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Christian Kiffner · Monica L. Bond ·
Derek E. Lee *Editors*

Tarangire: Human-Wildlife Coexistence in a Fragmented Ecosystem

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Part I
Introduction

Chapter 1
Human-Wildlife Interactions in the Tarangire Ecosystem

Christian Kiffner, Monica L. Bond, and Derek E. Lee

Abstract For millennia, people have lived alongside wildlife in the semi-arid savanna of the Tarangire Ecosystem (TE), northern Tanzania. The TE preserves one of the last long-distance wildlife migrations in Africa as well as a large and diverse human population. Initial wildlife conservation approaches, settlement politics, and changes in human livelihoods have created a fragmented coupled social-ecological system that currently faces serious challenges for both people and wildlife. In this introduction to the book "Tarangire: Human-Wildlife Coexistence in a Fragmented Ecosystem" we outline the environmental and climatic settings as well as the social, economic, and political structures and histories of the ecosystem. The combination of heterogeneous geology, variable rainfall, a historical focus on conserving dry-season ranges of wildlife, and an expanding human population brings people and wildlife in contact, often with negative consequences for humans and wildlife. From an anthropocentric perspective, large carnivores and elephants are perceived as particularly problematic. In this book, we adopt a social-ecological approach and present different perspectives on wildlife conservation in the TE as frameworks for integrated and effective solutions. The first section of the book addresses the human dimension in human-wildlife interactions, whereas the second section employs a more ecocentric perspective and summarizes the status and ecologies of key large-

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mammal populations in the TE. The third section addresses human-wildlife interactions explicitly with an eye towards solutions.

Keywords Coupled socio-ecological systems · Human-wildlife conflict · Human-wildlife coexistence · Tarangire Manyara Ecosystem · Maasai Steppe

1.1 Human-Wildlife Interactions Through a Diverse Set of Lenses

The aim of this book is to draw together human-centered, wildlife-centered, and interdisciplinary research in the Tarangire Ecosystem of northern Tanzania to understand the challenges, mechanisms, and processes underlying interactions between humans and wildlife in this dynamic landscape. The Tarangire Ecosystem (also known as the Maasai Ecosystem, Maasai Steppe, and Tarangire-Manyara Ecosystem) is a geographical location (Fig. 1.1) with diverse plant and animal communities and a human history going back to early hominins (Lamprey 1963; Keller et al. 1975; Prins 1987). By synthesizing case studies from multiple academic disciplines we highlight challenges of and options for shaping sustainable human-wildlife coexistence in terrestrial landscapes with growing human populations.

Wildlife conservation and human livelihoods in East Africa are facing complex challenges that can only be understood and eventually solved by addressing the interdependencies among humans, wildlife, and the environment. This book presents diverse perspectives on human-wildlife interactions in one well-studied, but often overlooked ecosystem. Similar books on this topic focused either on coupled socio-ecological systems in the neighboring and famous Serengeti ecosystem which represents a continuous ecosystem (Sinclair et al. 2015), are based on a single author's perspective (Reid 2012), or presented selected examples of human-wildlife interactions across the globe (Frank et al. 2019). This book focuses on a fragmented landscape and addresses theoretical and practical aspects of human-wildlife interactions from a range of perspectives and thus offers an integrated and pluralistic perspective on biodiversity conservation (Pascual et al. 2021). We posit that this book is a timely effort to summarize interdisciplinary aspects of wildlife conservation in the Tarangire Ecosystem, and we hope the resulting synthesis may prove helpful for guiding inter- and transdisciplinary conservation efforts in other fragmented coupled human-natural ecosystems (Liu et al. 2007).

Across the globe, human activities are causing unprecedented declines in biodiversity (Sala et al. 2000; Dirzo et al. 2014; Ceballos et al. 2017; IPBES 2019; IPCC 2019; Bradshaw et al. 2021). Biodiversity declines are particularly evident among global populations of large-bodied "megafauna" (i.e. mammal species >100 kg of body mass) (Ripple et al. 2016) of both herbivorous (Ripple et al. 2015) and carnivorous species (Ripple et al. 2014). East Africa is no exception to this worrisome global pattern. From 1970 to 2005, an aggregated index of wildlife populations in 43 protected areas of East Africa declined significantly over time, with populations in 2005 being reduced to approximately half of the 1970 baseline (Craigie et al. 2010).

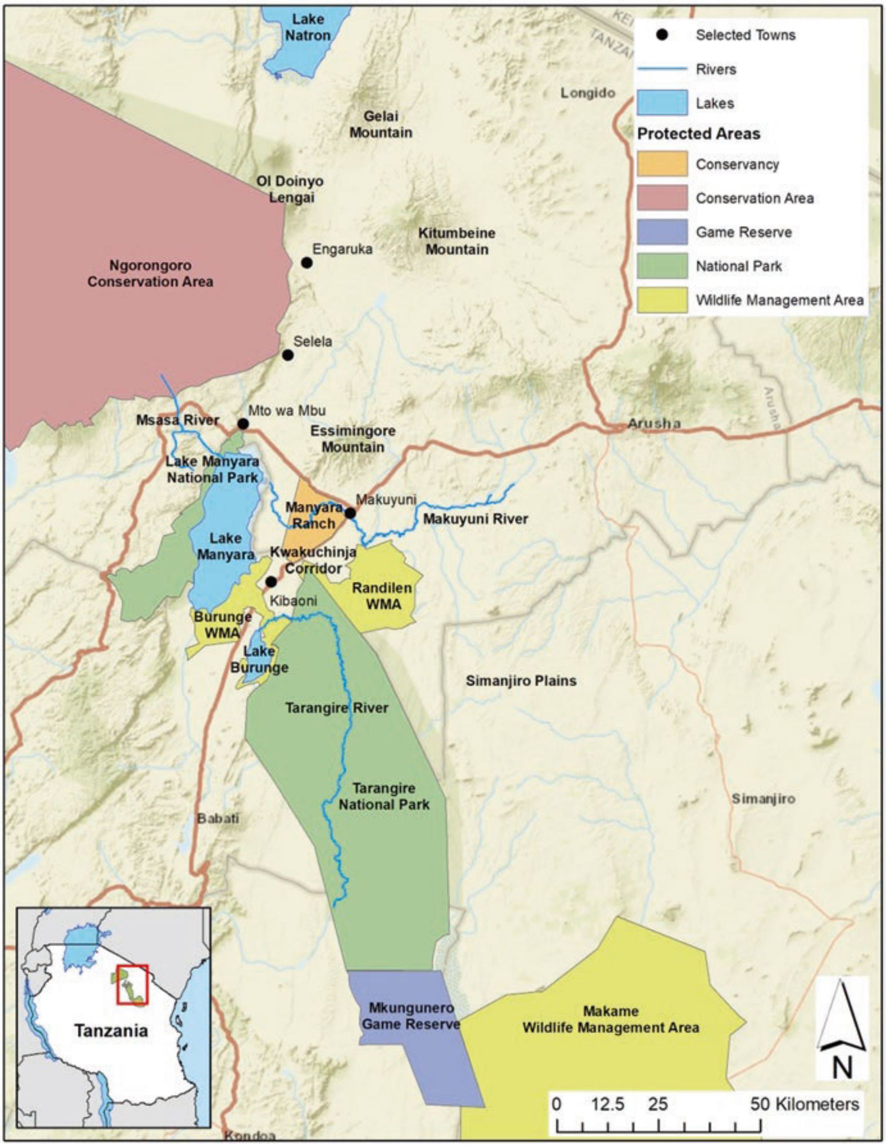

Fig. 1.1 Map of the Tarangire Ecosystem outlining the main protected areas (Lake Manyara and Tarangire National Parks; Mkungunero Game Reserve; Burunge, Makame, and Randilen Wildlife Management Areas; Manyara Ranch), the topography, rivers, district boundaries (gray lines), main roads (dark red lines) and main towns (Map created by Jason Riggio)

Box 1.1: The Evolution of Human-Wildlife Interactions

Wherever humans live on this planet, we constantly interact with wild animals. In one way or another, human-wildlife interactions shape human cultures, animal communities, ecosystem functioning, and species evolution. Wild animals are major protagonists of human evolution from early hominins to modern humans (Shipman 2010). Our ancestors did – and many contemporary humans still do – hunt wild animals for protein, fur, and bones; fear and kill carnivores and other taxa that threaten their safety and wellbeing; compete with wild animals for resources; and domesticate wild species to make use of their meat and products (Stringer and Andrews 2011). The diverse influences of wild animals on human life likely exerted a key selection pressure for the evolution of tool making, the control of fire, and symbolic behavior and language (Shipman 2010; Pontzer 2012). In turn, these new skills also allowed humans to domesticate plants and animals. One could thus argue that animals and human-animal relationships were a major driver of the Neolithic revolution, cultural development, and human evolution as a whole (Shipman 2010). However, this coevolution does not necessarily mean sustainable coexistence.

As much as animals impose selective pressures on human evolution, those very skills that largely differentiate us as humans from other mammal species (i.e. tool making, control of fire, symbolic behavior and language, domestication of animals), exert strong selection pressures on wild animal species. Humans have invented advanced tools to hunt wild animals, developed skills and machinery to convert large tracts of land into agriculture, introduced and kept domestic species at high densities, and emitted greenhouse gases at unprecedented rates. These human modifications of our planet cause biodiversity loss, alter the climate, and ultimately impair the ecosystem services on which human life depends (Sala et al. 2000; IPBES 2019; IPCC 2019; Bradshaw et al. 2021). Across the globe, human impacts are now a major driving force of ecosystem processes and our activities have far-reaching direct and indirect impacts on the distribution, abundance, and interactions of species as well as overall ecosystem functioning (Estes et al. 2011; Dirzo et al. 2014; Ripple et al. 2015; Young et al. 2016). Anthropogenic activities strongly impact ecosystems and associated functioning and (dis-) services. Ironically, yet consistent with the ingrained ties between the environment and humans (Díaz et al. 2015), biodiversity loss and subsequent changes in ecosystem processes and functioning has tremendous repercussions on humanity itself (Cardinale et al. 2012). Several wild animal species can cope relatively well in human-dominated landscapes, and there is evidence that densities of mammal species are positively scaled with the human footprint (Tucker et al. 2020). In turn, species well adapted to human-modified landscapes are often perceived as “nuisances”, “pests”, or “vermin” because they feed on crops, kill and feed on livestock, transmit pathogens, or are a threat to human wellbeing and life (Nyhus 2016) and thus impact human livelihoods in many ways.

The main strategy to stop or counteract wildlife declines is to delineate and establish protected areas (Geldmann et al. 2013; Coetzee et al. 2014; Lindsey et al. 2014; Dinerstein et al. 2017). Protected areas can be part of a land-sparing approach (Grass et al. 2019), that divides the land into zones for wildlife where human land use is restricted (e.g. national parks or other protected areas) and other zones where a variety of human land uses are allowed and little attention is paid to the needs of wildlife. Several conservationists are calling for half the area of every biome on Earth to be protected to avert the looming biodiversity and climate disasters (Kopnina 2016; Dinerstein et al. 2017; Kopnina et al. 2018). In a best-case scenario, a land-sparing approach would include land sharing and human-wildlife coexistence as guiding principles to ensure that wildlife and human populations are sufficiently separated to maintain ecosystem functioning and resource accessibility for both human and wildlife populations.

However, land sparing and protected areas can result in "fortress conservation" models (Brockington 2002) that can have multiple negative implications for both people and wildlife. Where people are evicted from areas or denied access to natural resources they had used in the past, serious harm to human wellbeing can result and can be a root cause for non-compliance with environmental laws (Goldman 2011; Reid 2012). Protected areas are also often too small to maintain viable wildlife populations, especially for wide-ranging animal species (Newmark 1996; Woodroffe and Ginsberg 1998; Fynn and Bonyongo 2011). Thus, a strict land-sparing approach could be a lose-lose situation that impairs human livelihoods and wellbeing as well as reduces the population viability of wildlife species and ecosystem functioning. While people are often the root cause of the biodiversity crisis (Soulé 1985), they could also be the solutions to these problems. Indeed, scholars have long realized that it is necessary to plan and manage landscapes in ways that work for people and biodiversity (Reid 2012; Kremen and Merenlender 2018). For such landscape planning and management to be effective, we believe that a place-based, social-ecological approach that illuminates the interactions between and within different human stakeholder groups and corresponding governance systems (i.e. the human system), the feedbacks between wildlife and other ecosystem components (i.e. the wildlife system), and the reciprocal and manifold couplings between the two systems is an essential prerequisite (Ostrom 2009; McGinnis and Ostrom 2014). As both the human and the wildlife systems are embedded in the context of the environmental and climatic settings of the Tarangire Ecosystem and the social, economic, and political histories and structures of the human population, we start with summarizing these aspects.

1.2 The Tarangire Ecosystem (TE)

The Tarangire River, one of the few year-round sources of fresh water in this arid to semi-arid savanna environment, gives the TE its name (Fig. 1.1). The TE can be defined by watershed boundaries of the Lake Manyara Basin and the Engaruka Basin, and the migratory ranges of the Tarangire populations of eastern white-bearded wildebeest (*Connochaetes taurinus albojubatus*) and plains zebra (*Equus quagga*) from their dry-season refuge along the Tarangire River north to Lake

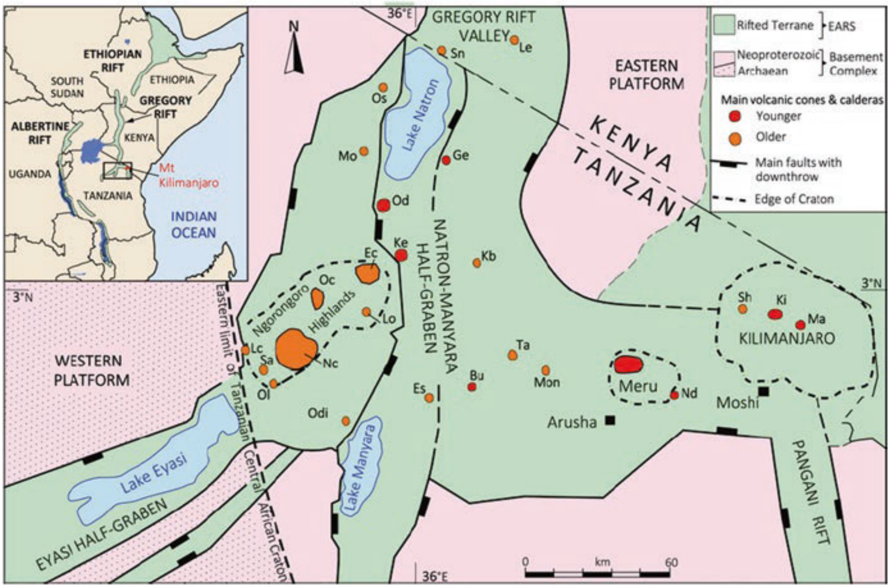

Fig. 1.2 Geology of the Tarangire Ecosystem (From Scoon 2018a)

Natron, east to the Simanjiro plains, and south on the Maasai Steppe (Lamprey 1964; Kahurananga and Silkiluwasha 1997). The TE encompasses roughly 30,000 km^2 situated at the southern end of the Gregory Rift, the eastern branch of the East African Rift System. Latitude is 2°S to 5°S, longitude is 35°E to 37°E, and elevation ranges from 950 m at Lake Manyara to 3000 m at its boundaries in the Ngorongoro Highlands.

Mean total annual rainfall at the Tarangire River was 650 mm for the years 1980–2009, with a coefficient of variation = 42.6% and range = 312–1398 mm (Foley and Foley 2014). There are three Indian Ocean monsoon-driven precipitation seasons per year [short rains = Oct–Jan, long rains = Feb–May, and dry season = Jun–Sep; (Prins and Loth 1988)]. Average monthly precipitation at the Tarangire River by season was: short rains = 63 mm, long rains = 100 mm, dry = 1 mm (Foley and Faust 2010; C. Foley, unpublished data). The adjacent Ngorongoro and Karatu highlands, Rift Valley escarpment, and volcanic mountain peaks that provide runoff water to the TE receive much more rainfall than the valley floor (Prins and Loth 1988).

The geographic configuration of the TE reached its present form approximately 1 million years ago (Le Gall et al. 2008). The Gregory Rift in northern Tanzania diverges into three arms (Eyasi, Natron-Manyara, and Pangani) within a 200-km-wide, structurally complex area with many volcanoes (Fig. 1.2, Dawson 2008). The Natron–Manyara half-graben (where much of the TE is located) is a continuation of the full graben in southern Kenya (Scoon 2018a). The volcanoes of northern Tanzania are divided into an older group, of which the Ngorongoro volcanic complex is the most well-known example, and a younger group, including the active cones of Mount Meru and Ol Doinyo Lengai (Dawson 2008). Localized sedimentary basins occur such as the Lake Manyara, Engaruka, and Natron basins.

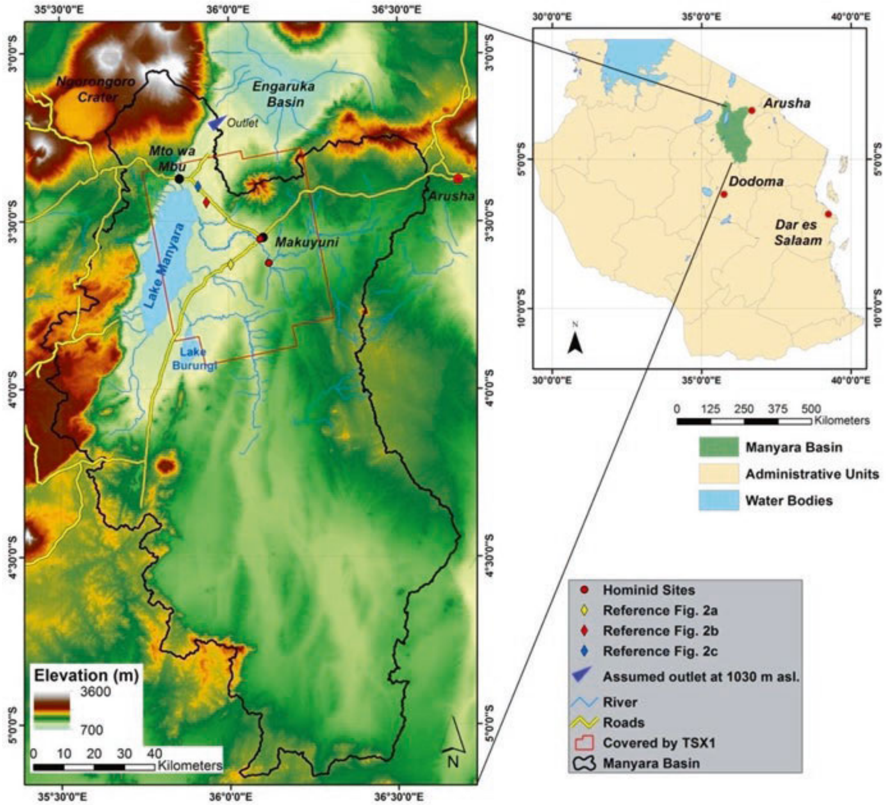

Fig. 1.3 Map of the Manyara hydrological basin (solid black line) (From Bachofer et al. 2014)

Hydrologically, the Lake Manyara Basin encompasses the core of what we consider to be the TE (Fig. 1.3), with the adjacent Engaruka Basin and Natron Basin to the north and Simanjiro plains to the east linked to the Tarangire River by migratory eastern white-bearded wildebeests and plains zebras that use those areas for calving grounds during the wet season. The Lake Manyara Basin is located in the Natron–Manyara half-graben with a 200–600 m high escarpment along the western shoulder with the highest elevation in the Ngorongoro highlands (Maerker et al. 2015). The eastern shoulder of the rift is lower in elevation and consists of tectonic blocks that dip toward the west. Basement material rises to form the eastern edge of the catchment as the Masai Plateau, location of the Simanjiro plains. The northeastern part of the Lake Manyara catchment area is dominated by the Essimingore volcano (2154 m). Lakes Manyara and Natron are shallow endorheic soda lakes with a maximum depth of 3 m (Deus et al. 2013).

In the arid lowlands that make up most of the TE, the only fresh water in the dry season is found in the Lake Manyara groundwater forest (Loth and Prins 1986), Silale and Gursi swamps, some sections in the Tarangire and Makuyuni rivers (Gereta et al. 2004), rivers coming down the escarpment from the Ngorongoro highlands, and some springs on the slopes of volcanic mountains. The three large lakes in the TE (Manyara, Natron, Burunge) are characterized by high levels of alkalinity and salinity because they have no or few outlets and evaporation exceeds inflow (Scoon 2018b).

Lowlands are covered by grasslands where drainage is poor due to volcanic ash, or by bush thickets and *Vachellia* (formerly *Acacia*) woodlands. The escarpment and upland areas are covered by *Commiphora* bushland and bush thicket. The highest (2000–3000 m) parts of the TE, all volcanoes, have *Podocarpus* and *Olea* forest (Prins 1987).

Animal and plant communities similar to modern savanna ecosystems became established in East Africa about 5 million years ago (Leakey and Harris 2003), and most of the existing forms of antelopes, giraffes, and zebras appeared a couple of million years ago (Lorenzen et al. 2012). Molecular estimates for the time of divergence between chimpanzees and hominins is about 5.5 million years ago (Kumar and Hedges 1998). Evolution of anatomically modern humans likely occurred in Africa about 300 thousand years ago (Bergström et al. 2021), and archaeological evidence of modern-equivalent cognition and behavior is also apparent from 300 thousand years ago (Spikins et al. 2021).

Plants and animals generally evolve through the process of adaptation, natural selection of random genetic changes that increase survival or reproduction. Humans (and other culture-creating animals such as elephants) also evolve via adaptability, behavioral self-modification (cultural change) in response to changing conditions. Humanity has existed for approximately 300 thousand years, and from our emergence to present, global climate cycles caused large changes in temperature, rainfall, and vegetation communities, which spurred social, behavioral, and technological innovations including the emergence 70–50 thousand years ago of a distinctly complex and advanced human culture based on foraging (Hetherington and Reid 2010).

Around 12 thousand years ago the most recent glacial period ended, and a period (called the Holocene) of warm and relatively stable climate continued until the present. The Holocene is a period of great human technological advancement including the domestication of plants and animals. In the TE, foraging economies and peoples speaking Khoisan languages have been present since about 50 thousand years ago. About 3300 years ago, specialized pastoralism with cattle, sheep, and goats arrived along with Cushitic and Nilotic language speakers (Marshall et al. 2011; Grillo et al. 2018), and between 2000 and 1200 years ago evidence of farming (sorghum, finger millet, yams, bananas) appears associated with Cushitic, Sudanic, and Bantu language speakers (Lane 2004; Crowther et al. 2018). For several centuries before the European colonial era began in the fifteenth century, peoples with foraging, pastoralist, and farming livelihoods coexisted in the TE in a dynamic mosaic of interaction, admixture, and coexistence with diverse and often overlapping ethnic, linguistic, political, economic, and social backgrounds (Crowther et al. 2018).

Human social organization in semi-arid areas such as in most of the TE was based on kinship in families and clans as well as age sets that linked people across large areas into social networks (Kimambo et al. 2017). Bantu-speaking farmers living on the highlands of Pare and Kilimanjaro were also kinship based, but they developed State societies about AD 1400 arising from iron-working clans that divided the mountain slopes like pie slices (Kimambo et al. 2017). Trade relationships for essential goods like obsidian, iron tools, salt, pottery, foodstuffs, and livestock within the TE and to outside areas existed for centuries, but remained at low levels until the late 1700s. In Tanzania, the human population has grown from approximately 3 million in AD 1800 to 60 million in AD 2020.

From the early 1800s, the Omani Sultanate in Zanzibar organized and monopolized large trading caravans bringing cotton cloth, beads, copper wire, and firearms into the East African interior, including the TE, in return for ivory and slaves for the international market dominated by traders from France, Britain, and the United States (Kimambo et al. 2017). In the 1840s, Ngoni-speaking people from Zululand in southern Africa entered southern Tanzania and disrupted central and southern Tanzania societies with widespread warfare and conquest that weakened and disrupted many communities. In the late 1880s the accidental introduction of the rinderpest virus decimated wild ungulate populations and livestock leading to famine, and smallpox also killed large numbers of people (Kimambo et al. 2017). The establishment of German and then British colonial administrations further disrupted traditional authorities and economies, and eventually led to the creation of the Tanzanian nation state (Kimambo et al. 2017).

Following independence in 1961, the United Republic of Tanzania experienced relative political stability. In the 1970s, Villagization policies of the Tanzanian government forcibly resettled millions of Tanzanians (approximately 70% of the population) and initiated large land cover changes in the TE (Nyerere 1977; Shao 1986). The human population has increased in the TE in accordance with the country as a whole (National Bureau of Statistics [NBS] and Office of Chief Government Statistician Zanzibar [OCGS] 2013), and land use in the TE and adjacent ecosystems has seen conversion of large areas of pastoral rangelands and dry-land wood and bush savanna to a landscape dominated by agricultural production (Börjeson et al. 2008; Homewood et al. 2009; John et al. 2014). In the late 1980s, Tanzania transitioned from Ujamaa Socialism to a more market-based economy, and in 1995, the nation transitioned from a "single-party democracy" to a multiparty political system (Lofchie 2014).

The current eco-socio-political makeup of the TE is a heterogeneous mosaic of diverse landforms, vegetation communities, human languages, land uses, and livelihoods (Fig. 1.1). The major ethnic group inhabiting the TE is the Maasai people. The Maasai largely depend on livestock (cattle, sheep, and goats) for their livelihoods, with some subsistence farming activities also practiced, predominantly in agro-pastoral communities. Most cattle kept by Maasai in the TE are zebu-type (*Bos indicus*). Other ethnic groups inhabiting the TE include Iraqw, Mbugwe, Gorowa, Irangi, Burunge, Wasi, Arusha, and Meru, who tend to practice farming of crops (i.e., maize, beans, pigeon peas) or mixed agro-pastoralism as a livelihood. In the early 1970s, several large-scale farming operations were established around Tarangire National Park (Borner 1985) and subsistence farming has increased markedly in the last decades (Msoffe et al. 2011). The economy of the TE is dominated by livestock keeping, agriculture, and ecotourism, with significant mining (phosphate at Minjingu) and charcoal making.

Arusha is a large city of >1 million inhabitants that, if not part of the TE is at least important to it, considering that migratory animals from Tarangire used to move to areas nearby (Lamprey 1964) and that it serves as the ecotourism and commercial hub of the region. Mto wa Mbu and Babati are the next largest urban areas in the TE. Other village centers tend to be along main roads and include Makuyuni, Kigongoni, Selela, Engaruka, Minjingu, Magugu, Vilima Vitatu, Lolkisale, Loiborsoit, Terat, and Loibor Serrit.

The TE is unfenced, allowing free movement of people, livestock, and wildlife throughout the ecosystem, but regulations exist that limit people and livestock activities in protected areas. There are two national parks (human activities are restricted to ecotourism and research) in the TE, Tarangire National Park and Lake Manyara National Park; the Mkungunero Game Reserve (where trophy hunting but no other forms of natural resource utilization are allowed); a multiple-use area called Manyara Ranch (where limited livestock grazing and ecotourism are permitted); and three Wildlife Management Areas (WMAs) called Burunge, Randilen, and Makame (Fig. 1.1). WMAs are set up by member villages and include zones delineated for specific land uses including areas reserved exclusively for wildlife and tourism. In Burunge and Randilen WMAs, the wildlife areas are currently dedicated to photographic tourism, whereas in Makame WMA trophy hunting is practiced. In addition, the TE contains four game controlled areas (GCAs): Lake Natron GCA, Lolkisale GCA, Mto wa Mbu GCA, and Simanjiro GCA. In GCAs, livestock keeping and cultivation are widespread: in some of the GCAs trophy hunting blocks exist. Forest reserves in the TE (e.g. Lossimingore Forest Reserve) also contain wildlife populations. In addition, the TE contains village lands which may also support wildlife. Several local communities in the TE hold certificates of customary rights of occupancy (CCRO) in which land use is restricted to pastoralism and other compatible uses. The villages Terat and Sukuro (both located in the Simanjiro area) receive money from tourism enterprises in exchange for setting land aside for wildlife; these arrangements are referred to as "conservation easements"; these easements have now been upgraded to CCROs to provide them with greater protection under Tanzanian law (Nelson et al. 2010).

