ($r^2 = 0.13$, $\beta = -0.36$, SE=0.18, p=0.059). *Pennisetum stramineum* was similarly predicted by elevation ($r^2 = 0.235$, $\beta = 0.48$, SE=0.17, p=0.009). *Cyperus spp.* was also predicted by elevation ($r^2 = 0.28$, $\beta = 0.53$, SE=1.667e-01, p=0.004). *Sansevieria volkensii* was strongly predicted by elevation ($\beta = 0.46$, SE = 0.14) and distance-to-river ($\beta = 0.47$, SE=0.46) with an r^2 value of 0.54 ($p < 0.001$). Elevation by itself had an r^2 of 0.339 ($\beta = 0.58$, SE=0.16, p=0.001) while distance-to-river had a comparable value of 0.35 ($\beta = 0.59$, SE=0.15, p=0.001). *Solanum incanum* was significantly predicted by cattle pressure estimates ($\beta = 0.50$, SE=0.17, r^2=0.25, p=0.007). Finally, and higher amounts of *Tragus berteronianus* was predicted by cattle range estimate gradients, with an r^2 of 0.249 ($\beta = 0.50$, SE=0.17, p=0.007, Figure 5.13). This small, unpalatable annual grass is typically found on bare, compacted soils, where very few other species can grow.

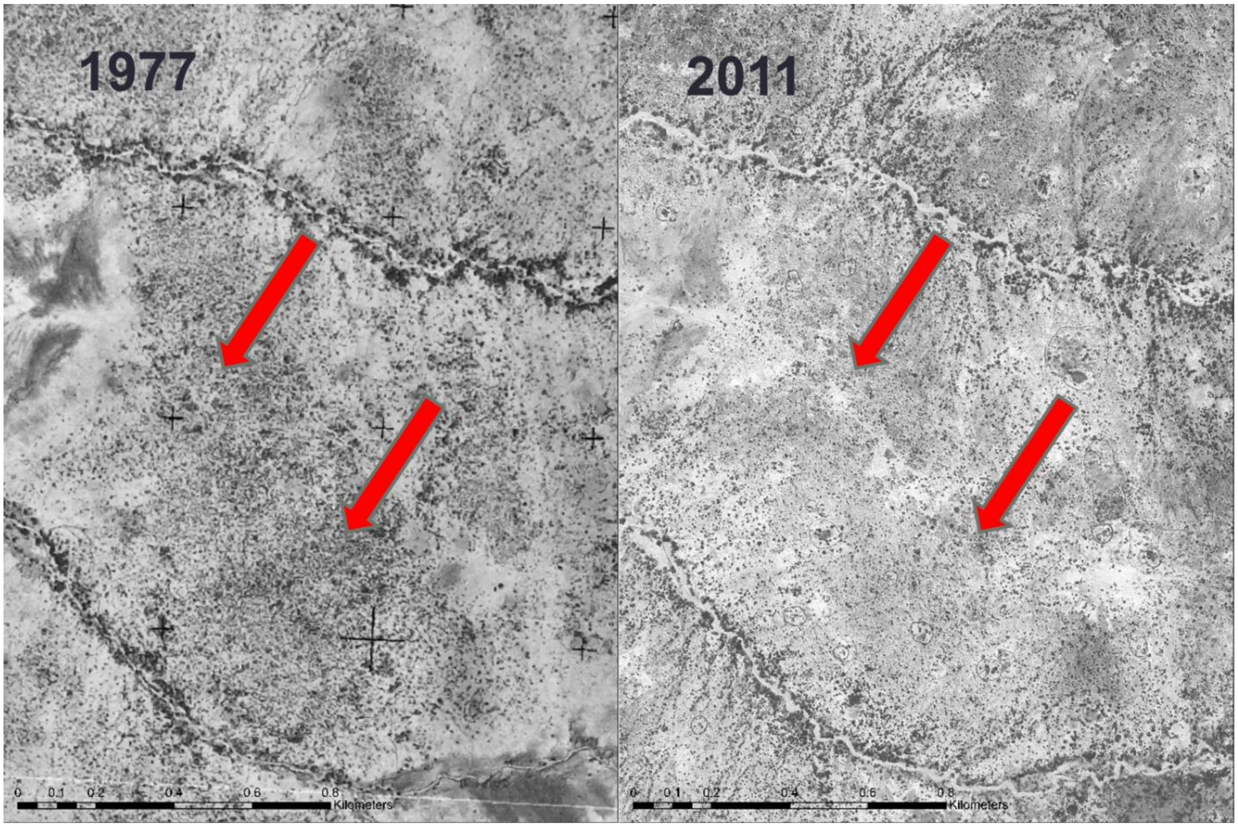

Figure 5.12. Area which formerly supported a *Euphorbia bussei* canopy, dense shrub cover, and vine species (arrows on left, 1977) that today has much more sparse vegetation and frequent bare areas (arrows on right, 2013)

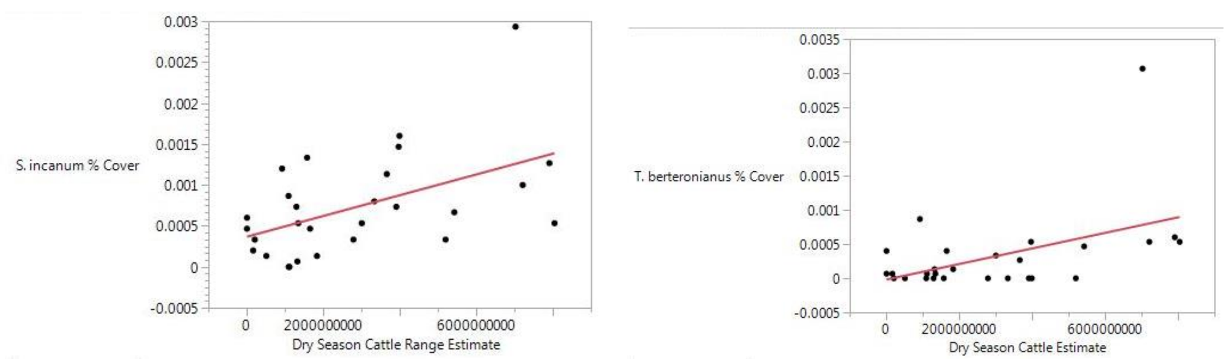

Figure 5.13. *Solanum incanum* and *Tragus berteronianus* % cover in hilltop plots, predicted by dry season cattle range estimates.

Finally, when only considering transitions to encroaching vegetation, by controlling for the state in 2013 according to classification in three classes with high percentages of three encroaching species of focus: *A. reficiens* (2013 class 23), *A. mellifera* (2013 class 29), and *S. volkensii* (2013 class 20, Table 5.2). Areas with high *A. reficiens* presence were fitted with a model for PC1 that indicated an increase in presence along rainfall gradients and wet seasons small stock pressure estimates (r^2=0.32, p<0.001), with a non-significant correlation to small stock when considered individually ($r^{2=}$0.00, p=0.40), and a strong correlation to rainfall gradients (r^2=0.30, p=< 0.0001). Change in NDVI had a lower correlation with distance to river and wet season small stock pressure estimates (r^2=0.203, p<0.001), and while correlated with distance to river (r^2=0.18, p<0.001), were not independently significant for small stock (r^2=0.01, p=0.13). Areas with dense canopies of *A. mellifera* in 2013 had a significant, robust correlation between increase in NDVI and the rainfall gradient (Nagelkerke r^2=0.378, β = -0.60, p<0.0001, p=0.025). There was also a similar correlation between PC1 and distance to river (Nagelkerke r^2 $^=$ 0.295, p=0.031) that included slope as a non-significant effect in the model (β = -0.07, p=0.261), that when removed yielded only a marginal decrease in the correlation (Nagelkerke r^2 = 0.291, p<0.001). In areas dominated by *S. volkensii* in 2013 there was a trend of increasing NDVI associated with areas of higher elevation (r^2=0.32, p < 0.001), and a different response when considering PC1 (β = 0.47, r^2=0.30, p< 0.001) that was most closely associated with the rainfall ($r^{2=}$0.27, p<0.001) and cattle (r^2=0.11, p<0.001) gradients.

3. Formerly vegetated hillslopes that currently are in a bare state (ldoroto).

According to elders, a number of hillslopes throughout Koija group ranch have experienced a recent loss of canopy species, shrubs, vines, grasses, and other herbaceous species. These areas in 1987 were formerly covered with dense shrubs and trees that were likely a mix of

Euphorbia spp., including E. bussei canopies, and according to elders, supporting shrub, grass, vine, and forb understories (Figure 5.12). *E. bussei* is a large, single-stemmed tree-like succulent plant with a candelabra-shaped canopy. Other *Euphorbia spp.*, such as *E. heterochroma*, were also located in the understories and areas with dense shrubs associated with these hillslopes. Like *E. bussei*, *E. heterochroma* is spiny, unpalatable, and toxic to cattle and small stock, but this species has a shrubby growth habit, usually 1.5 to 2.5 m tall. *Euphorbia bussei* formerly provided a semi-open canopy in many sites, but these hillslopes are today characterized by frequent bare areas, with sparse *S. volkensii* and *Sansevieria robusta* succulents, sparse *A. mellifera* and *A. tortillis* tree and shrubs, and a number of other woody species that are largely unpalatable to livestock. In our vegetation type classification, we were able to identify four areas that were characterized by *E. bussei* canopy cover in 1987 -- Classes 3, 9, 19, and 33 (Table 5.1) – but differed in understory and shrub composition. We analyzed changes in community composition and vegetation indices for those classes from 1987 to 2013.

In 1987 Classes 33 and 3, (Table 5.1) changes in productivity (NDVI) were relatively strongly correlated to environmental variables. NDVI change in Class 33 was correlated to distance-to-river (r^2=0.29), while PC1 was predicted by dry season cattle (r^2=0.29). Class 3 NDVI decreases and change in PC1 were both correlated with dry season cattle pressure estimates (r^2= 0.23 and r^2= 0.23). While it is difficult to interpret the exact mechanism relating to the change in composition detected, the correlations imply that the changes are related to livestock. Though there is a possibility that cattle impacted the roots of *E. bussei*, these detected correlations most likely indicate impacts on understory species. A combination of ethnographic evidence and our own experience at Koija indicates that this correlation more likely implies the cumulative effects of cattle on the exposed understory after *E. bussei* had collapsed. The impacts of cattle were never indicated in interviews as a cause of the loss of canopy, and the large area with all individuals in the canopy being lost evenly across the landscape and none present today, would not explain the correlation with gradient of estimated cattle pressure. Elephants are instead the most frequently cited cause of declines of *E. bussei*, and this is supported by their presence within fenced areas that cattle are grazing today, but the fences exclude elephants. The correlation detected therefore indicates to us that cattle grazing, potentially on an exposed, sensitive understory after elephants had eliminated the canopy, led to this relatively strong correlation between the gradient of vegetation change and the gradient of estimated cattle pressure being observed within these two classes.

Considering remotely-sensed analysis of vegetation types that were frequently located on hillslopes and classified as this community, in pixels classified as class 7 in 1987 (Table 5.1), which referred to areas of very dense shrub and *E. bussei*, there was a correlation between decreases in NDVI and wet season small stock pressure estimates (r^2=0.21). Change in PC1 was also correlated to dry season small stock pressure estimates ($r^{2=}$0.24), implying that goats had an impact on the shrubby understory and vines that it likely supported, based upon elder accounts. Decreases in NDVI (r^2=0.16) and changes in PC1 (r^2=0.17) were both also correlated with wet season small stock pressure estimates in class 26, another class associated with *Euphorbia spp.* understory, implying this decrease in productivity also represents goat impacts on woody vegetation and vines in these areas. Class 36, a mix of grass, forb, and shrub, change in NDVI was also correlated with dry season cattle pressure estimates and elevation (r^2=0.15), while these factors as well as slope were included in the best predictor of PC1 ($r^{2=}$0.22), with the strongest predictor being cattle for both NDVI and PC1, implying that cattle had an impact on the herbaceous vegetation in this community type during the dry season.

4. Loss of E. tirucalli (pushiruti) canopy (entim)

Euphorbia tirucalli (pushiruti) formerly constituted a very dense canopy near the Ewaso Ng'iro River, that disappeared mysteriously sometime prior to the 1990s (Figure 5.14). Today these areas are either semi-open-canopied, frequently largely dominated by shrubs and dense perennial grasses, or supporting dense, primarily *A. mellifera* forests with dense shrub and vine understories. It is unknown why the canopy species died. With class 10, referring to pixels that formerly had a *E. tirucalli* canopy in 1987, there was a correlation between NDVI and both dry season cattle pressure estimates and elevation, with cattle having a larger contribution to the model (r^2=0.13) and PC1 being most closely correlated with elevation (r^2=0.15). Class 14, which also indicates areas that formerly had *E. tirucalli* canopies, also indicated correlations between NDVI (r^2=0.14) as well as PC1 (r^2=0.1335) and dry season cattle ranges. Similar to the trends observed with *E. bussei*, these changes likely reflect the impacts of livestock on the understory, as the canopy loss across these areas were uniform, and these changes instead likely reflect the impacts of cattle in the context of canopy loss.

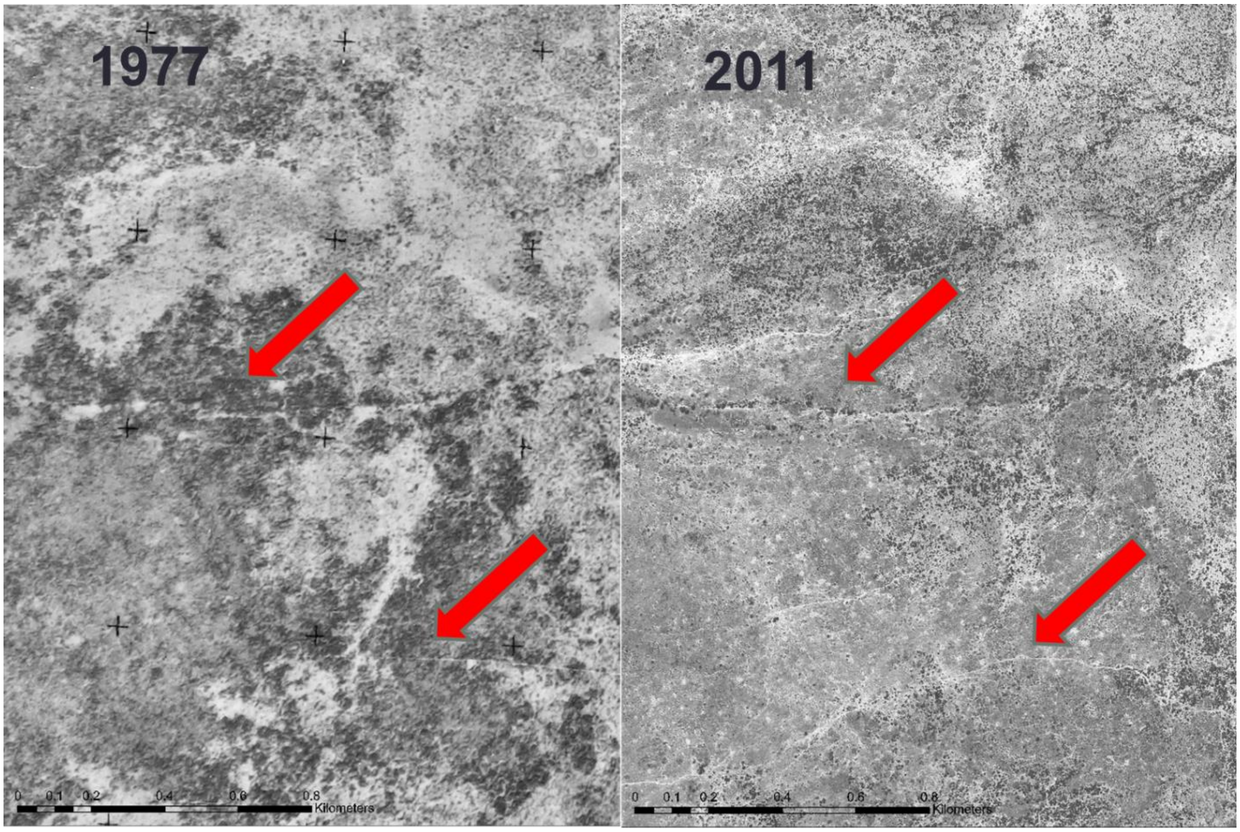

Figure 5.14. Area of formerly very dense *Euphorbia tirucalli* forest (1977, left) today supporting dense perennial grasses with sparse *Acacia spp., Euphorbia heterochroma,* and *Croton dichogamus* shrub cover (2013, right).

5. Seasonal streams (oreyiet)

Areas along seasonal streams which formerly had large amounts of perennial grass throughout them but have been altered greatly due to sand deposition. The canopies are not thought to have been greatly altered, but evidence of elephant damage is not uncommon. NDVI within Class 35, referring to dense perennial grass with *Acacia spp.* canopies in lagga (seasonal stream) bottoms, was correlated to distance-to-river and wet season cattle range to predict decreased NDVI (Nagelkerke pseudo-r^2=0.12), and a correlation between PC1 and distance-to-river, slope, and cattle all together (r^2=0.211), with cattle by itself having the largest correlation when considered individually (Appendix T). Within Class 12, also indicating past areas with dense trees in lagga, there was also a correlation between dry season cattle and change in PC1 (r^2=0.17). These results imply that cattle, which rely on these seasonal streams heavily during the dry season, according to elders, maybe be having impacts on the productivity these seasonally areas during dry season times when outside access is limited (Chapter 2).

Discussion

Widespread changes in vegetation have occurred across the landscape at Koija over the study period, to the extent that only grazing lawns, some other areas of dense perennial grasses, and bare areas have remained in a similar state to what was present in 1987. The most conspicuous changes that have occurred at Koija historically included the loss of two canopy species that dominated a great deal of the extent of land cover, and the encroachment of two shrub species that become extremely predominant. Recent ecological changes are often assumed to be related to the large changes in livestock husbandry that have occurred over the period of study. However, we found little evidence that livestock played a role in the main changes that have occurred in canopy species. Out of 37 landcover types considered in 1987, only 12 of these outcomes indicated a correlation between livestock and the response variable above a relatively generous cutoff point (r^2 = 0.10), and our study did not provide evidence that any of the numerous changes in loss of canopy species, as well as the establishment of novel canopy species, *A. mellifera* and *A. reficiens* were correlated with livestock. However, we did find evidence that small stock likely have had impacts on shrub and vine layers and perennial grasses, while cattle have likely impacted the understories that were exposed when canopy losses occurred. Additionally, cattle grazing intensity was correlated with the establishment of two encroaching understory species, and small stock densities may have played a role in the reduction of productivity. Finally, cattle have likely had negative impacts on perennial grasses that are sensitive to grazing and trampling during dry seasons. We emphasize that while this study does not rule out the role of livestock, it implies that there are likely much more complicated drivers of the overall vegetation changes, such as the cascading impacts of canopy loss, as well as multiple other factors that have worked in addition to, or perhaps even independently of livestock.

Regarding the four main categories of transitions that we analyzed, here we discuss the detailed observed transitions and the most likely explanatory factors, given our ethnographic and rangeland ecological understanding of the system:

1. Changes in grazing lawns (murua)

Grazing lawns are perhaps the most stable vegetation communities at Koija. Through the concentration of resources in former livestock enclosures, the increased nutrient concentrations in grazing lawns cascades and supports unique plant and animal communities (Young et al. 1995). These areas support perennial grasses that in turn increase infiltration of water and retain nutrients at the local scale. At the landscape scale, they also prevent water runoff during rainfall events, and lead to decreased erosion in areas that are positioned below them on hillslopes. These communities can persist for long periods of time, over 100 years, after removal of livestock enclosures. These areas often support highly nutritious forage, and also are frequently open due to manual bush clearing during homestead construction, providing areas where predators are more visible, leading to increased time spent by grazers in these areas, and subsequent nutrient deposition and continued maintenance of grazing lawns. These feedbacks likely lead to a greater stability of these areas over time relative to other vegetation types, however, our study indicated that livestock pressure might also influence the composition of these communities.

These communities generally did not appear to become unproductive over time, though there was a small change in NDVI that was predicted by distance to river, and change in PC1 predicated by cattle pressure estimates. A correlation between bare patches and small stock estimates in plant community analysis also implies that sheep and goats may be having detrimental impacts on the productivity of these communities, either damaging them through grazing or through trampling, both of which were suggested by elders to have more impact than the impacts of cattle grazing. Finally, a relative strong correlation between grazing lawns and cattle pressure estimates indicated that herbivory is potentially conferring some competitive advantage to encroaching succulent species, and that the historical impacts of cattle might have played an indirect, facilitative role in the establishment of *S. volkensii* within glade communities, perhaps through grazing over time shifting dominance away from palatable perennial grasses to these succulent species.

2. Changes in hilltops (ongatta)

Some hilltop areas showed patterns of transitioning from grass dominated communities to ones dominated by *S. volkensii and A. mellifera.* This however, was most closely predicted by a combination of increases of transitions to the landcover class in areas with placement along the gradient to the river, though one class showed an equal correlation to cattle pressure estimates. Areas where the NDVI signature conspicuously increased on the eastern side of Koija corresponded with *S. volkensii*. Thus while *S. volkensii* was predicted to be higher in density in areas with higher estimated livestock pressure in grazing lawns, when gradient analysis was applied to individual species in hilltop vegetation plots, similar patterns were not observed.

While higher densities of *A. mellifera* and *S. volkensii* appear to be unable to establish as readily in grazing lawns, which also also typically located nearby on hilltops, these two species have readily established in the less nutrient enriched areas near to grazing lawns. We found little evidence of livestock composition driving the observed increase of *A. mellifera*, or playing a primary role in facilitating *S. volkensii* establishment in these hilltop communities. Analysis of change indices, including only areas that have experienced intense encroachment by certain species, indicated that encroaching species mostly are establishing in more productive areas along a gradient where productivity increases. This strong indication that their productivity changes along the gradient to the river, implying limitation by soil moisture, indicates that in the past these species were perhaps regulated by a widespread process, such as fire, or that another more evenly distributed factor has been altered and is aiding their establishment.