In Tanzania, wildlife belongs to the state (Nelson et al. 2007), yet the corresponding jurisdiction for wildlife depends on the location (Caro and Davenport 2016). Inside a national park, Tanzania National Park Authority (TANAPA) is responsible for wildlife. In game reserves, game controlled areas, Manyara Ranch, forest reserves, and village lands, wildlife is under the jurisdiction of the Tanzanian Wildlife Authority (TAWA). Thus, a variety of authorities are involved with wildlife and hence with human-wildlife interactions. To further complicate matters, the TE spans across multiple districts (Babati, Kiteto, Monduli) and thus wildlife in the ecosystem is subject to different administrative units: each district has its own game officer who is responsible for wildlife in the district outside the national parks.

Wildlife populations in the TE are among the most abundant and diverse anywhere on Earth (Prins and Douglas-Hamilton 1990; Foley and Foley 2014). A nearly intact assemblage of large mammals including megaherbivores such as elephants (*Loxodonta africana*) (Foley and Faust 2010) and giraffes (*Giraffa camelopardalis*) (Lee and Bolger 2017), as well as large carnivores such as lions (*Panthera leo*), leopards (*Panthera pardus*), spotted hyenas (*Crocuta crocuta*), cheetahs (*Acinonyx jubatus*), and wild dogs (*Lycaon pictus*) (Pettorelli et al. 2010) roam in the savannas of the TE. Importantly, the TE features one of the few remaining large populations of migrating ungulates (Bolger et al. 2008) and the seasonal movements of the wildebeest and zebra populations define the spatial extent of the TE (Lamprey 1964) (Figs. 1.4 and 1.4).

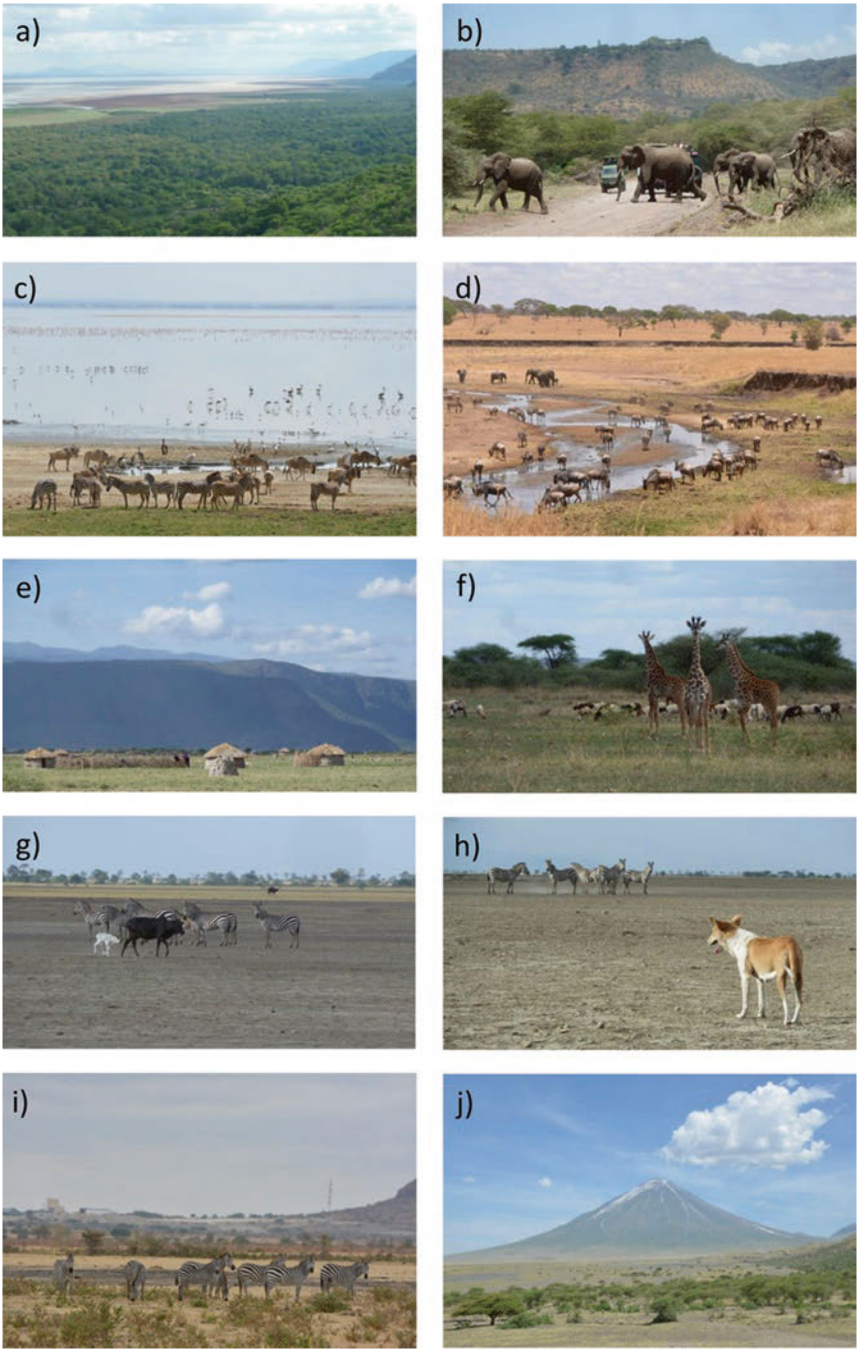

Fig. 1.4 Impressions of the Tarangire Ecosystem: (**a**) View from the escarpment over Lake Manyara; (**b**) Tourists taking pictures of elephants in Lake Manyara National Park; (**c**) Wildebeest, zebra, marabou stork (*Leptoptilos crumeniferus*), yellow-billed stork (*Mycteria ibis*), great white pelican (*Pelecanus onocrotalus*), and lesser flamingo (*Phoeniconaias minor*) at Lake Manyara; (**d**) Elephants, wildebeests, and zebras aggregate around the Tarangire River during the dry season; (**e**) Maasai boma and escarpment in the background; (**f**) Juvenile giraffe next to sheep in Manyara Ranch; (**g**) Cattle walking next to zebra along the eastern shore of Lake Manyara; (**h**) Domestic dog and zebra along the eastern shore of Lake Manyara; (**i**) Zebra near the phosphate mine in Minjingu; (**j**) Oldonyo Lengai – an active volcano in the north of the ecosystem (Photos: C. Kiffner)

1.3 The Nature of Human-Wildlife Interactions in the Tarangire Ecosystem

Historically, human-wildlife interactions have focused on “conflicts” and “costs” associated with wildlife (costs could be labelled as ecosystem disservices), yet it is increasingly being recognized that wildlife also provides benefits (which could be labelled as ecosystem services) and has strong intrinsic and cultural values (Lute et al. 2016; Ceauşu et al. 2019). This more integrated and nuanced view on the patterns and consequences of human-wildlife interactions suggests that human-wildlife interactions are not binary but rather align along a conflict-coexistence continuum (Frank et al. 2019). For this chapter and throughout this book we define human-wildlife coexistence according to the comprehensive concept outlined by Carter and Linnell (2016). Accordingly, we view human-wildlife coexistence as a dynamic process in which humans and wildlife “co-adapt to living in shared landscapes, where human interactions […] are governed by effective institutions that ensure long-term […] persistence [of wildlife populations], social legitimacy, and tolerable levels of risk”.

A first step towards assessing the sustainability of human-wildlife coexistence in the TE is to identify factors promoting and inhibiting interactions between humans and wildlife for a broad range of species (Carter and Linnell 2016). People living in the TE potentially interact with the full suite of wildlife species that occur in the ecosystem, although the wildlife species assemblage in areas that are permanently inhabited by people such as village lands and GCAs is depauperate compared to adjacent protected areas and compared to historical baselines in village lands (Kiffner et al. 2015). Through a human lens, interactions with wildlife species may be perceived as positive if benefits outweigh the costs associated with a wildlife species. Benefits include tangible assets such as the supply of animal protein and animal products, ecosystem services such as plant pollination, seed dispersal, control of pest species and pathogens, recreational experience, and income through wildlife-based tourism as well as intangible benefits manifested by an overall appreciation of animals and their central roles in human cultures (Power 2010; DeMello 2012). Humans may also perceive interactions with certain species as neutral. However, a major focus on human-wildlife interactions has been on interactions that humans perceive as negative, likely because such interactions incur costs to people. Tangible costs include damages to crops, livestock, and property, the transmission of pathogens to livestock and humans, actual threats to human wellbeing, indirect health impacts, and opportunity and transaction costs associated with the prevention or mitigation of negative interactions with wildlife (Naughton et al. 1999; Zhang et al. 2007; Barua et al. 2013; Dickman et al. 2013; Nyhus 2016; Kushnir and Packer 2019).

To provide insights on species that are considered to be problematic for people residing in the TE, Bencin et al. (2016) conducted structured interviews and asked 166 residents of the TE about their perceptions and attitudes towards wildlife species or groups of species (for animals that are often not identified to species level by lay people) that are widely distributed in the TE. Interviewees considered the following species to be frequently involved in negative interactions: rodents (the most widespread species in the TE is *Mastomys natalensis*), zebra, elephant, black-backed jackal (*Canis mesomelas*), spotted hyena, and birds of prey (Fig. 1.5a). The variation in perceived conflict frequency varied considerably across species and several species – particularly hippopotamus (*Hippopotamus amphibius*), wild dog, African civet (*Civettictis civetta*), caracal (*Caracal caracal*), giraffe, and Thomson's gazelle (*Eudorcas thomsonii*) – were rarely reported to be involved in negative interactions with humans. Interviewees showed considerable levels of tolerance for most wildlife species. However, a substantial proportion of interviewees wanted to see population declines in spotted hyena, rodents, elephant, birds of prey, lion, and black-backed jackal (Fig. 1.5b). As expected, the perceived frequency of negative interactions and the proportion of interviewees wanting to see a population decline were positively correlated (Fig. 1.5c). Further, this figure illustrates that some species are perceived as particularly problematic whereas coexistence with other species is rarely perceived as problematic. For example, large carnivores (leopard, lion, and spotted hyena) and elephant are above the regression line, suggesting that these species are considered particularly problematic by residents of the TE. As large carnivores and elephant are relatively well studied in the TE and present key challenges for coexistence in the TE and beyond (Di Minin et al. 2021), four chapters of this book focus on these species.

Coexisting with wildlife species crucially depends on people and may be partially related to the unequal distribution of wildlife-related costs and benefits among different stakeholder groups (Ceaușu et al. 2019). For example, listening to a lion roar in the near distance can be perceived quite differently whether you sit in a comfortable chair in the lounge of the Tarangire Safari Lodge as opposed to sitting on a wooden stool under the stars of the African savanna while guarding your livestock. If you are working in the tourism industry, showing your clients a pride of lions that is feeding on a wildebeest may provide you with a valuable source of income for you and your family. However, if you are Maasai and lions kill and feed on one of your cattle, lion predation reduces your wealth and food security. Thus, perceived or real consequences of human-wildlife interactions can determine how people perceive wildlife species. In turn, these perceptions may consolidate to attitudes that can eventually manifest as actions towards wildlife (Kansky and Knight 2014). As a case in point, interviewed Maasai were four times more likely to wish for population declines of lions compared to interviewees from other ethnicities (Bencin et al. 2016). Likely, this difference is at least partially related to the fact that Maasai (who mainly base their livelihood on livestock keeping) incur actual or perceived costs related to livestock depredation by lions whereas people who are less dependent on livestock may not perceive costs and may thus be more tolerant towards lions.

Such disparities in perceptions and attitudes among different stakeholder groups (e.g. pastoralists vs. conservationists, farmers vs. pastoralists, pastoralists vs. conservation authorities) can ultimately lead to human-human conflicts that are mediated by human-wildlife interactions (Redpath et al. 2013; Zimmermann 2020) and by strong power discrepancies among stakeholder groups (Reed et al. 2018). Thus,

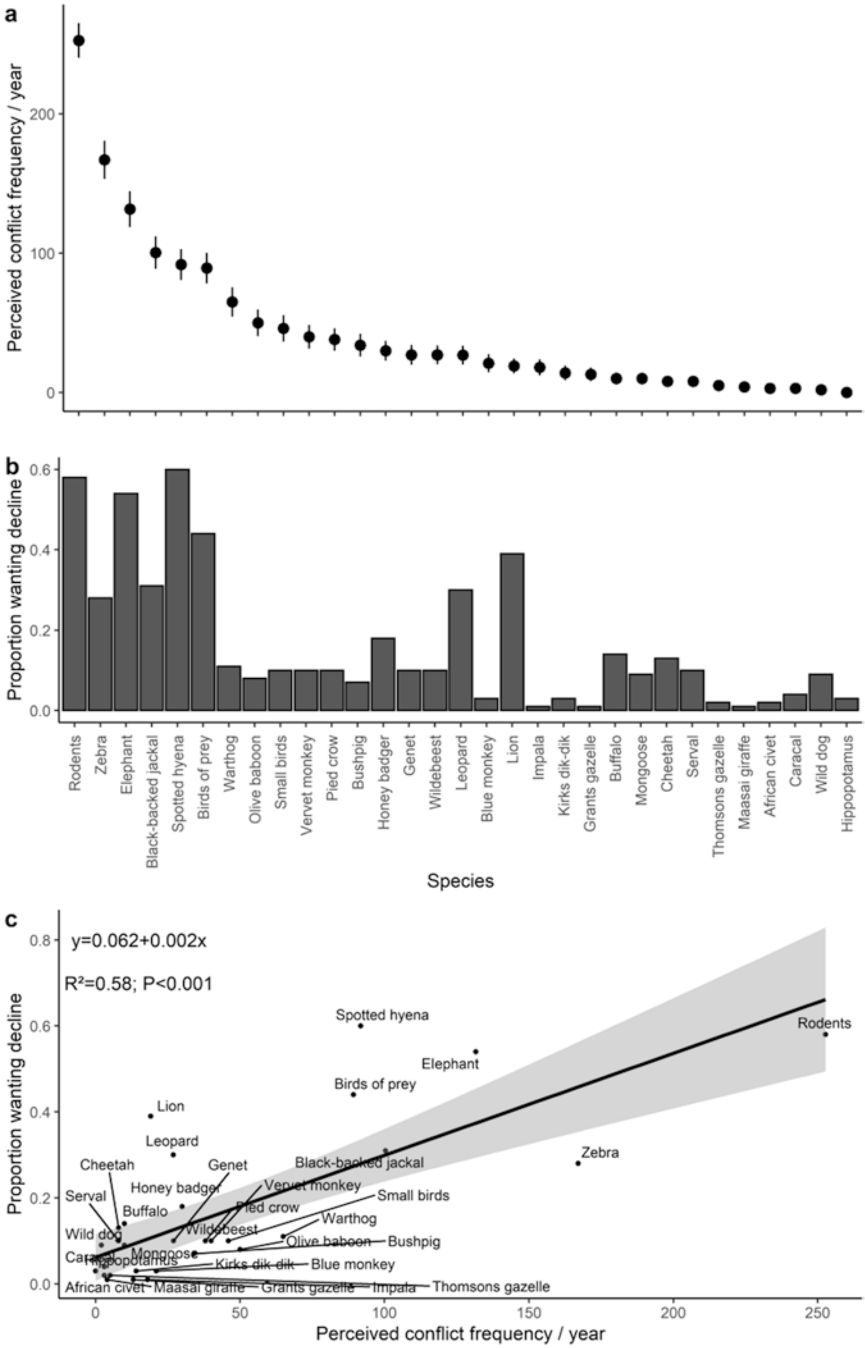

Fig. 1.5 Perceptions associated with 31 wildlife species (groups) in the Tarangire Ecosystem based on structured interviews (Bencin et al. 2016). (**a**) depicts the perceived frequency of conflict with animal species (groups), error bars indicate the standard error. (**b**) shows the proportion of respondents who wanted to see a population decline in each species (group). (**c**) illustrates the relationship between the perceived conflict frequency and the proportion of interviewees who wanted to see a population decline in each species

the dual role of wildlife in providing both services and disservices, and the unequal distribution of associated costs and benefits across multiple stakeholders, makes human-wildlife coexistence in coupled social-ecological systems a formidable and complex challenge with no simple and generic solutions (Redpath et al. 2013; Mason et al. 2018; Ceauşu et al. 2019).

1.4 Perspectives on Human-Wildlife Interactions and Coexistence in the Tarangire Ecosystem

The Tarangire Ecosystem in northern Tanzania offers an excellent opportunity to study different conservation models and the complexities of human-wildlife interactions in detail and from multiple perspectives. The ecosystem consists of a mosaic of two world-renowned national parks, several community-based conservation models, communal lands with people practicing traditional agricultural and pastoral livelihoods, and one of the last remaining long-distance migrations of large ungulates.

The first section of the book addresses the human dimension in human-wildlife interactions, with particular attention to the manifold human-human conflicts that have arisen in the TE among different stakeholders over wildlife-related policies. Chapters 2 and 3 give an anthropological political ecology perspective on recent conservation and human rights issues in the TE. Chapters 4 and 5 examine the perspectives of Maasai people in the TE on conservation, wellbeing, and livelihoods. Chapter 6 reports data from a survey of Maasai people's attitudes towards a community-based Wildlife Management Area.

The second section of the book shifts the lens to a more ecocentric perspective and discusses the spatial distribution and temporal dynamics of wildlife populations in the TE. Chapter 7 presents a reconstructed history of wildlife populations in the TE and draws together a wealth of historical sources. Chapters 8, 9, 10, and 11 summarize data on the ecosystem's populations of ungulates (excluding giraffes), giraffes, elephants, and large carnivores, respectively.

The third section addresses human-wildlife interactions explicitly with an eye towards solutions. Chapter 12 examines long-distance wildlife movements and landscape connectivity. Chapter 13 provides data on elephant-livestock interactions. Chapter 14 reports on efforts to mitigate human-carnivore conflicts in the TE. Chapter 15 describes a new model of habitat protection based on carbon credits to fund community-based conservation. Chapter 16 describes environmental education campaigns in the TE and their outcomes.

The unique combination of persisting wildlife populations, diverse wildlife conservation approaches, a growing human population, and comprehensive and interdisciplinary research on human and wildlife components makes the Tarangire Ecosystem an ideal system to study challenges and solutions to sustainable human-wildlife coexistence in fragmented, coupled human-natural landscapes. We hope readers of this book will be inspired to take the next steps and build upon the work presented here to advance our knowledge on these topics and to find and implement sustainable solutions that adequately address the needs of both humans and wildlife in the ecosystem.

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Part II
The Human Dimension

Chapter 2
Historical Political Ecology of the Tarangire Ecosystem: From Colonial Legacies, to Contested Histories, Towards Convivial Conservation?

Jevgeniy Bluwstein

Abstract This chapter outlines a historical political ecology of conservation initiatives in the Tarangire Ecosystem (TE). First, I turn to chronological history to highlight the origins and the evolution of key stages in the making and expanding of conservation initiatives in the TE. Through attention to chronological history, I show how dominant ideas about people and nature changed over time in the study area. Second, I revisit the TE as a site of contested histories to show how two environmental history narratives compete with each other – a *statist narrative* which is embraced by public authorities in government and conservation bureaucracies, and a *people's history* which represents lived experiences and bottom-up conservation practices of human-wildlife coexistence. I argue that by dismissing and marginalizing locally meaningful narratives, experiences and representations of the TE, a statist narrative continues animating conservation conflicts in the present. Drawing on these insights from the TE's environmental history and historical political ecology, the chapter concludes with an outlook on how people-wildlife coexistence in the region could be fostered through convivial conservation.

Keywords Political ecology · Environmental history · People's history · Convivial conservation · Coexistence

2.1 Introduction

In this chapter I chart a historical political ecology of the Tarangire Ecosystem (TE) by mobilizing two perspectives on the historical development of conservation initiatives in the region. Through a chronological perspective on Tarangire's environmental

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history, I highlight key historical moments and developments that led to the evolution of TE from colonial beginnings to the present. My goal is to highlight how colonial legacies pertaining to environmental management and control continue to linger on today, although some of the key ideas about people and nature have shifted significantly over time. Furthermore, Tarangire's chronological history illustrates how conservation in the region has been landscaped (expanded beyond core protected area boundaries) and neoliberalized (relying on market-based approaches to conservation) in the last decades.

Through a narrative-centered perspective, I highlight two presently dominant and contrasting narratives about Tarangire's history as they circulate in administration and conservation bureaucracies, and in local communities. By showing how a *statist narrative* of Tarangire is at odds with *people's history*, I highlight Tarangire's contested environmental history to explain how present conflicts over land and resources are underpinned by an antagonism between a state-centric logic of territorial control and the needs of pastoralists to maintain a vital geography of access to land and resources in a semi-arid environment.

Both perspectives on Tarangire's environmental history inform the rest of the chapter, where I highlight three key challenges to human-wildlife coexistence around Tarangire: the entrenched state-centric logic of territorial administration and conservation through fixed boundaries, the idea that rural livelihoods have to be compatible with western conservation ideals and initiatives, and the reliance on market-based initiatives for conservation. I end the chapter by offering a radically different – convivial – vision for conservation around Tarangire which goes beyond taken-for-granted aspects, such as protected areas and their boundaries, tourism and conservation finance, participation in decision-making and conservation expertise, and the role of conservation NGOs and the Tanzanian state.

2.2 A Historical Political Ecology of Tarangire Ecosystem

2.2.1 Chronological History

I begin the chronological history of conservation initiatives in the TE in the 1890s, when catastrophic epidemics caused human, livestock and wildlife depopulation in the study area. This development was amplified by the German colonial occupation and led to a collapse of many of the pre-existing human systems of ecological control through settlement, fire, cultivation and livestock grazing. In the years following European occupation, previously human-dominated territories were increasingly repopulated by wildlife and resulting bush encroachment favored the expansion of pathogen-carrying tsetse flies. Upon taking over control from the Germans after World War I, the British colonial administration responded with a large-scale bush-clearing and human resettlement campaign to separate people and livestock from tsetse-dominated areas (Arlin 2011; Rohde and Hilhorst 2001; Kjekshus 1996). Lacking awareness of African practices of containment, control and coexistence

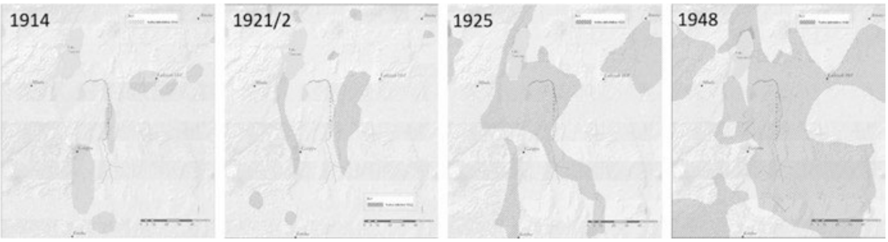

Fig. 2.1 Tsetse fly distribution (shaded areas) in the present-day Tarangire Ecosystem, from Arlin (2011), reprinted with permission from the author. 1914 distribution of tsetse is based on Dietrich Reimer (Ernst Vohsen) Berlin, red. P. Sprigade u. M. Moisel, *Deutsch-Ostafrika*, 1:5000000, Übersichtskarte über Rinderreiche- und Tsetse-Gebiete. 1921/22 distribution is based on *Annual report of the Department of Veterinary Science and Animal Husbandry*, Tanganyika, 1922, p. 11, PRO CO 736/1, 1:5000000. 1925 distribution is based on *Department of Veterinary Science & Animal Husbandry* (1:2000000) PRO CO 736/4 94396. 1948 distribution is based on *Atlas of Tanganyika Colonial Office* 1948

with the non-human environment, this colonial strategy of separation was underpinned by western ideas about nature being the opposite of culture, and human development being at odds with non-human ecologies (Kjekshus 1996; Adams and McShane 1992; Koponen 1988).

Spatial separation had direct consequences for what would later become the Tarangire Game Reserve, the first officially gazetted protected area in the study area. Drawing on different archival sources, Arlin (2011) mapped the dynamics of the tsetse distribution in the present-day TE in 1914, 1921, 1925 and 1948 (Fig. 2.1). Arlin shows how in 1914 tsetse flies were largely limited to parts of the Tarangire River, and areas around Galapo and Lolkisale, hence southwest and northeast of the area that would become Tarangire Game Reserve in 1957. In 1921–1922 tsetse flies had spread east of Tarangire River and south of Lake Manyara, an area settled by the Mbugwe. Arlin suggests that the spread south of Lake Manyara is due to the combination of a human population decrease and British imposition of game laws in 1920 which forbade locals to hunt wildlife and thereby to control the spread of the fly. Arlin further suggests that the dramatic spread of tsetse flies in 1925 – surrounding most of Lake Manyara, and extending south and southeast along Tarangire River – was due to British efforts to contain the fly through resettlement campaigns which allowed the fly take hold of previously human-dominated territories. By 1948, the fly had established itself in an area that had a striking resemblance with much of the extent of the present-day TE. This shifting tsetse fly distribution likely gave weight to calls to establish protected areas "where the interests of man and game do not clash", as expressed by a game warden in 1949 (ACC 69 275/1 Vol.1, cited in Arlin 2011).

In short, the birth of TE was underpinned by the dominant win-win promise at the time. Rural people would benefit from their farms being protected outside the reserves from marauding wildlife, people and livestock would be protected from tsetse flies, and white hunters would benefit from hunting opportunities inside the reserve. Importantly, the area in question was not a timeless tsetse and wildlife geography. It took active interventions by the British administration in the wake of disease outbreaks, colonial occupation and World War I, to transform a previously largely tsetse-free area of human-wildlife coexistence into a tsetse occupied protected area. Yet still, up until 1955 – 2 years before Tarangire Game Reserve was officially gazetted – voices within British colonial administration in support of present-day Tarangire and Lake Manyara National Parks *as agricultural areas* dominated debates over what should be done with these lands (Arlin 2011).

The idea of a wildlife container for the benefit of humans started to unravel as soon as reserve boundaries were drawn. Wildlife moved outside reserve boundaries during the wet season and returned during the drought. Supported by research by British biologist H. F. Lamprey in the 1960s (Lamprey 1964), the initial colonial idea of Tarangire as a wildlife container that protects human land use outside of reserve boundaries and offers opportunities to hunt inside started to shift. Increasingly, Tarangire was understood as a center of a much larger wildlife habitat that needed to be protected from humans (Bluwstein 2018).

The shifting perceptions about people and wildlife in and around Tarangire led to a territorial expansion and an upgrade of the Game Reserve to a National Park in 1970. However, it took another decade until conservation authorities started to problematize rural livelihoods around Tarangire in the name of conservation. Beginning in the 1980s, we see how the idea of Tarangire as part of a larger wildlife habitat is embraced by conservationists who propose different interventions – from changing human behavior and land use (Mwalyosi 1991a, b; Borner 1985) to resettlement of entire villages (Prins 1987) – to expand Tarangire's reach beyond its official park boundaries.

The example of Borner's (1985) proposal is insightful. Fearing the "increasing isolation of Tarangire", he advocated for a "new land-use authority" southeast of the National Park to ensure a strict livestock population management and control (Borner 1985). Borner reasoned that this "unpopular" move "would ultimately benefit the Maasai" while it would also preserve "tourism value" of the park (Borner 1985). Motivated by similar concerns over human land use around Tarangire, Tanzanian researcher Mwalyosi (1991b) suggested that "non-land based economies" should be encouraged in the TE "in order to reduce pressure on the land". To Mwalyosi, "sustainable development" was akin to limiting agropastoral practices, which 30 years later continue sustaining rural livelihoods for most people living in the TE.

In this sense, we see how the 1980s mark the beginnings of a neoliberal win-win narrative about conservation (Igoe and Brockington 2007) around Tarangire which entails the adoption of conservation-friendly land-use practices of limited livestock rearing and cultivation in return for market-based revenue generation through wildlife-based tourism. Tourism-based revenues would compensate and support "sustainable" = "non-land based" rural livelihoods (Mwalyosi 1991b). Maasai pastoralists were not convinced and successfully resisted attempts of land alienation in the name of conservation (McCabe and Woodhouse this volume; Igoe and Brockington 1999; Sachedina 2008).

Growing concerns over the future of Tarangire's wildlife led Tarangire park authorities to extend their focus in the 1990s from ensuring the protection of wildlife within the park boundaries (known as the fortress conservation approach) to also protect it outside. This was to be achieved by shaping people's behavior and land use through community-based conservation initiatives. The ultimate goal was to render human land-use practices around parks “compatible” with conservation objectives, as the 1994 National Park Policy put it (TANAPA 1994). To achieve this goal, park authorities tried to manage local people’s expectations in order to keep them as low as possible, assuming that people will simply give in to demands by government authorities even when little to no compensation was offered for foreclosed land-use opportunities. Unlike the park authorities, some tourism entrepreneurs offered tangible economic incentives to rural people in return for agreements not to convert grazing land to agriculture (Bluwstein 2018; Nelson et al. 2010).

It was in the late 1990s that another course of history was suddenly possible, but remained foreclosed for rural communities living around the park. In a multi-stakeholder planning workshop in 1998, NGOs and private sector (safari tourism) representatives recommended that rural people should be allowed to let their livestock graze inside Tarangire and Lake Manyara National Parks. These stakeholders asked park authorities to balance expected negative effects on rural livelihoods stemming from community-based conservation interventions that park authorities, NGOs and tourism entrepreneurs started to experiment with in the 1990s. These recommendations were ignored and the *de facto* expansion of the “fortress” Tarangire beyond its boundaries into rural spaces accelerated in the coming years (Bluwstein 2018).

Particularly the international conservation NGO African Wildlife Foundation (AWF) was successful in redefining the idea of Tarangire from being a conservation fortress to being the center of a much larger conservation landscape. Funded by USAID, Tarangire Ecosystem became AWF’s “Maasai-Steppe Heartland”, a brand that AWF employed to attract further funding, conservation initiatives and tourism flows (Sachedina 2008; Igoe 2017). From the late 1990s and well into the 2010s, AWF promoted – directly and indirectly – a set of interventions in rural communities living around the Tarangire, Lake Manyara and Mkungunero protected areas. These interventions, including resettlements, land-use zoning and land alienation, were often based on manipulation, coercion and false promises, and led to the militarization of wildlife and resource management in the villages (Bluwstein 2018; Davis and Goldman 2017; Bluwstein et al. 2016; Igoe and Croucher 2007).

Parallel to AWF’s efforts to arrest rural economic development and land use around the TE’s core protected areas, park and reserve authorities launched boundary re-survey projects in the mid-2000s. As a result, both Tarangire’s and Mkungunero’s official boundaries were expanded, producing new territorial claims by state authorities vis-à-vis villages and rural communities. Previously officially recognized, cultivated and grazed village territories became suddenly illegal for human use. The ensuing land-use conflicts have not been settled to this day, forcing people to live with a risk of violence, eviction and economic dispossession (Bluwstein 2018, 2019).

USAID did not extend funding to AWF in 2014, being frustrated with AWF’s antagonizing approach which created much hostility to conservation initiatives by rural communities (Bluwstein 2018). In 2015, USAID funded a new NGO consortium – the Northern Tanzania Rangelands Initiative (NTRI) – to support conservation initiatives in the TE. Led by another international NGO, The Nature Conservancy, NTRI set out to take a less antagonistic approach to community-based conservation, although it remains to be seen if the strategy of containing agricultural land use in favor of human-livestock-wildlife coexistence – NTRI's declared objective – will be successful.

To sum up, a chronological history of conservation initiatives in the Tarangire Ecosystem highlights how the TE has been subject to human-environment interactions and coexistence long before colonial administrators and western conservationists set their eyes on it (Rohde and Hilhorst 2001). Tarangire's history is part of a global history of conservation interventions to protect the very same landscapes from people as pristine nature and wilderness that people shaped through their land use practices in the first place (Reid 2012; Homewood et al. 2009; Shetler 2007; Brockington 2002; Rohde and Hilhorst 2001; Neumann 1998; Fairhead and Leach 1996; Adams and McShane 1992; Parkipuny 1991). This protection through separation may have even impoverished these ecologies rather than enhanced them (Western and Gichohi 1993; Goldman 2020).

TE's chronological history also illustrates how ideas about people and nature are historically contingent (Cronon 1993). Dominant ideas about people and nature, human-wildlife coexistence, conservation, preservation and rural development shifted over time pertaining to the area known today as the Tarangire Ecosystem. The initial colonial ambition was to keep humans and wildlife separate to provide hunting grounds for white hunters, to protect humans from wildlife ('problem animals') and tsetse flies, to encourage agricultural development and 'productivity', and to confine pastoralists to drylands with little access to permanent water and pastures (Hodgson 2001). Later, these concerns and priorities gave way to the idea that (by now perceived as 'charismatic') wildlife needs to be protected from humans across the entire wildlife habitat (understood as landscape or ecosystem) beyond Tarangire's protected area boundaries. To protect wildlife from humans at the landscape scale, human development was tied to international wildlife tourism, and – more recently – to carbon markets (Bluwstein 2018). Put differently, Tarangire as a conservation fortress became landscaped (expanded beyond core protected area boundaries) and conservation initiatives became neoliberalized (relying on market-based initiatives).

2.2.2 Contested Boundaries, Contested Histories

While insightful, a chronological history of the TE does not offer immediate lessons to understand and address contemporary conservation challenges and conflicts. A chronological history tells us little about how contemporary stakeholders understand Tarangire's evolution, environmental change and human development, much less how its environmental history may in fact be a contested terrain of different *histories* that are at odds with each other.

Drawing on my research in the southeastern part of the TE (Bluwstein 2019), I aim to show how there are at least two dominant and competing narratives at work around TE. On the one hand, there is a *statist narrative* (Sunseri 2000) that is largely embraced and shared by public authorities in government and conservation bureaucracies. On the other hand, there is a narrative akin to *people's history* (Zinn 1980), a history of lived everyday experiences of local people. In his study of the Maji Maji uprising in German East Africa, Sunseri (2000) highlights the power and the effects of statist narratives:

> While statist narratives have the function of empowering the state and its goals, they also serve to thwart criticism, opposition, or divergent political perspectives. By substituting a complex and sometimes contradictory history for one that is simplistic, they have the power to shut the door to historical enquiry. They furthermore tend to erase the stories and aspirations of common people, marginal ethnic and social groups, and women.

To recover stories and aspirations of common people living with wildlife in the TE, I show how a people's history can illustrate local practices of what could be called bottom-up conservation, how this perspective on human-wildlife coexistence is marginalized in statist, often hegemonic, representations of the TE, and how this contested past continues animating conservation conflicts in the present.

The divide between a statist and a people's environmental history manifests through the antagonistic relationship between pastoralists and the colonial and post-colonial state in Tanganyika/Tanzania (Ndagala 1990). A statist narrative of Tanzania's history is bound up with colonial and post-colonial practices of state-led mapping and boundary-making to designate different land uses, property regimes and ethnicities for purposes of administration and conservation. However, pastoralist livelihoods and land-use practices were poorly represented by administrative and conservation boundaries, given that pastoral mobility is tied to social networks and mutual obligations, cultural practices and material needs, changing environmental conditions, and a lack of durable structures on land such as farm plots and fences (Ndagala 1990). What is more, the state perceived pastoralists as free and uncivilized barbaric raiders who had to be brought under control and learn to respect the rule of law (Ndagala 1990). The colonial grid of administrative and conservation territories (e.g. customary territories for 'natives' such as *Maasai Reserve*, and forest and game reserves for 'nature') was a key technique of social control (Hodgson 2001).

To establish social control, three intersecting top-down processes of boundary-making produced a set of administrative and conservation boundaries that crisscross Tanzania today and continue being entangled in land conflicts: ethnicization, nature conservation, and villagization (Bluwstein 2019). Ethnicization was a colonial strategy of organizing territories along ethnic lines and institutions in order to govern colonial subjects through indirect rule (Anthias and Hoffmann 2020; Hodgson 2001). Nature conservation, too, was a colonial strategy of government through a spatial separation of human and non-human land use, with agriculture and pastoralism on one side and pristine nature on the other side of the boundary. Although introduced decades later under post-colonial socialist rule, villagization meant to further instill a "modern" way of living and using land and resources in rural areas (Schneider 2004; Ndagala 1990). Inevitably, all these boundary-making practices of spatial control lead to a simplification of complex socio-ecological arrangements (Scott 1998; Goldman 2003). Pastoralists have been particularly affected by the convergence of these three historical and ongoing statist strategies of land control and the simplifications that they engender. However, these strategies have remained partial, explaining the particularities of a troubled and contested history of the TE as a thoroughly but never completely and unambiguously mapped, bounded and territorialized space (Bluwstein 2019).

2.2.2.1 The Village Kimotorok as a Site of Contested Histories

I trace Tarangire's contested history through the lens of the village Kimotorok. The village is located within the Tarangire Ecosystem, bordering Tarangire National Park and Mkungunero Game Reserve (Fig. 2.2). Kimotorok is home to a majority Maasai community of around 3000 people, 60,000 heads of cattle, and 67,000 goats and sheep, according to a 2012 census. The village received its name due to its location around the Kimotorok swamp. During the dry season, people and livestock live close to the swamp which provides water and grass. When the swamp is flooded during the wet season, Kimotorok residents move to temporary homesteads away from the swamp, west- and eastwards (Bluwstein 2019).

These seasonal movements of people and livestock have coexisted with local wildlife habitats, movements and migration patterns, predating top-down conservation initiatives in the present-day TE. However, the combined effects of ethnicization, conservation and villagization have placed people-livestock movements at odds with state and conservation bureaucracies and boundaries. Beginning during

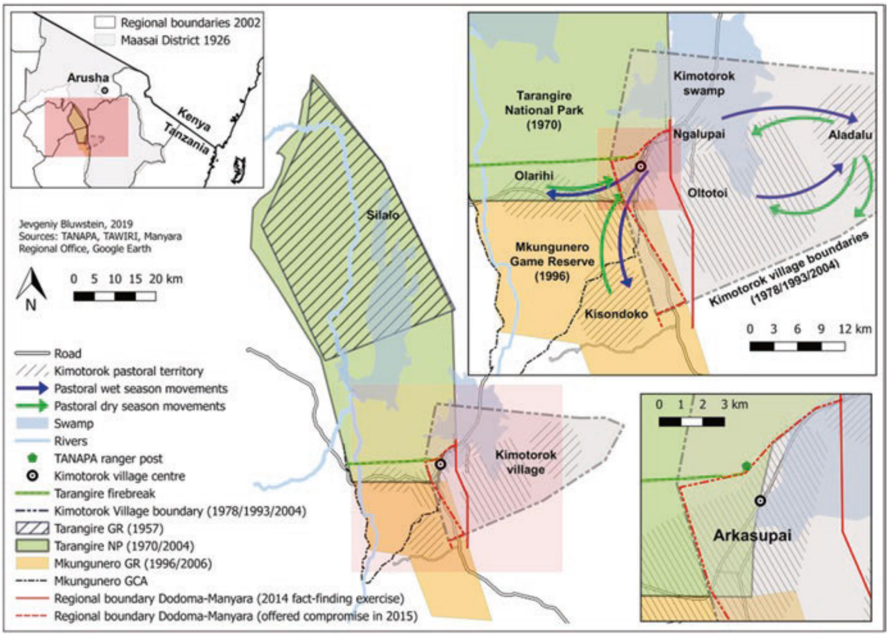

Fig. 2.2 Pastoral land use and boundary overlaps with protected and administrative areas. Red lines (solid and dotted) illustrate how the regional boundary cut through Kimotorok village and what compromise official state authorities considered in response. (For more details see Bluwstein 2019)

German colonial rule, ethnicized administrative boundaries were drawn to contain Maasai in a 'reserve' in 1905, which the British administration confirmed as the 'Maasai Reserve' in 1922. The goal was to isolate Maa-speaking people in an 'ethnological and economic sanctuary, rigidly closed to outside influence and to trade' (Provincial Commissioner Mitchell, 16 March 1927, cited in Hodgson 2001; Ndagala 1990). In 1926, the British gazetted the districts 'Maasai' and 'Kondoa' to further entrench the territorial division between Maasai pastoralists and Rangi farmers. Maasai District lacked adequate water resources and pastures for pastoral livelihoods in a semi-arid environment, forcing Maasai to transgress district boundaries to sustain their livelihoods and social networks.

Decades later, the boundary dividing both districts would become the dividing line between Tarangire Game Reserve (and later National Park) and Kimotorok village (Fig. 2.2). The establishment of the Tarangire Game Reserve in 1957 did not lead to a total loss of access to Tarangire's perennial swamp, Silalo (Igoe 2002). This happened 13 years later, when the reserve was upgraded to a national park in 1970 (Fig. 2.2). With this upgrade, Tarangire was expanded further south, and now overlapped with livestock grazing territories of Maasai living around the Kimotorok swamp. The later Prime Minister Sokoine recounted how this upgrade meant 'the loss of homes, grazing pastures and water points that [the Maasai] urgently needed for themselves and their cattle' (Hagen 1979).

In the coming decades, the expansion of Tarangire and its new status as a national park introduced a new era of top-down administrative and conservation boundary-making, whereby public authorities had little geographical knowledge about the very boundaries that they sought to draw on maps and enforce on the ground. To begin with, Tarangire park authorities (TANAPA) hardly knew where the new boundary of Tarangire National Park was on its southeastern end. Kimotorok Maasai continued to use their customary dry season territories for livestock grazing and agriculture, unaware of overlaps with the park. What is more, TANAPA created a fire break (Fig. 2.2) around 5 km within the park's southern boundary (as it was later resurveyed in 2007) that Kimotorok residents respected and came to understand as the actual boundary (Bluwstein 2019). In 1978, Kimotorok residents were officially recognized as villagers in the wake of the national villagization program. Here again, state officials lacked spatial knowledge of boundary overlaps between Kimotorok, its main village Loiborsiret (Kimotorok was still a subvillage), and Tarangire National Park. In 1983, Kondoa district officials decided to create a game reserve adjacent to Tarangire National Park on its southern boundary, hence deep into Kimotorok's wet season livestock grazing territory. The district officials were not aware of potential land conflicts with Kimotorok, because Kimotorok Maasai were officially not part of Kondoa district, illustrating how the colonial legacy of ethnicized administrative boundaries converged with a post-colonial villagization program. This convergence erased Maasai's customary land-use practices where these did not respect official administration and conservation boundaries which were unknown to local communities and official authorities alike. Hence, Kimotorok residents were not informed that their customary territories of wet season livestock grazing were swallowed by Mkungunero Game Reserve by the time it was gazetted in 1996 (Fig. 2.2). To complicate matters, 3 years earlier, in 1993, Kimotorok was officially upgraded from a subvillage to a village, and the wet season grazing area located deep into Mkungunero – called Kisondoko – was now an official subvillage (Bluwstein 2019).

Yet it was only in 2006, that game reserve officials surveyed the boundaries of Mkungunero and realized that it overlapped with Kimotorok's officially recognized and mapped village boundaries and its residents' customary land use practices. Mobilizing the state-centric logic of territorial control, which does not tolerate spatial overlaps and ambiguities, Mkungunero officials – state bureaucrats at the Ministry of Natural Resources and Tourism – declared two subvillages of Kimotorok to be illegal. Kisondoko was declared illegal in its entirety, criminalizing access to Kimotorok's wet season pasture reserves. The other subvillage, Arkasupai, was deemed partly illegal. Arkasupai is host to Kimotorok's public village infrastructure (school, dispensary) and small shops and businesses (Bluwstein 2019). Similar claims were advanced by Tarangire park officials who resurveyed park boundaries in 2004 and realized that Kimotorok's primary school – which TANAPA helped to build in 2003 – was inside the park (Masara 2005).

How do government and conservation authorities perceive these land conflicts? A parliamentary task force concluded that the history of official boundary-making of Mkungunero Game Reserve was marred by 'technical errors', a lack of involvement of village and district authorities, a poor 'interpretation' of already existing official boundary gazettements (called Government Notices), and years of delays in demarcating and enforcing reserve boundaries which only reinforced local claims to land (Bluwstein 2019). Similar conclusions can be drawn about Tarangire National Park. Yet, the insights by the parliamentary task force did not stop the authorities from insisting on new boundaries for Tarangire and Mkungunero which challenge customary land claims and officially mapped village lands.

A central point of contention between central government officials and conservation authorities on one side and village government and its residents on the other is the history of the Tarangire Ecosystem. Kimotorok's village leaders insist that they are "the natives of this village" which gives them the right "to show where are the boundaries and the hills". In a series of letters to officials, village leaders maintained that land claims by Tarangire and Mkungunero would interrupt "proper land use", erase three "legal subvillages" and their "economies" and undermine "sustainable development". The village leaders insist that they should decide where the boundaries are "according to their use". Further, the government is expected "to seek the truth by involving the community members and stop relying on maps which have been forged so as to create the current situation […] which does not consider human life and sustainable conservation". Ultimately, Kimotorok leaders insist that "legal authority" will be derived from the "right land use history" (Bluwstein 2019).

Tarangire and Mkungunero authorities have a different understanding of land-use history in the TE, as my interviews suggest. A TANAPA warden pointed out that "when Tarangire was established, Kimotorok was empty, there were no people here". Mkungunero's director insisted that the protected area predates people's land claims given that the reserve was already declared a Game Controlled Area back in 1954, and thus two decades before villagization gave official birth to Kimotorok.

"Villagization came in 1974. Before 1974 villages were not defined", expressed the director to me. Moreover, in separate conversations, the director of Mkungunero and a Kondoa district official claimed that hardly anyone lived in Kimotorok before the El Niño event in 1997–1998. According to the district official, "there was no village, just some grazing". Ultimately, a high-level official at the Ministry of Natural Resources and Tourism who was familiar with the land conflict pointed out that elephants were present in the area before the first humans arrived there (Bluwstein 2019).

2.2.2.2 Whose History Matters in Struggles Over Land in Nature Conservation Initiatives?

The juxtaposition of contested narratives of environmental histories of the TE highlights how different stakeholders compete for authority with recourse to history. To Kimotorok residents, people's environmental history is tied up with how people use land and resources, past and present. This is a socio-ecological history which is bound up with the rangeland ecology of a semi-arid environment, its seasons, and its plant and animal life (Goldman 2020; Reid 2012; Homewood et al. 2009). Put differently, a people's environmental history around Tarangire – as I have shown through the case of Kimotorok Maasai – is about maintaining a vital geography of access to water, land and pastures, in coexistence with wildlife.

To government authorities, history starts and ends with the state, and its past and present efforts to govern people and spaces through land laws and boundary-making practices, underpinned by three techniques of territorial government and control: ethnicization, nature conservation and villagization. What happened before the rise of the state has no history, does not require governmental attention, and thus is erased and void. By bounding spaces of human–non-human land use and interaction as protected areas, they would become timeless spaces of nature (conservation fortresses) in which human history would end, surrounded by rural territories in which history could continue to unfold.

Here, it is important to draw attention to the implications of how Tarangire was initially conceived as a conservation fortress and reconfigured into a conservation landscape later. Whereas the colonial beginnings of Tarangire saw rural territories around Tarangire as spaces of human development, contemporary attempts to landscape Tarangire see the same rural spaces as wildlife habitats fragmented and threatened by human land use, in need of connectivity. Whereas in the past it was human history that was supposed to unfold outside of protected areas which were to contain wildlife, today wildlife is to be conserved in its historical habitat in the midst of rural communities who are expected to restrict their land use and forego land-based development.

How do conservation scientists and NGO practitioners make sense of Tarangire's history and what lessons for conservation do they draw from it? In many ways, conservationists occupy and negotiate a complicated and at times contradictory position, sitting on the fence between recognizing a people's history and subscribing to a statist narrative. Conservationists tend to think of nature and people as separate entities, while many increasingly work to overcome a dichotomizing view towards people-wildlife coexistence (Büscher and Fletcher 2020). Conservationists are usually aware of a troubled history of conservation interventions and many are familiar with local practices of what I call bottom-up conservation. Yet, conservationists also tend to embrace state and conservation boundaries as important and necessary spatial practices for top-down conservation. In the next two sections, I draw on insights from Tarangire's environmental history and political ecology to highlight how conservationists can help foster human-wildlife coexistence in the TE in the future.

2.3 Political Ecology of Human-Wildlife Coexistence Around Tarangire Today

Today, people and wildlife within the Tarangire Ecosystem continue sharing land and resources in different ways *despite* a history of more or less successful attempts from above to separate people and wildlife into different territories. The key question for a political ecology of human-wildlife coexistence is thus not so much about whether coexistence is possible and how local communities can be taught coexistence. It is already practiced now and has been in the past, albeit practicing coexistence has become increasingly difficult due to various historical developments and present constraints (Goldman 2020; Cooke 2007; Anderson and Grove 1988). The key question in my view is to examine what these developments and constraints are. Here, I focus on three key aspects that continue to challenge human-wildlife coexistence in the TE today: state-centric logic of territorial administration and conservation, the idea that people have to adopt land-use practices that are compatible with western conservation ideals and initiatives, and the reliance on market-based initiatives to promote and incentivize conservation.

First, drawing on insights from environmental history I have argued that human-wildlife coexistence is at odds with a state-centric logic of territorial administration and conservation, regardless if this logic aims to separate protected areas from rural communities (the fortress conservation approach) or to separate different land-use zones through community-based conservation initiatives within a conservation landscape. This is not to say that only the state and conservation bureaucracies rely on spatially explicit practices to manage access to land and resources. Tarangire Maasai, too, rely on locally meaningful spatial practices and boundaries (place-names) to signify the location of numerous places at different scales, often cutting across administrative and conservation boundaries set from above (Goldman 2020). However, even though Maasai's place-names and boundaries foster bottom-up human-wildlife coexistence, their spatial practices often do not make it into official conservation land-use planning, much less are they known and acknowledged by official authorities (Bluwstein 2019; Lovell 2018).

To be sure, the initial motivation to separate people from the environment has begun to shift in the last years around Tarangire. The rise of the landscape approach to conservation is questioning the role of fixed boundaries in conservation, to some extent seeing them as part of a problem, not as a solution. However, so far, it has been wildlife that benefits from this realization. Human activities have experienced not less, but more boundary-making through conservation. Boundary-making continues at two levels, through territorialization of land and resource control on the ground, and through positivist epistemologies that continue underpinning conservation science. On the one hand, community-based conservation initiatives territorialize rural spaces into different land-use zones to contain different human land uses to particular, bounded areas in order to ensure that wildlife can roam freely in a quasi-unfragmented conservation landscape (Bluwstein 2018; Bluwstein and Lund 2018; Bluwstein et al. 2016; Goldman 2003). On the other hand, conservation science concerned with people-wildlife coexistence outside of protected areas continues to reproduce dichotomies of nature and society, scientific and local/indigenous knowledge, rights of nature and rights of people (Goldman 2020; Brehony et al. 2018). Landscape conservation efforts can thus be understood as a soft(er) expansion of the traditional fortress approach beyond the boundaries of core protected areas.

A second and related obstacle towards people-wildlife coexistence is the idea that people have to adopt land-use practices that are compatible with western conservation ideals and initiatives which hold that land use is to be extensive, largely pastoral and free from cultivation (Bluwstein 2018). However, there is little room for agriculture in this vision, an important income-generating opportunity and key to food security around Tarangire and in rural Tanzania more broadly (Ponte and Brockington 2020; **McCabe and Woodhouse** Chap. 4), including among Maasai pastoralists living around Tarangire (Goldman 2003; McCabe 2003). The idea to limit cultivation in the TE goes back to the 1980s, when Borner (1985) advocated to put rural territories east and south of Tarangire under a “new land-use authority”. Maasai NGOs successfully rejected this proposal, fearing food insecurity and land alienation (Sachedina 2008; Igoe and Brockington 1999). Similarly, TANAPA’s efforts – in line with ideas put forward by Mwalyosi (1991a) – to establish a wildlife corridor in the 1990s on settled and farmed village land between Manyara Ranch and Lake Manyara were rejected by the Maasai (Goldman 2020).

Today, conflict lines over agriculture around Tarangire have shifted a bit. Some agropastoral communities have welcomed conservation initiatives which would limit agricultural land use in order to protect their land claims against farmers and pastoralists, who are deemed as outsiders, migrants and nonresidents. Makame WMA, for instance, has supported residents’ land rights against farmers through a REDD+ project that generates substantial revenues for village development (Bluwstein 2018; **Baker et al.** Chap. 15). Some farming communities in Burunge WMA have supported the WMA to challenge pastoralists’ claims to land in return for tourism revenues that are generated through exclusive safari tourism on pastoral territories (Bluwstein 2017). In other words, a political ecology of coexistence highlights that people’s (at times competing) economic interests, land claims and livelihoods have to be taken seriously by conservation authorities and scientists who are interested in promoting human-wildlife coexistence. If conservation science continues dismissing human needs and local knowledge, its findings will poorly resonate with people who practice coexistence on the ground (**McCabe and Woodhouse** Chap. 4). When conservation authorities cannot be trusted, their legitimacy is often called into question. When people living with wildlife risk losing access to vital resources, they may express their grievances, distrust, resent and resistance by sabotaging conservation initiatives (Mariki et al. 2015; Benjaminsen et al. 2013; Goldman et al. 2013; Goldman 2003; Witter 2021). Conversely, trust and legitimacy foster coexistence (**McCabe and Woodhouse** Chap. 4).

Third, the reliance on market-based initiatives such as global tourism and REDD+ payments to promote and incentivize conservation does not only often fail to match, much less exceed opportunity costs of conservation (Poudyal et al. 2018; Bluwstein 2017; McCabe 2003), it also puts the economic sustainability of conservation initiatives on shaky foundations (Sandbrook et al. 2020; Fletcher et al. 2016, 2020). When global tourism or carbon markets shift or become disrupted, local communities may be inclined to disengage from conservation agreements which demand a foregoing of land-based income and subsistence opportunities in return for market-based revenues.

2.4 Towards Convivial Conservation Around Tarangire?

In Sect. 2.3 I have illustrated several challenges towards human-wildlife coexistence as an opportunity to rethink conservation around Tarangire. To rethink conservation, I draw on a recently introduced vision of "convivial conservation", a set of concrete but at the same time radical propositions for how to move conservation forward (Fletcher and Büscher 2020; Büscher and Fletcher 2019, 2020). Convivial conservation is a response to various impasses of fortress and community-based approaches to conservation. To overcome these impasses, Büscher and Fletcher propose several "governance principles" for convivial conservation which should be embedded in what they call a post-capitalism vision of equity, structural transformation and environmental justice (Büscher and Fletcher 2019). In what follows, I highlight some of these key principles and how they relate to insights from political ecological research conducted around Tarangire.

First, the authors suggest that conservation needs to overcome "privatized expert technocracy" towards "common democratic engagement" (Büscher and Fletcher 2019). In other words, conservation decisions should be democratized to include different forms of knowledge and expertise. This echoes well research by Goldman who has demonstrated how local ecological knowledge is ignored and marginalized by conservation actors despite its potential to foster human-wildlife coexistence (Goldman 2003, 2007). Democratizing conservation thus means to make space for pluralism as to how people and wildlife can coexist (Goldman 2020; Brehony et al. 2018).

Second, protected areas should be replaced by promoted areas, which Büscher and Fletcher equate with overcoming the entrenched idea that nature has to be protected from humans through boundaries that separate both from each other (Büscher and Fletcher 2019). Promoted areas would serve both humans and non-humans, and they would need to operate in a new economic (postcapitalist) context. Rather than being places for capital accumulation through exclusive tourism in non-human landscapes, promoted areas are to become places where "people are considered welcome visitors, dwellers or travelers" (Büscher and Fletcher 2019). This general call needs to be adopted to the TE's context. Drawing on insights from political ecologists and geographers who have studied nonequilibrium ecologies of East African rangelands (Reid 2012; Goldman 2020; Brockington and Homewood 2001; Mwalyosi 1992), I suggest that an important step towards making the TE a convivial conservation landscape would start with rethinking the role of conservation boundaries. Rather than enforcing hard boundaries for core protected areas and WMAs, the management of these boundaries should become more *democratic* (see the first point on democratic engagement instead of expert technocracy) and *adaptive* to changing environmental conditions and people's resource needs.

Third, a transition from protected to promoted areas would need to be underpinned by a replacement of market-based initiatives with a new "value system" for nature (Büscher and Fletcher 2019). This new value system should not be exclusively bound up with economic valuation, but embrace economic, social, political, ecological, and cultural dimensions (also see **McCabe and Woodhouse** Chap. 4). Positioned against neoliberalization and its discursive and material manifestations such as "natural capital", "ecotourism", and "payments for ecosystem services", convivial conservation does not reduce people living with wildlife to entrepreneurs (*homo economicus*) who can be swayed to conserve nature in return for monetary gains. Whereas neoliberal conservation is underpinned by the idea of technocratic management of *scarce* resources, conviviality can be understood as the management of *abundance* (Lewis 2008; Kallis 2019). Rather than relying on the capitalization of scarce natural resources and ecosystem services to finance their protection, convivial conservation builds on the realization that people take care of animals and the environment for reasons other than immediate, short-term economic gains (Lewis 2008; Singh 2013; Ferguson 1985). Conservation NGOs and ecotourism entrepreneurs working in the TE are yet to fully embrace these insights (Davis and Goldman 2017; Bluwstein 2017).