Our results do not indicate what factors may be driving shrub encroachment, whether fire suppression, changing climate, or herbivore interactions that are shifting competitive advantage over grasses. However, a number of mechanisms have been proposed to explain shrub encroachment (e.g. D'odorico et al. 2012, Stevens et al. 2017), that include fire suppression (van Langevelde et al. 2003), climate, loss of wild grazers (Sankaran et al. 2013), and indirect facilitation by domestic herbivore pressure (D'odorico et al. 2012). Though the encroaching shrub species *A. mellifera* may be beneficial as forage for goats in previously undocumented ways (Chapter 2), our analysis of community shifts and composition indicates that there are large accompanying changes in species diversity and decreases in the abundance of perennial grasses where *Acacia mellifera* has become dominant. Finally, DCA axes indicating plant composition, as well as increased densities of *Solanum incanum* and *Tragus berteronianus* were all correlated with cattle pressure estimates, indicating that though the main encroaching species are not related to livestock, there are other changes in species composition, for example those expected by Pringle et al. (2014), which are related to cattle grazing in these hilltop areas.

3. Formerly vegetated hillslopes that currently are in a bare state (ldoroto)

The strongest correlations of small stock density and vegetation change were seen in vegetation classes that were formerly dense, shrubby or forested vegetation. This aligns with oral histories of large decreases in shrubs and vines, which in personal experience are much more abundant in areas that have experienced restricted goat access in recent years. These robust trends imply decreases in shrubby vegetation and vines largely related to increases in goats and their known affinity for vine species. On the other hand, cattle densities, especially in dry season ranges, tended to be correlated with changes in vegetation indices toward lower productivity in areas with former *E. bussei* canopies. However, the correlations are likely due to cattle impacts on the understory, not directly on canopy cover. Elephants, who topple the mature trees, were said by elders to have large impacts on the canopies of *E. bussei*. While these changes due to elephants were not explicitly considered in our study model, these seem most likely to explain the initial loss of canopy, perhaps with cascading impacts on understory conditions. These feedbacks could have occurred by potentially increasing soil temperatures, reducing soil moisture, and creating less stable soils, as large changes in ecosystem characteristics have been documented with canopy loss (e.g. see D'odorico et al. 2012). However, there was also correlation of small stock pressure estimates with changes in these vegetation types, which may also have impacted this understory vegetation, and this is also an area where some elders indicated that sheep and goats may be recently impacting herbaceous vegetation (Appendix H).

Finally, the barest areas on Koija were already bare in 1987, and coincide with areas that have very high slope. These hillslopes tend to have extremely hard surface texture, indicating greatly reduced infiltration and increased runoff, also leading to large amounts of soil and nutrient loss and decreases in overall species diversity.

4. Loss of E. tirucalli canopy (entim)

E. tirucalli disappeared suddenly, in a dramatic extirpation of canopy cover, according to elder accounts. Because of the uniform nature of this loss of canopy, our methods would be unable to detect any gradient associated with this change. However, while *E. tirucalli* is a favorable goat forage in other areas, and the large number of goats on Koija could have prevented its regeneration, as indicated by elders (Appendix H), it seems unlikely that goats could have themselves caused complete loss of this canopy species, given the size of the trees that were present in the past, and their local abundance in other nearby areas with higher moisture. Potential causes of canopy loss most likely include drought and toppling by elephants, though these were suggested only infrequently by elders as drivers (See Appendix H). Cattle are also an unlikely candidate for a cause of canopy loss, and similar to the correlations observed between *E. bussei* and cattle, likely imply a heightened sensitivity occurred due to the cascading feedbacks caused by canopy loss.

5. Lagga (seasonal stream) changes

Changes in lagga vegetation was correlated with estimated cattle pressure, which is not surprising as these areas are relied on heavily during the wet season as they are near to homesteads, have high soil moisture, and support dense perennial grass in the dry season for cattle and sheep that were indicated as a preferred grazing site (Appendix H). Other changes, however, that would not have been detected by our landscape variables, include sand deposition, are thought to have dramatically influenced these communities as well.

Summary of impacts of livestock

Koija is under intense livestock pressure that has probably been historically unprecedented, prior to the 1980s. Though there is evidence that biomass of livestock owned by Koija residents is historically comparable in the 1980s (Chapter 2), at that time there was still frequent migration outside of Koija. From the mid-1980s on, as documented by others (Herren 1991), there have been drastic changes in the composition of livestock, and seasonal ranges have become increasingly constrained. Today there are channelized pathways and a largely denuded landscape surrounding homesteads, readily apparent in aerial photos. Estimated cattle grazing pressure was relatively strongly correlated with changes in vegetation indices from remotely sensed images within 1987 classes formerly dominated by *E. bussei*, forest or forest edge, shrubs, shrub/grass, and within laggas. Notably, however, the majority of these impacts seen in decreases in NDVI correlated with cattle occurred in areas of dry season grazing pressure, with only two classes' response variables predicted by wet season pressure. During dry seasons in the past, the young men would have typically left Koija with the cattle to find pasture elsewhere (Ch 2). However, today, a large percentage of the cattle remain in a state of precarity on Koija (Ch 2 and 3) at a time when the herbaceous vegetation has already been grazed and is extremely sensitive to trampling. The complex ways that drought and herbivory interact require more detailed study, especially to develop a better mechanistic understanding of these potential pinch-points where domestic herbivore pressures might be magnified. Additionally, the landscape-scale patterns of goat herbivore pressure were correlated with areas of loss of shrub and vine vegetation detected in 1987.

While there is evidence for some changes driven by livestock, and our results do not rule out the impacts of livestock by any means, our results stand in contrast to the explanations of vegetation change that are frequently referenced in the conservation and development grey literature we mentioned previously (e.g. Alexovitch et al. 2012, Fennessy 2009, Lent et al. 2002, NAREDA 2004, Sumba et al. 2007), but also in our own experience of discussions at regional meetings on wildlife conservation. In these explanations, the management of group ranches, human population, and stocking rates are cited as the main factors driving vegetation change. However, we found little evidence that the most widespread changes in vegetation, such as loss of canopy species, or shrub encroachment, which both have sweeping implications for vegetation communities, were correlated to our estimation of recent livestock pressure. Livestock are clearly only a part of the explanation and do not appear to be associated with the gross changes often attributed to livestock management. While elders do acknowledge the impacts of livestock on ecological change (Appendix H), the alternate explanations given, that drought has been one of the dominant factors driving landscape vegetation changes, and that elephants have had large impacts on canopy species, should also be considered as an equally plausible and important factor in designing future studies, given our results. Frequent droughts are the most conspicuous "pulse" events that occur frequently at Koija, potentially resulting in the past death of canopy trees, such as *E. tirucalli*. There are also climate "press" events at Koija, where the conditions of increasingly infrequent rainfall over time leave little time for forage to recover following these infrequent rains before livestock return to these sites, leading to a state that resembles a "permanent drought" (Chapter 2). The complex interactions of drought and herbivory have been emphasized by others (Hodgkinson 1995) but are relatively under-studied. A further potential interaction is that heavy rainfall events represent an additional "pulse" event, especially following long periods of low rainfall, causing recent erosion of soils and even loss of trees (pers obs.). This could have a strong impact on landscapes, especially when coinciding with the end of long droughts, where there is little vegetation to retain topsoil. Other changes in landscape process that we were not able to consider include changes in the fire regime, known to shape herbaceous species composition, reduce woody vegetation, and potentially play a role in shrub encroachment. Finally, the impacts of wild and domestic herbivores can produce browsing pressure on trees (Sankaran et al. 2013). Elephants in particular are known to frequently kill individual trees (Kimuyu et al. 2014, Western and Maitumo 2004). These canopy changes then could result in decreases in moisture in the understory, increased soil erosion, or alternately, an influx of nutrients into soils through decay. Future consideration should be given to the cascading impacts of canopy loss, with some of the currently least vegetated areas on Koija coinciding with areas that formerly had *E. bussei* and *E. tirucalli* canopies.

In designing this study, we recognized a hierarchy of additional factors that could potentially provide explanatory mechanisms for transitions between vegetation states in different sites at Koija (Bestelmeyer et al. 2017). These include climate, fire, wild herbivores, increased CO_2 levels and nitrogen deposition. This list of factors indicates that numerous studies are needed to find support among these potential alternative hypotheses. For this initial study we chose to focus on livestock herbivory, hypothesizing that novel spatial concentrations of livestock herbivory, which are embedded in socio-ecological change (Chapter 2), could explain some of the differential outcomes in vegetation between 1987 and 2013. While we have not assessed the relative impacts of drought (Huho et al. 2009, Franz et al. 2010), fire suppression (Augustine 2003), loss of browsing mammals (Sankaran et al. 2013), or cascading impacts from canopy loss (D'odorico et al. 2007), we did find that considering the potential role such ecological factors alongside estimated landscape-wide patterns of livestock herbivory (Chapter 4) contributed new ecologically and ethnographically supported insights into the complex trajectories of reduced biomass and shifts in species composition that have occurred on the landscape between 1987 and 2013.

In discussion of the methodology, we found that commonly used unsupervised classification approaches were unable to accurately capture the distinctions between the highly heterogeneous dryland vegetation types in our study system, leading to improper identification and confusion of a number of classes. This led, out of necessity, to two methodological developments that could potentially aid in the study of land-cover change in general, with specific application in historical vegetation analysis in complex landscapes. By using a classification-fitting technique to determine the most accurate unsupervised classification technique to use, we found that using supervised classification and hierarchical k-means unsupervised classification together can lead to highly replicable classifications that can be customized to the needs of a given project while maintaining the ability of the supervised classification analyst to cater the classifications to changes of interest. Secondly, combining categorical and continuous change detection methods aligned our methods with a continuous understanding of change, while working within a state and transition framing. Building upon this approach led to a more nuanced understanding of the direction and magnitude vegetation changes while controlling for abiotic context and historical contingency of vegetation than could have been achieved using pre-classification, post-classification, or continuous methods alone. Finally, it led to an ability to directly compare continuous metrics of change to continuous abiotic and biotic gradients.

While our study lacked the precision of long term manipulative experiments, it provides an initial understanding of the complexity of the broad-scale transitions that have occurred in Laikipia. An improved approach would benefit from added temporal frequency and analysis of phenology. One problem that could potentially be adjusted for in future analyses would be to adjust for multi-collinearity in livestock layers, which prevented analysis of the cumulative effects of individual species. Considering the large numbers of small stock today, and conspicuous overlap in pathways used by the different species, there are likely multiple overlapping impacts of the different livestock species. Additionally, detected changes were often very similar among classes when comparing PC1 and NDVI, and other continuous vegetation change metrics could be explored to gain a better understanding of changes in composition.

Conclusion

This study developed an integrative approach for analyzing the complexity of shifts in vegetation states over time in an ecologically heterogeneous pastoralist commons. To examine vegetation transitions with respect to environmental and herbivory gradients, we used a mixed-methods, multi-scalar approach that employed remote sensing and plant community ecology methods. Using a combination of Landsat images, high resolution Quickbird and Pleiades images, one aerial photograph from 1977, ethnographic descriptions of past changes, and our own knowledge of the area, were able to reconstruct an initial understanding of the nuanced trajectories of changes in the vegetation composition and productivity across Koija between 1987 and 2013. Though we were unable to analyze all underlying environmental variables, by using proxies for watershed position, soil moisture, soil types, and slope, we were able to control for how vegetation shifts and responses to herbivory are context-dependent. Our approach also allowed us to use continuous analysis of change while controlling for past vegetation states. This analysis indicates that transitions to different vegetation states have occurred across the majority of Koija. Only two types of vegetation did not appear to experience transitions to a different state over time: bare lands and grazing lawns. Taken together, the results indicate large implications for landscape-level process, herder livelihoods, and wildlife conservation.

This approach used remotely-sensed and plot-based data simultaneously to compare the outcomes in different landscape contexts, with different types of dependent variables (fine composition, pixel composition, pixel productivity). We used a state and transition framing and mixed methods that enabled triangulation between different states, but primarily relied on continuous metrics of change trajectories while controlling for different historical states at the landscape scale. Oral histories (Appendix H) and detailed on-the-ground knowledge of vegetation types enabled separation of specific types of current and historical vegetation using unsupervised classification, supervised classification, or continuous vegetation change indices alone.

Finally, elder oral histories of vegetation changes (Appendix H) were invaluable for our understanding of vegetation changes as they relate to herbivory, and situated these ecological changes within a complex social landscape, improving both the credibility of our findings and our understanding of the relevance of changes for livelihoods. This is crucial to consider, as recent changes in domestic herbivore pressure in Laikipia are situated within, and shaped by, the wider political and economic context of changing land use. Rather than assuming changes are solely due to uniform impacts of human population growth or livestock stocking rates, our results suggest that these changes are better understood as context-dependent and driven by multiple historical processes. Specifically, the novel landscape pressures that have occurred have been shaped by external political and economic changes that have created constraints on livelihoods (Chapter 2). This, together with herders' adaptations to these constraints (Chapter 3), have resulted in cattle being concentrated locally, as well as increasing densities of small stock (Chapter 2). This context of Koija with its complex history, and highly heterogeneous vegetation, create a high potential for misinterpretation of interrelated changes (Velasquez Runk et al. 2010, Robbins 2003, Turner 2003). The multiple framings of changes in landscape vegetation, along with the insights drawn from previous ethnographic work, led to an improved ability to critically and reflexively examine hypotheses about land cover change at the landscape scale. Such shifts in approaches are crucial in developing more pluralistic inquiries into coupled human and natural systems (Olsson 2015).

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CHAPTER 6

CONCLUSIONS

This dissertation was shaped by the integrative use of social ecological systems framings and political ecology side by side, as laid out in chapter 1. Followed by four interdisciplinary analytical chapters, this approach led to a number of novel insights and a more nuanced understanding of the recent history of inter-related social and ecological changes. We first documented the specific factors that have impacted the ability of pastoralists to respond to drought and to access forage, and how this access has become not just limited, but uneven over time in a pastoralist group ranch in central Kenya (Chapter 2). This approach revealed an understanding of access as situated in a context of historical loss of land, wider conflicts, and more recent changes in institutions related to wildlife conservation governance (Chapter 2). These latter changes have come about within a context where a focus on maintaining habitat and connectivity of the Laikipia landscape for wildlife is a primary concern of private landowners and NGOs. NGO efforts have focused on creation of conservation areas on spatial scales to manage for strategic wildlife corridors. However, at the same time these actors have not emphasized the importance of the connectivity of herding practices, leading to a tension between livelihoods and wildlife conservation (Chapter 2). To inform policy, by incorporating critical social science elements alongside detailed analyses of livelihoods and herding practices, we reframed the discussion when considering concepts such as adaptive capacity and vulnerability (Chapter 3) to include a consideration of the ultimate causes of vulnerability (Ribot 2014). In the setting of Koija, access and vulnerability are inherently intertwined, and so considering the vulnerability of livelihoods to drought and the individual abilities of herders' responses necessitated first understanding the historical sequence of limitation and restructuring of access (Chapter 2). We considered access as inherently political, considering who can access resources when, and how they access these resources (Ribot and Peluso 2003).

These critical insights also informed a nuanced, context specific approach to asking questions about the causes of vegetation changes. This linked explanation of ecological patterns to unevenness in the social realm, where changes in climate, ecology, and social factors have all likely influenced vegetation change in recent years. Specifically, we explored how wider political and economic factors have impacted herding ecology, concentrating the impacts of certain livestock within pastoralist group ranches. We used a landscape ecology approach and least cost corridor analysis to create estimates of how this pressure has become concentrated within one group ranch (Chapter 4), and then used a state and transition framing and to analyze the complex, context-specific vegetation changes that we observed (Chapter 5). Using these landscape approaches was then conducive to using remote sensing methods as well as vegetation plots across the landscape to understand past changes and how current patterns seen today relate to those changes.

Livelihoods were also a central concern of how we framed our landscape ecological analyses, and we argue that through understanding how livelihoods are structured, we were able to better inform and develop understandings of landscape processes. This was also greatly informed by an examination of ecological knowledge (in prep). This integrative understanding of landscape change from multiple perspectives led to modes of inquiry that drew attention to my own positionality and the potential to bridge epistemologies between two groups with situated knowledges (Haraway 1988, Goldman 2003) of the world. Further, this pluralistic perspective led to an understanding of vegetation change that would not have been possible from an ecological science perspective alone, leading to the generation of alternate hypotheses about landscape ecological processes and landscape change. Taken together, this provided a novel way to grapple with the study of complex interactions between changes in herding land use and landscape vegetation change. Finally, analyzing social and ecological processes side by side led to improved understanding of the way that changes in vegetation are in turn impacting livelihoods. These insights of access and vulnerability together also deeply informed the framing of the analysis of ecological process, where lack of mobility is exacerbating continuously drought-like conditions by preventing regrowth of grasses, and potentially driving some vegetation changes. In this light, contemporary interventions to attempt to restore rangeland productivity in group ranches once again echoes the past, where management interventions historically failed because they restricted areas internally in order to restore rangelands but did not offer external grazing access (Anderson 2002).

When considering livelihood challenges and vegetation changes, policy documents frequently ignore the existence of pastoralist institutions, and NGOs attribute the decreasing success of pastoralist livelihoods to factors such as "inadequate management" or "decreased forage resources" (Sumba et al. 2007). This leaves out consideration of underlying historical factors of land loss, increasing confinement, and unequal access to resources. Greater attention is required to accommodate the spatial complexity that these relationships take across the landscape to incorporate the ecological and social rationale of pastoralist institutions and customary land use in Laikipia. Further, historical environmental discourses emphasizing overstocking and overpopulation (Turner 1993, Anderson 2002) are commonly still heard in

Laikipia today (McIntosh 2016). Much of the historical dynamics of development initiatives, as well as more recent conservation initiatives, have been shaped by prior scientific discourses about pastoralist property regimes, herding economy, and rangeland ecology. These arguments are often based upon theories of common property regime inadequacy (Hardin 1968), over-accumulation of cattle due to “irrationality”, or overpopulation and subsequent overstocking as primarily driving sustainability challenges (Turner 1993, Nelson 2012). Many Kenyan policies have historically focused on the need to reduce livestock densities, and are tied to an equilibrium understanding of semi-arid ecology (Turner 1993, Roba and Oba 2008), and are compounded by misunderstandings of pastoralist institutions and impacts of livestock on land (McCabe 2003).

There is a marked lack of incorporation of local knowledge in the design and monitoring of conservation endeavors in Laikipia (Yurco 2011), a pattern that exists beyond Laikipia and throughout East Africa (Goldman 2011). Despite a growing body of work on the highly nuanced pastoralist understanding of rangelands (Roba and Oba 2008) and the increasing emphasis of local knowledge in CBNRM principles, there is a conspicuous divide between pastoralists’ knowledge of ecological change and that of conservation and development actors. Chapter 5, while not explicitly studying ecological knowledge, highlights the need for pluralistic understandings of landscape changes to inform analyses of vegetation change. Preliminary results of this analysis (in prep) on pastoralist knowledge which informed these four chapters, indicate that elder herders’ understandings of the ecological change landscapes based upon their lived experience at Koija contrasts sharply with those of conservation and development actors. The previous dissertation chapters were all motivated by a gap in knowledge and livelihoods that had previously been identified as absent from the perspective of researchers on private ranches in Laikipia (DePuy 2011). Further, as privately-owned ranches are located in areas with higher, more consistent rainfall (Franz et al. 2010), and have recently been managed with much lower livestock densities compared to pastoralist group ranches (Georgiadis et al. 2007) with due to historical appropriation of these large ranches, these lands do not experience the same social-ecological constraints that livestock husbandry does on Koija (Chapter 1). The production of knowledge about neighboring pastoralists and rangeland practices is shaped by these factors, as well as by a technical, distanced standpoint that accompanies being a hub for international researchers (DePuy 2011).

In closing, land management on pastoralist group ranches is frequently portrayed as resting solely upon how that land is utilized within boundaries of land held by collective title. This has a long line of reasoning leading back to historical colonial authority, and to the view widely held in the post-independence era that private tenure is necessary to prevent land degradation, in line with views such as Garret Hardin's (1968). In the case of Laikipia, the need for improved management within these boundaries is frequently emphasized in conservation and development grey literature (Chapter 5). These management recommendations usually involve 1. designation of an area that is dedicated as a wildlife conservancy, 2. introduction of a form of rotational grazing, and 3. introduction of exotic breeds of livestock. However, in the case of Laikipia, consideration of management solely within the confines of group ranch boundaries overlooks the very recent historical changes that have occurred within rangeland access and livestock husbandry practices there. As we showed elsewhere (Chapter 3), livelihood practices experience extreme constraints, especially during drought. Group ranches are the end product of externally initiated processes including: colonial dispossession, forced relocation onto reserves, privatization of and subsequent exclusion from formerly open lands, conflict with other pastoralist groups, and most recently, the establishment of conservancies (Chapter 1). In this context, speaking of management and local practices alone implies that changes in vegetation within Koija are due solely to the management of that collectively titled land. While we are confident that livestock practices have had impacts on certain types of vegetation (Chapter 5), there is a danger of a slippage to a mode of "intervention" in the name of conservation, that results in the demonization of pastoralist land use practices as was common throughout the colonial era (Anderson 2002).

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APPENDIX A

2013 FOCUS GROUP DISCUSSION SUMMARY

The following document was compiled following my first summer of research at Koija, and drew from focus group discussions conducted with elders from households across Koija. It was summarized to guide development of a survey instrument (Appendices B and C), to determine areas of focus for key-informant interviews (Appendices D and E), and to provide a future reference for general views.