Convivial conservation thus moves away from the neoliberal paradigm of conservation-through-ecotourism-and-spectacle – what Büscher and Fletcher call "touristic voyeurism" – to "engaged visitation". Conservation should thus not rely on short-term touristic trips into "the wild" by the wealthy and the elites in order to generate funding and revenues for nature protection. Instead of conventional tourism with all its problems, challenges and conflicts (Salazar 2009), decommodified long-term visitation with a focus on social and environmental justice should become the norm (Büscher and Fletcher 2019). Importantly, justice-oriented visitations would need to avoid the pitfalls of "voluntourism" and a "white savior" (or its reincarnation as urban savior) mentality (Freidus 2017; Bandyopadhyay 2019;

Mostafanezhad 2014). Research on the role of touristic voyeurism within the TE includes my study of two ecotourism investors in Burunge WMA (Bluwstein 2017), and Igoe's recent book, the *Spectacle of Nature* (Igoe 2017).

Fourth, convivial conservation may require a new relationship vis-à-vis the state. Büscher and Fletcher suggest that a transition to convivial conservation will have to work through organized structures, which will include the state in one form or another. Thinking through this challenge in the Tanzanian context, what is the role of the state in the TE? In what form do local communities, conservation NGOs and donors experience and engage with the Tanzanian state?

In a now famous interview, Bourdieu (1992) saw the (French) state as having a left hand of social institutions acting in the public interest, such as education and health care, and a right hand of the state, symbolized by the public-private sector of finance and banking. To Bourdieu, neoliberalization of the French state led to a regression of the left hand and a strengthening of the right hand (Bourdieu 1992). The Tanzanian state is, as any other state, a complex construct. In rural areas its left hand can be traced mostly through underfunded public infrastructure and service delivery such as schools, nurseries, housing for teachers and doctors, and emerging electricity lines. While there is virtually no investment in rural communities by the state, it extracts considerable revenues from the same (Brockington 2008). Rural people experience the right hand of the state in different ways, from corruption, to extraction of taxes, misappropriation and misallocation of funds, and policing, often in the name of conservation and natural resource management (Brockington 2008). Neumann (2001) goes as far as to argue – with a particular reference to Tanzania – that "state efforts to control wildlife resources in Africa are inherently violent or at least inherently conducive to violence". Clearly, these facets of the Tanzanian state are obstacles to an idea of human-wildlife coexistence that is based on democratic principles of conviviality. For now, a state-centric logic of fixed boundaries, and state-sanctioned neoliberal, heavily policed and militarized conservation prevails in Tanzania (Mabele 2017; Weldemichel 2020; Bluwstein 2017).

Büscher and Fletcher (2019) propose a two-step strategy to bring about the necessary transformative changes. In the short-term, the authors suggest that commodification of nature for conservation and tourism must be *subverted* through various smaller-scale initiatives and practices. In the medium to longer term, the authors point to larger-scale efforts to *overcome* commodification and expert technocracy towards democratization of conservation decision-making.

Applied to the case of Tanzania and the TE, I suggest that short-term efforts are already underway through everyday subversion of nature commodification and conservation boundary-making by people living with wildlife in the TE, as Goldman has shown in her research on Maasai communities living next to Manyara ranch (Goldman 2020), and as I covered through my research in Burunge WMA and in the village Kimotorok (Bluwstein 2017, 2019). Going beyond everyday acts of resistance, the state-centric logic of fixed boundaries for conservation can be subverted and eventually eroded in joint efforts by people living with wildlife, conservation

NGOs and scientists, and local governments. Put differently, the right hand of an extractivist, boundary-enforcing, and policing Tanzanian state (and tourism investors who act like a state, see Bluwstein (2017)) can wither away at the scale of villages, districts, community-based conservation initiatives and protected areas when stakeholders agree together that conservation should not be about enforcing boundaries but about building lasting relationships.

Medium to longer-term efforts will have to grapple with the dual challenge of revenues. Rural livelihoods depend on land-based resources for income generation (where the state is most absent), and conservation initiatives depend on revenues from donors, philanthropists, tourism, or payments from carbon offsets (here too the state is not playing a major role). In other words, the challenge is to address the lack of a strong left hand of the state. Here Fletcher and Büscher (2020) put forward "conservation-basic income" (CBI) as a non-market-based support for convivial conservation. CBI is understood as "a monetary payment to individual community members living in or around promoted areas that allows them to lead a (locally defined) decent life" (Büscher and Fletcher 2019). Importantly, these payments are not meant to incentivize people to behave in a particular way. They are *unconditional*, and hence contrary to the principles of neoliberal conservation that imposes a set of conditions on people in return for economic opportunities.

Where should the money for CBI come from? Büscher and Fletcher (2020) would like to see these funds to be generated from a collective pooling of resources to make conservation less dependent on global markets. The idea that the global community should pay people living with globally appreciated wildlife through substantial – and unlike in REDD+ schemes, *unconditional* – North-South transfers of governmental funds is not new (Balmford and Whitten 2003). Such transfers should not be understood as "aid or charity but as payment for developed countries' vast ecological debts" to the Global South (Bigger et al. 2021).

2.5 Conclusion

Environmental history and political ecology of the TE offer important lessons about the contested past and conflicted present of conservation. The TE's chronological history shows how dominant ideas about people and wildlife, nature and conservation have shifted over time, and how conservation has come to govern people and spaces around Tarangire. Attention to the TE's contested history highlights the mismatched and conflicting logics between state-led territorial administration and conservation, and pastoral land-use practices. Conservationists occupy a complicated and contradictory middle ground in this contested terrain.

Convivial conservation offers a vision for how to move conservation around Tarangire forward beyond its contested past and conflicted present by examining some of the root causes of conservation conflicts and challenges: the entrenched state-centric logic of territorial administration and conservation through fixed boundaries, the idea that rural livelihoods have to be compatible with western conservation ideals and initiatives, and the reliance on market-based initiatives for conservation.

However, convivial conservation is not a blueprint. It does not offer ready-made turnkey policy recommendations to problems that are at once political, economic, social, cultural, and ecological. The kind of solutions that convivial conservation puts forward require a radical overhaul of the political economic context in which conservation initiatives operate, a rethinking of how conservation science conceptualizes nature-society relations, and a turnaround as to how local ecological knowledge is included in conservation research and practice. In this sense, convivial conservation draws on a political ecological critique that is both a 'hatchet' to examine entrenched nature-society relations, and a 'seed' to envision something new (Robbins 2004). Simply put, a political ecological critique *can* be turned to solutions when we dare to embrace the possibility of alternative histories, futures, and ways of coexistence for people and wildlife (Goldman 2020).

A convivial approach is also promising to break with entrenched colonial legacies and neo-Malthusian anxieties that continue to underpin conservation science and practice in the global South (Adams and Mulligan 2003). During British colonial rule it was the colonial administration and the institution of indirect rule that sought to influence human behavior around protected areas in the present-day Tarangire Ecosystem to make it more compatible with goals of agricultural productivity and pastoral subsistence. Today, international NGOs, conservation scientists, and their local partners seek to guide rural communities and human behavior in ways that are compatible with western conservation objectives. While the objectives have changed, Malthusian concerns of human overpopulation in the TE remained. These were already raised in the 1920s by colonial administrators (Rohde and Hilhorst 2001) and have changed little in the last 100 years (Myers 1972; Mwalyosi 1991b; Prins 1992; Kideghesho 2009; Hariohay and Røskaft 2015). However, while human overpopulation has remained a major concern for administration and conservation authorities until this day, the ways in which people and wildlife have been problematized have changed significantly. While colonial administrators were more worried about wildlife impacts on people, conservationists have come to worry more about human impacts on wildlife (Bluwstein 2018). This change in perceptions has been underpinned by shifting discourses about people's and wildlife's moral standing (Neumann 2004). By and large, wildlife has ceased to be understood as "problem animals" and is today anthropomorphized into "charismatic species". People living with wildlife, on the other hand, are increasingly framed as potential poaching suspects and threats to conservation and environmental protection. Convivial conservation can be a path forward, beyond the zero-sum implications of colonial legacies and Malthusian anxieties.

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Chapter 3
A Conservationist Political Ecology in and for the Tarangire Ecosystem

Jim Igoe

Abstract Wildlife conservation in Tanzania is informed by the perceived inadequacy of protected areas relative to wildlife habitats and migration routes. It is important to note, however, that the creation of modern protected areas has been part of the ongoing processes of ecosystem fragmentation, with which conservationists are appropriately concerned. This chapter considers these concerns through the creation of Tarangire National Park and related protected areas between the 1950s and 1980s and the subsequent rise of NGO-driven conservation interventions in the 1980s and 1990s. It highlights the ongoing legacies of these histories, their implications for ecosystem fragmentation and human-wildlife coexistence, as well the challenges and opportunities they present for more holistic and equitable conservation alternatives. Its framework and object of analysis is a conservationist political ecology.

Keywords Political ecology · Maasai · Tanzania · Tarangire National Park · Fragmentation · Community-based land planning

3.1 Introduction

This chapter is best read in relation to Jevgeniy Bluwstein's contribution to this volume (Chap. 2), which also uses a political ecology framework (more on which below). Around the common ground of this shared framework, our chapters offer complementary information and analysis. Both engage colonial projects of territorialization, related conflicts and struggles around imposed boundaries and exclusions, and the continuing challenges they present to current and future conservation efforts. Bluwstein's chapter illuminates the state-centric logic of making and

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managing conservation territories. This one emphasizes the longstanding presence and influence of western conservationists in the landscape and particularly NGO-driven interventions across the turn of the millenium. Both chapters also argue that these processes and relationships continued to matter to current conservation interventions that turn on ideas of coexistence, collaboration, and connectivity. Finally, both are concerned with possibilities for convivial conservation, in the terms succinctly outlined by Bluwstein in his chapter.

Another shared feature of these two political ecology chapters is their foregrounding of what we might call hidden histories. Theories about why some historical narratives become prominent, while others are obscured, are beyond the scope of this chapter (but see Bluwstein's framing of a people's history of the TE). It is important to note, however, that hidden is a relative term. For instance, Ojibwe historian David Truer (2021) outlines the case for returning America's national parks to Native American peoples. His article reveals ongoing histories of violent displacements and exclusions, both caused and concealed by the U.S.'s national heritage. After nearly 150 years,[1] these histories are only now being addressed in mainstream media and remain overshadowed by prominent framings of parks as "America's best idea."[2] One thing the article makes abundantly clear, however, is that these histories are painful everyday realities for Native peoples excluded from their ancestral homelands and sacred sites.

My own research engages ways in which conservation and tourism have similarly displaced local communities in the TE and how they also operate to conceal those displacements and other important aspects of human and other-than-human coexistence. Accordingly, I am concerned with Tarangire National Park (TNP) as one of several externally imposed human projects, which dramatically transformed ecological and social relationships in this part of Tanzania in the twentieth century. The details of this argument concern the transformations of which the creation of TNP has been part, why certain portrayals and descriptions of these transformations have become powerful and prominent, and how TNP and related conservation spaces have operated as sites for the reproduction of these powerful and prominent narratives.

When considering TNP in these terms, it is important to bear in mind the ways in which Maasai herding systems evolved in relation to wildlife migrations (Homewood and Rogers 1991; Spear 1993). In short, this revolves around seasonal movements of domestic ungulates that mimic the movements of wild ungulates: concentrating near a permanent water source during the dry season and dispersing to new flushes of pasture and seasonal waters sources during the wet season. My book Conservation and Globalization (2004) describes and analyzes the ways in which the creation of TNP excluded Maasai and their livestock from a crucial dry season water source, a significant fragmentation of herding system ecology (also see Igoe 2002).[3] It also

[1] Yellowstone was established at America's first national park in 1872 and Native peoples were systematically removed and excluded across the ensuing years (Spence 1999).

[2] The title of a popular Ken Burn's documentary (2009), https://www.pbs.org/kenburns/the-national-parks/, accessed June 22, 2021.

[3] My understandings of these transformations are informed by numerous conversations and interviews with Maasai elders in the 1990s, who had previously brought their livestock to the Tarangire

disrupted longstanding systems for collectively managing the movement of people and livestock and in relation to wildlife migrations and related modes of environmental knowledge and resource management practices (see Igoe and Brockington 1999).[4]

While this is of only part of the TE fragmentation story, it is an essential part that has been consistently under considered. Of course, Maasai were not the only people present in and around the landscape that became TNP.[5] And of course there are other significant sources of ecosystem fragmentation, such as farming, mining, expanding human settlements, and tourism. However, nature parks are consistently portrayed as exceptional spaces, largely free of these fragmentation processes on their inside, but consistently threatened by them from their outside. By refiguring TNP and related conservation areas as part of the fragmentation of the TE, this chapter offers possibilities for improved understandings of fragmentation and coexistence, in relation to the pursuit of more convivial modes of conservation.

3.2 A Conservationist Political Ecology

My discussion in the following sections is organized in relation to a conservationist political ecology, which is at once a conceptual framework and an object of analysis. As such, it highlights the politics of conservation interventions in and around TNP, in relation to multi-faceted struggles over the disposition (the way things are placed and arranged in relation to other things), value (the regard that something is held to deserve) and meaning (the implied or explicit significance) of livelihoods, land, livestock, and wildlife. From the 1970s onward, conservation has been an important political force in this part of Tanzania. It thus makes sense to speak of a conservationist political ecology that exists and operates in the terms outlined above. As a conceptual framework, political ecology tends to the ways in which ecosystemic processes (fragmentation in this case) interact with inequities, power, and social struggles.

What might a conservationist political ecology focus help us understand about human-wildlife coexistence? Foundationally the concept of coexistence refers to the state of "existing together in the same place at the same time."[6] Significantly, a

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River prior to enforced exclusions beginning in 1970. I relate their accounts to a generalized model of Maasai herding systems was created by Maasai community leader Saruni Ndelalya in conservation with elders throughout what was then Kiteto District.

[4] During my field research in the 1990s, Maasai described and demonstrated to me the ways in which they built their bomas and managed their herds in relation to Wildebeest migrations and calving grounds in the Simanjiro Plains. Calving wildebeests can infect cattle with Malignant Catarrhal Fever, which can decimate household herds (also see Homewood and Rogers 1991: Chapter 9).

[5] Others included Datooga livestock herders, Dorobo hunter-gatherers, and a diversity of small-scale farmers, charcoal burners, and traders whose presence has largely been erased (Igoe and Croucher 2007; Årlin 2011, for comparable histories of the Greater Serengeti see Jan Bender Shetler's 2007 *Imagining the Serengeti).*

[6] According to the *American Heritage College Dictionary*.

major feature of mainstream conservation (Brockington et al. 2009: 9) has been the erasure and denial of local people's coexistence with conservation landscapes, tourists, and wildlife (Neumann 1998; Spence 1999; Gilio-Whitaker 2020). Recognizing the presence of local people in conservation landscapes is thus a potential improvement over previous modes of conservation. Of course, it matters a great deal how coexistence is admitted, imagined, portrayed, and pursued.

In the TE, as in many other conservation landscapes, relationships between people and animals are managed through zones and boundaries (Goldman 2009; Neumann 2016; Igoe 2017), down to chain link fences at the household level to mitigate predation on people and livestock.[7] Money is also a tool for managing coexistence, often through programs designed to help local people better appreciate the value of wildlife (recognize, be grateful for and contribute to its growth). Through local business opportunities and community development projects, these interventions seek to incentivize local people to care about, or at least tolerate, wildlife in their midst, refrain from activities that may impinge on wildlife habitats and migration routes, and perhaps even agree to relocate their lives and livelihoods away from designated wildlife conservation areas (Igoe 2004; Igoe and Croucher 2007; Goldman 2011).[8]

These modes of framing and pursuing coexistence are derived from fragmentation.[9] They focus on selected locations, relationships, and outcomes, while framing out others (c.f. Igoe 2010).[10] For instance, they usually have little to say about centuries of co-evolution between human lifeworlds (abiding practices, relationships, and experiences that makeup the world of a community of people) and wildlife habitats. They also revolve around claims about positive synergies between economic growth, human prosperity, and ecosystem health that are more often asserted

[7] Rod Neumann (2016) traces the phenomenon of buffer zones to philosophical and institutional framings of local people and conservation, in which coexistence requires separation. Mara Goldman (2009) describes and analyzes the phenomenon of wildlife corridors to imperatives of allowing animals to move across fragmented ecosystems, in yet another arrangement in which coexistence depends on separation. Drawing from Mbembe (2003: 25–26) I argue that conservation landscapes involve the imposition of complex zones and boundaries, which depend on separation that appears to restore holistic coexistence (Igoe 2017: 56). So, while chain link fences may facilitate the coexistence of people and carnivores in some contexts, they also exemplify the logic of these arrangements.

[8] Throughout my research on both sides of TNP, it was common to hear government officials and NGO representatives state that local people might find that moving would be in their own interests, as a way of gaining easement payments, tourist revenues, better land elsewhere, or some other incentives. Such statements were often accompanied with warnings about the dangers of living in proximity to wild animals and the ever-present possibility of forced relocation by the state.

[9] This fragmentation is related to, and sometimes the same as, the fragmentation of which conservation ecologists speak relative to the TE. For example, the imposition of protected area boundaries has transformed livestock ecologies in ways that contribute to wildlife habitat fragmentation – an example of why it is important to consider the co-evolution of human lifeworlds and wildlife habits (which could also arguably be considered the same thing). The presentation of fragments as wholes (e.g., Tarangire appearing as unfragmented African Nature) can moreover conceal and distort important aspects of habitat fragmentation as it concerns ecologists.

[10] While these practices have recently been refined in remarkable ways in the TE, works in this article's bibliography and illustrate that they have co-evolved with mainstream conversation and related modes of tourism.

than proven. In TE these are commonly presented in terms of tourism driving economic growth, creating non-land-based economic opportunities for local people, and generating resources for continued wildlife conservation (cf. **Bluwstein** Chap. 2). While continuously contested and fraught with contradictions (more on which below), these narratives appear to have compelling explanatory power in relation to the realities that are being portrayed. While quite literally only part of the story, they are made to appear as the whole of the story. The perspective of conservationist political ecology can help us to better understand these processes, the interests and values they reflect, realities and relationships they exclude, and the possible costs of these exclusions. Throughout the following sections, I relate this perspective through histories of fragmentation related to the TE.

3.3 Bounded Spaces and Boundless Visions: Tarangire and the Maasai Reserve (1923–1970)

Tanzania's world-famous Northern safari circuit promises authentic encounters with panoramic African nature and exotic African people, especially Maasai.[11] Indeed, Maasai are popularly portrayed as coexisting with wildlife in vast and timeless landscapes. This hegemonically popular vision is supported by the management of segregated spaces, through which encounters and interactions between people, as well as between people and wildlife, are managed and portrayed. For example, cultural villages are designed to facilitate encounters between tourists and select groups of local people in carefully managed situations (e.g., cultural performances and souvenir markets). Such arrangements are common throughout East Africa (for Maasai experiences see Collett 1987; Bruner 2001; Hodgson 2001; Igoe 2004).

At the same time, parks like Tarangire are organized around particular ways of seeing African nature. Typically, these involve commanding overviews of panoramic vistas, with tourist lodges being strategically sited in relation to such views. These are combined with managed up-close views, usually from within safari vehicles, between tourists and wildlife (c.f., Igoe 2010). These ways of seeing nature prescribe ways of being in nature, since they automatically exclude the possibility of lifeworlds and livelihoods that entail dwelling and laboring in the managed landscapes (Cronon 1995). Thus, the creation of national parks and related protected areas frequently entails forced exclusions of local people and their lifeworlds in the

[11] Although there has been some diversification of East African tourism experiences in recent years, including tours for Africans and other non-western people, the regions booming tourist economy is driven by internationally networked tourist enterprises, catering to westerners and turning on the visions briefly outlined above. This can be seen in the Tanzanian Governments branding of the country as "The Land of Kilimanjaro, Zanzibar, and the Serengeti" (Igoe 2017: 20). Significantly, this kind of hegemonic "mainstream tourism" coevolved with hegemonic "mainstream conservation" (Brockington et al. 2009) in East Africa, including the creation of Nature Parks (Neumann 1998; Igoe 2004, 2017; Lekan 2020). Also see the documentary film *A Place without People* (2009), https://vimeo.com/ondemand/place, accessed June 19, 2021.

establishment of wilderness areas and tourist infrastructure. Some local people are then selectively readmitted in the terms briefly outline above. Such arrangements are a problematic legacy of national parks and conservation in the United States (Spence 1999; Truer 2021), which have been reproduced in nature-based tourist circuits in Tanzania and around the world (for a global overview see West and Carrier 2004; West et al. 2006; Dowie 2009).

Though a signature destination in Tanzania's Northern Tourist Circuit, Tarangire is overshadowed by the world-famous Serengeti Plains and Ngorongoro Crater; its history is accordingly less well documented.[12] My own knowledge of this history is derived mainly from the accounts of elders (Maasai and others) living just outside Tarangire, with whom I interacted during two separate field projects: in villages bordering the park to the east (between 1992 and 1996) and in villages bordering the parks to the west (between 2005 and 2006). Elders living in the eastern villages were mostly Maasai herders, many of whom had herded and lived inside Tarangire prior to evictions and exclusions in the early 1970s (Igoe 2004). Elders living in the western villages were members of mixed ethnic communities (including Arusha, Datooga, and others). Many of them had been displaced from elsewhere, and sought to make a living through farming, hunting, and charcoal burning (Igoe and Croucher 2007).[13]

Maasai elders described the appearance of helicopters over their pastures and homesteads in the early 1970s, followed days later by armed rangers in Land Rovers, who ordered them to move east of newly placed border beacons. Elders living west of the park described rumors that their villages would be relocated, to make way for the unification of Tarangire and Lake Manyara national parks. They also describe their relief following a radio address from President Julius Nyerere. In the course of this address, they claim, Nyerere reassured them that they would not be relocated, since the country's policy of village-based socialist development (Ujamaa Vijijini) took precedence over wildlife conservation and tourism.

As a field researcher, I have been struck by the consistency of these accounts, independently related in many different locations across two decades. At the same time, they have been difficult to corroborate with official documents and written histories. I have been unable, for instance, to find documentation or recording of the

[12] The evictions of Maasai herders from Serengeti National Park in the 1950s and the so-called compromise that allowed some of them to continue living in the Ngorongoro Conservation Area are well documented historical events (Bonner 1994; Shetler 2007; Lekan 2020).

[13] Colonialism and post-independence villagization, along with numerous other conservation and development schemes, have entailed significant displacement and relocation over many years. As a result, many villages in northern Tanzania are precarious homes to marginal mixed-ethnic communities. Many of these communities are not officially recognized and so subject to further displacement on the grounds that they do not belong where they happen to be. Examples include Arusha and Meru people displaced from the slopes of Mt. Meru by Arusha National Park and coffee plantations (Igoe and Brockington 1999) and Datooga people from Hanang. Others are descended from plantation laborers, who came from as far away as Zambia (then Northern Rhodesia) and Malawi (then Nyasaland). There are many undocumented movements and settlements (e.g., Igoe and Croucher 2007).

aforementioned radio address, though it has been related to me in consistent detail by people who were living in different parts of Tanzania in the early 1970s. These accounts are moreover consistent with ongoing conservationist efforts to restrict and remove local people and their livelihood activities from between Lake Manyara and Tarangire national parks (Igoe and Croucher 2007; Bluwstein 2018), with the putative goal of restoring ecological connectivity (c.f., Goldman 2009).

Camilla Årlin's (2011) historical study, Becoming Tarangire, adds welcome archival detail to these oral accounts. Her study confirms that Tarangire was a site of diverse human communities and livelihood activities prior to the 1970s. It also reveals an ironic contradiction to contemporary conservationist imperatives: contrary to what contemporary conservationists might think, Tarangire was created as a game reserve in the 1950s that was "specifically designed to become isolated" (ibid: 201). Her study also demonstrates that a wilderness ideal was not suddenly and systematically imposed in this context, but was promoted and contested throughout the colonial period, and only became firmly established after Tarangire became a park in 1970 (c.f., Igoe and Brockington 1999; Igoe 2002, 2004).

Maasai elders who related these transformations to me in the 1990s frequently recalled their exclusion from Tarangire in the early 1970s. They wistfully described Tarangire as a place of pastoral flourishing, with abundant dry season pasture. In that place their livestock could graze and drink from the Tarangire River, allowing for the recovery of wet season pasture in the Simanjiro Plains.[14] It was also home to a large wetland, which ensured the sustainability of their herding systems during periods of extended drought (Igoe 2002). Painful memories of their exclusions from the park in the 1970s, and efforts to expand the park eastward in the 1980s, fueled local hostilities towards anything associated with conservation, which continued through the 1990s and into the 2000s (for details see Igoe 2004; Cooke 2007).

Significantly, the eastern boundary of Tarangire is historically coterminous with the western boundary of the now-defunct Maasai Reserve (Fig. 3.1). Containment of Maasai in this reserve during colonialism (1923–1961) justified removals of people from land slated for European acquisition and facilitated indirect rule. Criteria derived from noble savage stereotypes and racist ideologies of ethnic purity determined spatialized segregations of people deemed to be Maasai and members of other ethnic groups (Collett 1987; Hodgson 2001; Lekan 2020). Tarangire was consolidated as a wilderness adventure space a decade after the Maasai reserve was officially dissolved. However, colonial visions of Maasai people remain indispensable to the framing and funding of conservation in this part of Tanzania. They are also continuously reproduced in cultural performance spaces (Bruner 2001; Igoe 2004; Salazar 2012).

[14] These accounts are consistent with a herd survey undertaken by Anthropologist Alan Jacobs in 1957, which indicates significant concentrations of people and wildlife along the Tarangire River during the dry season of this year. Alan also described these settlements to me during conversations that we had in the early 1990s.

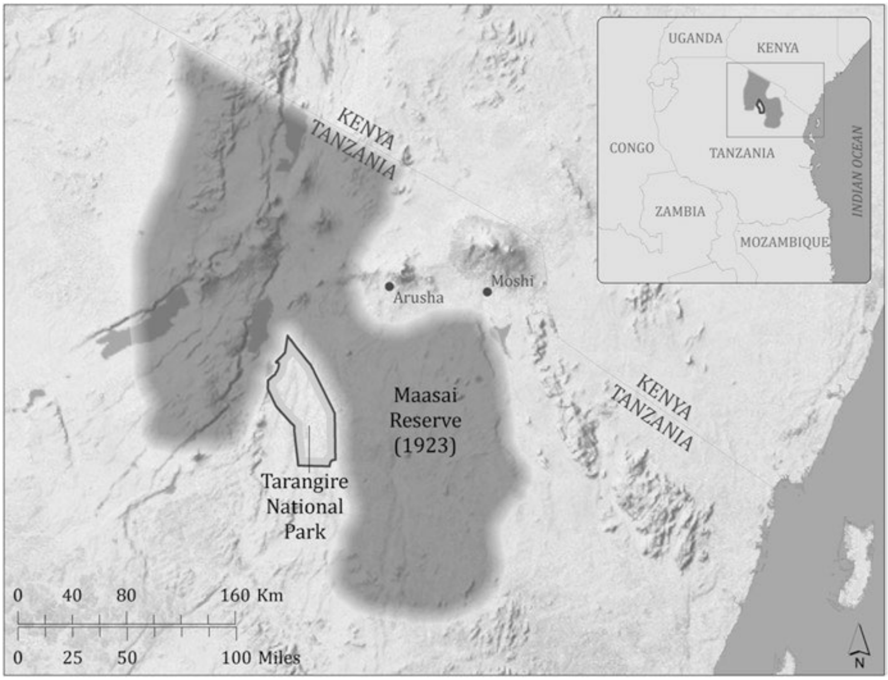

Fig. 3.1 Map showing the contemporary boundaries of TNP (present day) in relation to the historical boundaries of the Maasai Reserve (1923–1961). The two units of land coexisted from the mid-1950s through 1961. (This map was produced by Jonathan Chipman, who referenced Hodgson (2001) and ESRI (http://www.esri.com))

3.4 The Emergence of Conservationist Political Ecology In and Around TNP (1970–2000)

Western conservationists were present in Tarangire in the years leading up to independence, and even advocated for the removal of local people to make more room for wildlife (Årlin 2011: 187). Unlike their counterparts in the more famous Serengeti, however, they lacked the political clout to exclude local people from in and around designated conservation spaces. During the 1950s, western conservationists and the Maasai District Administration were in conflict over expanding Tarangire into Maasai Territory. Maasai and other local people were allowed access to Tarangire throughout the 1950s and 1960s (Årlin 2011: 188).