Grazing Practices

It was stated unanimously by focus-group participants that grazing decisions are made by Wazee (elders) within each neighborhood grazing cluster. It was also stated by everyone asked that that the neighborhood distinction is a formality and that individuals are permitted to graze in other neighborhoods. The Wazee are all called together, and then this decision is formalized by the Group Ranch committee (I now am uncertain whether there may be differences in this exact process, and the level of formality, between Koija and Il Motiok). At Koija, Wazee decisions were indicated to be made based primarily upon water levels in ponds. At Koija, these are geological features or depressions in soils created by elephants. At Il Motiok there are now two large dams that have created new watering point concentrations. The dams at Il Motiok were stated to have an impact on grazing practices, but the detail of how this is thought to be occurring is unknown and rules appear to remain based primarily upon the wet and dry season distinctions. In general however, for both group ranches, when this water supply runs low, neighborhoods switch to using watering points at river access points and utilizing a grazing area that is restricted during other times of the year. (However, I observed that when this switch occurs in practice appears to vary by household, and this appears to be correlated with herd size and composition).

At Il Motiok and Koija group ranch, a rotational grazing practice known as Holistic Management has been put into practice since 2010, but this practice has been abandoned at Koija group ranch. At Il Motiok, elders indicated that this has resulted in the majority of cattle being grazed in a single large herd in specific areas by the ilmurran, while sheep and goats herding remains organized primarily at the household level though within the wet and dry season restriction designations. An additional differing characteristic of Il Motiok's rangeland practices is that once dry season grazing areas are opened, wet season grazing areas are restricted, whereas dry season grazing areas can still be utilized at Koija during this time. It is unknown whether wet season grazing areas were restricted during the dry season by customary authority in the past.

According to participants, the beginnings of different seasons of rain are indicated by the appearance of constellations, and the main rainy seasons are Ingokwa (April/May), Lorikine (July/August), and Tumarin (November). These decisions are/were then communicated by Wazee throughout neighborhoods, and all individuals are/were informed of the areas where they can graze and cannot graze. In the past, as today, visitors to the area would have to seek permission for grazing, and would be informed of designated areas at this time. It was stated by some individuals that rules became flexible during dry periods in the past, and similarly today, if water becomes low certain routes may be opened and the restrictions may become flexible.

Some individuals added that decisions are also based upon the height of grass, but I was unable to gain further detail on how this assessment occurs. A second type of grazing restriction was also discussed where in the past specific areas within each of these may also be set aside to allow grasses to regenerate. Additionally, grasses were formerly burned historically, usually one or two weeks before the rains came in areas that weren't forested, but the scarcity of grass doesn't allow for this anymore. It was stated that this burning helped to control ticks, decreased the shrub cover, and increased the nutrition of the grasses. It was sometimes denied in discussions at Il Motiok that burning was used to manage the land in the past, and was instead stated to have only occurred accidentally.

Changes in timing of grazing decisions in more recent years have been occurring according to all interviewed, and these were primarily attributed to a reduction in rains, where more frequent rains in the past that would keep the ponds filled with water, but other impacts included sedimentation in ponds, the ephemeral streams not holding water in pools as long, and more livestock visiting the ponds today, in turn decreasing the water level more rapidly. Additional reasons stated by some were that elephants and zebras destroy watering points.

It was also stated that in the past there were differences in the timing of grazing because it took longer to graze an area due to the higher density of tree cover leading to greater water retention in the soil. Referring to forage, it was also stated that areas could have been restricted for longer in the past, but factors such as population pressure and increases in livestock have made this difficult, and similarly, areas that are set aside become degraded very quickly after lifting restrictions. In the past other areas could have been restricted again following grazing for a certain period, but today this is impossible (Follow-up questions are needed here).

It seemed to be consensus with most groups that the areas on Koija that were designated as reserve and wet season grazing were flexible prior to the AWF management plan but have not changed in general, with a valley serving as a divider between these two grazing areas. Through later interviews, however, there do seem to have been changes in the locations of residences on Koija , with many families in Nosarai neighborhood having been relocated from one hillside to another following the implementation of the African Wildlife Fund resource management plan (See Sumba et al. 2007 and Muthiani 2011 for detail on this agreement). Similarly, it was stated by some individuals that in the past from time to time all families could move to a completely different hillside nearby if this was deemed appropriate (the basis of how these decisions were made is unknown). At Il Motiok, families formerly lived in the current conservation area. In both ranches, there were a number of watering points in the conservation area that are no longer used (at Il Motiok these include Nenkinki, Nekiki, Nipiren, Noopoilitro, and Nongabishi). Both Group Raches were said to use their conservation areas as a reserve grazing area currently.

It was unanimously stated that prior to group ranch formalization, it was possible for the entire group ranch to move with families to another area, and the example was given that anyone considered Maasai would be able to move to and graze in neighboring Il Motiok. There appeared to be similar consensus that migration with the entire family occurs very rarely today, and some stated that this activity stopped in 1976. Stated reasons cited for no longer migrating with the entire family and manyattas include: children attending school, group ranch subdivision, wage employment, permanent house structures, changes in diet, and a decrease in hunting (numerous times it was stated that in the past it was common for the LeUaso to kill an elephant and to then move to that location).

Two types of migration were said to have occurred in the past: one with a family moving the entire manyatta, and one with the *ilmurran* traveling with livestock only. Formerly during periods when the rains would fail (long dry seasons), *ilmurran* moved with the livestock until they came to an area where rains had fallen. In February and March the *ilmurran* would typically be on *porr* (migration) with the livestock, and if the Tumarin rains failed they would go on *porr* in November too. It was stated that today that many areas are not available for access to grazing (locations to be covered in detail in surveys), for example, grazing is now available in Il Motiok only at a paid rate. It was stated that raiders would be a threat but in the past it was easier and less dangerous for *ilmurran* when they travelled together. All participants agreed that today there is greatly decreased migration together, while in the past everyone's cattle were the responsibility of the whole community, emphasizing that now people only travel together with their close relatives, close friends, or alone. Finally, some individuals stated that changes in herds have led to decreased migration, due to the decreased ability of small stock (goats and sheep) to walk long distances. In explaining this, it was stated that goats in particular have been increasingly favored due to fast reproduction, ease of sale, drought resistance, ease of slaughter, and rapid use of meat (this is in agreement with observations by Urs Herren in the mid 1980's throughout Mukogodo Division).

It was agreed upon by participants that the grazing committee now decides when to take cattle off of the Group Ranch, and last year the cattle were on private ranches from August to Jan at a rate of 350 shillings (~$4.25) per head at privately owned ranches. It was indicated that sheep and goats are not allowed, except for Sukutan, which allows sheep. Camels and goats usually remain on the Group Ranches while some cattle are moved off-site due to their requirement of grass. Some stated that sheep are moved off of the group ranch while others said they are not moved. In extreme drought, all types of animals may be moved off of the group ranch, but it was my understanding that goats will be taken if herding labor availability doesn't allow for separate herds. It was stated by a few individuals that 100 head of cattle per cluster can be sent to Loisaba from Koija (a total of 400 out of thousands), where individually owned cattle are all kept together in large herds where individuals are only permitted to see their animals at certain times. Cattle that are not sent from Koija to Loisaba are allowed to graze in the wildlife conservation areas. It is thought that Mpala primarily provides paid grazing to Il Motiok, but the details of this relationship are unknown.

It was stated by some that labor is still shared by those that migrate to non-paid areas by friends, relatives, and neighbors, though it occurs less than in the past and sharing of labor primarily occurs only between close relatives. Some also stated that this decrease in sharing of labor restricts movements, allowing for only short migrations today. One reason stated for a lack of sharing labor was children being in school, and that people who did not have children in school now expected payment for their children to help others, and that many people are paying herders on private ranches during drought now. Labor sharing has also decreased due to "individualization" where in the past labor sharing was a collective responsibility of the *ilmurran.*

It was stated that some people still send animals with family and friends in other areas, but these former networks are regarded as "risky", and it is considered better to pay for grazing if you are able. The "risk" of this was explained in terms that population has increased and people are becoming poorer, and subsequently there is a lot of mistrust and so livestock may be more at risk when loaned out. While considered much less frequent in comparison to the past, participants generally agreed that animals may still be shared if one family owns multiple cattle and another is in need. Some continue to share animals when one family has more lactating cows, one house will loan another house one to provide milk, and then give them a calf when they return to collect the cow. In the past this debt was repaid, but today reciprocation seems to have decreased. This exchange is primarily thought to occur within manyattas today, and it is much more likely for a family to give grains such as mahindi (corn). These changes in sharing and cooperation were sometimes attributed to monetization, individualism, and the "hard" economy. It was stated that there is a trend of selling animals instead of gifting, and an increasing preference for money over the prior exchange networks.

It was stated by some that the Wazee decisions do not carry the level of legitimacy today that they formerly did, as some individuals in the group ranches accept the rules but others, primarily educated youth, do not. The level of compliance with Wazee decisions was ambiguous, however. Some individuals mentioned people continuing to graze in restricted areas. Common explanations of this behavior were that it could be attributed to individualism, competitiveness, and the increasing population. This point was elaborated on by one family, stating that raising children is no longer being viewed as the responsibility of the community as a whole anymore, and that rather, individual families' values now take precedence.

Sustainability Challenges

Dramatic changes were stated in terms of decreased herd reproduction and decreases in milk yields per animal that were stated to be related to animal forage (type and abundance), drought conditions, and market factors. All individuals who participated in focus group discussions appeared to agree that population and livestock numbers were less in the past and it was easier to restrict areas and reach consensus about decisions. Additionally, there appeared to be agreement that rains have changed and become more variable in recent times. It also appeared to be agreed upon that following the rains, the high livestock numbers exhaust the forage supply much more quickly today. Some specifically stated that goats have adverse effects on grasses compared to cattle.

Several individuals stated that they felt that the conservation areas restrict available grazing and increase pressure on the current grazing areas, and some at Koija (not Il Motiok, likely due to the small area being farmed) felt that farming is also reducing grazing and reducing access to watering points. Some felt that conservation areas should be opened up to increased grazing. This group also stated that a larger grazing area is needed to decrease the pressure on vegetation, stating that Loisaba and Mpala used to be open for grazing without payment. In Koija, when asked what current problems related to grazing arrangements were, some stated that current arrangements resulted in grass being grazed too long before moving. In contrast, when considering lack of movement, a discussion involving the Il Motiok chairman was dominated by the idea that moving animals off group ranch doesn't matter now because of Holistic Management practices and that fact that under this practice, areas are restricted that were not in the past. One group in Il Motiok stated that largest difficulties to herding today are predation of animals, lack of herders, lack of medicine for livestock diseases, and inadequate grazing areas. One group in Koija stated, when asked about problems due to current grazing arrangements, that there is a lack of water on one side of the group ranch, so that one side is becoming more heavily grazed, and that the locations of the watering points are driving this pattern. When asked if current grazing arrangements were exacerbating inequality, several groups in both Koija and Il Motiok replied that inequality results in the poorest people and people who have mainly goats being unable to move their herds off of the group ranch during droughts while others with mainly cattle, and cash access, are able to access paid grazing areas.

Politics of Community Based Natural Resource Management

Many individuals appear to feel that they are adequately compensated, that the bursaries, employment, funds and services available for healthcare are more important than this grazing area being set aside. In terms of benefits of the wildlife conservation area, typical answers were that vegetation was regenerating, that there were economic benefits through employment and bursaries, and that it served as a reserve for poor people to graze during extreme drought. Some stated that one problem arising from conservation areas was that those living close to conservation areas may receive more frequent fines due to animals wandering into these areas. One difference between group ranches, is that Il Motiok currently does not have a partner that directly compensates them for their conservation area through benefits generated by the lodge. Interestingly, Il Motiok continues to maintain its conservation area despite not having an active lodge or income, and this was explained as being due to seeking an investor and its use as a reserve grazing area (other factors concerning their relations with Mpala that may explain this?). At Koija, with regard to their relationship with Loisaba, many stated that they feel their voice is heard in all affairs, but others feel that there is no transparency in the affairs of the lodge, and resent their lack of involvement in management. Distrust was frequently expressed at Koija (all confirming the findings of Muthiani et al. 2011), especially combined with the views that Loisaba takes a bigger portion from proceeds, or that Loisaba may be fundraising for the purpose of aiding the community, but instead actually keeping this money. There was also a very prominent view that neighboring ranches are using the community to fundraise for other projects, and that they exploit the community and Maasai culture as a trademark to market their businesses abroad.

A distinct difference emerged in Il Motiok in that it was continually emphasized how much novel management had improved rangeland conditions recently, especially due to Holistic Management. One group elaborated on their views of this, expressing that there were disadvantages due to lack of movement, but also emphasizing that Il Motiok was very degraded in the past due to a lack of exclusion of outsiders, who came and overused the grass resources. This group concluded that outsiders needed to be excluded, and that this then necessitated changes in management of the forage resources within the group ranch. (as a side note: afterwards, when I became very curious how this view could be so dominant in this group, and perhaps on Il Motiok, it was here pointed out to me that these men were all employed by Mpala, and this distinct historical view may be partly related to this relationship).

Some groups clearly expressed that historical tensions remain in the area, with one group stating that "big ranches should accept settlement", and that the Maasai "need to reclaim the land". One group stated that wildlife conservation could be improved if Loisaba could provide grazing areas to take pressure off of the land instead of charging, but currently they felt they only use the community herd to manage their land by targeting certain areas for grazing.

Others expressed that conservation areas serve as a tool for security in the area, emphasizing that "when people from Isiolo come to graze they use the conservancies first, and this acts as a barrier to Loisaba".

In regard to the recently established wildlife conservation area, it appears that some see it as having negative impacts, for example due to the reduction of available grazing area, and the lack of grazing within this area leading to changes in vegetation, while others concluded these areas are good for the land, good for vegetation density, and good for wildlife. Individuals frequently stated combinations of positive and negative aspects in their views. Many individuals stated that the conservation areas would be best maintained by occasional grazing, and some stated that grazed grass has higher nutritional value than the grass in the conservation area. The majority of participants seemed to believe conservation areas have created reserve grazing areas that are beneficial during drought ("our food in the store"), but have also decreased the available grazing. Some stated that a negative impact of the conservation area is that there is now greatly increased livestock pressure in settlement areas.

It was stated by one group that as a result of management decisions there was improve tree condition in conservation and grazing areas, while resulting in greater impacts near houses. Trees are taller in conservation areas because there is no pruning. It was stated by many that conservation areas are restricted for too long, and that this may cause an increase in diseases for livestock. It was frequently stated that due to a lack of grazing that the conservation areas now fostered increased livestock diseases such as East Coast Fever and parasites such as ticks. It was also frequently stated that conservation area has increased predators such as hyenas, jackals, and lions, as well as elephants, leading to human-wildlife conflict. Holistic Management appeared to be viewed favorably by most interviewed at Il Motiok, and the rationale for this is that the practice is improving land conditions, has created more uniform grazing, and that the cattle hooves break up the soil in concentrated areas, aiding grass growth.

Vegetation Changes

Large changes in dominant vegetation were expressed by many focus-group participants. The dates were variable, but it appeared unanimous that in the early 1980's this change started to occur, and was marked by an increase in certain trees (e.g. *Acacia melliferia*) and a loss of others (e.g. Pushiruti). Before this, the canopy was said to be more open in some areas (e.g. Il Motiok), and more closed in other (Koija). At Il Motiok, it was stated that there were only three *Acacia mellifera* trees on the entire group ranch in the 1980's, it was very open before, Bobongi (*Euphorbia candelabrum*) and Pushirooti were dominant trees, there was more grass then as well. It was said that it used to be difficult to find pathways to pass through certain areas that are now bare, that the Bobongi trees are all very short, and that specific grasses like Il Peresi Uwaas

(4-6 seed heads) have greatly decreased, and that there are now increases in gullies and decreased nutrients. It was said that Pushirooti trees used to be widespread and were as large as a Tepes tree, so large that they were formerly used to hide cattle stolen from neighboring ranches. The encroachment of two cactus species (*Opuntia stricta* And Unknown *sp*.) at munichoi (a neighborhood in Koija) was emphasized as an extremely pressing problem, preventing grazing and causing a number of other problems. This same group emphasized that the entire valley between Munichoi and Il Motiok was formerly forested, but the deposition of sand from areas uphill (one person stating this occurred following "El Nino" years with heavy rains) that have eroded have completely changed the lagga, and these trees do not survive there now.

In terms of changes in specific species, the destruction of Bobongi tries was attributed unanimously to elephants. It was stated numerous times that Pushirooti and Matundai (unknown plant, not cactus but shares the name) were likely killed by, a neighboring owner of a cattle ranch (note: this belief was also stated during the Sardep interview). Considering more broad changes, prolonged drought due to changes in rain pattern, destruction by elephants, and goats feeding on trees have were all stated to have contributed to the more general changes in vegetation. In areas were the canopy has opened, this change was stated to be mainly caused by inadequate rainfall, beginning with a series of droughts in 1997, where droughts have caused the canopy to become more open, but other causes stated include increased population, as well as increased cutting of trees for house construction. The main reasons listed in one group for the area not being forested currently was lack of rain, though following being asked, some attributed impacts to increased numbers of goats as a catalyst, along with overpopulation. Goats and sheep were stated by one group to be the main reason for lack of vegetation within 500 meters of manyattas. The impact of goats and sheep on vegetation was seen by some as an explanation of overall changes in vegetation due to hoof movement breaking roots, and feeding impacting grass and trees. One group in Il Motiok emphasized that goats eat certain trees during the dry season, and that the loss of certain grass species (Il Peresi Uwaas) was thought to be due to drought, but also due to being eaten by goats. Specific tree species that goats feed on were listed as Tepes, Pushirooti, Giloriti, Munishoi, Girigiri, Shushei, and Nasurai. Decreases in grasses were explained by some as due to goats and sheep, emphasizing that this impact is magnified when it is dry. Some stated, when asked, that lack of animal movements has an impact, elaborating that the roots of grasses are gone now and that annual grasses do not respond as quickly following rains.

APPENDIX B

2013 SURVEY INSTRUMENT

Survey #__________________Manyatta___________Person Interviewed___________

Date________________

BOMA GPS Coordinates Lat____________________ Long______________________

1. What specific watering points by name do you use for each type of animal when there are grazing restrictions near the river? (Fill in table below with responses)

2. Do you have preferred sites, and if so, how often in a given month do you use these sites on average compared to the others? (Fill in table below with responses)

3. Are these preferred sites the same for all animals? (Fill in table below with responses)

Watering Points During Periods of Grazing Restrictions

Watering Points Used	**Cows**	**Goats**	**Sheep**	**Camels**	**Number of Times Used on Average Relative to Other Points (Specify If Different for Each Animal)**

4. Why do you go to these watering points that you use?

5. How often do you take each type of animal to water during this time?

6. How far away and in which direction do you take each on days you don't take them to water during this time?

7. Do you take the animals farther than the watering points on days that you take them to the water? (Try to specify distance or specific area if farther than the watering point)

8. How even is the grazing to the farthest point and back? Do you visit some areas more frequently than others?

9. What specific watering points by name do you use for each animal type when there are no grazing restrictions (or for Il Motiok when restricted away from the river)? (Fill in table below with responses)

10. Do you have preferred sites, and on average, how often in a given month do you use these sites compared to the others? (Fill in table below with responses)

11. Are these preferred sites the same for all animals? (Fill in table below with responses)

Watering Points (Dams, Laggas, and Rivers) When There Are No Grazing Restrictions

Watering Points Used	Cows	Goats	Sheep	Camels	Number of Times Used on Average Relative to Other Points (Specify If Different for Each Animal)

12. Do you first use the ponds and then the river with any animals? How do you decide where to go?

13. How often do you take each type of animal to water during this time?

14. How far away and in which direction do you take each on days you don't take them to water during this time?

15. Do you take the animals farther than the watering points on days that you take them to the water? (Try to specify distance or specific area if farther than the watering point)

16. How even is the grazing to the farthest point and back, for example, do you visit some areas more frequently than others?

17. Are there any watering points you would like to go to but can't, for example due to the conservation areas or farming?

18. Why don't you go to each of these places?

19. How often would you have gone there in the past?

20. Do you ever send animals to sites off of the group ranch during the long dry season? If so, where do you send them? (Fill in table below with responses)

21. In a typical year, how many of each type of animals do you send to each place? (use table below)

22. Are each of these places paid grazing, places you know someone, or open areas? (use table below)

Name of Grazing Site	#Cows	#Sheep	# Goats	#Camels	Paid Grazing, open area, or know someone?

23. Are there some years you don't take animals off of the group ranch during the long dry season? If so, how frequently is this?

24. If you leave animals on the group ranch during the long dry season, which watering points do you use then?

25. Do you graze in any different places during the long dry season?

26. Are there places for the long dry season grazing you would like to go off of the group ranch or used to go to but can't go to anymore? (use table below)

27. Why can't you go to each of these places now? (use table below)

28. When did you stop going to each? (use table below)

Place	Reason for not going there now	When this changed

29. Have any changes in movements of your animals without moving the whole manyatta occurred due to group ranch formalization?

31. What season would you have used these areas?

32. Have any changes in movement of your whole manyatta occurred due to group ranch formalization?

33. Where would you have moved to?

34. In what season would you have moved the manyatta in the past?

35. Were you required to move your manyatta following group ranch subdivision and neighborhood designation?

36. Have you moved from a different neighborhood cluster recently?

37. How many others in addition to you have animals in this manyatta?

38. If so, do you herd all of the animals together within this manyatta?

39. Does this apply to all types of animals?

40. How many people each murran, women, mzee, or children are typically available to herd?

41. Do you hire anyone to help?

42. Do you ever combine herds with other manyattas? If so, how many manyattas?

43. If so, when and why do you combine herds?

44. Do you or did you have any cows in the Holistic Management herds, and if so, how many?

45. How much have milk yields (by the standard cup) from each type of animal changed over the years?

46. If you don’t mind telling me, how many of each cattle, camels, sheep, and goats do you own?

47. What about the manyatta as a whole?

48. Are any of these borrowed?

49. Have you lent any to friends or family?

50. How has your herd changed in number of each animal throughout your life (get specific dates)?

51. How many of each animal do you sell in a typical year?

52. What are the main barriers to herding for you, for example changes in vegetation, access to former sites, or animal loses due to drought or disease?