Conservation did not become a sustained political force in this part of Tanzania till after Tarangire was well established as a national park. Elders I talked with in the 1990s frequently referred to schemes to create a conservation area to the east of Tarangire, which would have effectively "swallowed all their villages" (Igoe 2004: 64–65). A village official provided me with a copy of the proposal, which he claimed was leaked from a secret meeting of Western conservationists and Tanzanian wildlife officials (Igoe 2004: 64–65).[15] The proposal is strikingly similar to "The Increasing Isolation of Tarangire National Park" (1985) by Markus Borner. In that article, Borner avers that Maasai in Simanjiro "still favor the coexistence of game and livestock" (emphasis mine) and that "there seems to be no serious objection to the dual use of the Simanjiro Plains and the Lolkisale Game Controlled Area for livestock and wildlife" (ibid: 95). Accordingly, he recommends that Simanjiro, along with Lolkisale and Mkungunero[16] be united under a new land use authority modelled after the Ngorongoro Conservation Area (ibid: 95).

Two of Borner's points bear elaborating here. The first is his brief acknowledgement that Maasai (or at least their livestock) coexisting with wildlife is not new in Tarangire. Indeed, he emphasizes that it is something that Maasai continue to favor. The second is that "there seems to be no serious objection to the dual use … for livestock and wildlife" of landscapes outside of Tarangire. Considering the long coevolution of human lifeworlds and wildlife habitats in East Africa, the implications of his first point call for a great deal more careful attention before we can arrive at meaningful formulations of coexistence for contemporary conservation efforts. For example, the exclusion of livestock from Tarangire disrupted patterns of human-wildlife coexistence and undermined the annual recovery of wet season pasturelands, to the detriment of livestock and wildlife (Igoe 2004; Cooke 2007; also c.f. Msoffe et al. 2011: 270–271).

With regards to Borner's second point, there is significant variation in local attitudes towards wildlife on village lands (Cooke 2007; Igoe and Croucher 2007; Sachedina 2008; Benjaminsen et al. 2013; **Kiffner et al.** Chap. 1). However, Maasai community leaders were less concerned about wildlife per se than they were with conservation as they were experiencing it during this period. In the late 1980s, they formed an intervillage entity called The Simanjiro Anti-Conservation Committee, which lobbied the Prime Minister to block the proposed conservation area. The committee was not opposed to environmental care and sustainable management of natural resources, and this is not what they understood conservation to mean. Rather they were opposed to conservation as a political force that was seeking to control their territories and to regulate their lives and livelihoods (see Igoe 2014; also c.f., West 2006; Dowie 2009).

These events roughly mark the beginning of an intensifying conservationist political ecology in this part of Tanzania, as outlined in the introduction. By the early 1990s conservation interventions were a matter of public concern in villages throughout the Simanjiro Plains. Conservation outreach by international conservation NGOs and their Tanzanian partners in these villages had a campaign-like feel,[17]

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[15] A conservationist who was at the meeting confirmed that he leaked the proposal to relevant local officials.

[16] Mkungunero, previously a Game Controlled area and since 1996 a Game Reserve, lies due south of Tarangire National Park. For details of current boundary disputes between Mkungunero and Maasai communities see Jevgeniy Bluwstein's chapter in this volume.

[17] These arrangements often confounded distinctions between governmental and non-governmental actors. Some NGO partners were Maasai leaders who were not from the TE, which contributed to

as they sought to win the hearts and minds of local people, to enroll them conservation-defined agendas, and promised beneficial outcomes to their intended local constituents. These politics intensified in relation to wider struggles over the disposition, value, and meaning of livelihoods, land, livestock, and wildlife. A rising generation of community leaders fostered opposition to these interventions, in pursuit of an increasingly popular agenda of community-based land rights and cultural self-determination.

The presence of conservation agendas and interventions meanwhile continued to intensify on all sides of the park, accompanied by a proliferation of conservation areas and associated political struggles (Goldman 2011; Davis 2011; Bluwstein 2018). Significantly, the logic of these interventions consistently turn on Borner's (1985) vision of extending conservationist control of landscapes for the protection of wildlife habitats beyond parks and the restoration of connection between them (c.f., Goldman 2009). While this vision sets its sites on village lands, it revolves around Tarangire as an ecological and economic center for conservation and community development beyond its borders (Igoe 2017: 37).

From the perspective of this book TNP can be accurately described as an especially large and significant fragment in a larger fragmented landscape. It was created by as a game reserve by colonial conservationists in the 1950s (Årlin 2011), transformed into a national park with significant influence and support from western conservation organizations (Igoe 2004: 99–100), and is now managed under the authority of Tanzania National Parks (TANAPA). Along with other Tanzanian parks, TNP supports distinctive modes of coexistence, most visibly between tourists and wildlife and between scientists and wildlife. A conservationist political ecology further highlights the realities of Tarangire as an enclave of amenities, infrastructure, and high-end real estate. It is also a celebrated site of less tangible, but coveted, values – including experiences, narratives, and images (c.f., Buscher 2010; Igoe et al. 2010).

This arrangement turns on stark inequities, which are consistently deemphasized in conservation discourses and policies. For many people living around Tarangire, however, these inequities are an unignorable feature of their everyday lives. They are denied access to territories and resources, which they regard as part of their ancestral lifeworlds and heritage. In galling contrast, people from faraway lands, with no apparent or imaginable claims to these places, enjoy easy access to them and all that they have to offer. Christine Noe (2019) refers to the continuing legacies of colonial boundary making in such contexts as The Berlin Curse.[18] Bluwstein

conflicts with community based Maasai NGOs in target villages (Igoe 2004). Other partners were government officials who worked international conservation NGOs (Igoe and Brockington 2007: 440; Igoe and Croucher 2007: 544–547)

[18] With reference to the Berlin conference of 1884, where European powers began to draw/map boundaries by which they would subdivide Africa in order to rule the continent. With reference to the Selous Game Reserve, Noe argues that these techniques for managing space and people – and we can add wildlife – continue to the present day. This insight is consistent with Mbembe's (2003: 25–26) formulations of colonial space-making techniques.

(Chap. 2) shows how the contested dynamics of this kind of boundary-making continue in Kimotorok Village in relation to TNP and the Mukungunero Game Reserve.

Conservationist political ecology analyses begins with these historically produced inequities, as they are related to the creation of conservation territories. They are also related to a diversity of non-local relationships, interests, and agendas, which Tarangire has attracted and concentrated in the TE. As James Ferguson (2006: 42–48) has shown, many African parks are spaces that are locally enclaved (cut off from contiguous places and communities) and globally networked. Their relationships to Western governments, development agencies, corporations, investors, philanthropists, and researchers are often far stronger than their relationships to local people. Tarangire and its satellite conservation spaces exemplify this larger pattern of relationships (Igoe 2017; **Bluwstein** Chap. 2). They are also spaces in which prominent narratives and portrayals of Maasai, wildlife, and possibilities for their coexistence are produced.

3.5 The Reemergence of the Maasai Steppe (1999–2009)

Since the turn of the millennium, conservation politics in the TE have been increasingly and improbably connected to politics at other scales and in completely other places. During her bid for the U.S. presidency in 2016, Hillary Clinton appeared on the Ellen DeGeneres show and revealed that the elephant was her spirit animal, even though it is the official symbol of the Republican Party. Clinton attributes her connection with elephants to a visit she made to northern Tanzania in 1997, accompanied by her daughter Chelsea. This and subsequent elephant encounters inspired Clinton's efforts to combat poaching, both as First Lady and U.S. Secretary of State (Hance 2016). In 2013, Chelsea Clinton visited Tarangire in support of the Clinton Foundation's Partnership to Save African Elephants (Igoe 2017).

The Clintons' visits were associated with conservation initiatives led by American conservation NGOs, which were funded and supported by the US Agency for International Development (USAID). At the turn of the millenium, the African Wildlife Foundation (AWF) introduced the Maasai Steppe Heartland (MSH) (Cooke 2007: 25), as part of its African Heartlands Program and related 10-year capital campaign (Igoe 2017: 65). Through maps, images, and narratives, the MSH was represented as a conservation landscape of approximately 22,000 square miles (Igoe 2017: 125), with Tarangire and Lake Manyara national parks as its ecological anchors (Igoe 2017: 37 and 125). These portrayals revitalized colonial visions of Maasai as proud, traditional people, adding green capitalist visions of optimizing economic growth and ecosystem health. Specifically, it was presented as "the vast plains of northern Tanzania where wildlife and Maasai people live side by side" and as one of several "vast landscapes that function ecologically and economically, so that the needs of people and wildlife can be balanced" (Igoe 2017: 35).

The MSH vision elaborates Borner's (1985) recommendations for increasing connectivity, which is now shared imperative of international conservation NGOs and Tanzanian government agencies. Indeed, one of the central imperatives of NGO-driven large landscape conservation, in general, is to expand conservation territories, by creatively combining a variety of approaches and mechanisms.[19] These include NGO-managed protected areas and for-profit private conservancies. Growing conservation NGOs, in relation to growing conservation territories, has been accompanied by capital campaigns (Chapin 2004; Sachedina 2010) lobbying for increased political influence (Garland 2008; Corson 2016), increases in corporate partnerships (Chapin 2004; MacDonald 2010a) and a growing emphasis on commodifying and monetizing nature (McAffee 1999; Igoe and Brockington 2007; Fairhead et al. 2012).

A vision of a landscapes that "function ecologically and economically" valorizes these transformations through imagined spaces in which economic growth and ecosystem health are reciprocally enhancing. Associating Tarangire with the Maasai Steppe adds compelling specificity to this generic vision. To quote Thomas Spear's introduction of his book Being Maasai (1993): "Everyone 'knows' the Maasai." Spear's sentence encapsulates the power of particular ways of "seeing Maasai" (Hodgson 2001) rooted in colonial-era segregation and reproduced through tourism (Bruner 2001). Popular western ways of seeing African wildlife are similarly powerful, popular, and partial (Bonner 1994; Adams and McShane 1992).

Such portrayals of people and wildlife are major sources of value for Tanzania's tourist industry. They are also valuable to specialized portrayals of conservation in the Maasai Steppe, which turn on the idea that expanding and improving conservation territories will drive economic growth for the benefit of African nations and people (e.g., Muruthi 2005: 2–3). Through the logic of these portrayals, money appears as a tool for mediating the coexistence of people and wildlife. Increased local economic opportunities, this logic goes, will enhance local people's appreciation of the value of wildlife and associated conservation territories. Local conservation capacity can be enhanced through educational programs and operationalized through community-based land use plans, which typically involve setting aside conservation areas that people will refrain from using for other purpose. Claims that local people benefit from resulting tourist revenues consistently overlook costs of foregone access to these spaces and the thorny question of benefit distribution (Igoe 2006; Igoe and Croucher 2007; **Bluwstein** Chap. 2).

While such portrayals and interventions have been continuously contested in this part of Tanzania (e.g., Igoe and Croucher 2007; Goldman 2011; Bluwstein et al. 2016), they have nevertheless gained prominence that extends far beyond the Maasai Steppe. Both Hillary and Chelsea Clinton narrate their passion for African

[19] I use the term conservation territory to indicate spaces that are organized for the purposes of nature conservation and to the exclusion of other activities. Ideally, from a mainstream conservationist perspective, these spaces should be continuous (e.g., the abiding idea that Tarangire National Park and Lake Manyara National Park should be connected together in a single contiguous territory). While conservation practices may also be undertaken in other kinds of spaces (sometimes described as mixed-use), the mapping and making of conservation territories is a prominent practice of mainstream conservation (e.g., the Fifty Percent for Nature Movement).

conservation around life-changing encounters with elephants, in relation to U.S. foreign policy, promoting a foundation, and presidential bid. In the years between the two Clinton visits (1997–2013) other U.S. officials paid similar visits to the Maasai Steppe, including lawmakers and a cabinet secretary (for details see Sachedina 2008: 22–23; Igoe 2021: 39–44).[20] Other influential visitors have included philanthropists, celebrities, filmmakers, and corporate volunteers (Igoe 2017, 2021).

Images and narratives of these visits were derived from a limited selection of relatively small spaces – again fragments – but were consistently made to represent a much larger whole of the MSH itself, such as it was (Igoe 2017: 70). High-profile visitors encountered these spaces as part of a carefully orchestrated touristic experience (Sachedina 2008: 19–23; Igoe 2017: 63–69). And their visits were documented via photographs, videos and textual narratives that are remarkably consistent in their subject matter and perspectives and the stories that they tell. They are also remarkably consistent with green capitalist policy discourses that have been globally influential since the turn of the 1990s (MacDonald 2010b; Corson 2016; Igoe 2021).

This turn-of-the millenium revitalization of the Maasai Steppe portrays people and nature in ways that relegate other important realities and relationships to outside of its frame. This includes its own entanglements with NGO fundraising, celebrity branding, and geopolitics and how these may influence actual conservation practice. It also includes contradictions and negative ecological effects of growth, such as the tourist industry's carbon footprint and tourists crowding into wildlife habitats. When local people appear in the frame of these portrayals, it is almost always in the limited roles of timelessly traditional Maasai, grateful beneficiaries, or (rarely) modern wage workers and entrepreneurs. How might these arrangements be changing in relation to subsequent and ongoing conservation interventions?

3.6 Boundaries to/for Coexistence; What Next for the TE?

Chelsea Clinton's visit to the TNP in 2013 coincided with significant shifts in the conservation political ecology in the TE. USAID discontinued its support for the AWF's work in the TE in 2014. It in turn funded The Northern Tanzanian Rangeland Initiative, led by the Nature Conservancy, in 2015 (Bluwstein 2018). This new initiative sought to take a less antagonistic approach to conservation, while valorizing pastoralist livelihoods and better coexistence of people and wildlife. In pursuit of this vision, to whatever extent it may ultimately be achieved, it will be necessary to tend to the ongoing processes and relationships outlined in the previous sections.

Most notably, the spatialized colonial legacies of the TNP and the Maasai Reserve continue. While the Maasai Reserve has been officially dissolved for

[20] Henry Paulson visited the Maasai Steppe Heartland in 2007, as Secretary of the Treasury in the second term of the administration of President George W. Bush, the younger.

decades, the boundary between game-reserve and human-reserve remains a defining feature of contemporary conservation political ecology in the TE. As in previous decades, communities living just East of those boundaries continue to struggle against their renewed expansion and negative impacts on their livelihoods (**Bluwstein** Chap. 2). As in previous decades, conservationists remain concerned with conservation territories and practices beyond the TNP, in the former Maasai Reserve.

A key difference in the current moment is the community land titling movement, led by UCRT (see **Brehony et al.** Chap. 5), which builds on key recommendations of the Presidential Land Tenure Commission (URT 1994).[21] The movement's approach to collective resource management and stewardship makes it attractive to conservationists.[22] One of the its main attractions for conservationists, is that it seeks to restore and protect rangelands, which are also important corridors and habitats for wildlife migrating from TNP during the wet season.

As compatibilities between wildlife conservation and pastoralism reflect centuries of coevolution between pastoralist lifeworlds and wildlife habitats, it is heartening that conservationists are supporting pastoralist land rights and resource management in the TE. It is important not to forget, however, that conservation has been one of the biggest threats to pastoralist lifeworlds in this part of Tanzania for the past several generations. Indeed, one of the movement's leaders, Edward Loure, describes it as a long-term response to Tarangire evictions of his childhood. One of its key goals, therefore, is to protect Maasai lifeworlds from further disruptions by tourism and conservation area expansion.[23]

Along these lines there is more that could be done to repair the mutually reproducing relationships between pastoralist lifeworlds and wildlife habitats in the present day. However, this would entail admitting and engaging the disruptions that TNP has presented to livestock herding ecologies, and how these are related to the agricultural transformations that conservationists are working to reverse in the TE. Flourishing pastoralist lifeworlds require substantial and dependable dry season water and pasture and permanent drought reserves. And restoring local control of land in Simanjiro cannot change the fact that the best dry season pastures and drought reserves in this part of Tanzania are inside TNP. The ecological benefits of these arrangements would be greatly enhanced by reopening parts of TNP for livestock herding on a seasonal permit basis (see Igoe 2002, 2004: Chapter 2).

As politically difficult as this arrangement would be, it would in step with arrangements that are currently being promoted for national parks in the United States and other settler colonial societies. An internet search for "decolonizing

[21] Led by Professor Issa Shivji, one of the main recommendations of the Land Tenure Commission was that land be held in common by rural communities, under the authority of village assemblies and local elders' councils.

[22] To the extent that Edward Loure was awarded the Goldman Prize for Environmental Stewardship in 2016, https://www.goldmanprize.org/recipient/edward-loure/, accessed on March 12, 2021.

[23] See *Made for Minds,* Pastoralists in Africa Fight for their Rights, https://www.dw.com/en/pastoralists-in-africa-fight-for-their-rights/a-19262085, accessed March 12, 2021.

conservation" and "decolonizing environmentalism" reveals the extent to which repairing the colonial legacies of nature conservation is becoming a global project. In the TE this kind of work would need to admit the diversity of other-than-Maasai African people, who were excluded and marginalized by the creation of the Maasai Reserve and other similar territories (Årlin 2011: 188), and who are less often recognized as right-holding constituents of conservation in this part of Tanzania.

This is a matter of equity, of course, since these people also deserve to benefit from conservation and development. More elaborately, it is a matter of recognizing these peoples' place-based lifeworlds and ways of caring for more than human environments. Accordingly, it will be important to better understand historical synergies between different peoples' resource management practices, to help repair these when possible, and to support emergent practices when old practices have been lost. Such approaches would be more open-endedly coalitional and collaborative compared with those of mainstream conservation today, and they will require more openness to diversity and uncertainty. They may also bring forth a new kind of conservationist political ecology, extending possibilities for coexistence – being together in the same time and place – to conviviality – being "with life" in ways that support human and other-than-human flourishing (see **Bluwstein** Chap. 2 for details of convivial conservation).

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Chapter 4
Maasai Wellbeing and Implications for Wildlife Migrating from Tarangire National Park

J. Terrence McCabe and Emily Woodhouse

Abstract A multi-dimensional, locally grounded conceptualisation of human wellbeing provides a way to understand the complexity of people's lives, incentives and aspirations with the potential to inform socially just conservation interventions that have local legitimacy. Based on semi-structured group interviews and a survey at the household level, we discuss how wellbeing is conceptualized among the Maasai of Simanjiro, how this differs between social groups, and how social aspirations have implications for conservation interventions in the ecosystem. We highlight how communal grazing land which aligns with conservation priorities is of paramount importance, but agriculture is also central to people's lives and there is a growing emphasis by younger men on securing private land. Social unity also constitutes wellbeing, but is jeopardized by land disputes and party politics, and is tied up with mistrust of external actors rooted in a history of land and resource alienation. Land insecurity is viewed as a threat to wellbeing, and partly drives the conversion of land to agriculture as well as other aspirations such as education. The findings suggest that future interventions will need to increase land security, work to establish trust in conservation processes and institutions, and provide equitable alternatives to agriculture to meet subsistence needs.

Keywords Maasai · Conservation · Well-being · Age · Gender · Wildlife

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4.1 Introduction

There is increasing acknowledgement in the conservation literature that narrow economic indicators are inadequate for describing poverty and human wellbeing, failing to capture local priorities and the varied ways that local lives may be impacted by interventions (Loveridge et al. 2020; Woodhouse et al. 2018). Ecologically relevant practices are motivated by not only improved income and livelihoods, but a broad range of values such as identity, social relationships, and a sense of fairness (Coulthard et al. 2011; Chaigneau and Brown 2016). A conservation project that is designed to reduce poverty may be successful in achieving that goal, but can also result in increased social stratification and conflict within local communities (West 2006; Gurney et al. 2014). Instead, more inclusive alternative approaches are being explored that capture multiple dimensions of a good life, with the aim of improving the design and evaluation of conservation to better meet ecological and social goals (Woodhouse et al. 2015). Taking a more holistic approach through the lens of multidimensional wellbeing has the potential to improve social justice, as well as the local legitimacy of conservation, and ultimately its success. Adopting a wellbeing approach presents challenges in defining, understanding, and measuring wellbeing in a way that is relevant to external policy-makers, practitioners, and local communities. There is, however, a growing recognition that understanding wellbeing for conservation must be grounded in locally relevant conceptions even within a broadly comparable universal framework (Woodhouse et al. 2018). In this chapter we discuss how wellbeing is conceptualized among the Maasai of northern Tanzania, how this differs among men and women, and how social aspirations have implications for wildlife migrating out of Tarangire National Park (TNP) and conservation interventions in the dispersal area in Simanjiro, a key wet season range for wildlife species in the Tarangire Ecosystem (Kahurananga and Silkiluwasha 1997). Although some of the results of this study have been previously published (Woodhouse and McCabe 2018), we are incorporating unpublished materials here including quantitative data and a synthesis of our learning about Maasai wellbeing from previous and ongoing research projects on risk and livelihoods.

4.1.1 Wellbeing

Although there are a number of definitions of wellbeing, the one we used in our research is based on the framework developed by the Wellbeing in Developing Counties project based at the University of Bath. Here wellbeing is defined as: "the state of being with others, which arises where human needs are met, where one can act meaningfully to pursue one's goals, and where one can enjoy a satisfactory quality of life" (McGregor 2007). McGregor and Sumner (2010) identified three interacting dimensions of wellbeing: (1) the objective material circumstances of a person; (2) a person's subjective evaluation of his or her life; and (3) a relational component based on how individuals interact with each other to meet their own goals and needs. What is different about a wellbeing approach, and what makes it difficult to put into use for policy and practice is that each community is different, and each community is not a homogenous group of people but may be split along ethnic, political, wealth, religious, gendered or other lines (see Agrawal and Gibson 1999). This means that one cannot rely on a predefined set of categories or indicators as communities will differ in their conceptualization of wellbeing, and segments of the community may also differentially conceptualize what wellbeing means to them (Daw et al. 2011).

4.2 Maasai Social Organization and the Diversification of Livelihoods

There is a rich body of literature concerning the Maasai, and here we present a brief overview to help the reader contextualize this study. Although the commonly depicted image of many African pastoral people emphasizes mobility and an almost exclusive dependence on livestock for subsistence, this romanticized depiction does not represent the complexity of pastoral livelihoods today or even in the past (Homewood 2008; Homewood et al. 2009). This is especially true for the Maasai, whose image of brave warriors driving herds of cattle though the dust appears on billboards, in movies and on television shows and is shown throughout the world.

The Maasai occupy approximately 150,000 km^2 spanning Tanzania and Kenya. The Maasai population in Tanzania tends to be less economically developed than that in Kenya, primarily due to government policies, especially with respect to land tenure. While land in Tanzania remains as property of the state, with management rights devolving to the village, in Kenya land is privatized and households have title deeds and the rights to buy and sell property.

Maasai social organization is based on three articulated institutions: family, territory, and the age grade/age set system. Households are generally polygamous with each married woman having her own household or *enkaji*. A man and his wives, children, and dependents form the next organizational unit called the *olmarei*; this is the basic family unit and is responsible for the management of livestock. A number of *olmarei* may live together forming the *enkang*, but it is increasingly common to find an *enkang* consisting of a single *olmarie* (McCabe et al. 2010). All Maasai men are members of a clan, in which members help each other in times of stress. All men also pass through a set of age grades from boy, warrior, junior elder, elder, and retired elder. During the warrior age set young men forge a corporate identity, have a leadership structure, and adopt an age set name. The age set acts as a self-help institution and members of a particular age set have responsibilities to each other and will redistribute livestock to poor families if needed.

The largest territorial unit is the *olosho*, in which all members of the *olosho* have access to grazing resources. In general, livestock movements are confined to the *olosho* but in times of drought *olosho* boundaries may be crossed with permission. Within the *olosho* are smaller units in which local communities have defined wet and dry season grazing areas. Both wet and dry season grazing areas are opened and closed seasonally, and according to climatic conditions. Traditional enforcement of rules governing who can use grazing resources and when dry season areas are opened was the responsibility of spiritual leaders or *laibons*. Those trespassing into restricted areas would be cursed and removed. This traditional system has now been replaced by a livestock management system governed at the village level (see below). Some water resources are open to all, but wells, and some streams, are owned by clans.

Previous research has demonstrated that Maasai in Tanzania began to adopt cultivation beginning in the 1950s (O'Malley 2000), and that cultivation became widespread among Maasai communities in northern Tanzania during the 1970s and 1980s (McCabe et al. 2010). The process began with planting small gardens which expanded into farms over time. The most commonly grown crops were maize and beans in farms and vegetables in home gardens. The plots were small, and fields were plowed primarily by hand, although in certain areas ox plows were used to till the land. The Maasai have traditionally incorporated grain in their diet, and it has been argued that wealthier people adopted cultivation to avoid selling livestock to buy grain and poorer people were pushed into agriculture because their livestock holdings were too small to provide enough food (McCabe et al. 2010).

The Tanzanian villagization program of the 1960s and 1970s encouraged people to settle in defined villages but this was not as successful in Maasailand as in other parts of the country. Nevertheless, mobility began to decrease, and the population increased exponentially. Schools, health clinics, small shops, and churches were built in village centers, but village boundaries did not inhibit the movement of people and livestock (Homewood and Rodgers 1991). As the human population grew, villages divided into sub-villages. With the expansion of cultivation and a decrease in mobility village leaders felt pressure to allocate individual plots to households. This process began in the 1980s and continues today. Plots were generally small consisting of one or two acres, but as we discuss below land allocations in Simanjiro – our study site – were larger than in other parts of Maasailand.

During the 1980s many young men, seeking to make money to purchase livestock, began to migrate into urban areas to find jobs, primarily as watchmen. Wages were very low and often no shelter was provided, yet young men continued to migrate. Some were able to buy enough livestock to serve as a basis for an independent household, but most young men only made enough money to survive (McCabe et al. 2014). It has been argued that, despite this lack of success, migration of young men away from Maasailand is transforming into a rite of passage, similar to that of young men guarding cattle in remote areas during the dry season (McCabe et al. 2014).

The Village Land Act of 1999 established the village administration authority to manage village land and resources. This act was seen as a critical juncture in shifting the responsibility of rule enforcement related to livestock management from the *laibons* to the village government. This coincided with a weakening of the authority of *laibons* and the expansion of Christianity (McCabe et al. 2020). Village government consists of a village assembly that includes all members of the village over the age of 18. The village assembly elects a chairperson (*mwenyekiti*), secretary (*katibu*), and treasurer (*mweka hazina)*, while the village executive officer (*mtendaji*) is appointed by the District. Villages are divided into sub-villages, each of which has a chairperson and secretary. A village council, consisting of the chairperson, sub-village chairpersons, and elected members (which must include women) is the governing body of the village. Within the village council a series of small committees are formed to deal with legislation that may include how livestock and water are managed. The livestock committee is responsible for defining and enforcing rules concerning the opening and closing of wet and dry season grazing areas, and if and when outsiders will be allowed to use village resources (McCabe et al. 2020).

4.3 Simanjiro Case Study

The population of Simanjiro District is primarily Kisongo Maasai who now engage in a set of diversified livelihoods, incorporating cultivation, migration to urban areas for wage labor, and acting as middlemen in the Tanzanite trade, along with their traditional economic and social emphasis on the raising of livestock (Fig. 4.1). Cultivation among Maasai communities in Simanjiro followed a very different process of adoption than Maasai communities in other areas of northern Tanzania. Beginning in the 1980s people began to cultivate in Simanjiro but not by using handheld hoes in small plots; rather, the use of tractors and ox-plows were common, and the cultivated plots were large, often averaging 10 or more acres.