53. Do you or your family gain any personal benefits from the conservation areas? How?

54. Do the conservation areas have any negative impacts on you? How? (If they answered earlier that grazing and watering points have become restricted, but then answered no to this, ask them to explain this).

55. What do you think the private ranches like Mpala and Loisaba gain from their relationships with the group ranches?

56. Does anyone in your family work for the private ranches or the lodges?

57. What about in the past?

APPENDIX C

2014 SURVEY INSTRUMENT

Name________________________SurveyCode______Date________________Consent Y / N

1.) In an earlier interview, you told us what watering points you use when you herd today. Have there been any major changes in the places you go to take animals to water or to find forage in your lifetime?

2.) If yes, how did your herding differ in the past? (Please get as much detail about watering points and former areas as possible)

3.) If your herding differed in the past, why did it change?

4.) What are the last three places that you have moved your boma to and when did you move?

5.) Are there any areas you specifically avoid taking your animals to on Koija? Why?

6.) Do you ever combine animals with others today? When?

7.) If you combine animals, who does the herding labor at this time?

8.) Do you currently have someone that does the herding labor for you regularly and that you feel confident about their abilities?

9.) What are the main costs involved in maintaining your herd ? (please get as much detail as possible)

10.) How often do you give each type of animal medicine (get detail for each type here)?

11.) Do you keep medicine for animals on hand? What kind?

12.) At the moment, how many of your sheep or cows do not have secured areas they can be taken to for forage?

13.) Are you planning to do anything differently than usual if this drought continues? What are your plans if it continues?

14.) Do you have any form of outside employment, income, or remittances? Please list this in as much detail as possible over the past 5 years.

15.) If you have income from sources other than herding, where do you allocate these funds? How much of it goes back into livestock?

17.) Have you given or loaned any animals to relatives or friends within the past year? If so, who and how many?

18.) Please list the main reasons why you have sold animals within the past two years.

19.) Which market or other place do you go to to sell animals?

20.) How has your herd size and composition changed since formalization (since 2002)? (Please try to get detailed numbers of animals in 2002 and today)

APPENDIX D

2015 POST-DROUGHT SURVEY INSTRUMENT

Name____________________2013SurveyCode______Date________________Consent Y / N

1.) Where did each type of different animal you keep stay during the drought in Feb-April 2015?

2.) How many of each type of animal were in each place, and for what period of time? Why did you move to different places?

3.) How many of each animal type died during the drought?

4.) What did each type of animal die from, and how many of each animal type died from the cause you listed?

5.) Of these that died, what breed of animal (sheep, goats, cows) were they?

6.) Of these that died, what ages were they?

7.) If you don’t leave Koija anymore, why do you not leave?

APPENDIX E

EMPIRICAL QUESTIONS USED TO GUIDE KEY INFORMANT INTERVIEWS

1. What historical factors have led to the different herding strategies and outcomes observed today, why have some accumulated animals and others have not, and how does this relate to differential access?
2. What were the range of herding options in the past (pre-2002, pre-1984), and what are the range of options today?
3. Where there "safety-nets" in the past that were relied upon during droughts that do not exist today? Are there other safety nets that have emerged for some during droughts today?
4. How are social relations and institutions currently modifying the themes discussed above (access, choice sets, and response capacities of households)?
5. How do individuals within each of these groups perceive their comparative vulnerability? What types of logistical challenges and ecological changes provide the greatest hurdles to livestock production?
6. Are there proxies that individuals use to indicate either high ability to respond or vulnerability of livelihoods to droughts and ecological changes?
7. Have grazing rules and norms changed in how herders track ecological conditions?
8. Have grazing rules and norms changed in their flexibility, for example with climate variation? What about in comparison to the past rules and norms?
9. Do these changes pose perceived constraints upon grazing access and ability to respond to ecological conditions, especially during drought?

10. If constraints are perceived, how do herders optimize within these constraints? What, if any, adaptive strategies are employed by herders, and do these strategies vary among different social strata?

APPENDIX F

SAMPLE KEY INFORMANT INTERVIEW QUESTIONS 1

1. What are the main factors that have influenced changes in people's herds over the years?

2. What about size of herds and animal type?

3. How has drought impacted herds over the years?

4. Were any factors different during the 2009 drought?

5. Why have some's herd increased since the 2009 drought and others have not?

6. Do different people have different abilities to maintain their herds during drought? Are there any things in particular that stand out to you about these people? Are some people more susceptible to the effects of drought? Are there things that stand out about these different groups of people?

7. What about to exploit vegetation within Koija, do people have different abilities? Are there any things in particular that stand out to you about people that are able to access some vegetation more and those that cannot?

8. What does the ability to send animals to unpaid areas off of Koija depend on? Could you please list all current options, and past options for this? Are there differences in people's ability to access this forage? Does it lead to greater success in herding for some people?

9. What does the ability to send animals to paid forage depend on? Could you please list all current options, and past options for this over the years (1984, 2002, 2015)? Are some people more able to access this? Does it lead to greater success in herding for some people?

10. What does free grazing on private ranches depend upon? Could you please list all current options, and past options for this over the years (1984, 2002, 2015)? Are some people more able to access this? Does it lead to greater success in herding for some people?

11. What does contested grazing depend upon? Are there differences in people's ability to access this? Does it lead to greater success in herding for some people?

APPENDIX G

SAMPLE KEY INFORMANT INTERVIEW QUESTIONS 2

Part 1 – Key-informant interviews

1. Do certain positions on the landscape support certain vegetation or soils?

2. Do certain soils support certain types of vegetation?

3. Are there any changes that occur in these types?

4. Are any of the above types more sensitive to changes at certain times?

5. What are the main changes in vegetation that have been observed throughout your lifetime?

6. What are the main factors you see driving these changes?

7. Do certain practices result in certain types of vegetation?

8. Are any of the above types able to withstand certain uses more than others?

9. What are the different responses of these types of vegetation to:

    a. Drought

    b. Livestock Forage (different species)

    c. Bomas

    d. Fire

    e. Other

10. Are certain vegetation types, landforms, or soils more susceptible to impacts from certain types of livestock?

Part 2 – Transect-walk Interviews and Worksheet for Abbreviated Response Summaries

1. What landscape type or landform is this place classified as?

2. What type of soils are found here?

3. What are the most important vegetation indicators of this type, and are these species found here?

4. Does this place have a certain condition? E.g. suitability for livestock, stability of soil?

5. What indicates this condition to you?

6. Did the type, soil, or condition differ in the past?

7. What type of soil and vegetation would have been found here?

8. Are some vegetation species increasing or decreasing?

9. If it has changed, what are the underlying factors that have driven these changes?

Mzee / Date	Mzee / Date
Place	Place
Landform / type	Landform / type
Soil Type	Soil Type
Past Landform / type	Past Landform / type
Past Soil Type	Past Soil Type
Indicators	Indicators
Soils and Veg Comparison	Soils and Veg Comparison
Past Indicators	Past Indicators
Increasers	Increasers
Decreasers	Decreasers
Soil Changes / Indicators	Soil Changes / Indicators
Drivers of Changes	Drivers of Changes
Livestock / Practices / Seasonality?	Livestock / Practices / Seasonality?
Condition in Past / Today	Condition in Past / Today

APPENDIX H

ELDER UNDERSTANDINGS OF ECOLOGICAL CHANGE: METHODS AND PRELIMINARY RESULTS

Introduction

There is a marked lack of robust incorporation of local knowledge in the design and monitoring of conservation endeavors in Laikipia (Yurco 2011). This is a pattern that exists beyond Laikipia throughout East Africa, where the rhetorical embrace of indigenous knowledge by International Non-Governmental Organizations is rarely incorporated into conservation and development practice (Goldman 2007). Despite a growing body of work on the highly nuanced pastoralist understanding of rangelands (Roba and Oba 2008) and the emphasis on local knowledge in CBNRM principles, there is a marked lack of robust incorporation of this knowledge in the design and monitoring of conservation and development practice in East Africa (Goldman 2007). Further, ecological assessments based upon an equilibrium understanding of rangelands are still regularly used (Roba and Oba 2008).

There have been numerous critiques of the ways that indigenous knowledge under various acronyms has been merely made legible and then systematically filed under a category of little consequence (Nadasdy 2005), as a cornerstone of the rationale of conservation practices. In our framing we use indigenous understandings of ecological process alongside conservation actor understandings of process to explore how different knowledges are priviledged or suppressed knowledge, for example as seen by Davis (2005) in their exploration of the role of knowledge and power in interactions between Moroccan range managers and pastoralists, or Goldman (2007) in her consideration of wildebeest knowledge among pastoralists and wildlife biologists. Being primarily a natural scientist by training, I attempt to frame this study as a way of incorporating a large body of critical work into ecological analysis, as well as an exercise in acknowledging the partiality and situatedness (Goldman 2003, Haraway 1988) of knowledge. From an ecological perspective, we take great pains to remove bias, understand biological significance, consider statistical assumptions, and ensure comparison to neutral models, but we rarely consider the social factors underlying why we ask a question in a certain way, an understanding that is common in the domain of science and technology studies (STS) and political ecology. I use ethnoecological methods, supplemented by previous analyses of landscape ecology and institutional theory.

There have been detailed ethnobotanical studies conducted in Laikipia, documenting a large number of plants, their uses, and local names (Brenzinger et al. 1994). However, there is little information on how understandings of ecological processes relate to past and current herding practices, and no exploration to date of whether and how herder understandings compare with the views of rangeland ecologists and development practitioners. Building upon previous studies that have sought nuanced understandings of the relationship between perception, biophysical data, and natural resource management politics (Dahlberg and Blaikie 1999, Goldman 2007), I explore the convergences and contradictions between pastoralists in one group ranch's understandings, and conservation and development actors' understandings of ecological process. In order to further explore how these knowledges differ and converge, as well as explore their potentially differing management implications, I explore the question, "How do elder herders understand recent local vegetation dynamics?"

Methods

To explore herder understandings of ecological dynamics, we used a combination of focus-group discussions and key-informant interviews with elders who make herding decisions or are responsible for herding labor and were known among community members for their extensive knowledge of plants and historical changes in ecological factors. The objective of this component was to better understand the way that herders think about the land, talk about the land, and associate certain environmental conditions with certain histories, contextual factors or processes. Additional themes explored how herders refer to areas of differential quality for livestock use, perceive changes in land conditions, and determine factors which changes in land conditions are attributed to.

Interview topics were identified from previous interviews, and informal discussions of salient landscape features. Semi-structured discussions occurred while referring to ten specific places on the landscape, selected for the landform it was thought to represent, and repeating a set of questions below for each location. Fourteen elders were selected purposively for their reputation across the group ranch for having strong knowledge of vegetation and soils. We first conducted focus group discussions and in-depth interviews on landform taxonomies and terminology in Maa, and translated these to English. These included discussion of different land cover types, landscape positions, vegetation species assemblages, soils, salient changes in these variables over the past 30 years, and the different possible transitions between vegetation states that can occur. We visited these locations with a subset of the elders, and refer to the location by place-name (locations are nearby and extremely well-known to all elders interviewed).

Focus group discussions and key-informant interviews

These followed a semi-structured approach that touched upon the following themes:

- Environmental changes observed over time
- The terms for grouping of ecological landscape zones and vegetation assemblages
- Ranking of each of the above assemblages in terms of importance for different livestock species
- Soil properties, landforms, or patterns of land uses that certain assemblages may be associated with
- How have the relative abundances of vegetation assemblages changed throughout their lifetime
- What changes in vegetation are attributed to

Landform Histories

Building upon answers and terms used above, we located ten areas on the landscape that represented the landform types. We interviewed herders while walking along a transect about the landforms and vegetation present. I will also ask about the indicators that herders use to distinguish certain ecological zones or vegetation assemblages from others (e.g. soil properties, plant species composition, physiognomy of vegetation structure, etc.). I will ask about how these ecological zones and vegetation assemblages have changed in relative abundance throughout their lifetime, and the underlying factors that are thought to shape the current distribution of and features associated with these classes

Participant observation of herder grazing practices

I asked a family of herders to accompany them on their herding routes, to observe the relative importance of different individual plant species or plant assemblages for livestock and also to "ground-truth" how the concepts revealed in interviews are practically applied and revealed in tacit knowledge that is not verbalized.

Analysis

Interviews were then coded in NVIVO. The main landform, vegetation assemblages, soil concepts, landscape positions, transitions, drivers, and interrelation/interdependence of these factors was systematically analyzed.

Preliminary Results

I. A hierarchy of typical vegetation assemblages with corresponding taxonomy was explained, describing the possible types of vegetation that occur on specific soils and in specific landscape positions.

II. Possible transitions between these types were reported. Examples of common transitions that were reported included: open areas (*ongatta*) becoming forested (*entim*). Forested areas being cut, and the livestock manure being mixed with the soil creating different soils when the livestock pen is moved (*medjoni)* that develops into a grazing lawn (*muurua*). Other forests have disappeared (*pushirut*i) only to be replaced by open grassy areas on hilltops (*ongatta*) or novel types of forests *(entim*).

III. Rankings of vegetation assemblage types in terms of preference as forage for each livestock species are summarized in Table 1

IV. A number of drivers of changes were indicated. These are summarized here, to provide context of an analysis of the impacts of livestock changes.

   a. Drought is a very frequently cited reason for ecological change. It was frequently stated that most vegetation that had decreased would return if rains were more frequent. Drought was indicated to have caused decreases in a number of grasses, vines, and trees. The most frequently mentioned were *pushiruti, kimanjoi, and loingwaroi.*

b. Elephants were very frequently stated to be responsible for the loss of *Euphorbia bussei.*

c. Two increasing shrub/tree species (*munichoi, nchurai*) were frequently said to cause declines in grasses

d. Sheep were frequently said to be having negative impacts on grass, through uprooting it, unlike cattle, and also trampling it.

e. Goats were said to have impacts on two vine species (*loingwaroi, ldenja*), and two shrubs (*kimanjoi, sambungike*)

f. One succulent (suguroi) was said to have been impacted by elephants

g. A number of vines were said to have been lost because they were supported by *mpoponi* and *pushirooti* trees

h. Some emphasized that if they were able to do restrictions then many of the species would return.

i. It was frequently said that livestock are impacting regrowth of vegetation when it does rain, because long droughts are followed by direct foraging on grass soon after it rains and before it matures

j. Vegetation is more sensitive during droughts

k. When at the ends of interviews, we asked about the impacts of livestock, especially if they had not been mentioned, elders indicated that drought is worse than livestock for grass

l. Livestock paths explain bare lands (*ldoroto*)

m. It was emphasized by some that the differential impacts of goats and sheep was due to the impact of their hooves and their direct impacts when grazing the grass.

n. Cutting around cattle pens and homesteads decreases trees

o. Manure in the soils are needed to make a *muurua*

p. During droughts tree are cut in the laggas

q. Sand is being deposited in the laggas

r. Water is carrying away tree in the laggas

s. Munichoi is also causing bare areas because it doesn't allow grass to grow

t. Elephants have eaten *marapari, ngoki,* and maybe *siteti*

u. Decreases in cattle grazing are causing muuruas to have trees establish within them

v. *Ldupaisero* are increasing because nothing is feeding on them

w. Elephants are killing *ltepes*

x. *Ldupaisero* is spreading and occupying areas grass could grow on

y. Soil has become too hard and grass can no longer grow

z. Trees no longer block the sun and the ground is hot

aa. When there is grass the sheep and cattle don't have an impact, but when there is less grass they do

bb. *Munichoi* changes the soil

cc. *Sulubei* (referring to a former large patch of shrubs in vertisol soils (*ngisero*)) died due to drought

dd. Trees that had "milk" (referring to Euphorbiaceae and other species with milky sap) have all gone so maybe one thing affected them ee. Livestock spend a lot of time in laggas and so have impacts there

ff. Sand under ltepes in laggas are drying them out

gg. Two impacts in particular had responses that elders usually discussed at length and said they were not sure about, these included the increases of Munichoi and Nchurai, and the loss of Pushirooti. The only hypothesis for increase of shrubs was "livestock spread the seeds", or that "god" caused them to increase. One suggested these species (along with *Ltepes*) were not impacted by drought.

    i. Multiple people suggested drought killed *pushirooti*

    ii. Multiple people suggested elephants killed *pushirooti*

    iii. One suggested that individuals from white settlement communities had perhaps killed *pushirooti*

    iv. One suggested an insect may have killed *pushirooti*

    v. One suggested that disease may have killed *pushirooti*

hh. It was stated by some that fire was used in the past and this helped to decrease shrub cover, increase nutrition of grasses, and to control the density of ticks.

ii. It was also emphasized that now that when it rains following a drought there is very little vegetation on the ground, and so the water runs off rapidly and clears even more vegetation away, with one elder emphasizing that today in many areas the vegetation must start from bare ground where in the past the grass would have just dried out and would regrow immediately following rains. Repeatedly when asked if the grass had decreased, this complex interrelation between factors was emphasized jj. While it was emphasized that the rains have begun to fail and are a dominant driver of changes, and that all of these factors may be listed by herders as a first response, we saw repeated indication following in-depth interviews, that all of these process are cognized as interacting in a complex manner.

kk. With laggas (seasonal streams) it was stated by many that the water now runs more rapidly than it did in the past. This increased flow of water has caused trees to fall, and also sand has been deposited in the lagga.

ll. Some emphasized that the areas with dense trees (entim), are usually avoided, but these places except for during drought, so maybe livestock have an impact then.

	Cattle	Sheep	Goats
Grazing Lawn (muurua) #1	3.111111	4	3.888889
Lagga	4	4.222222	4.444444
Vertisol Soils	5.444444	4.444444	7.888889
Open Shrubby Area (ongatta)	2.333333	3.777778	3.888889
Grazing Lawn (muurua) #2	5.888889	4.666667	6.111111
Grazing Lawn (muurua) #3	5.777778	5	7.555556
Forest (Entim) #1	3.111111	4.888889	2.222222
Forest (Entim) #2	6.333333	5.444444	2.444444
SparseVegetation and Bare Soil #1	9.222222	8.777778	9
SparseVegetation and Bare Soil #2	9.777778	9.777778	7.555556

Average Rank (1-10) in order of preference of use for each livestock species (n=9)

APPENDIX I

STRATEGIC COMMUNICATION

The following document consists of a summary of research results from Chapters 2 and 3, intended for distribution among conservation and development actors in Laikipia.

Consequences of Changes in Herding Mobility for Livelihoods and Sustainable Landscapes

A Case Study from Laikipia, Kenya

ICON

Integrative Conservation Research Brief

Decreases in dryland herding mobility are thought to have large impacts on the sustainability of herding practices in East Africa. To understand how changes in forage access impact herder livelihoods in semi-arid Laikipia, I took a landscape-level, historical approach. I used focus-group discussions, interviews, and surveys of most households in a Maa-speaking herding community to investigate how household assets, employment, and social factors shape herding strategies and responses to drought events. I found that three recent historical changes have combined to reduce historical grazing access: restriction from private wildlife conservation ranches, pastoralist conflict, and group ranch conservancy formation. The restructuring of herding practices due to these changes has implications for vegetation, livelihoods, and ongoing conflicts over forage access.

INTRODUCTION

Mobility in Semi-arid Lands

Pastoralist herding, or livestock husbandry, is generally considered to be a well-suited livelihood for semi-arid environments that do not support farming. Due to high variability in rainfall (see Figure 1), accessing water and pasture resources requires seasonal flexibility. Customary pastoralist rules and norms were thought to facilitate access in much of East Africa historically[1]. However, when seasonal grazing movements cannot occur, vegetation has less chance to recover seasonally, and there is also a decreased ability to meet subsistence needs[2,3]. Therefore, while agreements that allow for mobility are thought to prevent overuse and degradation of forage during dry seasons, negative impacts on perennial grasses can occur when there is a lack of movement[4,5].

In Laikipia, Kenya, historical changes have shaped pastoralist land use and mobility[6,7]. Most Maasai pastoralists were forcibly removed from Laikipia by the British government in the early 1900's. The remaining herding communities living in Laikipia are mainly descended from Maa-speaking pastoralists that intermarried with five hunter-gatherer groups[6].

These communities live within Mukogodo Division, an area with boundaries dating to the colonial era. Other work has documented how decreasing access to external areas during the dry-season has led to low numbers of livestock per person compared to other pastoralists in the region[6]. There has also been a recently increased reliance on goats and sheep that is driven by their greater drought-hardiness, fast reproduction, and ease of sale[6].

July 11, 2014

April 21, 2015

June 16, 2015

Figure 1 – Seasonal Grass Variability

RESEARCH FRAMING AND METHODOLOGY

This study was undertaken to understand the interaction between changes in herding livelihoods and changes in landscape-level vegetation. In the subset of research reported here, I focus on the impacts of changes in mobility on livelihoods. This work involved first establishing a timeline of changes in forage access and herding practices. I then explored how these changes have affected herder livelihoods. This study, combined with other ecological work in progress, aims to inform sustainable landscape conservation.

Between 2012 and 2015, at Koija Group Ranch, Mukogodo Division, in Laikipia County (Figure 2), focus-group discussions with elder herders were used to determine salient ecological and livelihood changes that have occurred over recent history (1980-2015). Household surveys with the majority of households (225 out of 227) were completed with an elder who makes herding decisions (male or female, average age estimated at ~48.2 yrs). Data collected included information on livestock, household assets, herding practices, labor, seasonal herding location place names, and household response to drought.

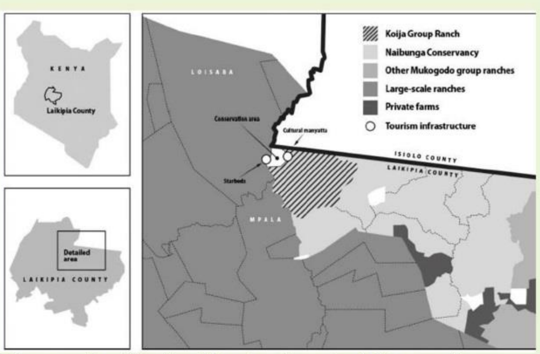

Figure 2 – Study Site in Central Kenya[8]

RESULTS AND DISCUSSION

Historical Losses of Mobility

Results indicate that customary cattle grazing access has been lost over the past 30 years due to three main reasons: 1) restriction from private wildlife conservation-oriented ranches to the west, 2) pastoralist conflict to the north and east, and most recently 3) land titling and restricted access driven by wildlife conservancy formation within pastoralist group ranches (Table 1).

Stated Reason	Time Period	# of Households Reporting
Exclusion from Private Ranches	1980 -1985	90
Conflict Between Other Pastoralists	1990 - 1995 1995 - 2000 2000 - 2005	145 48 34
Group Ranch Conservancy Formation	1995 - 2000 2000 - 2005	20 59

Table 1 – Most Frequent Reasons Stated for Loss of Seasonal Grazing Access
(only includes reasons with >20 households reporting)

Changes in Herd Composition and Local Concentration of Livestock

Herding remains the dominant livelihood at Koija. Cattle have decreased since the early 1980's (Table 2), while small-stock (sheep and goats) have increased. Overall livestock (TLUs) have increased in the past 30 years (Table 2), but at the same time population has doubled due to both influxes from other areas and reproduction. There are few animals per person overall, with an average overall value of 2.04 Tropical Livestock Units (TLUs) per person (1 TLU = 10 sheep or goats, 1.42 cattle, or 1 camel). Just 37 out of 225 households had above 4 TLUs per person, the number considered adequate for subsistence in the region[9]. Cows are distributed unevenly, with no cows in 53 out of 225 households, but fourteen households having modestly-sized herds of 45 to 170 cattle. On the other hand, seventy households had less than 0.7 TLUs per person (the equivalent of 10 goats or sheep per person).

	1980	2016
Camels	0	299
Cows	5,357	3,530
Sheep + Goats	2,692	28,386
TLU	4,042	5,624

Table 2 - Livestock Holdings (1980-2016)

Cows require grass, necessitating dry season movements to areas outside of Koija. Sheep require less grass than cows, and goats can survive on diverse vegetation. The dry-season woody vegetation and grass within Koija is able to support small-stock but not cattle.

Increasing Dependence on Few Areas

Cattle are increasingly dependent on a limited number of areas accessible either through customary rights, paid access to private ranches, or through employment on private ranches. Factors such as access to cash, household assets, and herding labor also determines who can gain access. Herders with larger herds utilize a disproportionate share of access to the reserve forage resources for cows. Just seven families accounted for 26.53% of the paid and employed access to private wildlife conservation ranches. Many herders are less able to gain access through the above pathways, and so remain reliant on grazing in areas they formerly accessed customarily, but are currently not permitted to access. This unpermitted grazing involves physical danger to herders due to wildlife, conflict with private land owners/managers, and monetary penalties imposed when caught. Due to these factors, many households rely primarily on goats and sheep that can better survive on the forage resources on Koija. This has resulted in a localized concentration of small-stock, creating a novel ecological pressure on vegetation. Most herders indicated these changes were also decreasing regrowth of grass during the wet season.

Inequality in livestock holdings has increased since 2002. Herders with outside employment had higher rates of cattle increases between 2002 and 2016 (t-Ratio=1.748, prob>t=0.431). Drawing from interviews, herders attributed this not just to the direct benefits from income, but to individual grazing resource access on private ranches and other benefits gained from employer relationships.

CONCLUSIONS AND RECOMMENDATIONS

Complexity of Livelihood Changes

The main model of wildlife conservation on pastoralist lands in Laikipia involves partnerships between wildlife conservation organizations and pastoralist group ranch lands where a portion of land is set aside as wildlife habitat in exchange for direct involvement in ecotourism. Within conservancies a number of additional practices are often adopted, including market-oriented solutions intended to make herding practices more profitable, and governance reforms intended to promote specific types of rangeland management. While these approaches emphasize adjustments within group ranch boundaries, this study's results point to external forage access as a matter of higher relevance for livelihood concerns.

Insufficiency of Forage within Group Ranches

Forage areas within Koija are unable to support cattle year-round due to the high variability of rainfall. This variability in rainfall is especially high compared to neighboring private ranches[10], leading to this factor perhaps being overlooked in policy discussions. The current amount of permitted reserve grazing access on private lands appears to provide benefits mainly to herders that are employed or have modest-sized herds, while other herders rely overwhelmingly on herding of small-stock. Illicit grazing on private wildlife conservation ranches by herders from Koija can thus be best understood as a response to the limitations of access to reserve grazing during drought. Further, mobility is known to prevent degradation of rangelands[4], and sedentism is indicated by herders to be decreasing forage regrowth in the wet season. While internal management of forage resources through practices such as rotational grazing may yield benefits, my results imply these practices are unlikely to offset the need for outside forage.

History's Role in Current Land Use

This study demonstrates how understanding the historical sequence of forage access loss is necessary to understand changes in livelihoods and ecological impacts. These results indicate that human population growth and increases in livestock do not sufficiently explain current livelihood challenges. The findings reported here provide insight that can inform conversations about grazing access with a sensitivity to historical loss of land from herder perspectives. Recognizing the complexity of these inter-related historical changes may also identify previously overlooked ways of improving livelihoods. Finally, while conflicts such as those occurring at the time of writing (early March 2017) are related to complex regional and political factors, these findings could potentially help shape policy relevant to such conflicts.

Seeking Policies for Sustainable Landscape Based Upon Herding Ecology

Others[11] have recommended that legally recognized rights of pastoralists to access seasonal grazing lands might lead to enhanced flexibility and improved relations. Such policies, if structured with explicit regard to seasonal variability, have a high potential to minimize locally negative impacts of livestock on vegetation and to enhance livelihoods more equally across pastoral communities. Private ranches and conservation organizations have played a historical role in the titling of group ranch lands as a prerequisite of conservancy formation[12], but the results of this study show conservancy plans have not accounted for robust consideration of seasonal access. In the absence of policy changes, it should be expected that localized degradation as well as conflicts will continue, especially as variation of rainfall increases with climate change[13].

Acknowledgements

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APPENDIX J

IRB APPROVAL FORM

APPROVAL OF PROTOCOL

June 3, 2013

Elizabeth King

egking@uga.edu

Dear Elizabeth King:

On 6/3/2013, the IRB reviewed the following submission:

Type of Review:	Initial Study
Title of Study:	Pastoralist Livelihood Transitions in Laikipia Kenya
Investigator:	Elizabeth King
IRB ID:	STUDY00000011
Funding:	None
Grant ID:	None
IND, IDE, or HDE:	None
Documents Reviewed:	None

The IRB approved the protocol from 6/3/2013.

In conducting this study, you are required to follow the requirements listed in the Investigator Manual (HRP-103).

Sincerely,

Larry Nackerud
University of Georgia
Institutional Review Board Chairperson

Page 1 of 1

APPENDIX K

ARCPY SCRIPT FOR CHAPTER 4 LEAST COST PATH NESTED LOOP

```
# Iterates through variable start points, end points, and resistance surfaces
# and calculates the least cost path
# Final Version October 2, 2017
# Written by Tom Prebyl and Ryan Unks

import os
from multiprocessing import Pool, freeze_support
import arcpy
from arcpy.sa import *
import itertools
from arcpy import env
arcpy.CheckOutExtension("Spatial")

#set directory for temporary files
arcpy.env.workspace = "C:\\Data\\Ryan\\tempdirs"
arcpy.env.overwriteOutput = True
#set raster snapping
arcpy.env.snapRaster = ""
#set extent
arcpy.env.extent = "MINOF"
#set cell size
arcpy.env.cellSize = "MINOF"

# create a nested loop to iterate through and calculate least cost paths for all combinations of inputs
# define a function to process the data where the input is a combination tuple from the "combs" list, where
# each "comb" is a variable from one of three data sources: 1. resistance surfaces 2. bomas 3. water points

def lcp_koija(comb):

    try:
        i = comb[0]
        j = comb[1]
        k = comb[2]

        #creates temporary directories for multiprocessing
        sname = os.path.join(arcpy.env.workspace,"td%s_%s_%s" %(i,j,k))
        os.mkdir(sname)
        arcpy.env.scratchWorkspace = sname

        costsurface = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\R3LU7.gdb\\RES{0}".format(i)
        boma =
"C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\SeparatedBomasAbr.gdb\\Object_ID_{0}".format(j)
        wp =
"C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\SeparatedRiverPointsAbr.gdb\\Object_ID_{0}".format(k)
```

```
        costpathname = 
os.path.join(r"C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\LCP\\","res%s_boma%s_wp%s.tif" % (i,j,k))
        costdistname = 
os.path.join(r"C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\CostDist\\","res%s_boma%s_wp%s.tif" % (i,j,k))
        tbacklink = arcpy.CreateScratchName("bl%s_%s_%s" % (i,j,k),",",arcpy.env.scratchWorkspace)

        tcostdist = arcpy.sa.CostDistance(boma,costsurface,"",tbacklink)
        tcostdist.save(costdistname)

        tcostpath = arcpy.sa.CostPath(wp,tcostdist,tbacklink,"BEST_SINGLE", "OBJECTID")
        tcostpath.save(costpathname)

        return "Success!"
    except:
        return "Fail!"


# enable multiprocessing to run processes in parallel
if __name__ == "__main__":
    freeze_support()
    pool = Pool(processes=12)
    print "runnin!"


    # Make a list of tuples of all possible combinations of data sets
    combs = []
    for i in range (1,42):
        for j in range(1,101):
            for k in range(1,22):
                combs.append((i,j,k))


    results = pool.map(lcp_koija,combs)

    print "finished!!!"
    pool.close()


    #Collect Cost Distances of all Least Cost Paths in a table

    #------Inputs
    ingeo = r"C:\Users\ryanunks\Desktop\\ModelFit\Python\LCP"
    #input geodatabase that contains all of the rasters you want to extract values from
    outtable = r"C:\Users\ryanunks\Desktop\ModelFit\Python\Pathcosts.gdb\R3LU7"
    #where you want to save the output table

    #-----Processes
    env.workspace = ingeo   #set the workspace, this is where you will search for all rasters
    all_rast = arcpy.ListRasters()

    # Loop through each raster to extract cost values
    print "looping through each raster..."
    costlist = []   #empty list to store path costs
    for r in all_rast:
        print "on raster: " + str(r)
```

```
        costvals = [row[0] for row in arcpy.da.SearchCursor(r, ('PATHCOST'))]   # get all the PATHCOST values
from the attribute table
        costlist.append(max(costvals))  #append the max value of the path to the list

    print "saving results..."
    # Make a temporary results table
    temptab = arcpy.CreateTable_management("in_memory",'temptab')   #make atemporary table (stored in memory)
    arcpy.AddField_management(temptab,'raster','TEXT',field_length = 65)    #add field to store raster name
    arcpy.AddField_management(temptab,'PATHCOST','DOUBLE')  #add a field to store path costs

    # Add values to table
    for i in range(len(all_rast)):
        with arcpy.da.InsertCursor(temptab,['raster','PATHCOST']) as icurs:
            icurs.insertRow([all_rast[i],costlist[i]])

    # Copy table to output location
    arcpy.CopyRows_management(temptab,outtable)

    print "done!"
```

APPENDIX L

ARCPY SCRIPT FOR CHAPTER 4 VARYING THE WEIGHTS OF COMPOSITE RESISTANCE SURFACE RASTERS

```
#Reclassifies multiple classes in a given raster
#Adds together multiple rasters with variable weights and writes them to a database

# Import arcpy module
import arcpy
arcpy.CheckOutExtension("Spatial")
arcpy.env.overwriteOutput = True

# Local variables:
Slope1 = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1"
House1 = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\RESLayers\\CompositeBuildingBlocks.gdb\\House1"
LandUseReclassApril2017_Resamp =
"C:\\Users\\ryanunks\\Desktop\\ModelFit\\NewModelFitLayers.gdb\\Reclass_Land_August2017"

RES1 = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Composites.gdb\\RES1"
RES2 = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Composites.gdb\\RES2"
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RES10temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES10temp"
RES11temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES11temp"
RES12temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES12temp"
RES13temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES13temp"
RES14temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES14temp"
RES15temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES15temp"
RES16temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES16temp"
RES17temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES17temp"
RES18temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES18temp"
RES19temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES19temp"
RES20temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES20temp"
RES21temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES21temp"
RES22temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES22temp"
RES23temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES23temp"
RES24temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES24temp"
RES25temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES25temp"
RES26temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES26temp"
RES27temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES27temp"
RES28temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES28temp"
RES29temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES29temp"
RES30temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES30temp"
RES31temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES31temp"
RES32temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES32temp"
RES33temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES33temp"
RES34temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES34temp"
RES35temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES35temp"
RES36temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES36temp"
RES37temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES37temp"
RES38temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES38temp"
RES39temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES39temp"
RES40temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES40temp"
RES41temp = "C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES41temp"
```

```
# Set Geoprocessing environments
arcpy.env.extent = Slope1
arcpy.env.cellSize = "30.836464173042"
arcpy.env.snapRaster = Slope1

arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES1temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 10000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES2temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 7500;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES3temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 2500;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES4temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 100;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES5temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 10000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES6temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 7500;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES7temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 2500;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES8temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 100;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES9temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 10000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES10temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 7500;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES11temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 2500;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES12temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 100;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES13temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 10000;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES14temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 7500;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES15temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 2500;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES16temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 100;7 5000;31 5000;32 5000;33 5000;34 5000;35 5000", RES17temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 10000;31 5000;32 5000;33 5000;34 5000;35 5000", RES18temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 7500;31 5000;32 5000;33 5000;34 5000;35 5000", RES19temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 2500;31 5000;32 5000;33 5000;34 5000;35 5000", RES20temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 100;31 5000;32 5000;33 5000;34 5000;35 5000", RES21temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 10000;32 5000;33 5000;34 5000;35 5000", RES22temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 7500;32 5000;33 5000;34 5000;35 5000", RES23temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 2500;32 5000;33 5000;34 5000;35 5000", RES24temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 100;32 5000;33 5000;34 5000;35 5000", RES25temp, "DATA")
```

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arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 10000;33 5000;34 5000;35 5000", RES26temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 7500;33 5000;34 5000;35 5000", RES27temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 2500;33 5000;34 5000;35 5000", RES28temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 100;33 5000;34 5000;35 5000", RES29temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 10000;34 5000;35 5000", RES30temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 7500;34 5000;35 5000", RES31temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 2500;34 5000;35 5000", RES32temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 100;34 5000;35 5000", RES33temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 10000;35 5000", RES34temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 7500;35 5000", RES35temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 2500;35 5000", RES36temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 100;35 5000", RES37temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 10000", RES38temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 7500", RES39temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 2500", RES40temp, "DATA")
arcpy.gp.Reclassify_sa(LandUseReclassApril2017_Resamp, "Value", "1 5000;2 100;4 5000;5 5000;6 5000;7 5000;31 5000;32 5000;33 5000;34 5000;35 100", RES41temp, "DATA")

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES1temp Value 1", RES1)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES2temp Value 1", RES2)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES3temp Value 1", RES3)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES4temp Value 1", RES4)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
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1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES5temp Value 1", RES5)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES6temp Value 1", RES6)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES7temp Value 1", RES7)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES8temp Value 1", RES8)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES9temp Value 1", RES9)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES10temp Value 1", RES10)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES11temp Value 1", RES11)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES12temp Value 1", RES12)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES13temp Value 1", RES13)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES14temp Value 1", RES14)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES15temp Value 1", RES15)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value

1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES16temp Value 1", RES16)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES17temp Value 1", RES17)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES18temp Value 1", RES18)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES19temp Value 1", RES19)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES20temp Value 1", RES20)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES21temp Value 1", RES21)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES22temp Value 1", RES22)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES23temp Value 1", RES23)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES24temp Value 1", RES24)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES25temp Value 1", RES25)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES26temp Value 1", RES26)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value

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1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES27temp Value 1", RES27)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES28temp Value 1", RES28)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES29temp Value 1", RES29)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES30temp Value 1", RES30)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES31temp Value 1", RES31)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES32temp Value 1", RES32)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES33temp Value 1", RES33)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES34temp Value 1", RES34)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES35temp Value 1", RES35)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES36temp Value 1", RES36)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value
1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES37temp Value 1", RES37)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBloc
ks.gdb\\Slope1 Value
```

1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES38temp Value 1", RES38)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES39temp Value 1", RES39)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES40temp Value 1", RES40)

arcpy.gp.WeightedSum_sa("C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\Slope1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\CompositeBuildingBlocks.gdb\\House1 Value 1;C:\\Users\\ryanunks\\Desktop\\ModelFit\\Python\\RESLayers\\Temp.gdb\\RES41temp Value 1", RES41)

APPENDIX M

CHAPTER 4 ARCPY CODE FOR PROCESSING LEAST COST CORRIDORS

 # ArcPy code for normalizing the minimum of corridors

```
import arcpy, os
from arcpy import env
arcpy.CheckOutExtension("Spatial")
from arcpy.sa import *


# Define input workspace and create list of rasters
arcpy.env.workspace = r'C:\Users\ryanunks\Desktop\TrialofHeatMapGen\PreMinus_WPs_Wet_Corridors.gdb'
rasters = arcpy.ListRasters()

out_workspace = r'C:\Users\ryanunks\Desktop\TrialofHeatMapGen\Minimum_WPs_Wet_Corridors.gdb'

# Run conditional


for raster in rasters:
    ras = Raster(raster)
    minRaster = arcpy.GetRasterProperties_management(raster, ("MINIMUM"))
    OutRas = (ras - (int(float(minRaster.getOutput(0)))))
    OutRas.save(os.path.join(out_workspace, raster))
Print = "Done"

print('Done Processing')


# Define input workspace and create list of rasters
arcpy.env.workspace = r'C:\Users\ryanunks\Desktop\NgombeHeatMaps\PreMinus_WPs_Dry_Corridors.gdb'
rasters = arcpy.ListRasters()

out_workspace = r'C:\Users\ryanunks\Desktop\NgombeHeatMaps\Minimum_WPs_Dry_Corridors.gdb'

# Run conditional


for raster in rasters:
    ras = Raster(raster)
    minRaster = arcpy.GetRasterProperties_management(ras, ("MINIMUM"))
    OutRas = (ras - (int(float(minRaster.getOutput(0)))))
    OutRas.save(os.path.join(out_workspace, raster))
Print = "Done"

print('Done Processing')
```

 # ArcPy code for extracting only the 100th quantile of a corridor

```
import arcpy, os
from arcpy import env
import pandas as pd
arcpy.CheckOutExtension("Spatial")
from arcpy.sa import *
import random
import numpy
import numpy as np

# Define input workspace and create list of rasters
arcpy.env.workspace = r'C:\Users\ryanunks\Desktop\NgombeHeatMaps\Bomas_WPs_Wet_Corridors.gdb'
rasters = arcpy.ListRasters()

out_workspace = r'C:\Users\ryanunks\Desktop\NgombeHeatMaps\Thresholded_WPs_Wet_Corridors.gdb'