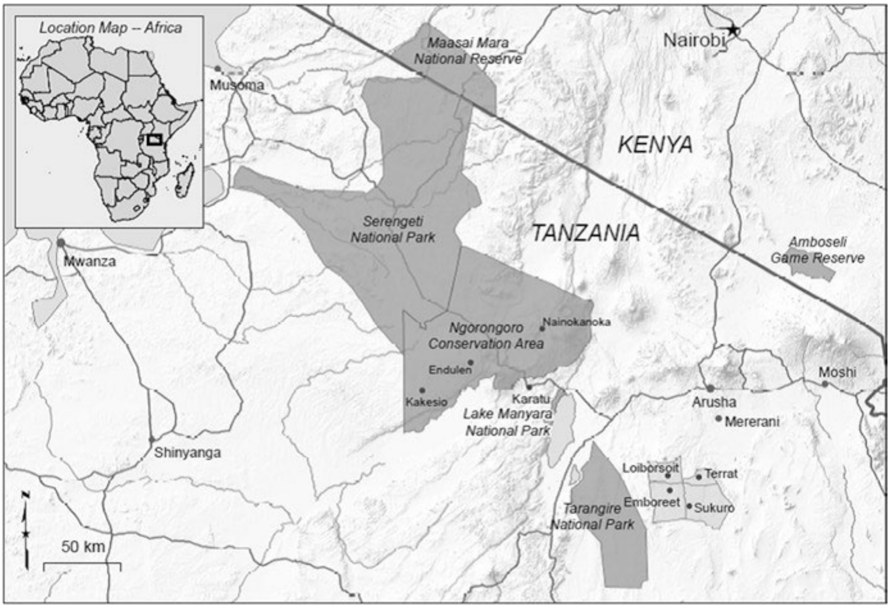

Fig. 4.1 Map of the study area showing focal communities in relation to Tarangire National Park

Although individuals are not allowed to sell these allocations, it is not uncommon to see large plots of land "leased" to outside interests for commercial agriculture. The regional government declared a moratorium on new farms on the Simanjiro plains in 2006, fuelling fears among local people that they would lose land (Davis 2011). Alongside processes of land allocation, village administrations have developed land-use plans often supported by NGOs, to specify areas for cultivation, herding, and sometimes conservation. Just as the process of adopting cultivation was different in Simanjiro than in other places in Maasailand, the pattern of rural-urban migration is also unique with implications for the social-ecological system. The gemstone Tanzanite was discovered in the hills behind Kilimanjaro airport in 1967, close to the town of Mererani. In Simanjiro, like in the rest of Maasailand, young men were migrating to urban areas during the 1980s, but the lack of success and the danger inherent in guarding influenced elders to encourage young men to migrate to Mererani rather than to urban areas, where they learned to evaluate gemstones and began to buy and sell Tanzanite (Smith 2012; McCabe et al. 2014). Although there were men from other parts of Tanzania engaged in the gemstone business, men from Simanjiro began to dominate as middlemen buying gemstones and selling them to local brokers and brokers in the city of Arusha. Unlike those who migrated to seek wage labor, many men engaged in the Tanzanite trade became wealthy. With this newfound wealth many built modern houses and purchased land and tractors. This resulted in the acceleration of land being converted from rangeland to cultivated land, a process which is continuing today.

Finally, following a severe drought in 2008 and 2009, in which tens of thousands of cattle from the border areas in northern Tanzania and southern Kenya migrated into the rangelands of Simanjiro, many villages in Simanjiro began to redefine access to resources to outsiders. The process was different among villages but the common theme was that informal institutions based around a common ethnic identity that facilitated access to village land to outsiders during times of stress transitioned to formal village-based institutions limiting access to village lands and strengthening village boundaries (McCabe et al. 2020).

Research for the wellbeing study took place in four villages located to the east of Tarangire National Park (TNP). Two villages, Emboreet and Lobor Soit border the park, while the other villages, Sukuro and Terrat, are located 40 and 45 kilometers east of the park boundary. Prior to the establishment of TNP Maasai moved with their livestock, especially small stock, west to utilize the water and forage resources in what is now the park during dry seasons and times of drought (Fig. 4.1).

Simanjiro District as well as TNP are located within the Tarangire Ecosystem which is considered one of the most biodiversity rich ecosystems in Africa (Olson and Dinerstein 1998). TNP is famous for its large population of elephants and the migration of ungulates that is second only to the Serengeti-Ngorongoro ecosystem in the size of the annual migration (**Bond et al.** Chap. 8; **Foley and Foley** Chap. 10). The Tarangire River and the swamps are among the only permanent sources of water in this semi-arid landscape. TNP is a dry season refuge, and many of the animals migrate out of the park in the wet season and back into the park as the dry season progresses. The major migration routes lead eastward to the Simanjiro plains and north to the plains located just south of Lake Natron (**Lohay et al.** Chap. 12). The soils in the Simanjiro plains are rich in phosphorus which is important to eastern white-bearded wildebeests (*Connochaetes taurinus albojubatus*) and plains zebras (*Equus quagga*) during the calving season (Kahurananga and Silkiluwasha 1997). The Simanjiro area is considered vital to maintaining the wildlife populations of TNP but is also an area without any formal protected status. It has thus been the focus of contention between wildlife conservationists, human rights advocates, and the Maasai.

There have been numerous efforts to confer some type of protected status to the Simanjiro plains, probably the most famous of which was Markus Borner's call for parts of Simanjiro to be managed in a similar fashion to the Ngorongoro Conservation Area (Borner 1985), and the latest being a recent attempt to establish a Wildlife Management Area by the government and promoted by the African Wildlife Foundation, but these efforts have all been opposed by many Maasai living in Simanjiro (Benjaminsen et al. 2013).

One exception to this was the establishment of the Simanjiro Conservation Easement in the village of Terrat in 2006. A number of conservation NGOs, tour operators, and lodge owners agreed to contribute approximately $4500 per year to the village government in exchange for setting aside a section of the plains within the village boundaries (Nelson et al. 2010). The Easement prohibits cultivation and permanent settlement but allows the area to be used for livestock grazing. In 2010 the village of Sukuro also joined the Easement. This attempt to set aside an area for wildlife, especially during the calving season, has been successful to a large extent due to the transparency of the process of distributing funds and maintaining an open area for livestock. However, based on conversations we had with villagers in Terrat, there are factions within the Terrat village government that are challenging the Easement's continuation, and its future remains uncertain.

4.4 Methods

We conducted 26 semi-structured group interviews with groups of men and women between January and August 2014. A qualitative approach which is flexible and open to unexpected findings, was appropriate to understanding the nuances of local conceptions of wellbeing as well as the historical, political, and cultural issues shaping these ideas (Woodhouse et al. 2018). These interviews were centered around a series of questions but allowed flexibility in following up answers with further questions and themes that emerged to be explored further in subsequent interviews. Due to the challenges of recruiting participants randomly in a dispersed population, we used local contacts to access participants and to include different age sets, subvillages, and wealth categories. Because participants were not randomly selected, results cannot be extrapolated to the community as a whole. Validity of the data was supported by establishing trust through long-term fieldwork, sampling a range of people, using culturally appropriate forms of communication (group meetings), and trained and experienced field assistants. In total 76 men were interviewed in 14 groups, and 72 women in 12 groups. McCabe was primarily responsible for the interviews conducted with men and Woodhouse primarily responsible for interviews conducted among the women. Both McCabe and Woodhouse were aided by male and female Maasai assistants.

As mentioned above, one of the principal organizing features among the Maasai is the age-set system, in which men pass through a series of stages from boys, warriors, junior elders, senior elders, and retired elders. There is some overlap between the closing of one age set and the opening of another and in the context of this research the age sets represented were *Esuri* (estimated age 59–78), *Makaa* (48–60), *Landess* (37–50), and *Korianga* (21–35). The newest age set *Nyangulo* had just been opened at the time of the research and was not included in the interviews. Women do not go through the structured age system as the men but are identified as *Endoyie* (unmarried girls), *Siangiki* (married with a few children, approximately 20–32 years old), *Endasati* (married women between 33 and 49 years of age), and *Koko* (women beyond reproductive years). Women of each status were also included in the interviews.

One of the first challenges was trying to translate the concept of wellbeing into Maa. Early attempts based on what we called a "good life" often produced a wish list that was more aspirational than realistic. We settled on translating a "normal life" (*engishui e kawaida*), which seemed to capture the wellbeing concept fairly well. Group interviews were guided by the following key questions:

- What is important for you to feel that you have a normal life?
- Why is this important?
- How has this aspect of your life changed over the last 10 years?
- Why has it changed?
- Are the changes the same for everyone in the village?
- What threats do you see for the continuation of this aspect of your life?

Insights gained from the qualitative data into locally meaningful concepts of wellbeing informed the design of a survey to further understand patterns of wellbeing and perceived changes through time. The survey sample (n = 149), carried out in 2015 after the group interviews were analyzed, and was structured to capture variation in household wealth, sub-village location (i.e., near and far from the village center), and age of the household head, a sample which is part of an ongoing study of land use in the region (McCabe et al. 2014; Baird et al. 2009). We also surveyed 148 women (wives of household heads) within these households. The surveys were piloted and then conducted by a trained team of Maasai field researchers. Descriptive statistics from a preliminary analysis of these data are presented here.

Data analysis of transcripts and notes from the group interviews was guided by the three dimensions of wellbeing but was inductive in that the specific components of wellbeing and the reasons given for their importance emerged from the data through coding carried out by Woodhouse. We tested emerging ideas iteratively, inspecting for recurring instances and differences to ensure comprehensive treatment of the data set (Silverman 2006). The analysis presented here also draws upon ongoing ethnographic fieldwork including participant observation and informal conversations focused on livelihoods and land use carried out over the last 16 years in the study villages by McCabe.

4.5 Results for Men

Summaries of the results of the group interviews for percentage of Maasai men reporting on what components (structured in family, land, and livestock) are necessary for a "normal life" (*engishui e kawaida*). Percentages are broken down by age set [*Korianga* (estimated age: 21–35 years; *Landess* (37–50), *Makaa* (48–60), *Esuri* (59–78)] are presented in Tables 4.1 and 4.2. We are not including the following components in the tables as all respondents said that these were important to a normal life: livestock, wives, children, cultivation, having one's own land, and common grazing areas.

Table 4.1 Components of a normal life for Maasai men of different age sets in Simanjiro: family, land, and livestock. Families were considered small if a man said two or one wife, medium if the man said 3–5 wives, and large if the man said that 6 or more wives were necessary for a normal life. Land allocations were considered small if they consisted of less than 10 acres; medium if the land allocation was between 11 and 30 acres, and large if the desired land allocation was more than 30 acres. Herds were considered small if cattle numbers were less than 30, medium if between 31 and 50 cattle, and large if the desired herd exceeded 50 cattle. The numbers in parentheses are the number of respondents

	Korianga (N = 25)	Landess (N = 27)	Makaa (N = 9)	Esuri (N = 4)
Small family	36% (9)	44% (12)	22% (2)	0
Medium family	8% (2)	15% (4)	67% (6)	0
Large family	16% (4)	11% (3)	0	100% (4)
Small land allocation	8% (2)	7% (2)	22% (2)	0
Medium land allocation	40% (10)	19% (5)	11% (1)	0
Large land allocation	0	30% (8)	33% (3)	75% (3)
Small herd	16% (4)	7% (2)	11% (1)	0
Medium herd	8% (2)	30% (8)	0	0
Large herd	20% (5)	37% (10)	44% (4)	50% (2)
Improved breed	12% (3)	11% (3)	0	0

Table 4.2 Components of a normal life for Maasai men of different age sets in Simanjiro: infrastructure and development

	Korianga (N = 25)	Landess (N = 27)	Makaa (N = 9)	Esuri (N = 4)
Modern house	36% (9)	59% (16)	33% (3)	0
Motor bike	44% (11)	0	0	0
Services (transportation, shops, clinics etc.)	16% (4)	52% (14)	0	0
Development	24% (6)	7% (2)	0	0

There were clear distinctions among the men according to age sets, but for all the men issues pertaining to land and livestock were paramount. Although the Maasai practice diversified livelihood strategies, livestock – and in particular cattle – are not only a source of food and wealth but are integral to the Maasai sense of identity. Galaty points out the Maasai are "people of the cattle" (Galaty 1982) and this certainly was evident among men from all age sets. Traditionally, wealth among the Maasai was measured by the number of livestock, the number of wives, and especially by the number of children a man had. The older men in this study reflected this conceptualization of what constitutes wealth. Based on discussions that McCabe had on other research projects with younger Maasai men in the Ngorongoro Conservation Area (McCabe et al. 1992; McCabe 2002), it was evident that within Maasai society there were now two different ideas about what constituted wealth. While an older man described himself as wealthy, some younger men viewed the same person as poor. Younger men would commonly say: yes, he (referring to the same older man) does have a lot of cattle, but if you divide the number of cattle he has by the number of wives and children, you can see that he is actually poor. Although this sentiment was not explicitly expressed in this study, most young men (*Landis* and *Korianga* age sets) said that they wanted fewer cattle, fewer wives, and fewer children (Table 4.1). It is now considered normal to have two wives, each expected to have four or five children. Older men also felt that the zebu breed of cattle was fine, but for younger men cattle breeds such as Boran and Sahiwal were preferable. Both of these latter breeds are larger and produce more milk and meat per animal than zebu cattle allowing fewer animals to be kept, but may be more vulnerable to stress during drought years.

Land was also a concern for all men in the study, but again older men and younger men differed in what they saw as the amount of private land necessary for wellbeing. Older men stressed the need for large areas set aside for communal grazing, although they also recognized the importance of individual plots for houses and areas for sick animals and calves. Younger men had some of the same concerns, but priorities were reversed. Privately held land was considered extremely important for housing, cultivation, and for sick animals and calves. However, younger men did agree with the older men that communal grazing was also important. In arid and semi-arid lands precipitation is highly variable, both spatially and temporally, and all men agreed that in any one year parts of a village may receive adequate rainfall while other areas within the same village may not. The need to be able to move livestock from privately held plots to common grazing areas was therefore considered extremely important.

It has been evident for some time that Maasai society has been transitioning from a more cooperative form of social organization in which many families lived together within a compound and shared herding responsibilities and sometimes food, to a more individualized form of social organization in which it is not uncommon for an *enkang* to consist of a single family and its livestock (McCabe et al. 2010). The emphasis on private land allocations among younger men in this study was reflective of this transition.

In terms of wellbeing, older men stressed the importance of being able to help others, while younger men expressed having enough resources to have control of their own lives, and this related to both land and livestock. Education of children was also an issue agreed to as important for all men, but more so among the younger men, and that included the education of girls. The maintenance of tradition was also viewed as important especially by the older men, while the younger men felt that traditions were important but if there was some loss of traditions because children were being educated, then the tradeoff was worth it. A modern concrete house is increasingly important across age sets. Younger men in particular recognized the importance of being close to services such as education, water sources, and veterinary care, and were the only ones to specifically mention the concept of 'development' (Table 4.2). 'Development' includes things like a modern house and increased services, but also refers to being a modern Tanzanian.

One issue that was expressed by men of all age sets, and which was surprising to us, was an emphasis on having unity (*enaiboshu*) within the village. This is especially related to the Maasai tradition of mutual assistance including the practice of restocking poorer households with livestock within clans (*ewoloto*), but was also raised in relation to the importance of intrahousehold harmony particularly by women (see below). Arguments within households can be among wives; between wives and the husband over the education of children, or the allocation of resources; and between sons and their fathers over the allocation of livestock, and the desire for sons to establish their own households. Disputes over the boundaries of land allocations often result in conflict among households and disputes over village boundaries often cause conflict between and among villages. The survey results showed that 54% of women and 76% of men were of the opinion that unity had worsened within their village and between villages respectively, largely attributed to party politics and land issues. Indeed, village-level politics were discussed by men as the major cause of conflict within villages. This often related to national-level politics as the party which has ruled Tanzania since independence, Chama Cha Mapinduzi (CCM), was being challenged by a newly formed party, Chama cha Demokrasia na Maendeleo (Chadema). It was sometimes expressed that in the past it was easy to call for a village meeting when necessary, but now if a village leader calls for a meeting and he is a member of one party, villagers who are members of the opposing party will not attend.

Another issue that was strongly expressed relating to a lack of harmony at the village level, was a strong distrust of external actors who can take advantage of weak and corrupt leaders. This almost always concerned issues relating to land, and in some ways goes all the way back to the formation of Tarangire National Park. Stories about a local leader who put his thumb print on a document in 1970 that ceded land that the Maasai considered theirs is well known to Maasai living in Simanjiro today, and fuels suspicion about leaders giving away or selling land to conservation interests or private investors. This distrust is not without merit. A few years ago a local leader in Emboreet illegally sold 6000 acres to outsiders without approval of the village government, and court cases trying to resolve disputes arising from this illegal sale are ongoing. Our survey data of Maasai men corroborate

Table 4.3 Responses by Masaai men to the statement: I trust what people from the *following organizations* tell me about land issues

	Don't know them	Strongly disagree	Disagree	Undecided	Agree	Strongly agree
Leaders (N = 145)		5	5		118	17
Conservation NGOs (N = 144)	20	14	91	4	13	2
National Park officials (N = 145)	19	17	94	5	10	0
Development NGOs (N = 145)	0	10	46	1	81	7
Private investors (N = 144)	16	12	102	4	7	3
Photographic tourist companies (N = 144)	20	3	83	3	30	5
Hunting tourist companies (N = 144)	19	5	92	2	24	2

the expressions of distrust in our qualitative interviews. Although people generally expressed support of local leaders, there was a concern that corruption posed a threat to wellbeing and there are high levels of distrust for external organizations, especially private investors, tourist companies, and conservation organizations, but with the exception of development organizations (Table 4.3). Concerns were particularly raised about the establishment of a Wildlife Management Area (WMA) in Simanjiro, or selling land to private investors, both of which everyone we talked to opposed. During the time of our interviews, WMAs were generally viewed negatively due to their alignment with conservation priorities and organizations, and some people had heard other's experiences (but see **Raycraft** Chap. 6 for more recent attitudes about Randilen WMA in this region). For example, we talked to a number of people in the village of Lobor Soit who were taken to visit the Burunge WMA to see how that WMA worked. Lobor Soit has already previously made agreements with tour companies and joining a WMA was viewed as risking the dilution of revenue. This would put villagers in Lobor Soit in a similar position as those in the village of Minjingu, who have been attempting remove themselves from the Burunge WMA for many years (Bluwstein et al. 2016).

Based on previous research we knew that men were worried about TNP expanding into the villages in our study area (Baird et al. 2009), and there was concern about the establishment of wildlife corridors connecting TNP to the Simanjiro plains. Despite these concerns there was almost universal support for the Simanjiro Wildlife Easement, with protection of grazing land rather than funds given as the most important benefit by 86% of men in Terrat and Sukuro (n = 72). This suggests that if trust is established in creating a protected area, with tangible valued benefits provided, then initial suspicion concerning the motivations of the actors involved can be overcome (Davis and Goldman 2019, **Raycraft** Chap. 6).

4.5.1 Results for Women

One of the striking differences between responses of men and women concerning what constitutes wellbeing was that there was consistency of responses across all age groups for women. Many aspects of having a good life were shared by both women and men, but in many cases, it was the relational aspects of a good life that were emphasized by women (36 separate instances where this was raised compared with 17 for men). Because women are primarily responsible for the care of children, having enough livestock was seen as important in order to provide milk for children. Cultivation was seen as important for the same reasons—to provide food security so that the children would not go hungry. Cultivation was also seen as a source of income when money was needed for hospital costs, school fees, clothes, etc.

Children were seen as important, not as a component of wealth, but because women desired to live close to their children and sons and daughters are a critical component of their social lives. Having a son was important because when women pass their reproductive years, they often will leave the household of their husband to move to the household of their eldest son. Although the government of Tanzania has passed laws saying that inheritance should be shared equally between women and men, among the Maasai women will rarely, if ever, inherit livestock when a man dies and thus, they are dependent on sons following the death of their husband.

Women were also concerned about losing land to outsiders, and often stated that outsiders could not be trusted. There was less concern about losing land to conservation than to "unscrupulous" or "clever" people. Like with the men, the education of children was important and like with the men education was seen as important in securing land rights. Many expressed vulnerability in relation to external actors due to their lack of formal education and their inability to read. They were suspicious when asked to approve or sign documents that they do not understand, and they felt that the education of their children will prevent them from being exploited by more educated members of other ethnic groups who can "steal our land by the pen". There was more emphasis among the women for the education of girls than among the men. Both men and women felt that defending their rights to land will fall to their educated children. A Maasai friend of McCabe once told him that education is now "the tip of the spear" in defending land rights.

Unity was also a component of having a good life for women, but for women the emphasis was harmony within the household. This was true for the relationship of wives to their husband, but also among the wives in a single household. First wives tend to have more authority and access to resources than other wives, but a husband can also have a favorite wife who may not be the first wife. The difference in status among wives can cause conflict within the household, and this is much more of a concern among the women than among the men. Conflict within the village was also a concern, but primarily as it impacts dynamics within the household.

Women also emphasized the importance of being listened to and having their opinion respected concerning household decisions. Coupled with this was having some degree of economic independence and a focus on money which was not present in discussions with men about wellbeing. A number of NGOs and church groups have facilitated "merry go round" activities among women's groups where a number of women contribute money to the group who then give this money to one of the group's members, often used to buy livestock, that would be her own property. The next round would go to another woman in the group allowing all the women to have access to resources that were independent of the husband. Although there is little written on this for Maasai of Tanzania, Taeko discusses the success of this practice among Maasai women in Kenya (Taeko 2019) and economic schemes have been shown as one route towards empowerment for Tanzanian Maasai (Goldman and Little 2015).

One of the major differences between women and men was their attitudes towards the Church and Maasai traditions. Although there are many different denominations among the churches in Simanjiro, all are Christian. Men, unlike women, voiced concerns about Church leaders preaching about equality between men and women. Dorothy Hodgson has written on the particularly powerful relationship between the Church and Maasai women and its role in the negotiation of gender (Hodgson 2005). Some traditional aspects of Maasai life, such as the importance of *laibons* (spiritual leaders) has been undermined by church leaders, and many women agreed that these aspects of Maasai traditional life are outdated, perhaps reflecting that *laibons* have historically been aligned with men as ritual leaders of cattle raids and spiritual advisors of elders (Hodgson 2005). Other Maasai traditions remained highly valued, especially those designed to share resources and help those in need.

4.6 Threats to Wellbeing and Changes of the Last 10 Years

The majority of men (75%) and women (67%) expressed the view that their access to grazing land had worsened in the last 10 years. Two factors stand out as major causes for the loss of grazing land: the expansion of cultivation, and the shifting of village boundaries (Table 4.4). A minority also attribute change to private investors and the Tanzania National Park Authority (TANAPA).

Table 4.4 Maasai men's responses to the question 'Which groups or issues contributed to the worsening of access to communal grazing land?' (question only asked if they had answered that land access had worsened)

	Number of respondents (N = 112)	% of respondents
Boundary changes	95	85
Agriculture	58	52
Private investors	49	44
TANAPA	22	20
Local government	7	6
Photographic tourism	4	4
Hunting tourism	3	3
National government	3	3

As previously mentioned, when a village exceeds approximately 5000 residents a process of dividing the village into two separate villages is initiated, and as village boundaries become barriers to the free movement of livestock, people have to rely on grazing resources within the village. In addition, there are frequently disputes between and among villages concerning where the boundaries should be following village division. Large areas for grazing become divided and as the human population grows there is more pressure to allocate land to individuals. As discussed earlier the younger generation of men highly value individual land allocations and there is increasing pressure on village leaders to grant land allocations as young men move from warriors to junior elders.

Conversion of land to agriculture and the leasing of land in Simanjiro has been shown to be influenced by proximity to TNP because of perceived threats of park expansion (Sachedina and Trench 2009; Baird et al. 2009), a concern we heard raised in these park-adjacent villages. A large percentage of men (73%) view loss of grazing land as a continuing threat, and worry that there will be further losses in the next 10 years. A sense of security for the future, especially with regard to land, is fundamental to wellbeing in Simanjiro and drives people's actions.

Villages, however, are not helpless and efforts are underway to help preserve the rangelands from further fragmentation. The Simanjiro Wildlife Easement is one example. Another is the granting of Certificates of Customary Rights of Occupancy (CCROs), which formalizes customary land tenure registered by the national government. In Simanjiro, this means setting aside land for livestock and wildlife, by preventing cultivation or settlement. Organizations such as the Ujamaa Community Resource Team (UCRT), the Dorobo Fund, and the Northern Tanzania Rangeland Initiative have been particularly influential in helping villages with the establishment of CCROs. Once established, village leaders are able to resist pressure, both internally and externally, to further divide land.

One of the major obstacles to overcome in setting aside communal land is the lack of trust among the Maasai of Simanjiro towards private investors and many NGOs, in particular conservation-oriented NGOs. There is also a notable lack of trust towards the federal government. Davis (2011) has written about how the TANAPA "Good Neighborliness" (*Ujirani Mwema*) initiative, which aimed to reduce animosity by providing social services to villages, has not been as successful as envisioned in communities adjacent to TNP as the lack of access to resources and perceived threat of TNP expansion counteracts any positive benefits. The perceived lack of willingness on the part of TNP to allow limited grazing within park boundaries during times of drought contributes to the lack of trust between TANAPA and Maasai communities. This was particularly evident during the 2008–2009 drought when thousands, perhaps tens of thousands, of cattle had migrated just to the east of the TNP border. The only grazing available was located inside the park, but was inaccessible to the cattle, many of which died just outside the park boundary. Although this is the law of Tanzania, for the Maasai the relationship is between "neighbors", one of which is TNP.

What is unusual here is that the organizations that worked on the Easement and CCROs have achieved a level of trust that is not shared by some of the other NGOs or TANAPA representatives in this region. The establishment of the Simanjiro Wildlife Easement was initially accepted because the arrangement was presented to village leaders in Terrat by representatives from UCRT, one of the few NGOs that had gained trust from the communities because of years of work and the fact that UCRT was considered an honest broker. As an important relational aspect of wellbeing one should not underemphasize the importance of trust in establishing relationships among conservation organizations and local communities, something that is broadly recognized as a key enabler in the collaborative and adaptive governance of ecosystems (e.g. Hahn et al. 2006).

Men and women both agreed that small motorcycles (*piki piki*) and cell phones have made great improvements in the lives of people living in Simanjiro and contribute to their sense of wellbeing. Much of the positive impact of both motorcycles and cell phones were mentioned in relationship to health. Instead of remote households being isolated, far from health clinics and hospitals, a motorcycle and driver can be called, and a sick or injured person transported to a health facility in a matter of hours rather than days. Cell phones are widely used to transfer money, collect information about agricultural activities and grazing conditions, and to both strengthen and expand social networks (Quandt et al. 2020; Summers et al. 2020). Our surveys also suggest that people see improvements to services over the last 10 years, in particular water access (80% of men), healthcare (73%), and education (65%). Respondents were largely satisfied with access to services, highlighting that the access to valued services that comes with a more sedentary life is increasingly desirable.

4.7 Discussion

4.7.1 What Does This Mean for Tarangire?

Because many wildlife species migrate from TNP to the east into the Simanjiro plains during the wet season, what happens to the people and their livestock living there, and the extent of their cultivation, is of direct relevance to the sustainability of wildlife in TNP. It is not just numbers of people and livestock, or acres cultivated, but also the aspirations of people and what they find important to their lives. In looking at the results of the wellbeing study, it is apparent that some of the goals of wildlife conservationists and those aspects of Maasai conceptualizations of wellbeing overlap. This is especially true with the preservation of large areas of communal grazing land. The establishment of CCROs and the Simanjiro Conservation Easement are examples of how these components of Maasai wellbeing have been aligned with conservation in practice. These areas are shared by livestock and wildlife and can been seen as win-win scenarios for wildlife conservation and what contributes to a “good life” for the Maasai. Leaders and village councils in Terrat and Sukuro were not forced into implementing these land-use arrangements but were willing participants in the negotiations that resulted in the land being set aside, so that further fragmentation of the rangelands was prevented. In addition, among younger men, the desire for smaller families, and smaller but more productive herds is also consistent with conservation goals in Simanjiro.

On the other hand, some aspects of Maasai wellbeing are not consistent with conservation goals. One message that Maasai discussed as being promoted by conservationists is that people should depend on their livestock and revenues from tourism as their primary livelihood strategies and abandon cultivation. A frequent response to this message was: 'they want us to be like our grandfathers'. Cultivation is now a key component to Maasai wellbeing across all segments of Maasai society in Simanjiro, and revenue from tourism is often captured by village elites. Advocating for reduced dependence on cultivation, without viable alternatives for subsistence needs and equitable income generation, will likely not be acceptable to many people and is inconsistent with the Maasai ideas of what having a good life means.