# Run conditional statement for all rasters within the input workspace, output a new raster with values outside of the 100th quantile set to 0


for raster in rasters:
    ras = Raster(raster)
    data = arcpy.RasterToNumPyArray(ras)
    flat = data.flatten()
    cut = numpy.percentile(flat, 99, axis=None)
    OutRas = arcpy.sa.Con(ras >= cut, ras, 0)
    OutRas.save(os.path.join(out_workspace, raster))
Print = "Done"

print('Done Processing')
```

APPENDIX N

CHAPTER 4 RESISTANCE RASTER OPTIMIZATION RESULTS BY THEME

Avoided areas initial optimization process results:

Raster Layer	Iteration	Percent Increase	Cumulative Increase
Avoided Areas	1	0.182334	0.182334
	2	0.078441	0.246473
	3	0.017779	0.25987
	4	0.006041	0.264341
	5	0.003052	0.266586
	6	0.03587	0.292894
	7	0.006087	0.297198
	8	0.00919	0.303657
	9	0.0076	0.308949
	10	0.002262	0.310512
	11	0.001759	0.311725
	12	0.00948	0.318249
	13	0.005669	0.322115
	14	0.001311	0.323003

Slope initial optimization process results:

Raster Layer	Iteration	Percent Increase	Cumulative Increase
Slope	1	0.253016	0.253016
	2	0	0.253016
	3	0	0.253016
	4	0	0.253016

House buffer initial optimization process results:

Raster Layer	Iteration	Percent Increase	Cumulative Increase
House	1	0.009454	0.009454
	2	0.004124	0.013539
	3	0	0.013539
	4	0	0.013539

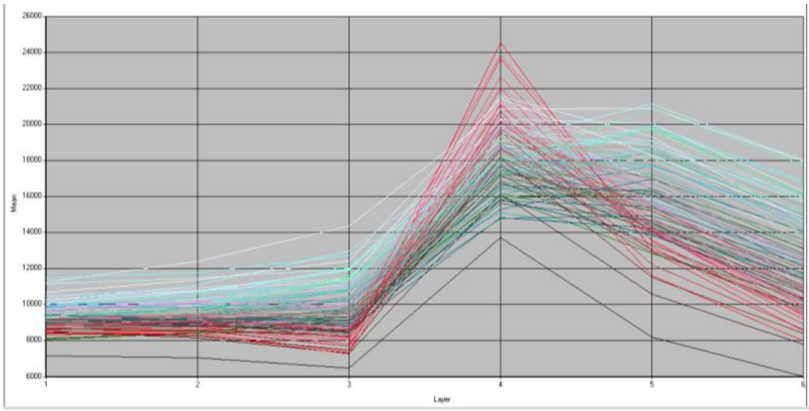
Decreases in overall model RMSE values due to each optimization of composite weights and subsequent optimization of individual raster layers

Raster Component Varied	Total Iterations	Percent Decrease in RMSE	Cumulative Decrease in RMSE
Composite1	22	-	0.256951
Composite2	23	0.131691	0.354804
Composite3	24	0.024368	0.370527
Composite4	25	0.019546	0.382831
Composite5	26	0.011909	0.390181
Composite6	27	0.003113	0.392079
Composite7	28	0.002415	0.393547
Composite8	29	0	0.393547
Composite9	30	-	0.393547
Round2LandUse1	31	0.013544	0.401761
Round2LandUse2	32	0.012437	0.409201
Round2LandUse3	33	0.00917	0.414619
Round2LandUse4	34	0.007598	0.419066
Round2LandUse5	35	0.00477	0.421837
Round2LandUse6	36	0.00459	0.424491
Round2LandUse7	37	0.003323	0.426403
Round2LandUse8	38	0.001993	0.427547
Round2LandUse9	39	0.001721	0.428532
Round2LandUse10	40	0.00332	0.430429
Round2LandUse11	41	0.00287	0.432063
Round2LandUse12	42	0	0.432063
Composite1	43	0	0.432063
Composite2	44	0	0.432063
Composite3	45	0	0.432063
Composite4	46	0	0.432063
Round2Slope1	47	0	0.432063
Round2Slope2	48	0	0.432063
Round2Slope3	49	0	0.432063
Round2House1	50	0.007018	0.436049
Round2House2	51	0.001392	0.436834
Round2House3	52	0	0.436834
Round2House4	53	0	0.436834
Composite1	54	0.001434	0.437642
Composite2	55	0	0.437642
Composite3	56	0.005089	0.440503

Composite4	57	0.00121	0.441181
Composite5	58	0	0.441181
Round3LandUse1	59	0.003361	0.443059
Round3LandUse2	60	0.001704	0.444008
Round3LandUse3	61	0.007152	0.447984
Round3LandUse4	62	0	0.447984
Round3LandUse5	63	0.003087	0.449688
Round3LandUse6	64	0	0.449688
Round3LandUse7	65	0	0.449688
Round3LandUse8	66	0.002857	0.45126
Round3LandUse9	67	0.001756	0.452224
Round3LandUse10	68	0	0.452224
Round3LandUse11	69	0	0.452224
Composite1	70	0	0.452224
Composite2	71	0	0.452224
Composite3	72	0	0.452224
Composite4	73	0	0.452224
Composite5	74	0	0.452224
Round3Slope1	75	0	0.452224
Round3Slope2	76	0.002044	0.453344
Round3Slope3	77	0	0.453344
Round3Slope4	78	0	0.453344
Round3Slope5	79	0	0.453344
Round3Slope6	80	0	0.453344
Composite1	81	0	0.453344
Composite2	82	0	0.453344
Composite3	83	0.000157	0.453429
Composite4	84	0	0.453429
Composite5	85	0	0.453429
Composite6	86	0	0.453429
Round3House1	87	0	0.453429
Round3House2	88	0	0.453429
Round3House3	89	0	0.453429
Round3House4	90	0	0.453429

APPENDIX O

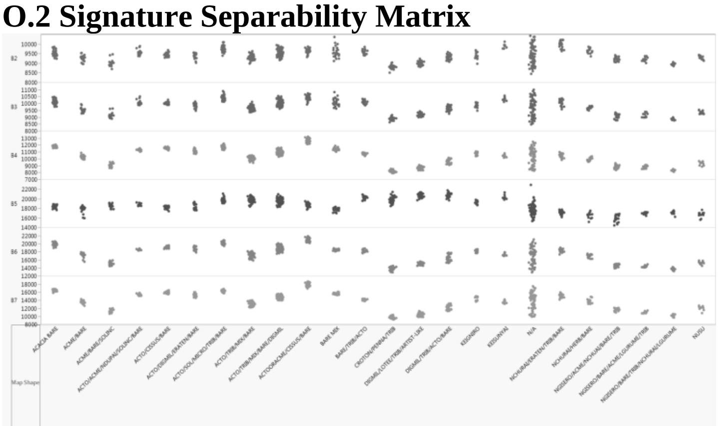
SUPERVISED CLASSIFICATION SPECTRAL SIGNATURE ANALYSIS



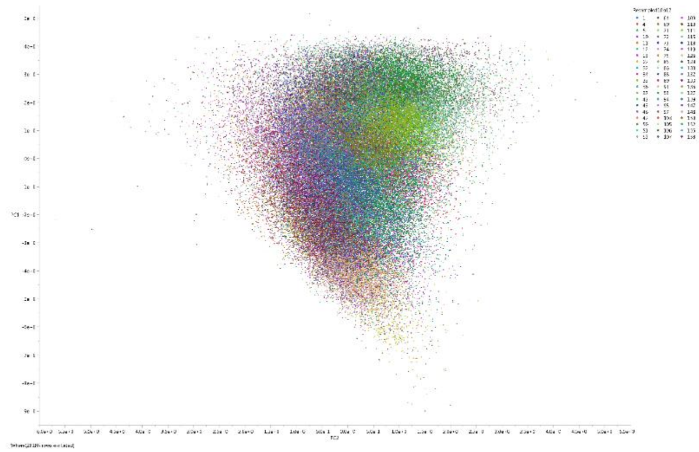
O.1 Spectral Signatures Graph

Signature Name		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
ACTO/BARE/MIX1	1	0	8384.5	1047.64	2426.44	1155.73	4236.47	1195.83	1712.44	2200.13	2176.16	4278.02	7902.97	6345.11	3759.23	4316.06	2921.17	8622.85	10300.2	3833.37	6052.8	8238.19	7020.12
PERENNIAL/CROTON	2	8384.5	0	9044.21	10746.1	7328.31	4242.26	9542.64	6692	6296.75	6550.73	5900.41	1093	2068.04	4692.67	4188.52	5831.3	1391.15	3120.47	5398.14	3408.48	2362.24	2408.21
ACTO/BARE/MIX2	3	1047.64	9044.21	0	1786.21	2112.74	4851.94	1072.92	2448.52	2933.69	2886.68	4477.98	8638.95	7026.39	4512.11	4893.13	3316.27	9271.05	11046.2	4595.55	6822.15	8945.24	7640.39
ACTO/BARE/MIX3	4	2426.44	10746.1	1786.21	0	3571.33	6536.29	1353.7	4099.04	4522.07	4494.27	5981.36	10308.8	8718.19	6166.59	6629.87	5074.78	10991.1	12711.5	6169.76	8452.69	10628.6	9358.67
ACTO/BARE	5	1155.73	7328.31	2112.74	3571.33	0	3279.74	2296.85	824.356	1328.11	1347.29	3854.52	6800.76	5283.2	2682.32	3364.04	2259.51	7575.55	9194.34	2819.86	4957.45	7163.98	6014.69
BARE/ACME/MICRO	6	4236.47	4242.26	4851.94	6536.29	3279.74	0	5385.15	2606.4	2225.92	2754.97	2458.44	3976.42	2274.84	1243.43	707.18	1757.86	4708.14	6644.18	2634.58	3084.09	4716.33	3414.73
BARE/ACTO/MIX4	7	1195.83	9542.64	1072.92	1353.7	2296.85	5385.15	0	2876.25	3281.85	3233.12	5158.98	9055.21	7514.87	4910.84	5486.2	4033.76	9759.96	11429.2	4911.95	7174.57	9362.63	8132.05
BARE/ACTO/MIX5	8	1712.44	6692	2448.52	4099.04	824.356	2606.4	2876.25	0	762.941	863.658	3262.35	6219.5	4661.78	2073.35	2646.64	1499.43	6916.44	8638.26	2394.41	4441.5	6571.89	5334.1
BARE/ACTO/MIX6	9	2200.13	6296.75	2933.69	4522.07	1328.11	2225.92	3281.85	762.941	0	1111.62	2898.09	5846.26	4312.5	1786.09	2416.55	1503.35	6549.6	8327.11	2464.85	4249.09	6289.39	4998.84
BARE/NCHURAI/MIX	10	2176.16	6550.73	2886.68	4494.27	1347.29	2754.97	3233.12	863.658	1111.62	0	3734.73	6005.45	4575.48	1920.11	2671.17	1831.96	6607.68	8259.93	1884.18	4032.04	6159.06	4946.97
KEISUNYAI/DIGMIL	11	4278.02	5900.41	4477.98	5981.36	3854.52	2458.44	5158.98	3262.35	2898.09	3734.73	0	5937.43	4310.59	3330.81	2849.78	2329.41	6486.6	8643.99	4503.19	5383.36	6796.16	5325.02
MIX2	12	7902.97	1093	8638.95	10308.8	6800.76	3976.42	9055.21	6219.5	5846.26	6005.45	5937.43	0	1765.68	4156.18	3898.65	5564.15	1443.51	2825.26	4685.14	2529.68	1710.36	2042.05
MIX3	13	6345.11	2068.04	7026.39	8718.19	5283.2	2274.84	7514.87	4661.78	4312.5	4575.48	4310.59	1765.68	0	2686.75	2211.83	3879.68	2654.55	4488.5	3531.44	2130.57	2809.92	2004.29
MIX4	14	3759.23	4692.67	4512.11	6166.59	2682.32	1243.43	4910.84	2073.35	1786.09	1920.11	3330.81	4156.18	2686.75	0	1191.76	1880.22	4898.72	6591.82	1483.5	2523.01	4565.93	3407.86
MUURUA1	15	4316.06	4188.52	4893.13	6629.87	3364.04	707.18	5486.2	2646.64	2416.55	2671.17	2849.78	3898.65	2211.83	1191.76	0	1683.03	4516.6	6434.97	2330.38	2772.61	4430	3118.93
MUURUA2	16	2921.17	5831.3	3316.27	5074.78	2259.51	1757.86	4033.76	1499.43	1503.35	1831.96	2329.41	5564.15	3879.68	1880.22	1683.03	0	6119.19	8063.97	2694.94	4162.08	5995.78	4592.68
NGISERO/LGURUME/ACME	17	8622.85	1391.15	9271.05	10991.1	7575.55	4708.14	9759.96	6916.44	6549.6	6607.68	6486.6	1443.51	2654.55	4898.72	4516.6	6119.19	0	2357.8	5335.46	3175.28	1377.18	1791.79
NGISERO/TRIB/BARE/NCHURAI	18	10300.2	3120.47	11046.2	12711.5	9194.34	6644.18	11429.2	8638.26	8327.11	8259.93	8643.99	2825.26	4488.5	6591.82	6434.97	8063.97	2357.8	0	6700.28	4321.42	2171.68	3738.98
NGISERO1	19	3833.37	5398.14	4595.55	6169.76	2819.86	2634.58	4911.95	2394.41	2464.85	1884.18	4503.19	4685.14	3531.44	1483.5	2330.38	2694.94	5335.46	6700.28	0	2387	4654.73	3798.05
NGISERO2	20	6052.8	3408.48	6822.15	8452.69	4957.45	3084.09	7174.57	4441.5	4249.09	4032.04	5383.36	2529.68	2130.57	2523.01	2772.61	4162.08	3175.28	4321.42	2387	0	2342.14	2039.99
NGISERO3	21	8238.19	2362.24	8945.24	10628.6	7163.98	4716.33	9362.63	6571.89	6289.39	6159.06	6796.16	1710.36	2809.92	4565.93	4430	5995.78	1377.18	2171.68	4654.73	2342.14	0	1716.37
NUSU	22	7020.12	2408.21	7640.39	9358.67	6014.69	3414.73	8132.05	5334.1	4998.84	4946.97	5325.02	2042.05	2004.29	3407.86	3118.93	4592.68	1791.79	3738.98	3798.05	2039.99	1716.37	0

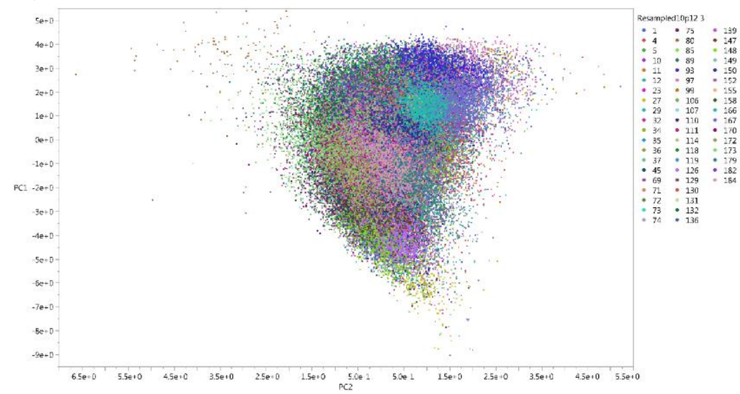
O.2 Signature Separability Matrix



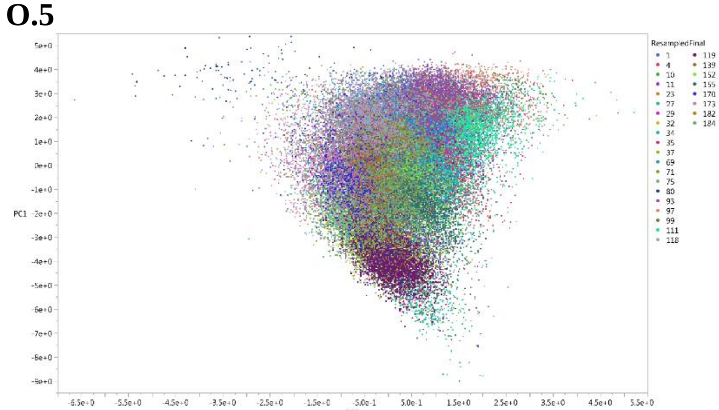
O.3 Example Comparison of Spectral Signature Distribution Overlaps (y axis=reflectance value by band#, x axis = class) Among Six Bands



O.4



O.5



O.6

APPENDIX P

R CODE FOR VARYING PARAMETERS AND CREATING MULTIPLE

CLASSIFICATIONS OF THE SAME SCENE

```
#script to compare many different classification parameter outcomes to a given classified image
library(cluster)
library(randomForest)
library(sp) # spatial/geographfic objects and functions
library(rgdal) #GDAL/OGR binding for R with functionalities
library(spdep) #spatial analyses operations, functions etc.
library(gtools) # contains mixsort and other useful functions
library(maptools) # tools to manipulate spatial data
library(parallel) # parallel computation, part of base package no
library(rasterVis) # raster visualization operations
library(raster) # raster functionalities
library(forecast) #ARIMA forecasting
library(xts) #extension for time series object and analyses
library(zoo) # time series object and analysis
library(lubridate) # dates functionality
library(colorRamps) #contains matlab.like color palette
library(rgeos) #contains topological operations
library(sphet) #contains spreg, spatial regression modeling
library(BMS) #contains hex2bin and bin2hex, Bayesian methods
library(bitops) # function for bitwise operations
library(foreign) # import datasets from SAS, spss, stata and other sources
library(gdata) #read xls, dbf etc., not recently updated but useful
library(classInt) #methods to generate class limits
library(plyr) #data wrangling: various operations for splitting, combining data
library(gstat) #spatial interpolation and kriging methods
library(readxl) #functionalities to read in excel type data
library(psych) #pca/eigenvector decomposition functionalities
library(dplyr)#dplyr, summarize, mutate functions
library(snow)#enables parallel processing
library(ggplot2)#enables plotting tools
library(broom)#enables tidy functions
library(factoextra)
library(mclust)
library(mcclust)

in_dir <- "C:/Users/ryanunks/Desktop/L8StackForAutomate/L8DCA/Masked"
CB <- list.files(path=in_dir, pattern="*.tif",full.names=T)
CB_stack <- stack(CB)
CB_stack
v <- getValues(CB_stack)
v[is.na(v)] <- 0

PCA30m <- prcomp(v[,4:9],
            center = TRUE,
            scale. = TRUE)
v2 <-cbind(v,PCA30m$x[,1:3])

set.seed(1234)
nsamples <- 36250
sdfAll <- subset(v2[sample(1:nrow(v2), nsamples), ])
sdfAll <- as.data.frame(sdfAll)
sdfAll <- sdfAll[!(apply(sdfAll, 1, function(y) any(y == 0))),]
```

```
mydata <- data.frame(sdfAll$band2mask, sdfAll$band3mask, sdfAll$band4mask, sdfAll$band5mask, sdfAll$band6mask,
sdfAll$band7mask)


#hierarchical k-means, complete
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "complete", iter.max = 300)
  f <- paste0('hkmeansComplete', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f

}
#hierarchical k-means, ward.D2
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "ward.D2", iter.max = 100)
  f <- paste0('hkmeansWard', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, manhattan
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.metric = "manhattan", iter.max = 100)
  f <- paste0('hkmeansMAN', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, canberra
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.metric = "canberra", iter.max = 100)
  f <- paste0('hkmeansCAN', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, single
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "single", iter.max = 100)
  f <- paste0('hkmeansSingle', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, average
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "average", iter.max = 100)
  f <- paste0('hkmeansAverage', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, "mcquitty"
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "mcquitty", iter.max = 100)
  f <- paste0('hkmeansmcquitty', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f

}

#hierarchical k-means, "median"
```

```
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "median", iter.max = 100)
  f <- paste0('hkmeansmedian', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, "centroid"
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "centroid", iter.max = 100)
  f <- paste0('hkmeanscentroid', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

# complete Method, Hierarchical Clustering
for (i in 1:100){
  d <- dist(mydata, method = "euclidean") # distance matrix
  fit <- hclust(d, method="complete")
  groups <- cutree(fit, k=i)
  f <- paste0('complete', i)
  sdfAll <- cbind(sdfAll, groups)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

# centroid Method, Hierarchical Clustering
for (i in 1:100){
  d <- dist(mydata, method = "euclidean") # distance matrix
  fit <- hclust(d, method="centroid")
  groups <- cutree(fit, k=i)
  f <- paste0('centroid', i)
  sdfAll <- cbind(sdfAll, groups)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}


# Ward's Method, Hierarchical Clustering
for (i in 1:100){
  d <- dist(mydata, method = "euclidean") # distance matrix
  fit <- hclust(d, method="ward.D2")
  groups <- cutree(fit, k=i)
  f <- paste0('ward', i)
  sdfAll <- cbind(sdfAll, groups)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#Kmeans
for (i in 1:100){
  e <- kmeans(mydata, algorithm="Lloyd", i, iter.max = 400, nstart = 10)
  f <- paste0('kmeans', i)
  sdfAll <- cbind(sdfAll, e$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#Clara method - improved k means
for (i in 1:100){
  clus <- clara(mydata,i,samples=100,metric="manhattan",pamLike=T)
  f <- paste0('clara', i)
  sdfAll <- cbind(sdfAll, clus$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, ward.D
```

```
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "ward.D", iter.max = 100)
  f <- paste0('hkmeansWard.D', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, median
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "median", iter.max = 100)
  f <- paste0('hkmeansmedian', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#KmeansHart
for (i in 1:100){
  e <- kmeans(mydata, algorithm="Hartigan-Wong", i, iter.max = 400, nstart = 10)
  f <- paste0('kmeansHart', i)
  sdfAll <- cbind(sdfAll, e$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, complete, LLOyd
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.method = "complete", iter.max = 200, km.algorithm = "Lloyd")
  f <- paste0('hkmeansCompleteLloyd', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}

#hierarchical k-means, complete, LLOyd, minkowski
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.metric = "minkowski", hc.method = "complete", iter.max = 200, km.algorithm = "Lloyd")
  f <- paste0('hkmeansCompleteLloydmink', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f

}

#hierarchical k-means, complete, LlOyd, max
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.metric = "maximum", hc.method = "complete", iter.max = 200, km.algorithm = "Lloyd")
  f <- paste0('hkmeansCompleteLloydmax', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f

}

#hierarchical k-means, complete, LlOyd, binary
for (i in 2:100){
  p <- hkmeans(mydata, i, hc.metric = "binary", hc.method = "complete", iter.max = 200, km.algorithm = "Lloyd")
  f <- paste0('hkmeansCompleteLloydbin', i)
  sdfAll <- cbind(sdfAll, p$cluster)
  g <- ncol(sdfAll)
  colnames(sdfAll)[g] <- f
}


sdfAll_subs <- sdfAll[!(apply(sdfAll, 1, function(y) any(y == 0))),]
rm(dummy)
dummy <- matrix(, nrow = 1, ncol = 0)
#calculate adjusted Rand's index across all columns, comparing all classifications to
#centroidResampledFinal
```