Conservationists should also be cognizant of the importance of both trust and social unity to the Maasai of Simanjiro. Many Maasai are aware that TNP does have programs that help in village development, but local people rarely view these projects as having relevance to their lives. Mistrust of conservation is rooted in historical land and resource alienation, as well as continuing perceived threats by a range of external actors (Goldman 2011). Many Maasai also are aware that revenue is being generated by tourist-related activities but the Maasai themselves rarely see any of these funds. In talking to a lodge owner a number of years ago McCabe was told: "I give 10% of bed night revenues to the village leaders, what happens then is up to them". We understand that it is not the responsibility of lodge owners to oversee the distribution of benefits so that they are distributed equally and transparently, but the lack of transparency on the part of the village leaders contributes to the lack of trust and the erosion of harmonious relationships. The results of the wellbeing study articulate well with a previous study on Maasai perceptions and the influence on TNP. In a study conducted by McCabe and colleagues conducted in 2004 and 2005, 240 household surveys were conducted in eight villages in Simanjiro; four were adjacent or close to TNP and four were far away from the park (80–115 km). The four villages close to the park were the same as those in the wellbeing study. The objective of that research was to examine how Maasai perceived risk and what could be done to mitigate the risk (Baird et al. 2009). For the villages located close to the park the risks that were considered of both high incidence and high severity were human disease, livestock disease, drought and conservation (risk that the park will expand or conservation policies will limit land use). Wildlife predation on crops and livestock were considered as high incidence risks but less severe. In the villages located far away from the park, water, hospital health services, and human disease were risks listed as both of high incidence and high severity. Wildlife-related risks were not mentioned in the distant villages. Ways to mitigate risks in the villages closer to TNP included increased leasing of the land to help secure land tenure, and planting of crops in a way that would inhibit the migration of wildlife directly into the Simanjiro plains and push the migration routes further to the south. In this way, ideas of risk and efforts to improve security are negatively impacting wildlife conservation but are partly driven by the impacts of conservation itself, highlighting that future interventions will need to reduce human-wildlife conflict, increase land security, and establish trust in conservation processes and institutions.

4.7.2 The Larger Picture

Protected areas are critical components of various attempts to stem the loss of biological diversity (Gray et al. 2016). However, there is increasing acceptance that for conservation to be sustainable and just, local communities must be more involved. Aichi target 11 of the Convention on Biological Diversity stresses that protected areas should be effectively and equitably managed. Equity in this sense refers to a fair distribution of cost and benefits to wellbeing, participation in decision-making, and the recognition of social and cultural differences (Schlosberg 2013). Although the situation is not yet irretrievable the world may be experiencing the sixth major extinction episode. As Sir Robert Watson, the chair of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystems Services (IPBES) stated in a recent media release: "The overwhelming evidence of the IPBES Global Assessment, from a wide range of different fields of knowledge, presents an ominous picture. The health of ecosystems on which we and all other species depend is deteriorating more rapidly than ever. We are eroding the very foundations of our economies, livelihoods, food security, health and quality of life worldwide."

The need for preserving current levels of biodiversity cannot be overstated, and protected areas are critical to achieving this goal. In this chapter we emphasize that to ensure the ecological sustainability of TNP, we need to understand the wellbeing and aspirations of the communities in the wildlife migration/dispersal area of Simanjiro and work to achieve equitable approaches to conservation in this area.

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Chapter 5
Land Tenure, Livelihoods, and Conservation: Perspectives on Priorities in Tanzania's Tarangire Ecosystem

Peadar Brehony, Alais Morindat, and Makko Sinandei

Etejo enkiteng': "mikintaaya, nchooyioki!" – The cow said "do not lend me out, give me away!" sensu The owner treats their own property better

Kipury (1983:183)

Abstract Research on conservation efforts demonstrates that local community support is critical to achieving conservation goals. In this chapter, we highlight innovative approaches which are currently being taken in the Tarangire Ecosystem to combine access to secure land tenure rights with landscape scale access to functional heterogeneity, governed through both formal and informal institutions. Informed by the concepts of social-ecological systems and just conservation, this chapter begins by considering the recent history of natural resource governance institutions in the Tarangire Ecosystem, where traditional systems were matched to the social-ecological context of that time. We go on to discuss how modernisation has resulted in significant changes to these systems over time, with a focus on the ways in which changes in land tenure have resulted in a loss of flexibility and shifts in local livelihoods. We highlight how, in this context, land tenure rights can play a critical role in community-based conservation efforts in the Tarangire Ecosystem to benefit both people and wild animals. We follow this with a description of the ongoing process in the Tarangire Ecosystem to secure rights to land and resources through spatial planning at a local scale, and how this can be expanded to the landscape scale. Finally, we reflect on some of the challenges with such an approach,

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particularly given the significant social-ecological variability and uncertainty that lies ahead.

Keywords Tanzania · Land tenure · Livelihoods · Community-based conservation · Pastoralism · Social-ecological systems

5.1 Introduction

The notions which underpin Western ideas of conservation have grown from natural resource management in the eighteenth century, to the regulation of hunting and spatial based protected areas in the nineteenth century, to the protection of ecosystem processes and conservation of all biodiversity in the twentieth century (Hölzl 2010; Watson et al. 2014; Western et al. 2020).

In East Africa, over the twentieth century, conservation interventions focussed on protecting large mammals and landscapes (Bennett et al. 2009; Fynn and Bonyongo 2011; Jenkins et al. 2013; Western and Gichohi 1993), sometimes at the expense of resident and migratory people (Adams 2004; Brockington 2002; Lindsay 1987). Over recent decades, conservation interventions have often sought to include local communities, recognising the important role they play in protecting and managing ecosystems (Western et al. 1994). Vast areas of the world have long been managed and shaped by local people, under various property regimes (Ellis et al. 2021). Currently, many local indigenous communities manage and practice sustainable rural livelihoods, while also conserving nature under a diverse set of stewardship practices successfully (Díaz et al. 2019a) on at least 25–28% of the Earth's land surface (Garnett et al. 2018).

In the Tarangire Ecosystem, national parks were originally set aside to protect people from wild animals (Chap. 2) and to allow certain people to hunt wild animals (see Fig. 5.1). These parks were set aside in areas that were deemed (by outsiders) to be marginal for development, a pattern recorded elsewhere (Joppa and Pfaff 2009). However, based on our best current understanding, many conservationists recognise that the areas which have been set aside are too small to avoid losing biodiversity, due to habitat fragmentation, insularisation, and future uncertainty (Fynn and Bonyongo 2011; Newmark 2008). For instance, in the Tarangire Ecosystem, Tarangire National Park does not cover the entire annual ranges of wide-ranging large mammals such as elephants, zebras, and wildebeests (Chap. 12). To overcome these shortcomings, conservationists and governments have sought to engage with landowners (we shall call them community members or communities) outside of protected areas, like national parks, often on terms set by conservationists. These arrangements tend to overlook the fact that those who contribute the greatest in this arrangement, and yet have the most to lose in terms of access to resources, are the community members. As others in this volume (Chap. 2) and elsewhere (Bluwstein et al. 2018; Brockington 2002; Igoe 2004) have demonstrated from research in Tanzania, ultimately, a history of land alienation together with

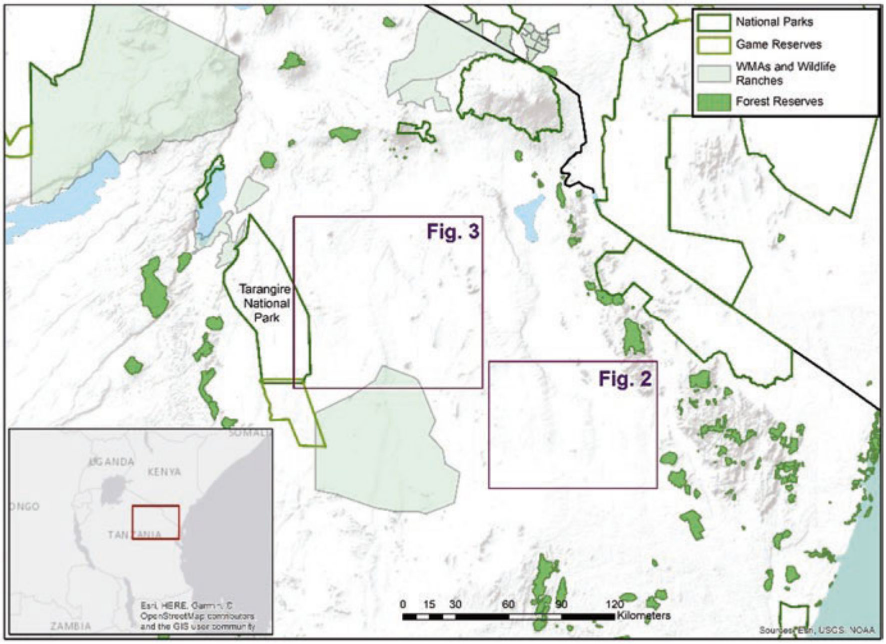
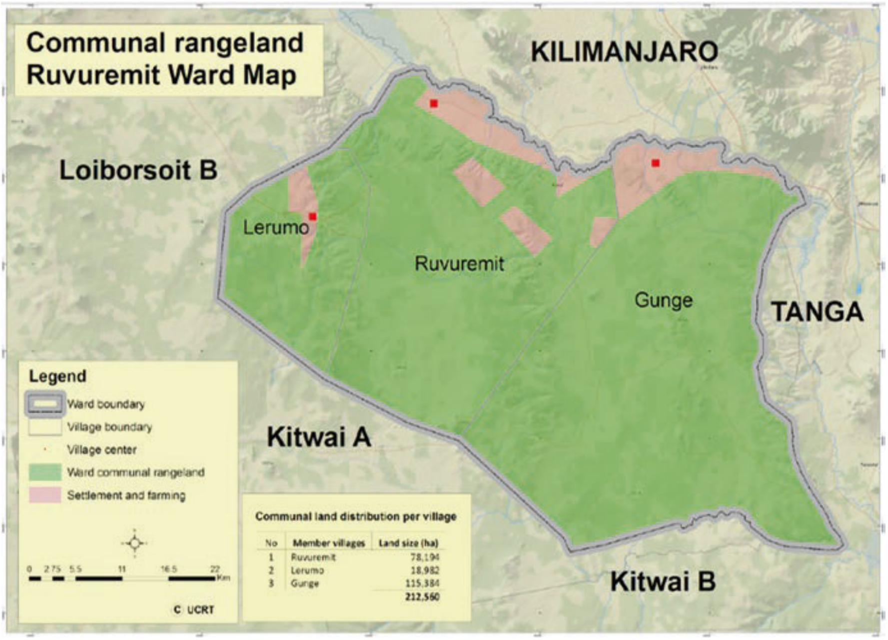


Fig. 5.1 Overview map of the Tarangire Ecosystem in northern Tanzania with the areas covered by Figs. 5.2 and 5.3 shown. Conservation area extents are from world database on protected areas (UNEP-WCMC and IUCN 2020) (Some boundaries are disputed, and some areas are missing from this database, including Randilen WMA)



No	Member villages	Land size (ha)
1	Ruvuremit	78,194
2	Lerumo	18,982
3	Gunge	115,384
		212,560

Fig. 5.2 Examples of village land use plans and joint rangeland plans from the eastern part of the Tarangire Ecosystem – see Fig. 5.1. (Map from UCRT and Simanjiro District Council 2019)

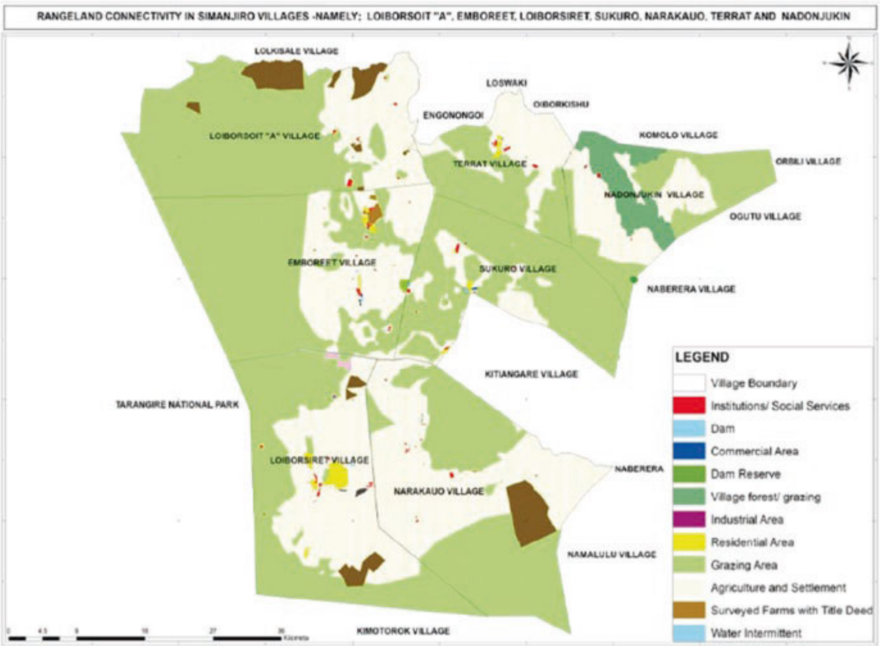


Fig. 5.3 Village land use plans and joint rangeland plans secure larger open rangelands in the eastern part of the Tarangire Ecosystem – see Fig. 5.1. (Map from UCRT and Simanjiro District Council 2019)

projects which fail to meet expectations, have resulted in local distrust towards conservation. Yet, at the same time, there is also an abundance of research which demonstrates that local community trust and support is critical to achieving conservation goals (Chaps. 4 and 6; Hulme and Murphree 1999; Persha et al. 2011; Adams and Hulme 2001; Oldekop et al. 2016).

Research from across the globe shows that: secure rights to resources; clear rules about the control of land and natural resources; transparent enforcement and revision of these rules and rights (Agrawal et al. 2008; Brehony 2020; Persha et al. 2011) together with robust governance institutions (Brehony 2020; Brockington et al. 2018; Kremen and Merenlender 2018; Ostrom 2009), and processes of engagement which are based on trust and respect (Davis and Goldman 2019; Kremen and Merenlender 2018), are all critical to successful natural resource management and conservation. At a more local scale, the approaches taken to achieve successful natural resource management and conservation will necessarily vary, as they are tied to locally relevant cultural and economic realities. Yet, there remains a dearth of research on the place-based ways in which secure rights to land, and support to local livelihoods can work in specific social-ecological contexts. In this chapter, we will highlight the innovative approaches which are currently being taken in the Tarangire Ecosystem to secure land rights and in turn support the resilience of culturally relevant local livelihoods (Davis and Goldman 2019). The approaches we describe are suited to the social-ecological context of the Tarangire Ecosystem, as well as the particular constitutional and legislative setting of the United Republic of Tanzania. They also recognise the important role that local communities continue to play in maintaining the ecological health of the land (see Berkes et al. 2012) on terms that community members dictate.

This chapter begins by considering the recent history of natural resource governance institutions in the Tarangire Ecosystem and how these have changed over time. We describe in detail how pre-existing traditional natural resource management systems were tailored to the social-ecological context of that time. We look at the current social and ecological context of the Tarangire Ecosystem, with a focus on how land tenure ties to local livelihoods. We then highlight the importance of rights to land tenure in this context, and show how such rights can play a critical role in the conservation of the Tarangire Ecosystem for people and wild animals. We follow this with a description of the ongoing process to secure rights to land through spatial planning at a local scale, particularly where there are natural resources that are communally significant. We also discuss how this process can be scaled up across a landscape. Finally, we reflect on the challenges of spatial planning in the context of current and future social-ecological variability and uncertainty. Although we deliberately limit our focus geographically to the Tarangire Ecosystem, we expect that our findings are likely to be relevant to other social-ecological systems which face similar challenges, particularly those in other sub-Saharan rangelands.

5.2 Research Approach

Our research approach is based on the concept of social-ecological systems which, as described by Berkes and Folke (1998:4), is a concept which can be used as an analytical structure to study local natural resource management systems by "match[ing] the dynamics of institutions with the dynamics of ecosystems for mutual social-ecological resilience and improved performance". Taken in this sense, linking social and ecological systems allows us to link two different streams of resource management theory. Firstly, there are systems thinking and adaptive management, where there is an emphasis on linkages and feedback controls across social and ecological systems. Secondly, there are people-oriented institutions and property rights. Using this as our conceptual framework allows us to order material, unveil patterns, and think clearly about relevant phenomena, all in a manner which emphasises the importance of, and links between, coupled and interdependent social and ecological dimensions (Folke et al. 2005). Furthermore, we look beyond the notion of simple panaceas, and instead remain open to a multitude of opportunities towards the amelioration of undesirable social and ecological outcomes (Ostrom and Cox 2010).

Such an approach has previously been used in other research on natural resource management to highlight the most significant elements that affect the likelihood of users' self-organizing to sustainably manage resources. These include: (a) communities have clear devolved rights over resource management; (b) institutions function at the correct social and ecological scales; (c) governance mechanisms can link across scales; (d) communities experience benefits from managing their resources; (e) strong social norms of collaborative governance and management are present (Brehony 2020; Cumming 2011; Ostrom 2007, 2009; Reid et al. 2014).

Our research approach is also informed by Martin's (2017) notion of "just conservation," where local perceptions of social justice mediate conservation outcomes. Indeed, local perceptions of social justice can determine how legitimate an intervention (like a conservation project) is considered to be and therefore the extent to which there will be local support for the intervention (Pascual et al. 2014, 2021). If this is not considered and the legitimacy of an intervention is questioned, then there will be a much higher compliance cost, an increased likelihood of conflict, and a decreased chance of achieving intended outcomes (ibid.). On the other hand, inclusive approaches involve appropriate access to resources, equitable distribution of costs and benefits, participatory decision making and respect for local cultures and knowledges.

Finally, due to the significant restrictions to travel caused by COVID-19, this chapter was principally a desk-based review of recent literature, together with our own knowledge and experiences. Alais Morindat has gone from herding his father's livestock, to now owning his own herd of livestock in the Tarangire Ecosystem. Makko Sinandei has spent decades working in the Tarangire Ecosystem for Ujamaa Community Resource Team. Peadar Brehony has been a regular visitor to the Tarangire Ecosystem for over 20 years. Aside from this, the research we present is also informed by conversations with, and readings of the works of many others.

5.3 Land and Livelihoods in the Tarangire Ecosystem

5.3.1 Livelihoods and Land Management in Semi-arid Rangelands

The Tarangire Ecosystem lies in East Africa's semi-arid rangelands, where pastoralism[1] combined with subsistence hunting and fishing, or small scale cultivation, has been a way of life for at least the past 4000 years (Marshall 1990; Marshall et al. 2018; Spear and Waller 1993a). Indeed, associations between people and their domestic grazing animals, as well as between people and wild animals, have allowed people to thrive in the arid and semi-arid rangelands of East Africa, where interannual rainfall regularly varies by more than 30% and ecological shocks are common (Collett 1987; Homewood 2008; Spear and Waller 1993a). East Africa's

[1] Pastoralism encompasses both those who are directly dependent on livestock for their livelihoods, as well as, in a broader sense, the entire system that is built around this people-livestock culture and economy (Homewood 2008).

unique bimodal rainfall patterns, and the introduction of *Bos indicus* cattle breeds, provided optimal conditions for the emergence, between 3000 and 2000 years ago, of specialised, milk-based pastoralism (ibid.).

These livelihood systems were built on detailed and locally rooted environmental knowledge which were passed on from generation to generation. The focus of these systems was on minimizing risk, particularly from drought losses, for long term resilience (Butt et al. 2009; Mwangi and Ostrom 2009). These flexible systems were not some imagined harmony with nature, but rather an effective strategy to adapt to the local social-ecological circumstances.

The people living in East Africa's rangelands have therefore, for millennia, maintained and created ecosystems and landscapes *for* pastoralism; ones which maximize seasonal grazing resources that can also be taken advantage of by wild animals (Allan et al. 2017; Illius and O'Connor 2000; Keesing et al. 2018; Russell et al. 2018; Tyrrell et al. 2017; Western 1982). For instance, historically, large scale burning was an important part of landscape management to both prevent bush and woodland encroachment, but also to control parasites (directly, such as ticks, and indirectly, such as tsetse flies that thrive in bush land; Kjekshus 1977). Likewise, limiting crop cultivation to areas with predictable rainfall or the potential for irrigation, also ensured that vast rangelands were maintained principally for livestock and wild animal grazing and browsing.

5.3.2 Maasai Social-Ecological Systems

Over recent centuries, much of the Tarangire Ecosystem has been managed by the Maasai people.[2] We will now examine the ways in which they manage their social-ecological systems. The Maasai people are transhumant pastoralists (and agro-pastoralists) who speak a Nilotic language (Maa), and live in southern Kenyan and northern Tanzanian rangelands. Maasai rely on their livestock for cultural, spiritual, and economic reasons. For instance, livestock are used as food, to sell, or in culturally and spiritually important rituals. As such, the management of grazing commons to ensure herd productivity and resilience is deeply rooted in Maasai governance and herding practices (Spear and Waller 1993b). Although pastoralism is of paramount importance to Maasai, they also have a complex relationship with cultivation-based people and hunter-gatherers where each group traded with each other, relied on each other, fought against one another, or assimilated people from different groups during times of hardship (Berntsen 1976; Sutton 1993; Waller 1993).

The Maasai people constitute thirteen politically semi-autonomous and geographically distinct sections, with all sections tied together under the same moieties (*inkajijik*), clans (*ilajijik* and *ilgilat*) and age-set groupings (*olaji* and *ilporori*), as

[2] Other communities, including Barbaig, Warusha, Mbugwe, Rangi and many others, also managed, or still manage, parts of this landscape. In this chapter we deliberately focus on the Maasai whose influence is the most geographically extensive over this area.

well as language and culture. Maasai sections are social and political units that regulate access to large territories and thereby limit the use of grazing areas and water resources to people within broad geographical areas (Homewood and Rodgers 1991). Although diminished today, overlapping clans and age-sets allowed Maasai to use reciprocal clan and age-set arrangements to move with livestock over large areas, including into the lands of other Maasai sections, largely in synchrony with wild animal migrations (Brehony 2020; Western and Nightingale 2004). These movements, often in times of severe droughts, expand the scale of use from an annual range of a few thousand, to tens of thousands of square kilometres (Western and Finch 1986).

In landscapes such as the Tarangire Ecosystem, this scale of movement is no longer possible because of changes in land tenure, including the introduction of Tarangire National Park where access to resources by local communities is no longer permitted (Igoe 2004; but see Miller et al. 2014 for other research on the historical importance of the park for grazing).

At the household level (*ormarei*) a family's social standing, wellbeing, and survival are intimately bound to the welfare of its livestock through the conservation of pasture and water. Nevertheless, there is no word for "conservation" in Maa. Instead, the link between rainfall, pasture production, herd productivity, family welfare and the maintenance of commons resources is incorporated in the concept of "*urematare"* (Western et al. 2020). *Erematare* is best described as an "ethos", the interconnectedness of Maasai husbandry practices, cultural customs and systems of household, livestock and land management (Godfrey 2018). *Erematare* linkages stretch across landscapes through social networks (as described above), giving households access to the resources needed to sustain them through the seasons and in times of drought.

Erematare also extends to the management of land for all life, including wild animals, which holds many values and uses among the Maasai, including for food, clothing, medicine, sacred ornamentation, utensils, clan symbols, environmental indicators and aesthetic appeal (Chap. 13; Kioko et al. 2015; Roque De Pinho et al. 2014; Western et al. 2019). Concepts which approximate this ethos include Nicolay Vavilov's "biocultures[3]" (Nabhan 2012) and Aldo Leopold's "land ethic" (Leopold 1949).

These systems of land management are rooted in local traditional ecological knowledge, and the ways in which this was passed on was a critical part of Maasai political and social organisation. For instance, young boys were sent out by their fathers and elders, to herd livestock in the pastures near home, learning about their livestock, what they needed, what plants they ate, how often they were to be watered, and so on. Then, as they became young men, they became warriors (*ilmoran*) who were given responsibility for herding cattle over longer distances, learning about the landscape at a broader scale, the locations of different patches of resources,

[3] Nicolay Vavilov describes biocultures as evolved husbandry practices and cultures sustained the health of the land for generations in the face of environmental perturbations and climate change.

including vegetation, water, and salt licks, interacting with neighbouring people, and understanding how livestock moved over both wet and dry seasons. This was "*eleenore,*" in the sense described by Ole Mpaayei in 1954: "when Maasai wish to migrate, they send scouts [*ilaleenok* who go to *eleenore*] to first see the land. When they return … they tell you how much grass and water there is" (Ole Mpaayei 1954:60).

Following this, warriors become junior elders who decide, in collaboration with their traditional age-set spokesmen (*ilaigwanak loonkishu*), fathers, elders, and warriors, where livestock should move, and how grazing resources should be managed (Brehony 2020). The role of elders is particularly important in drought seasons when knowledge about, and ability to negotiate access to distant resources is critical.

5.3.3 Changes in Land Institutions Over Time

What we describe above represents a picture of the traditional organisation of Maasai (Brehony 2020; Jacobs 1965). However, over recent decades, Maasai livelihoods, as well as their systems of land and livestock management have changed. Traditional systems have not suddenly been rejected, but instead, in many parts of Maasailand, the traditional and the modern have formed a dynamic combination with trade-offs and battles for legitimacy and morality (Brehony 2020).

For people living in the Tarangire Ecosystem, the process of modernisation has come at a significant cost. Over the past few decades, slowly and surely, millions of acres of land have been alienated from the management of traditional institutions, to other land uses, from large-scale commercial farms, which were primarily allocated to expatriate farmers, to national parks, and other land uses (Bluwstein et al. 2018; Kauzeni et al. 1993). Following independence in 1961, Tanganyika nationalised all land in 1962, and in 1963 the role of traditional chiefs in administering local affairs was abandoned (Kauzeni et al. 1993), and instead government committees at the regional, district, and village level were formed (ibid.). This fundamentally altered the rural land management that had once relied on traditional leadership, and commons land in particular.

Around the same time, the Tarangire Ecosystem landscape was being fragmented with new institutions which control land-use and management. We will not cover the history of Tarangire National Park here, as this has been covered by others in this volume (Chap. 2).[4] For the purposes of this chapter, we will revisit a couple of key points. An area around the Tarangire River which was being used and managed by pastoralists and other groups was initially declared a game reserve for hunting wild animals, called Tarangire, in 1957 (Igoe 2004). While a game reserve, the area continued to be used for grazing by local pastoralists, including during a severe drought

[4]These histories are similar to those we have expressed in other published work (Brehony 2020) for other parts of what was considered Maasailand, which includes the Tarangire Ecosystem, up to central Kenya at its northern extent.

in 1961 when elders revealed that access to Silale (or also Silalo) swamp was a critical drought grazing refuge used by some families who lived in the surrounding areas[5] (ibid.).

Then, in 1970 the area was gazetted as a National Park (Igoe 2004), controlled and managed by a government institution, and local people were no longer permitted to use or manage the land or resources within the boundaries of the park (see Fig. 5.1). Practices which once took place, such as accessing water, grazing, foraging, cultivating, or simply walking through to a neighbouring area, were no longer permitted (Goldman 2003; Igoe 2004; Igoe and Brockington 1999; Sachedina 2008). Roads, bridges, hotels, and offices were now built in areas where there were once none. Or that is how the area which became Tarangire National Park was perceived and understood to local communities who saw this form of conservation as no different to any other process of land alienation. To them, the park was not a public resource, but rather an area from which they were excluded, while wealthier local and international elites benefited (Igoe 2004; Sachedina 2008).

Indeed, over the past decades, a plethora of other pressures, including large-scale commercial agriculture developments (for example for wheat, barley, and flowers) and international development programs have further reduced resource availability and alienated local citizens from large tracts of land (Bluwstein et al. 2018; Igoe 2004; Igoe and Brockington 1999; Kauzeni et al. 1993).

Today, the Tarangire Ecosystem spans three of Tanzania's government administrative regions, and the landscape is mosaiced by two national parks (Lake Manyara National Park and Tarangire National Park), a game reserve, several game controlled areas, several forest reserves, a wildlife ranch, several community-based conservation initiatives, including Wildlife Management Areas (WMAs), Certificates of Customary Rights of Occupancy (CCROs), and easements, as well as numerous small towns and urban centres, and vast areas of village land (see Fig. 5.1).

5.3.4 Hardening of Lines and Loss of Flexibility

A social-ecological systems perspective demands that we understand links, feedbacks, and dynamic relationships which are constantly evolving (Liu et al. 2007; Ostrom 2007). Yet, the process of modernisation we describe above is predominantly about anchoring things in space, which in effect creates spatial separations and hardens boundaries (Watson 2010).[6]

For instance, Tarangire National Park has, together with a multitude of other forms of land alienation and government policies, accelerated the rate of growth of cultivation areas in surrounding lands, for two main reasons. Firstly, cultivation is

[5] Other research suggests that the swamp was indeed used in the past, not on a regular annual basis, but rather during the most severe droughts (Miller et al. 2014).

[6] This process happens for a number of reasons, not least of which is territorial and resource control (Scott 1998).

fixed in space and when people decide to cultivate a particular piece of land, there is no debate over who owns the land. In a context where the alienation of commons land (such as when creating national parks) is commonplace, there is therefore a strong incentive for people with a lack of secure land tenure to turn towards the more readily observable land-use, cultivation (see Weldemichel and Lein 2019 for a similar effect in the Maasai Mara with fencing).