```
for (i in 23:2107){
  n <- adjustedRandIndex(sdfAll_subs$kcentroidResampledFinal, sdfAll_subs[[i]])
  dummy <- cbind(dummy, n)
  h <- ncol(dummy)
  t <- colnames(sdfAll_subs)[i]
  colnames(dummy)[h] <- t
}
```

APPENDIX Q

GRAZING LAWN AND HILLTOP DCA AXIS CORRELATIONS AND SPECIES

Grazing Lawn DCA Axis Correlations

		axis 1			axis 2			axis 3	
	r	r-squared	tau	r	r-squared	tau	r	r-squared	tau
Abutilon mauritianum	-0.392	0.154	-0.298	-0.141	0.02	-0.054	0.226	0.051	0.379
Acacia etbaica canopy	0.318	0.101	0.366	0.509	0.26	0.164	-0.148	0.022	0.038
Acacia mellifera canopy	0.718	0.516	0.528	-0.133	0.018	-0.034	0.088	0.008	0.124
Acacia mellifera shrub	0.868	0.753	0.746	-0.412	0.169	-0.373	0.078	0.006	0.08
Acacia nilotica	0.172	0.03	0.145	-0.099	0.01	-0.029	-0.205	0.042	-0.262
Acacia reficiens canopy	-0.101	0.01	-0.203	0.069	0.005	0.016	0.047	0.002	-0.047
Acacia reficiens shrub	0.621	0.385	0.378	-0.138	0.019	-0.087	0.07	0.005	0.087
Acacia tortilis canopy	-0.448	0.201	-0.243	-0.011	0	-0.11	-0.348	0.121	-0.376
Acacia tortilis shrub	0.819	0.671	0.589	-0.281	0.079	-0.122	0.049	0.002	0.011
Aloe secundiflora	-0.14	0.02	-0.087	0.105	0.011	0.145	0.061	0.004	0.029
Balanites aegypitiaca canopy	0.172	0.03	0.145	-0.099	0.01	-0.029	-0.205	0.042	-0.262
Balanites aegypitiaca shrub	0.296	0.087	0.28	0.224	0.05	0.035	-0.205	0.042	-0.14
Barleria eranthemoides	0.205	0.042	0.301	-0.155	0.024	-0.301	-0.314	0.098	-0.549
Blepharis ciliaris	0.776	0.602	0.658	-0.376	0.142	-0.389	-0.242	0.059	-0.094
Cadaba farinosa	0.042	0.002	0.043	0.737	0.543	0.514	0.03	0.001	0.043
Cissus rotundifolia	0.239	0.057	0.206	0.348	0.121	-0.012	-0.025	0.001	0.133
Commelina spp.	0.275	0.076	0.185	-0.383	0.147	-0.433	0.151	0.023	0.155
Commicarpus plumbagineus	0.28	0.078	0.109	-0.238	0.056	-0.266	0.099	0.01	0.109
Craterostigma plantagineum	-0.083	0.007	-0.16	-0.324	0.105	-0.206	-0.787	0.619	-0.275
Cyathula orthocantha	0.154	0.024	0.087	0.674	0.455	0.378	-0.17	0.029	-0.204
Cyperus spp.	0.214	0.046	0.262	-0.268	0.072	-0.32	0.239	0.057	0.32
Digitaria milanjiana	-0.672	0.452	-0.495	0.051	0.003	0.121	0.218	0.048	0.143
Eragrostis tenuifolia	0.016	0	0	0.418	0.175	0.046	-0.258	0.066	-0.324
Euphorbia bussei	0.011	0	0.07	0.137	0.019	0.175	-0.72	0.519	-0.245
Euphorbia sp.	-0.406	0.165	-0.385	-0.216	0.047	-0.128	-0.079	0.006	-0.171
Indigofera spp.	0.318	0.101	0.167	-0.078	0.006	-0.1	0.219	0.048	-0.033
Ipomoea kituiensis	0.117	0.014	0.021	0.692	0.479	0.398	-0.153	0.023	-0.147
Ipomoea spatulata	0.465	0.216	0.361	-0.161	0.026	-0.168	-0.166	0.027	-0.289
Kyllinga spp.	-0.258	0.067	-0.256	0.061	0.004	0.189	0.267	0.071	0.278
Lycium europaeum	0.398	0.158	0.226	-0.221	0.049	-0.158	-0.377	0.142	-0.317
Microchloa kunthii	-0.001	0	0.056	-0.138	0.019	-0.078	-0.296	0.087	-0.322
Opuntia stricta	0.301	0.091	0.278	-0.393	0.154	-0.216	0.52	0.27	0.402
Oxygonum sinuatum	0.154	0.024	0.087	0.674	0.455	0.378	-0.17	0.029	-0.204
Pennisetum mezianum	0.179	0.032	0.21	-0.258	0.066	-0.349	-0.337	0.113	-0.175
Pennisetum stramineum	0.316	0.1	0.337	-0.05	0.003	-0.089	0.443	0.196	0.337
Pentanisia ouranogne	0.214	0.046	0.262	-0.268	0.072	-0.32	0.239	0.057	0.32
Pollichia campestris	-0.116	0.014	-0.11	-0.298	0.089	-0.16	0.04	0.002	0.012
Portulaca oleracea	0.154	0.024	0.087	0.674	0.455	0.378	-0.17	0.029	-0.204
Portulaca quadrifida	0.179	0.032	0.175	-0.147	0.022	-0.175	-0.164	0.027	-0.035
Sansevieria robusta	0.224	0.05	0.175	0.352	0.124	0.384	-0.246	0.061	-0.175
Sansevieria volkensii	0.678	0.459	0.663	-0.393	0.154	-0.297	-0.144	0.021	-0.16
Solanum coagulans	0.062	0.004	-0.055	-0.616	0.379	-0.409	-0.48	0.231	-0.166
Solanum incanum	-0.125	0.016	-0.133	-0.244	0.06	-0.177	-0.219	0.048	-0.177
Tragus berteronianus	-0.157	0.025	0.108	0.191	0.036	0.217	-0.034	0.001	-0.162
Tribulus terrestris	-0.137	0.019	-0.253	0.401	0.161	0.451	-0.089	0.008	-0.099
Unknown 1	-0.127	0.016	-0.185	0.032	0.001	0.031	0.055	0.003	-0.093
Zaleya pentandra	-0.353	0.125	-0.216	-0.249	0.062	-0.185	0.011	0	0.031

Hilltop Vegetation Transects DCA Axis Correlations

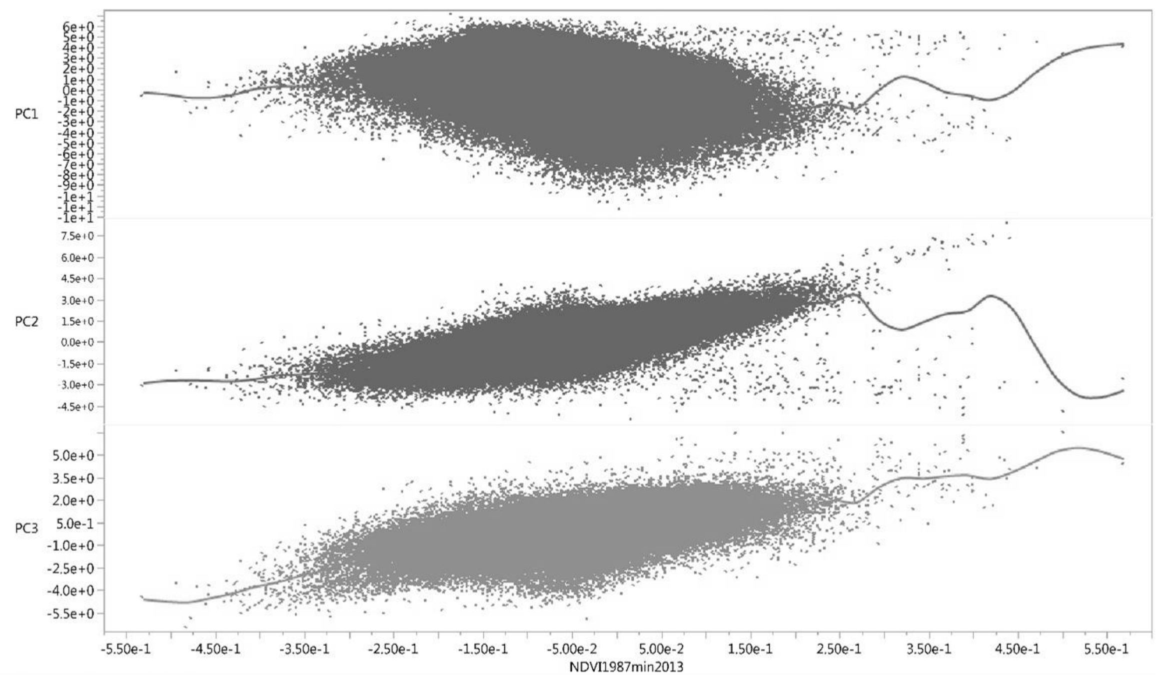
		axis 1			axis 2			axis 3	
	r	r-squared	tau	r	r-squared	tau	r	r-squared	tau
Abutilon mauritianum	0.088	0.008	0.099	-0.318	0.101	-0.225	0.015	0	0.027
Acacia brevispica	0.353	0.125	0.388	-0.053	0.003	0.243	0.373	0.139	0.098
Acacia etbaica	0.429	0.184	0.273	-0.457	0.209	-0.389	-0.195	0.038	-0.102
Acacia mellifera	-0.854	0.729	-0.702	-0.067	0.005	0.037	0.065	0.004	0.059
Acacia tortilis	-0.362	0.131	-0.308	0.453	0.205	0.345	0.175	0.031	0.069
Adenia volkensii	0.102	0.01	0.007	-0.066	0.004	-0.12	-0.088	0.008	-0.035
Adenium obesum	0.218	0.047	0.191	0.431	0.186	0.162	-0.028	0.001	-0.021
Asparagus africanus	0.296	0.088	0.257	-0.487	0.237	-0.399	0.434	0.188	0.262
Balanites aegypitiaca	0.29	0.084	0.162	-0.213	0.045	-0.244	0.223	0.05	0.214
Barleria eranthemoides	0.67	0.448	0.412	0.433	0.187	0.293	-0.064	0.004	-0.008
Barleria sp. 1	-0.256	0.065	-0.112	0.114	0.013	-0.081	0.358	0.128	0.102
Barleria sp. 1	0.05	0.003	-0.007	-0.038	0.001	-0.049	0.093	0.009	0.191
Becium obovatum	0.497	0.247	0.281	0.401	0.161	0.21	-0.242	0.059	-0.302
Blepharis ciliaris	-0.255	0.065	-0.186	-0.367	0.134	-0.235	-0.126	0.016	0.142
Brachiaria jubata	0.147	0.022	0.124	-0.219	0.048	-0.063	0.211	0.044	0.228
Cadaba farinosa	-0.064	0.004	-0.058	-0.136	0.018	-0.093	0.037	0.001	0.093
Capparis tomentosa	0.156	0.024	0.148	-0.257	0.066	-0.208	0.274	0.075	0.228
Caralluma dummeri	-0.2	0.04	-0.092	0.32	0.103	0.249	0.078	0.006	0.009
Caralluma foetida	0.348	0.121	0.247	-0.127	0.016	-0.109	0.522	0.272	0.267
Cenchrus ciliaris	0.223	0.05	0.003	0.372	0.139	0.142	-0.241	0.058	-0.135
Chloris virgata	0.144	0.021	0.145	-0.069	0.005	-0.158	-0.046	0.002	-0.106
Cissus rotundifolia	0.111	0.012	-0.064	0.23	0.053	0.261	-0.302	0.091	-0.261
Commelina spp.	0.014	0	0.011	0.262	0.069	0.209	0.214	0.046	0.182
Commicarpus plumbagineus	-0.401	0.161	-0.357	-0.044	0.002	-0.083	-0.197	0.039	-0.069
Commiphora sp.	-0.156	0.024	-0.149	-0.222	0.049	-0.177	-0.034	0.001	-0.019
Cordia monoica	0.229	0.053	0.204	-0.198	0.039	-0.244	0.154	0.024	0.244
Cordia sinensis	0.374	0.14	0.083	0.287	0.082	0.231	-0.186	0.035	-0.083
Craterostigma plantagineum	0.062	0.004	0.093	-0.12	0.014	-0.093	0.136	0.018	0.058
Croton dichogamus	0.172	0.03	0.268	0.346	0.12	0.274	0.468	0.219	0.2
Cyathula orthocantha	0.129	0.017	0.062	-0.091	0.008	-0.046	0.164	0.027	0.062
Cyperus sp. 1	0.316	0.1	0.042	0.206	0.042	0.048	-0.197	0.039	-0.078
Cyperus sp. 2	0.314	0.099	0.034	0.283	0.08	0.122	-0.306	0.094	-0.331
Cyperus sp. 3	-0.142	0.02	-0.038	-0.442	0.196	-0.351	-0.214	0.046	-0.026
Dactylonctenium aegyptium	0.186	0.035	0.046	0.503	0.253	0.261	-0.042	0.002	0.046
Digitaria milanjiana	0.163	0.026	0.135	-0.055	0.003	-0.04	-0.41	0.168	-0.225
Echidnopsis sharpei	-0.014	0	0.03	-0.133	0.018	-0.129	0.254	0.065	0.208
Edithcolea grandis	0.038	0.001	0.049	0.078	0.006	0.109	0.082	0.007	0.168
Enteropogon macro	0.016	0	-0.153	-0.386	0.149	-0.22	0.192	0.037	0.159
Eragrostis sp. 1	-0.121	0.015	0.077	0.194	0.038	0.261	-0.21	0.044	-0.261
Eragrostis tenuifolia	0.039	0.002	0.016	0.586	0.344	0.372	-0.453	0.205	-0.367
Euphorbia bussei	-0.209	0.044	-0.143	-0.228	0.052	-0.265	0.116	0.013	0.183
Euphorbia heterochroma	0.74	0.548	0.57	-0.168	0.028	-0.151	0.031	0.001	0.047
Euphorbia sp.	-0.408	0.166	-0.442	0.354	0.125	0.297	0.129	0.017	0.056
Grewia bicolor	0.14	0.02	0.181	-0.182	0.033	-0.171	0.279	0.078	0.222
Grewia sp.	0.095	0.009	-0.014	-0.428	0.183	-0.34	0.159	0.025	0.048
Guttenbergia boranensis	-0.375	0.14	-0.271	0.287	0.082	0.333	-0.167	0.028	-0.189
Harpachne schimperi	0.011	0	0.176	0.181	0.033	0.073	-0.236	0.055	-0.375
Heliotropium steudneri	0.063	0.004	0.017	-0.39	0.152	-0.329	0.262	0.069	0.128
Heliotropium steudneri	-0.152	0.023	-0.119	-0.167	0.028	-0.018	-0.081	0.007	-0.191
Hibiscus sp. 1	0.082	0.007	-0.047	-0.066	0.004	0.142	0.21	0.044	-0.013
Hibiscus sp. 2	-0.306	0.093	-0.221	0.07	0.005	0.14	-0.203	0.041	-0.21
Hibiscus sp. 3	-0.007	0	0.032	-0.223	0.05	-0.188	-0.257	0.066	-0.24
Hibiscus sp. 4	0.524	0.275	0.31	-0.054	0.003	-0.003	0.099	0.01	0.192
Indigofera sp. 1	0.177	0.031	0.134	0.335	0.112	0.112	-0.269	0.072	-0.246
Indigofera sp. 2	0.149	0.022	0.112	-0.251	0.063	-0.167	0.099	0.01	0.118
Indigofera sp. 3	-0.006	0	-0.018	-0.021	0	-0.041	0.082	0.007	0.135
Ipomoea kituiensis	0.539	0.29	0.306	0.349	0.122	0.263	-0.156	0.024	-0.011
Ipomoea mombassana	-0.272	0.074	-0.159	-0.002	0	0.006	0.153	0.024	-0.011

Ipomoea obscura	-0.189	0.036	-0.152	-0.108	0.012	-0.238	-0.131	0.017	0.082
Ipomoea sp. 1	-0.09	0.008	-0.109	0.234	0.055	0.208	-0.029	0.001	-0.03
Ipomoea sp. 2	0.338	0.114	0.228	-0.111	0.012	-0.089	-0.171	0.029	-0.188
Ipomoea sp. 3	0.156	0.024	0.148	-0.257	0.066	-0.208	0.274	0.075	0.228
Ipomoea sp. 4	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Justicia odora	0.493	0.243	0.304	-0.369	0.136	-0.275	0.462	0.214	0.089
Justicia sp. 1	0.351	0.123	0.219	-0.108	0.012	-0.007	0.531	0.282	0.318
Justicia sp. 2	0.461	0.212	0.359	0.162	0.026	0.176	0.428	0.183	0.12
Justicia sp. 3	-0.334	0.111	-0.29	0.027	0.001	0.035	-0.243	0.059	-0.276
Justicia sp. 4	0.268	0.072	0.252	-0.12	0.014	-0.111	0.442	0.195	0.381
Kleinia kleinioides	-0.201	0.04	-0.245	-0.067	0.005	0.012	-0.097	0.009	-0.105
Kleinia sp. 1	-0.244	0.06	-0.103	-0.223	0.05	-0.316	-0.515	0.265	-0.236
Kleinia sp. 2	-0.28	0.079	-0.182	-0.084	0.007	0.088	-0.002	0	-0.111
Kleinia sp. 3	0.505	0.255	0.298	-0.209	0.044	-0.065	0.082	0.007	-0.007
Kyllinga sp.	0.345	0.119	0.236	0.155	0.024	0.07	-0.278	0.077	-0.316
Lippia javonica	0.06	0.004	-0.023	-0.207	0.043	0.07	-0.101	0.01	-0.209
Lycium europaeum	0.173	0.03	0.149	-0.189	0.036	-0.312	-0.438	0.192	-0.283
Microchloa kunthii	0.061	0.004	-0.003	0.269	0.073	0.269	-0.294	0.087	-0.296
Ocimum americanum	-0.191	0.037	-0.105	-0.264	0.07	-0.153	0.077	0.006	0.039
Opuntia stricta	-0.181	0.033	-0.148	-0.097	0.009	-0.069	0.036	0.001	0.069
Ornithogalum gracilium	0.21	0.044	0.027	0.007	0	0.086	-0.102	0.01	0.068
Oxygonum sinuatum	0.169	0.029	0.153	0.503	0.253	0.301	-0.001	0	0.07
Pennisetum mezianum	0.038	0.001	0.049	0.078	0.006	0.109	0.082	0.007	0.168
Pennisetum stramineum	0.597	0.356	0.527	-0.458	0.21	-0.326	0.364	0.133	0.135
Phyllanthus ovalifolius	0.062	0.004	0.111	0.096	0.009	0.1	-0.29	0.084	-0.334
Plectranthus barbatus var. grandis	0.253	0.064	0.141	0.278	0.077	0.004	-0.106	0.011	-0.091
Pollichia campestris	-0.488	0.238	-0.336	-0.162	0.026	-0.082	-0.14	0.02	-0.006
Polygala sp.	0.454	0.206	0.362	-0.075	0.006	0.004	-0.06	0.004	-0.028
Portulaca oleracea	-0.183	0.034	-0.173	-0.144	0.021	0.024	-0.368	0.136	-0.335
Portulaca quadrifida	-0.095	0.009	-0.091	-0.362	0.131	-0.287	-0.474	0.225	0.026
Ruellia patula	-0.254	0.065	-0.07	-0.356	0.127	-0.282	0.27	0.073	0.321
Sansevieria frequens	0.231	0.053	0.233	-0.095	0.009	-0.134	-0.089	0.008	-0.092
Sansevieria robusta	0.415	0.172	0.333	-0.299	0.089	-0.188	0.468	0.219	0.231
Sansevieria sp. 1	0.195	0.038	0.208	-0.008	0	-0.01	-0.107	0.011	-0.148
Sansevieria volkensii	-0.321	0.103	-0.18	-0.476	0.226	-0.432	0.193	0.037	0.213
Sarcostemma viminale	0.352	0.124	0.233	-0.19	0.036	-0.148	-0.157	0.025	-0.115
Schkuhria pinnata	-0.181	0.033	-0.148	-0.097	0.009	-0.069	0.036	0.001	0.069
Sericocomopsis pallida	0.101	0.01	0.092	-0.064	0.004	-0.12	0.069	0.005	0.177
Solanum coagulans	-0.199	0.04	-0.039	-0.066	0.004	-0.111	-0.043	0.002	0.011
Solanum incanum	-0.058	0.003	-0.134	0.649	0.422	0.418	-0.128	0.016	-0.107
Solanum sp. 1	-0.007	0	-0.102	0.02	0	-0.017	-0.077	0.006	0.068
Solanum sp. 2	-0.168	0.028	-0.176	0.12	0.014	0.009	-0.094	0.009	-0.092
Solanum sp. 3	0.338	0.114	0.228	-0.111	0.012	-0.089	-0.171	0.029	-0.188
Sporobolus pyramidalis	0.034	0.001	0.085	0.156	0.024	0.077	0.103	0.011	0.093
Tragus berteronianus	0.166	0.028	0.134	0.675	0.455	0.402	-0.01	0	0.07
Tribulus terestris	0.252	0.063	0.184	0.129	0.017	0.028	-0.222	0.049	-0.122
Unknown A	-0.107	0.011	0.023	0.297	0.088	0.196	-0.163	0.027	-0.15
Unknown AA	0.35	0.122	0.29	-0.019	0	0.007	-0.029	0.001	-0.021
Unknown AB	-0.137	0.019	-0.129	-0.035	0.001	-0.03	-0.1	0.01	-0.129
Unknown AC	-0.299	0.09	-0.267	0.047	0.002	0.069	-0.218	0.048	-0.247
Unknown AD	0.338	0.114	0.228	-0.111	0.012	-0.089	-0.171	0.029	-0.188
Unknown B	-0.09	0.008	-0.109	0.234	0.055	0.208	-0.029	0.001	-0.03
Unknown C	0.338	0.114	0.228	-0.111	0.012	-0.089	-0.171	0.029	-0.188
Unknown D	-0.014	0	0.03	-0.133	0.018	-0.129	0.254	0.065	0.208
Unknown F	-0.267	0.071	-0.228	0.128	0.016	0.148	0.043	0.002	0.089
Unknown G	-0.09	0.008	-0.109	0.234	0.055	0.208	-0.029	0.001	-0.03
Unknown H	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Unknown I	-0.192	0.037	-0.064	0.045	0.002	0.076	-0.204	0.042	-0.17
Unknown J	0.188	0.035	0.188	0.051	0.003	0.089	0.08	0.006	0.148
Unknown K	-0.194	0.038	-0.168	-0.164	0.027	-0.148	0.004	0	-0.01
Unknown L	-0.075	0.006	-0.069	0.118	0.014	0.129	-0.214	0.046	-0.228
Unknown M	-0.023	0.001	-0.01	0.28	0.078	0.228	0.007	0	0.01
Unknown N	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208

Unknown O	0.344	0.118	0.314	-0.152	0.023	-0.058	0.237	0.056	0.058
Unknown P	-0.075	0.006	-0.069	0.118	0.014	0.129	-0.214	0.046	-0.228
Unknown R	0.188	0.035	0.168	0.496	0.246	0.267	-0.044	0.002	-0.069
Unknown S	0.188	0.035	0.168	0.496	0.246	0.267	-0.044	0.002	-0.069
Unknown T	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Unknown U	0.17	0.029	0.114	0.054	0.003	0.134	-0.55	0.302	-0.334
Unknown V	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Unknown W	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Unknown X	0.369	0.136	0.267	0.324	0.105	0.247	-0.179	0.032	-0.208
Unknown Y	-0.288	0.083	-0.247	0.226	0.051	0.188	0.337	0.113	0.247
Unknown Z	0.338	0.114	0.228	-0.111	0.012	-0.089	-0.171	0.029	-0.188
Unkown E	0.038	0.001	0.049	0.078	0.006	0.109	0.082	0.007	0.168
Unkown Q	-0.176	0.031	-0.064	0.43	0.185	0.318	0.286	0.082	0.134
Ximenia americana	0.195	0.038	0.208	-0.008	0	-0.01	-0.107	0.011	-0.148
Zaleya pentadra	0.153	0.023	0.081	0.202	0.041	0.186	-0.006	0	-0.032

APPENDIX R

CORRELATION BETWEEN NDVI DIFFERENCE AND FIRST THREE PRINCIPLE COMPONENTS



APPENDIX S

CHAPTER 5 MULTIPLE REGRESSION R CODE

```
library(tibble)
library(rcompanion)
library(splitstackshape)
library(lmtest)
library(car)
library(AICcmodavg)
library(SDMTools)
library(ROCR)
library(raster)
library(rgdal)
library(caret)

in_dir <- "C:/Users/ryanunks/Desktop/SEM/StackForNateLab"
LC <- list.files(path=in_dir, pattern="*.tif",full.names=T)
LCC_stack <- stack(LC)
#v <- getValues(LCC_stack)


Koija.poly <- rasterToPoints(LCC_stack)
Koija.poly <- as.data.frame(Koija.poly)
Koija.poly <- SpatialPointsDataFrame(Koija.poly[1:3], Koija.poly[4:16], coords.nrs = numeric(0), proj4string = CRS(as.character(NA)),
match.ID = TRUE)


set.seed(1)
head(Koija.poly)

#create a stratified random sample, then create a subset according to 1987 landcover type
Koija.poly.sub <- stratified(Koija.poly, c("X1987hkmeans37"), 200)
Koija.poly.sub <- subset(Koija.poly.sub, X1987hkmeans37==21)

#create spatial covariate layer
rm(coords)
coords<-coordinates(Koija.poly.sub)
head(coords)
coords<-coords[,14:15]
IDs<-row.names(as(Koija.poly.sub, "data.frame"))
koija_kd1<-dnearneigh(coords, 30, 2000, row.names=IDs)
koija_kd1_w<- nb2listw(koija_kd1)

#examine normal quantile plots and test transformations
qqPlot((Koija.poly.sub$NDVI1987min2013))
qqPlot((Koija.poly.sub$LCC_PC1))
qqPlot(sqrt((Koija.poly.sub$NDVI1987min2013+0.00001)-min(Koija.poly.sub$NDVI1987min2013)))
qqPlot(log((Koija.poly.sub$NDVI1987min2013+0.00001)-min(Koija.poly.sub$NDVI1987min2013)))
qqPlot(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)
qqPlot(log((Koija.poly.sub$LCC_PC1+1)-min(Koija.poly.sub$LCC_PC1)))
plotNormalHistogram((Koija.poly.sub$NDVI1987min2013))

#ordinary least squares
lr01 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr02 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
```

```
lr03 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr04 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)
lr05 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(ElevationUTM)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr06 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(ElevationUTM)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
lr07 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(ElevationUTM)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr08 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(ElevationUTM)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)
lr09 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(P99AllDryEntareResampled),
data=Koija.poly.sub)
lr10 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(P99AllWetEntareResampled),
data=Koija.poly.sub)
lr11 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(P99AllDryCattleResampled),
data=Koija.poly.sub)
lr12 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(P99AllWetCattleResampled),
data=Koija.poly.sub)
lr13 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM),
data=Koija.poly.sub)
lr14 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient), data=Koija.poly.sub)
lr15 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(ElevationUTM), data=Koija.poly.sub)
lr16 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope), data=Koija.poly.sub)
lr17 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(ElevationUTM),
data=Koija.poly.sub)
lr18 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(SteepSlope),
data=Koija.poly.sub)
lr19 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(SteepSlope)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr20 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(SteepSlope)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
lr21 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(SteepSlope)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr22 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(SteepSlope)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)
lr23 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr24 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
lr25 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr26 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)
lr27 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr28 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
lr29 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr30 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)
lr31 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(ElevationUTM)+scale(P99AllDryEntareResampled), data=Koija.poly.sub)
lr32 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(ElevationUTM)+scale(P99AllWetEntareResampled), data=Koija.poly.sub)
lr33 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(ElevationUTM)+scale(P99AllDryCattleResampled), data=Koija.poly.sub)
lr34 <- lm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(SteepSlope)+scale(ElevationUTM)+scale(P99AllWetCattleResampled), data=Koija.poly.sub)

#model selection procedure
lrModels<- list(lr01, lr02, lr03, lr04, lr05, lr06, lr07, lr08, lr09, lr10, lr11, lr12, lr13, lr14, lr15, lr16, lr17, lr18, lr19, lr20, lr21, lr22, lr23, lr24,
lr25, lr26, lr27, lr28, lr29, lr30, lr31, lr32, lr33, lr34)
lrNames <- c("Model1", "Model2", "Model3", "Model4", "Model5", "Model6", "Model7", "Model8", "Model9", "Model10", "Model11",
"Model12", "Model13", "Model14", "Model15", "Model16", "Model17", "Model18", "Model19", "Model20", "Model21", "Model22",
"Model23", "Model24", "Model25", "Model26", "Model27", "Model28", "Model29", "Model30", "Model31", "Model32", "Model33",
"Model34")
aicWt<-aictab(cand.set=lrModels, modnames=lrNames, sort=TRUE, c.hat=1)
```

```
aicWt

#regression results with correlation of coefficients
koija.ols <- lr12
summary(koija.ols, correlation = TRUE)

#calculate variance inflation factors
vif(koija.ols)

#calculate moran's i
lm.morantest(koija.ols, koija_kd1_w)

#LaGrange tests for type of spatial dependence
lm.LMtests(koija.ols, koija_kd1_w, test="all")

#calculate Breusch-Pagan
bptest(koija.ols)

#check graphically for heteroskedacticity
par(mfrow=c(2,2)) # init 4 charts in 1 panel
plot(koija.ols)

#calculate nagelkerke R-sq for ols
nagelkerke(koija.ols, null = NULL, restrictNobs = FALSE)

#BoxCox transformation
trans <- (((Koija.poly.sub$LCC_PC1+0.0001)-min(Koija.poly.sub$LCC_PC1))^2)
NDUPBCMod <- caret::BoxCoxTrans(trans)
print(NDUPBCMod)
BCM_new<-predict(NDUPBCMod, trans)
Koija.poly.sub[["transf_var"]]=BCM_new


#spatial error model example
lr01 <- errorsarlm(scale(((Koija.poly.sub$LCC_PC1+0.0001)-
min(Koija.poly.sub$LCC_PC1))^2)~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllDryEntareResampled), data=Koija.poly.sub,
listw=koija_kd1_w)

#spatial lag model example
lr01 <- lagsarlm(scale(log((Koija.poly.sub$NDVI1987min2013+0.00001)-
min(Koija.poly.sub$NDVI1987min2013)))~scale(RainGradient)+scale(ElevationUTM)+scale(P99AllDryEntareResampled),
data=Koija.poly.sub, listw=koija_kd1_w)

#spatial error model results
summary(koija.sem, correlation=TRUE, Nagelkerke=TRUE)

#calculate moran's i following model fitting
Koija.poly.sub$residuals <- residuals(koija.sem)
moran.mc(Koija.poly.sub$residuals, koija_kd1_w, 999)

#studentized Breusch-Pagan test, tests for heteroskedasticity
bptest.sarlm(koija.sem)
```

APPENDIX T

CHAPTER 5 MULTIPLE REGRESSION RESULTS

1987 Vegetation Class	1	1	3	3	4	4	5	5	6	6	7	7	8	8	9	9	10	10
Response Variable	PC1	NDVI	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1
Transformation	None	None	None	None	None	None	None	None	Square Root	None	None	None	None	None	None	None	Box-cox	None
Autoregressive Model	Spatial Lag	None	None	None	None	None	None	Spatial Error	None	None	None	None	None	None	None	None	None	None
Elevation β	-0.12									-0.10						0.21	-0.25	
Elevation Standard Error	0.07									0.07						0.07	0.07	
Elevation *p*-value	0.003									0.147						0.002	0.000	
Elevation Standard Error	0.10							0.27					-0.45	0.39	-0.34			
Rain β								0.07					0.07	0.06	0.07			
Rain *p*-value								0.000					0.000	0.000	0.000			
Slope β			-0.05	0.04	-0.07	0.02	0.05				-0.01							
Slope Standard Error			0.06	0.06	0.07	0.07	0.07				0.06							
Slope *p*-value			0.388	0.493	0.317	0.763	0.475				0.897							
Dry Season Small Stock β	-0.16	0.17										-0.49						
Dry Small Stock Standard Error	0.07	0.07										0.06						
Dry Small Stock *p*-value	0.022	0.015										0.000						
Wet Season Small Stock β													-0.11		0.15	-0.21		
Wet Small Stock Standard Error													0.07		0.07	0.07		
Wet Season *p*-value													0.086		0.027	0.002		
Dry Season Cattle β			0.47	-0.48	0.15	-0.25	0.12		0.22								0.33	-0.38
Dry Season Cattle Standard Error			0.06	0.06	0.07	0.07	0.07		0.07								0.07	0.07
Dry Season Cattle *p*-value			0.000	0.000	0.042	0.001	0.094		0.002								0.000	0.000
Wet Season Cattle β										-0.22	0.46			-0.20				
Wet Season Cattle Standard Error										0.07	0.06			0.06				
Wet Season Cattle *p*-value										0.002	0.000			0.003				
Nagelkerke's pseudo R-squared	0.06							0.08										
Multiple R-squared		0.03	0.23	0.23	0.03	0.06	0.02		0.05	0.06	0.21	0.24	0.20	0.22	0.15	0.10	0.13	0.15
Adjusted R-squared		0.02	0.22	0.23	0.02	0.05	0.01		0.04	0.05	0.20	0.24	0.19	0.21	0.14	0.09	0.12	0.14
F		6.00	29.47	30.08	2.98	6.52	1.71		9.89	6.53	26.01	62.96	24.34	27.43	17.61	10.67	14.83	34.07
DF		1 and 198	2 and 197	2 and 197	2 and 197	2 and 197	2 and 197		1 and 198	2 and 197	2 and 197	1 and 198	2 and 197	2 and 197	2 and 197	2 and 197	2 and 197	1 and 198
Model *p*-value	0.064	0.015	0.000	0.000	0.053	0.002	0.184	0.115	0.002	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
R-squared Model Without Rain													0.00	0.08	0.04			
Model Without Rain *p*-value													0.758	0.000	0.006			
R-squared Model Without Elevation	0.05									0.05						0.05	0.07	
Model Without Elevation *p*-value	0.031									0.001						0.001	0.000	
R-squared Model Without Cattle			0.01	0.01		0.00				0.01	0.00			0.18			0.03	
Model Without Cattle *p*-value			0.156	0.205		0.423				0.101	0.652			0.000			0.014	
R-squared Model Without Small Stock	0.03												0.19		0.13	0.05		
Model Without Small Stock *p*-value	0.036												0.000		0.000	0.001		
R-squared Model Without Slope			0.23	0.23	0.02	0.06					0.21							
Model Without Slope *p*-value			0.000	0.000	0.027	0.000					0.000							

1987 Vegetation Class	11	11	12	12	13	13	14	14	15	15	16	16	17	17	18	18	19	19
Response Variable	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1
Transformation	None	None	None	None	None	None	None	None	None	Box-cox	None	None	None	None	None	Box-cox	None	None
Autoregressive Model	None	None	None	None	None	None	None	None	None	None	None	None	Spatial Error	Spatial Error	None	None	None	None
Elevation β	-0.19			0.10			-0.18										0.20	-0.26
Elevation Standard Error	0.07			0.07			0.07										0.07	0.07
Elevation *p*-value	0.008			0.134			0.007										0.006	0.000
Elevation Standard Error											-0.40	0.28	-0.21	0.12	-0.33	0.39		
Rain β											0.06	0.07	0.07	0.07	0.07	0.07		
Rain *p*-value											0.000	0.000	0.003	0.087	0.000	0.000		
Slope β																-0.11		
Slope Standard Error																0.07		
Slope *p*-value																0.098		
Dry Season Small Stock β	0.18	-0.15																
Dry Small Stock Standard Error	0.07	0.07																
Dry Small Stock *p*-value	0.012	0.037																
Wet Season Small Stock β																		
Wet Small Stock Standard Error																		
Wet Season *p*-value																		
Dry Season Cattle β			0.31	-0.38	0.18		0.35	-0.42	0.26	-0.26							0.18	-0.25
Dry Season Cattle Standard Error			0.07	0.07	0.07		0.07	0.06	0.07	0.07							0.07	0.07
Dry Season Cattle *p*-value			0.000	0.000	0.011		0.000	0.000	0.000	0.000							0.011	0.000
Wet Season Cattle β						-0.16					0.16	-0.24		-0.15				
Wet Season Cattle Standard Error						0.07					0.06	0.07		0.07				
Wet Season Cattle *p*-value						0.020					0.016	0.000		0.034				
Nagelkerke's pseudo R-squared													0.06	0.07				
Multiple R-squared	0.07	0.02	0.10	0.17	0.03	0.03	0.13	0.18	0.07	0.07	0.21	0.16			0.11	0.16	0.06	0.10
Adjusted R-squared	0.06	0.02	0.09	0.16	0.03	0.02	0.12	0.17	0.06	0.07	0.20	0.15			0.11	0.15	0.05	0.09
F	7.81	4.41	21.36	19.49	6.52	5.53	15.17	42.69	14.02	14.91	25.41	18.34			24.40	18.30	5.77	10.80
DF	2 and 197	1 and 198	1 and 198	2 and 197	1 and 198	1 and 198	2 and 197	1 and 198	1 and 198	1 and 198	2 and 197	2 and 197			1 and 198	2 and 197	2 and 197	2 and 197
Model *p*-value	0.001	0.037	0.000	0.000	0.011	0.020	0.000	0.000	0.000	0.000	0.000	0.000			0.000	0.000	0.004	0.000
R-squared Model Without Rain											0.05	0.08				0.01		
Model Without Rain *p*-value											0.002	0.000	0.213	0.087		0.283		
R-squared Model Without Elevation	0.04			0.16			0.10										0.02	0.04
Model Without Elevation *p*-value	0.005			0.000			0.000										0.055	0.008
R-squared Model Without Cattle				0.02			0.02				0.18	0.10					0.02	0.04
Model Without Cattle *p*-value				0.030			0.081				0.000	0.000					0.029	0.004
R-squared Model Without Small Stock	0.04																	
Model Without Small Stock *p*-value	0.003																	
R-squared Model Without Slope																0.14		
Model Without Slope *p*-value																0.000		

1987 Vegetation Class	20	20	21	21	22	22	23	23	24	24	25	25	26	26	27	27	28	28
Response Variable	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1
Transformation	None	None	Log	Squared	Box-cox	Box-cox	None	None	Square Root	Squared	Squared	Squared	None	Squared	Cubed	None	Box-cox	Box-cox
Autoregressive Model	None	None	Spatial Error	Spatial Error	None	None	None	None	Spatial Error	Spatial Error	None	None	None	None	Spatial Error	Spatial Error	None	None
Elevation β		-0.12	-0.17														0.26	-0.31
Elevation Standard Error		0.07	0.07														0.07	0.07
Elevation *p*-value		0.088	0.014														0.000	0.000
Elevation Standard Error					0.28	-0.24			0.44						-0.14			
Rain β					0.07	0.07			0.06						0.07			
Rain *p*-value					0.000	0.001			0.000						0.037			
Slope β							0.12	-0.12		0.10								
Slope Standard Error							0.07	0.07		0.07								
Slope *p*-value							0.098	0.077		0.147								
Dry Season Small Stock β																		
Dry Small Stock Standard Error																		
Dry Small Stock *p*-value																		
Wet Season Small Stock β													0.39	-0.42		-0.25		
Wet Small Stock Standard Error													0.07	0.06		0.07		
Wet Season *p*-value													0.000	0.000		0.000		
Dry Season Cattle β	0.26	-0.31					0.13			-0.27	0.23	-0.32			-0.24			
Dry Season Cattle Standard Error	0.07	0.07					0.07			0.07	0.07	0.07			0.06			
Dry Season Cattle *p*-value	0.000	0.000					0.062			0.000	0.001	0.000			0.000			
Wet Season Cattle β				-0.25				-0.19										
Wet Season Cattle Standard Error				0.07				0.07										
Wet Season Cattle *p*-value				0.000				0.006										
Nagelkerke's pseudo R-squared			0.09	0.10					0.18	0.12					0.06	0.07		
Multiple R-squared	0.07	0.10			0.08	0.06	0.03	0.05			0.05	0.10	0.15	0.17			0.07	0.10
Adjusted R-squared	0.06	0.09			0.07	0.05	0.02	0.04			0.05	0.10	0.15	0.17			0.06	0.09
F	13.86	11.05			16.28	12.19	2.82	4.96			11.20	22.91	36.20	41.32			13.92	21.21
DF	1 and 198	2 and 197			1 and 198	1 and 198	2 and 197	2 and 197			1 and 198	1 and 198	1 and 198	1 and 198			1 and 198	1 and 198
Model *p*-value	0.000	0.000	0.039	0.020	0.000	0.001	0.062	0.008	0.413	0.025	0.001	0.000	0.000	0.000	0.146	0.169	0.000	0.000
R-squared Model Without Rain									0.05						0.04			
Model Without Rain *p*-value									0.339						0.179			
R-squared Model Without Elevation		0.07																
Model Without Elevation *p*-value		0.000																
R-squared Model Without Cattle		0.00						-0.10		0.05					0.00			
Model Without Cattle *p*-value		0.976						0.141		0.013					0.990			
R-squared Model Without Small Stock																		
Model Without Small Stock *p*-value																		
R-squared Model Without Slope								0.03	0.21	0.11								
Model Without Slope *p*-value								0.010	0.139	0.014								

1987 Vegetation Class	29	29	30	30	31	31	32	32	33	33	34	34	35	35	36	36	37	37
Response Variable	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1	NDVI	PC1
Transformation	None	None	None	None	Squared	None	None	None	None	None	None	None	Log	Box-cox	None	None	Square Root	Squared
Autoregressive Model	None	None	None	None	None	None	None	None	None	None	None	None	Spatial Lag	None	None	None	None	None
Elevation β															0.14	-0.24		
Elevation Standard Error															0.07	0.06		
Elevation *p*-value															0.034	0.000		
Elevation Standard Error			-0.26		-0.23	0.20			0.00		-0.26	0.16	0.23	-0.21				
Rain β			0.07		0.07	0.07			0.06		0.07	0.07	0.07	0.07				
Rain *p*-value			0.000		0.001	0.004			-0.543		0.000	0.026	0.000	0.002				
Slope β		0.10												0.12		0.11		
Slope Standard Error		0.07												0.07		0.06		
Slope *p*-value		0.167												0.068		0.076		
Dry Season Small Stock β												-0.20						-0.30
Dry Small Stock Standard Error												0.07						0.07
Dry Small Stock *p*-value												0.004						0.000
Wet Season Small Stock β							-0.17											
Wet Small Stock Standard Error							0.07											
Wet Season *p*-value							0.017											
Dry Season Cattle β								-0.07		-0.53	0.14				0.35	-0.37	0.19	
Dry Season Cattle Standard Error								0.07		0.06	0.07				0.07	0.06	0.07	
Dry Season Cattle *p*-value								0.301		0.000	0.049				0.000	0.000	0.007	
Wet Season Cattle β	-0.23			-0.27									0.16	-0.33				
Wet Season Cattle Standard Error	0.07			0.07									0.07	0.06				
Wet Season Cattle *p*-value	0.001			0.000									0.016	0.000				
Nagelkerke's pseudo R-squared													0.11					
Multiple R-squared	0.05	0.01	0.07	0.07	0.05	0.04	0.03	0.01	0.29	0.29	0.08	0.06		0.21	0.14	0.22	0.04	0.09
Adjusted R-squared	0.05	0.00	0.06	0.07	0.05	0.04	0.02	0.00	0.29	0.28	0.07	0.05		0.20	0.14	0.21	0.03	0.08
F	10.66	1.93	13.94	16.00	10.83	8.35	5.84	1.08	82.74	79.14	8.81	6.21		17.47	16.68	18.13	7.43	19.34
DF	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	1 and 198	2 and 197	2 and 197		3 and 196	2 and 197	3 and 196	1 and 198	1 and 198
Model *p*-value	0.001	0.167	0.000	0.000	0.001	0.004	0.017	0.301	< 2.2e-16	0.000	0.000	0.002	0.097	0.000	0.000	0.000	0.007	0.000
R-squared Model Without Rain													0.06	0.17				
Model Without Rain *p*-value													0.070	0.000				
R-squared Model Without Elevation															0.13	0.16		
Model Without Elevation *p*-value															0.000	0.000		
R-squared Model Without Cattle													0.09	0.10	0.02	0.08		
Model Without Cattle *p*-value													0.070	0.000	0.050	0.000		
R-squared Model Without Small Stock																		
Model Without Small Stock *p*-value																		
R-squared Model Without Slope														0.20		0.20		
Model Without Slope *p*-value														0.000		0.000		