Secondly, restricting access to a variety of key grazing resources (functional heterogeneity) eroded the mobility and flexibility that is crucial for livestock and wild animals to thrive in semi-arid rangelands (Butt et al. 2009; Fynn et al. 2016; Owen-Smith 2004; Western et al. 2020). Although there are inevitable trade-offs when livestock and wild animals compete for food, water, and other critical resources, and share parasites and pathogens (Herrero et al. 2009; Keesing et al. 2018), the integration of livestock and wild animals can also provide social and ecological benefits under particular conditions (Keesing et al. 2018; Kimuyu et al. 2017; Odadi et al. 2011).

Conditions such as flexibility and mobility allow pastoralists in highly variable semi-arid rangelands to track the richest pastures, often in tandem with wild animals. This minimizes exposure to drought, pathogens, local pasture degradation and perturbations (Boone 2005; Fynn et al. 2016; Wang et al. 2006). These ecological benefits of mobility are reflected in the energy bonus of improved digestive efficiency, growth rates and milk yields for both mobile livestock and wild animals (Butt and Turner 2012; Illius and O'Connor 2000; Owen-Smith 2004; Wang et al. 2006). However, these benefits are not realised when livestock and wild animals are increasingly confined to much smaller spheres of more intense grazing as mobility and flexibility are eroded (Butt 2010; Butt et al. 2009). Without the rest periods that were part and parcel of traditional grazing management, more regular intense grazing results in reduced grazing productivity, as multi-decadal rangeland research in other parts of Maasailand has demonstrated (Western and Mose 2021; Western et al. 2021). These factors result in exacerbated losses during drought periods, which in turn drive pastoralists to diversify into other livelihoods, including cultivation (Homewood 2008; Homewood et al. 2009). Indeed, an increasing number of Maasai are diversifying livelihoods into cultivation and wage-labour in urban and peri-urban areas (Homewood et al. 2009; McCabe et al. 2010). To exacerbate matters, cultivation tends to stabilise more readily in higher rainfall areas, which are often also key grazing areas for livestock and wild animals. This vicious cycle became obvious to pastoralists and conservationists alike, but different groups proposed different solutions to mitigating the loss of open rangelands.

Between 2010 and 2019 the percentage of global land covered by protected areas expanded from 14.1% to 15.3% (Maxwell et al. 2020). Some conservationists advocate for these area-based conservation targets to increase. For instance, there have been calls to set aside 20% of the globe for conservation by 2020, and even 50% by 2050 (Maxwell et al. 2020; Watson et al. 2014; Wilson 2016). Yet thus far, the success of such targets remains unclear (Maxwell et al. 2020). More importantly, area-based targets ignore the fact that land is a critical asset for people, particularly in rural areas and in the global south, to prosper, while simultaneously failing to recognise the important role that local communities play in conservation (Díaz et al. 2019a; Garnett et al. 2018). In Tanzania, the state and its manifold arms of power have regularly relocated people, disrupted rural livelihoods, or claims to land, in the name of conservation,[7] with the backing of global narratives that area-based targets can solve global wild animal declines and extinctions (Weldemichel 2020). Yet, at the same time, Tanzanians, particularly in poorer rural areas, rely on access to land. Over 72% of the population derive their livelihoods from cultivation, livestock, or related activities – all of which depend on land. As of 2020, the agricultural sector continued to be the biggest contributor to national GDP, at 26.5% (Bank of Tanzania 2020).

Other conservationists have proposed various forms of community-based conservation. In the Tarangire Ecosystem these have, for instance, taken the form of easements (see Davis and Goldman 2019; Northern Tanzania Rangelands Initiative 2019), wildlife ranches (see Goldman 2006), village based eco-tourism partnerships (see Dorobo Tours and Safaris and Oliver's Camp Ltd. 1996), and Wildlife Management Areas (see Keane et al. 2019). We will not go into detail about each of these approaches, but there are two common themes. Firstly, these models are reliant on revenue from international eco-tourism. Aside from in a minority of cases, conservation and eco-tourism alone cannot adequately compensate for loss of access to resources, or overcome other opportunity costs (Keane et al. 2019, Tyrrell, in press). Any revenue that is generated is often woefully inadequate and rarely reaches local people, particularly the poorest, who need it most (Keane et al. 2019). The COVID-19 crisis has further demonstrated the lack of resilience in relying on single external sources of funding, such as international eco-tourism (Lindsey et al. 2020).

The second common theme in all community-based conservation efforts in the Tarangire Ecosystem, is that the fate of wild animals depends heavily on the future of pastoralism (see Northern Tanzania Rangelands Initiative 2019). Wild animals find refuge in protected areas, but particularly in the case of large mammal, for significant populations to persist, they must range beyond protected areas to access seasonal pasture and nutrients (Fynn and Bonyongo 2011; Owen-Smith 2004). These areas are community land, often owned and managed by pastoralists. Yet, as we have described above, pastoralism faces many threats from both an ecological and social perspective. Many of these threats are shared with wild animals. For both livestock and wild animals, ecological adaptability to environmental perturbations is being eroded through the loss of space and mobility; land-use changes and land degradation; decreasing rangeland productivity; decreasing resilience to droughts; and the climate crisis (Boone et al. 2005; Haile et al. 2020; Hobbs et al. 2008a; Western et al. 2015, 2021). Socially, the erosion of traditional governance institutions which regulated pasture use and minimized risk to drought and other perturbations, is exacerbating these processes (Mwangi and Ostrom 2009). However,

[7] In early 2021 the Ngorongoro Conservation Area Authority attempted to evict thousands of people in the name of conservation and there have been several violent relocations in the areas surrounding Serengeti National Park (Currier and Mittal 2021).

because these are commonalities, redressing them can alleviate threats to both pastoral systems and wild animals.

For instance, by finding ways to include both informal traditional land-use practices and formal (e.g. state) practices, there are opportunities to support governance processes which are participatory, legitimate and effective (Folke et al. 2005). Indeed, social-ecological systems theory and landscape governance theory suggest that negotiated combinations of the formal and informal can help to solve problems of common resource management through social networks, rule-based institutions, and devolved management rights, at appropriate social and ecological scales (Cumming 2011; Ostrom 2007; Reid et al. 2014).

5.3.5 Conservation Inside-Out

There are therefore opportunities to build on this understanding in pastoral landscapes, such as the Tarangire Ecosystem. Western et al. (2020) show how space and mobility for sustaining large mammals can be secured indirectly through an approach the authors term 'conservation from the inside-out'. This approach draws on the aforementioned husbandry (*erematare*) and conservation practices used to maintain the productivity and resilience of pastoralism or other land uses, that also directly or indirectly maintain large free-ranging wild animal movements in the process. Whereas community-based conservation is founded on direct incentive-based approaches tied to wild animal conservation, an "inside-out" approach uses primary livelihood considerations to win space for wild animals indirectly (ibid.). At the heart of this approach are support for local citizens' rights to land and natural resources,[8] with support for thriving and ecologically important livelihoods, and *erematare,* a place-based land ethic to hold it together.

The traditional grazing and land-use practices, social networks, and governance arrangements that such an approach is reliant on to sustain natural resource management from an ecosystem to landscape and regional level, are changing. However, as we shall now describe, new tools and institutions which explicitly consider the current social-ecological realities also exist.

5.3.6 Wildlife Management Areas, Village Land, and Certificates of Customary Rights of Occupancy

The current effectiveness and constraints of Wildlife Management Areas (WMAs) have been covered in detail elsewhere (social: Homewood et al. 2020; Keane et al. 2019; Nelson et al. 2021; Sulle et al. 2011; Wright 2017; ecological: Kiffner et al.

[8] These already exist in constitution and legislation, but not always in practice.

2020; Lee and Bond 2018a). However, what we wish to focus on here are the opportunities. WMAs can offer secure rights to land tenure, and livelihoods, through land-use planning. For instance, Wright (2017) details the cases of Enduimet WMA and (the once nascent) Lake Natron WMA where people turned modern and formal WMAs into spaces which fit their traditional ideals by using new land management tools, primarily *for* grazing of livestock, which can potentially also benefit wild animals (Fynn et al. 2016; Keesing et al. 2018; Russell et al. 2018; Tyrrell et al. 2017). Similarly, it appears that Randilen WMA in the Tarangire Ecosystem is now[9] regarded by the participating communities as community-based, and have come to view the WMA as centrally important to their livelihoods (Chap. 6).

Likewise, Gardner (2016) and Nelson and Ole Makko (2005) describe the ways in which people in Loliondo, another part of Maasailand in Tanzania, turned modern state governance in the form of Village Land into legitimate political entities capable of securing livelihood and partnerships with eco-tourism operators on strong terms.

In the rest of this chapter, we will focus our attention on alternative innovative approaches, particularly Village Land Use Planning and Certificates of Customary Rights of Occupancy (CCROs).[10] To do this, we will focus on the work of Ujamaa Community Resource Team (UCRT) who have pioneered this process in the Tarangire Ecosystem (Lekaita et al. 2014).

Over the past two decades UCRT have, in collaboration with local communities, District Councils and other development partners, supported villages in the Tarangire Ecosystem in their efforts to put in place participatory land-use plans and natural resource management plans. These cement traditional land-use and governance practices with legal requirements, under Tanzania's National Land Policy (URT 1995), to plan village land-use. This participatory process, together with a robust legal instrument, provides an effective, legitimate and participatory tool for land management (Folke et al. 2005).

The result is a system where local citizens have clear authority and rights over their land, and where land uses which are critical to local livelihoods are formalised (Lekaita et al. 2014). These land-use plans are then managed by village councils who are the most powerful form of local government authority in Tanzania (URT

[9] In the initial implementation of Randilen WMA, empirical research by Loveless (2014) recorded conflicts in the villages of Naitolia and Mswakini. Subsequently, there were debates about the extent to which the implementation of Randilen WMA represented a community-based approach (Brehony et al. 2018; Lee and Bond 2018a, b). It should be celebrated that, based on the recent research presented by Raycraft (Chap. 6), community members themselves now report Randilen WMA as a "community-based" approach.

[10] Most land in Tanzania is held under Customary Rights of Occupancy. These are land rights exercised through the organs of local governance administration, Village Councils and Village Assemblies. "Customary lands are defined as 'Village Lands' in the Land Act (URT 1999a, b), and the Village Land Act (URT 1999b) provides the legal basis for management and governance of these lands. Village lands held through customary rights of occupancy may be apportioned to individuals or groups through Certificates of Customary Rights of Occupancy (CCROs), which effectively formalizes their rights to that land" (Lekaita et al. 2014).

1999b), through the implementation of natural resource governing bylaws that stipulate penalties for misuse.[11] Even more importantly, where planning results in common land-use areas which were once more easily alienated from local citizens, such as communal grazing areas, UCRT are assisting villages to secure group Certificates of Customary Rights of Occupancy (CCROs).

5.3.7 Scaling Up

In social-ecological systems and landscape theory, scales of governance are critical (Arts et al. 2017; Cumming 2011; Ostrom 2007). In semi-arid rangelands, it is in the vested interests of pastoralists and agro-pastoralists to expand the scale of management to much larger landscapes (Sayer et al. 2013; Scarlett and Mckinney 2016). As we have described above, this is vital to sustain livelihood productivity and resilience, as well as to avoid the negative impacts of rangeland fragmentation (Groom and Western 2013; Hobbs et al. 2008b; Western et al. 2020).

As UCRT developed the concept of participatory land-use planning and natural resource management further, they realised that through participatory governance structures, several villages could join grazing lands that were already secured with communal CCROs, through Joint Rangeland Committees, which can then be formalised through legal Memorandums of Understanding[12] (see Figs. 5.1, 5.2, and 5.3).

Furthermore, scaling up once more, at the District level, a higher level committee, the District Rangelands Governance Advisory Committee exists to advise and coordinate the efforts of local Joint Rangeland Committees (see Figs. 5.1, 5.2, and 5.3). At each level, from the Village Rangeland Management Committees, to the Joint Rangeland Committees, and up to the District Rangeland Governance Advisory Committee, there are very clear rules about the composition of these committees (with regards to representation and justice) and their roles and responsibilities.

[11] In Tanzania, according to the 1977 Constitution (URT 1977), a village forms the Local Government Authority and once registered under the Local Government Act No.7 (URT 1982), it is mandated to enter into agreements which benefit village members, on behalf of village members, subject to the consent of the Village Assembly. The Village Land Act (URT 1999a, b: No. 5) mandates that village authorities manage and protect village lands on behalf of village members, and Section 11 mandates that village authorities can enter into joint agreements to use village lands.

[12] Legislation of land tenure in Tanzania provides the opportunity for two or more villages to share natural resources across village boundaries. The Village Land Act (URT 1999a, b), under section 11 and through Regulation 2002 No. 26–35, empowers village councils to enter into joint land-use agreements with other villages, to jointly plan, manage, and use joint natural resources. Furthermore, the Land Use Plan Act section 18 (URT 2007) provides for the formation of a Joint Village Land Use Plan authority, and in section 33 (1) (b), provides for the preparation of a joint "resource management sector plan" for the use and management of shared natural resources. Furthermore, once the Joint Village Land Use Plan has been finalised, the association of land owners can seek customary rights of occupancy over the land, in order to secure their rights to land tenure.

Importantly, these structures do not supersede other customary institutions of land management, but aim to work in tandem with them (UCRT and Simanjiro District Council 2019, and see Brehony 2020 for examples elsewhere).

These secured and connected lands are managed for local livelihoods, principally grazing for livestock. Indeed, as described by the Simanjiro District Council (which falls within the Tarangire Ecosystem), the primary aim of this scaling up in connectivity is driven by self-interest; to ensure access to resources beyond a single village's land (access to habitat heterogeneity), to mitigate land-use related conflicts, and to reduce food insecurity (UCRT and Simanjiro District Council 2019; UCRT 2010; Western et al. 2020). However, in so doing, provided villages continue to accept the presence of wild animals on their land, they also allow wild animals to access large landscape functional heterogeneity. As of 2019, in Simanjiro District alone, over 1.5 million acres of land have been secured under certificates of customary rights of occupancy as open, communal land. This is more than double the area of Tarangire National Park (UCRT and Simanjiro District Council 2019).

From this starting point, other stakeholders, such as conservation organisations can collaborate with local communities, to achieve joint goals, on the terms of the local citizens who stand to bear the greatest costs if things do not work out – those with the most skin in the game. Indeed, as Davis and Goldman (2019) discuss, such a starting point is more likely to result in achieving joint outcomes, for instance when proposing payments for ecosystem services. Furthermore, the approach we describe above is place-based and tailored to the particular social-ecological context in Tanzania and semi-arid rangelands more generally. It is driven by local challenges of land security and recognises the importance of local management and local livelihoods.

5.3.8 Limitations

Although we believe that this approach shows great promise, we are also cautious for a number of reasons. Firstly, Bluwstein (Chap. 2) calls into question the distinction between state controlled land-use planning which separates people and wild animals, and local level land-use zonation. We should not forget to ask: "who benefits most from these arrangements?" We cannot answer this convincingly, but hope that further research will examine this in greater detail. Nevertheless, we believe that if the primary concern remains meeting people's material needs, through a diversity of culturally and economically important livelihoods, by securing access to land and natural resources, then this approach will remain effective and legitimate.

Secondly, we acknowledge that these systems are necessarily less flexible than the aforementioned traditional systems. Particularly in semi-arid rangelands with significant spatial and temporal variability which is likely to increase (Haile et al. 2020), any land-use which is fixed in space can result in fragility, as opposed to resilience. Although Maasai governance systems have institutions which are well suited to thriving within these landscapes (Goldman 2006), we are yet to see whether the aforementioned systems of land tenure and management which combine the formal and informal, will perform in a world of increasing social-ecological uncertainty. Greater attention needs to be paid to how approaches such as those we describe above, can maintain flexibility and become part of a more adaptable governance system (Brehony 2020).

Finally, from a governance perspective, the approaches we describe are founded on negotiation and consensus, an important starting point towards achieving the good governance trilemma of participation, legitimacy and effectiveness (Folke et al. 2005). However, even processes like these should not blind us to the reality that institutions of authority create power imbalances which can be abused for personal gain.

5.4 Rounding Off

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Service (IPBES) specifically recognises the critical role that local communities play in conservation, through their practices and detailed knowledge of ecosystems and biodiversity (IPBES 2021).[13] Indeed, at least 28% of the world's land surface is effectively managed to meet global conservation goals by local, indigenous communities (Garnett et al. 2018) under diverse forms of place-based stewardship (Díaz et al. 2019b). The future success of conservation efforts in East Africa's rangelands depends on these communities.

In the Tarangire Ecosystem, although processes of modernisation have resulted in land alienation and have eroded traditional landscape scale management systems, in this chapter we have highlighted some ongoing innovative approaches to overcome these challenges. Through spatial planning, local citizens have managed to secure land rights to communal land and resources, from the local to the regional scale. These approaches build on the local social-ecological context and provide a mechanism for continued access to landscape scale functional heterogeneity, which is critical for pastoralists and wild animals to overcome current and future social-ecological variability. Ultimately, this model creates a mixed-use coexistence landscape, where biodiversity conservation moves towards a land sparing-sharing continuum with a range of land-use options (Phalan 2018).

Although we have limited our focus to the innovations taking place in the Tarangire Ecosystem, these approaches are likely to be relevant to other social-ecological systems where solutions which combine informal and formal governance and practices, together with securing rights to land and resources for local livelihoods, are needed or are emerging. From a conservation, land ethic or *erematare* perspective, such tools can act to prevent the conversion of communal land and resources to other land uses and instead support local institutions to maintain rights

[13] A commitment captured under IPBES Objective 3 (b) "Enhanced recognition of and work with indigenous and local knowledge systems" (IPBES 2021).

to land and resources, and effectively manage these from the local to the landscape scale (Leopold 1949; Western et al. 2020). In so doing, they can continue to maintain resilient livelihoods, while also making a significant contribution to protecting ecosystems and wild animals (Reid et al. 2014).

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Chapter 6
Community Attitudes Towards Randilen Wildlife Management Area

Justin Raycraft

Abstract Wildlife Management Areas (WMAs) are a particular conservation strategy in the Tarangire Ecosystem. WMAs aim to balance wildlife conservation with community livelihoods through the implementation of land use plans at the village level that restrict some human activities while allowing others. They also enable the central government to extract revenue from conservation tourism that occurs on village land. The creation of WMAs can lead to tensions among local communities, private investors, and government authorities as a consequence of competing interests within and across these stakeholder groups. On these grounds, WMAs have been criticized by social scientists, particularly in such instances where the resource rights of rural communities are marginalized. Few case studies to date, however, have employed representative sampling procedures and quantitative methods to assess community perspectives on WMAs. This chapter presents results from a proportionately weighted and randomly sampled survey of community attitudes towards Randilen WMA (n = 678) administered in 2020. The results speak to high levels of community support for Randilen WMA, and highlight people's lived experiences of inclusion in conservation governance and management. Drawing from these findings, this chapter forwards an alternative perspective on WMAs, suggesting that they can show promise as mechanisms for reducing rangeland fragmentation and supporting people, livestock and wildlife.

Keywords Wildlife Management Areas · Community-based conservation · Rangeland management · Wildlife conservation · Attitudes · Lived experiences · Governance · Management

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6.1 Introduction

In northern Tanzania, Wildlife Management Areas (WMAs) represent a key strategy for conserving wildlife habitat outside of national parks. WMAs are often located adjacent to national parks in seasonal wildlife dispersal areas that overlap village land. The process of establishing a WMA involves reclassifying some village land as a reserve area to be managed through land use plans. These management plans generally prohibit some human activities to support conservation objectives, while allowing others for the sake of community livelihoods. WMAs must be contextualized in relation to a political history of centralized resource governance in colonial and post-independence Tanzania. They were established following reformation of the country's wildlife sector in the late 1990s to serve three main aims (Nelson et al. 2007): First, WMAs created a legislative framework for the central government to extract revenue from wildlife-related tourism occurring on village land. Second, they served to protect wildlife habitat outside of protected areas that was at risk of fragmentation through land use change. And third, WMAs were meant to reflect a form of decentralization that empowered local communities by allowing them to access the benefit streams associated with wildlife resources through participatory governance and management institutions (Wilfred 2010; Songorwa 1999). WMAs were thus microcosms of wider sectoral reform in that they were conceived to address key social, political, economic, and ecological concerns that had arisen from the national park model of conservation.

While WMAs have been touted by the government as a community-based model of wildlife conservation, this rhetoric often does not translate into practice (Goldman 2003). WMAs have been heavily criticized by social scientists for reconfiguring jurisdictions of authority in favour of the central government and private investors at the expense of village communities (Kicheleri et al. 2018a; Kicheleri 2018; Green and Adams 2015; Moyo et al. 2017). They have been framed as a form of state sanctioned rent seeking to 'grab' tourism revenues from communities, which may otherwise enter into direct agreements with private investors (Benjaminsen et al. 2013; Sulle and Banka 2017). Like other types of conservation areas, WMAs can be implemented in ways that are exclusionary and rife with conflict, depending on the distributions of power across WMA stakeholders, the livelihoods of local communities, and the governance and management mechanisms at play (Francis 2019; Bluwstein et al. 2016; Kicheleri et al. 2018b, 2021). Rather than contributing to 'sustainable development' at the village level, several studies have suggested the contrary, drawing connections between WMAs and persistent conditions of poverty in local communities (Homewood et al. 2020; Kaswamila 2012; Igoe and Croucher 2007; Keane et al. 2020). Despite these significant criticisms, however, WMAs may also represent opportunities for communities (see Wright 2017, 2019), particularly in such instances where they secure community access to rangeland resources vis-à-vis external actors with competing interests in the land. WMAs create an institution for managing land that is formalized through law, making it difficult for outside actors to encroach into community areas or grab land for other purposes. As such, WMAs may also directly protect the interests of resource-dependent communities. This consideration highlights the importance of engaging with the perspectives and lived experiences of community members on a case-by-case basis, rather than defaulting to broad-stroked critique of all WMAs in Tanzania. Currently, however, there are few studies that assess community attitudes towards WMAs through quantitative methods and representative sampling procedures. Consequently, the generalizability of qualitative studies of the social impacts of WMAs may be overstated in the literature.

This chapter presents data from a survey of community attitudes towards Randilen WMA, located to the northeast of Tarangire National Park (TNP) in the Monduli District. It draws attention to high levels of community support for the WMA across the member villages, and community perceptions of participatory governance and management institutions. Based on the findings of this study, I offer an alternative perspective on WMAs, maintaining that they can show potential for reducing rangeland fragmentation in such cases where they safeguard the interests of village communities and succeed in garnering local support for conservation. The remainder of the chapter is organized as follows: I first provide some background on Randilen WMA, including some basic descriptions of its governance structure, management divisions, and ethnographic context. After this, I lay out the methods for data collection and present the results of the survey. In the final section, I discuss these findings and arrive at my key conclusions in relation to the wider discourse on the social impacts of WMAs in Tanzania.

6.2 Randilen WMA Governance Structures and Management Units

Randilen is one of the newest WMAs in Tanzania, gazetted in 2013 under Regulation 32(2), and the Sixth Schedule of the Wildlife Conservation (Wildlife Management Areas) Regulations of 2012 (MNRT 2012, 9–10, 21; see also schedule 13(1–2) on page 12). It constitutes a key wildlife habitat area adjacent to TNP (Bond et al. Chap. 8; Foley and Foley Chap. 10), including a sizeable portion that has been managed for decades by TNP authorities as if it were part of TNP, though the area is technically in village land (Fig. 6.1). In total, Randilen WMA encompasses 31,201 hectares and includes eight member villages: Oldonyo, Lolkisale, Nafco, Lengoolwa, Lemooti, Naitolia, Mswakini Juu, and Mswakini (Chini). For the sake of clarity, ‘Chini’ is included here to differentiate Mswakini from Mswakini Juu, though some community-members have cautioned that referring to Mswakini Chini as such may be considered pejorative in some contexts (*juu* translates as up/above, while *chini* translates as down/below). Pursuant to the legislative framework of WMAs, Randilen comprises a council of elected representatives from each of the member villages, which together form an Authorised Association (AA) that is accountable to the village councils of the member villages. The AA council is authorized to govern the WMA on behalf of its constituents as per the Wildlife Conservation Act (WCA) No. 5 of 2009 and the most recent Wildlife Management Areas Regulations (the WMA regulations have been amended and updated several times since 2012). The AA is made up of 40 members in total, with five representatives from each of the eight member

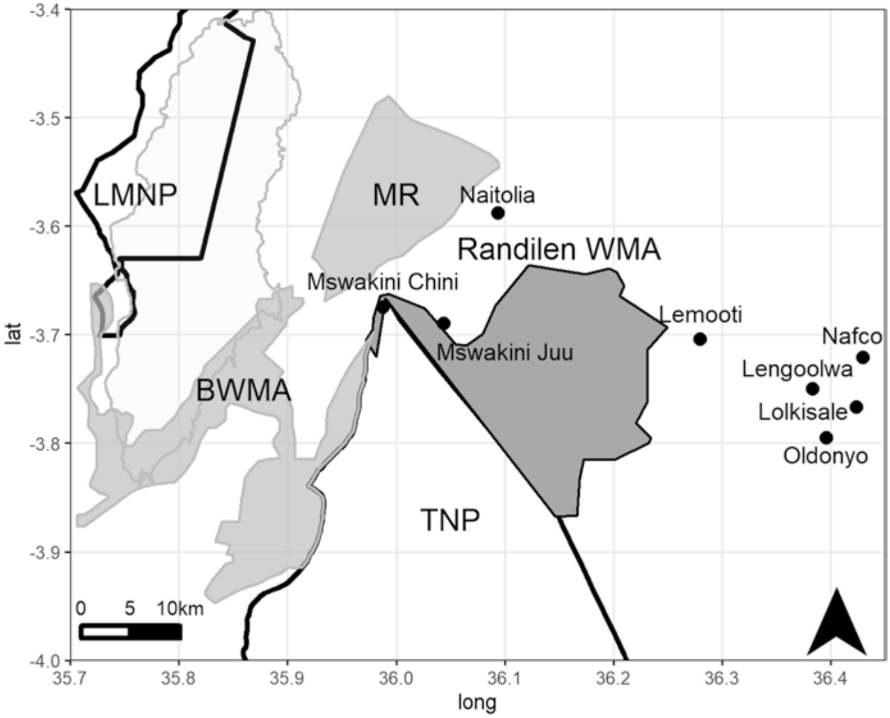


Fig. 6.1 Map of the central part of the Tarangire Ecosystem. The dark grey area shows the outline of the wildlife area of Randilen WMA. Member villages are marked as large dots for general reference. In reality, the villages cover larger areas and are administrative political units with boundaries. For context, the map also shows the outline of Lake Manyara National Park (LMNP), Tarangire National Park (TNP), Manyara Ranch (MR) and Burunge Wildlife Management Area (BWMA)

villages, including the WMA chair who is currently (2020/2021) from Mswakini Chini. Within the AA, there are sub-committees including the Finance and Planning Committee, and the Discipline Committee. Unlike other WMAs, Randilen WMA has a professionally trained manager and finance officer, both of whom are from member villages (Mswakini Juu and Lengoolwa, respectively).

The WMA also has a Board of Trustees and a District Advisory Board. The Board of Trustees holds the WMA accountable and addresses any external conflicts that are beyond the capacity of the executives and AA councils. The Board of Trustees currently comprises six individuals from the member villages. Two of the most recently subdivided villages (Lengoolwa and Oldonyo) do not yet have representatives on the board. The WMA constitution directs how the AA obtains its Board of Trustees. The District Advisory Board was established in accordance with section 33(1) of the Wildlife Conservation Regulations of 2012 for the purpose of advising the AA on matters relating to the coordination and administration of the WMA in collaboration with government and other external stakeholders (see MNRT 2012, 27). It currently includes 5–10 members.

At the top of the WMA's management umbrella is the manager, who is guided by the workplan and budgets approved by the AA. The manager oversees four management divisions: the protection unit, the tourism unit, the community unit, and the financial team. WMA regulations are enforced on the ground by Village Game Scouts (VGS), who are disaggregated into a zonal group, a camp group, and an entrance gate group. Zonal VGS fall under the protection unit, and entrance gate VGS are part of the tourism unit. Funding, training, and guidance for VGS are provided by Honeyguide (a Tanzanian NGO focusing on community-based conservation efforts), with support from the Nature Conservancy (an international NGO). Currently, there are 26 VGS from the member villages (3–4 from each village). At the head of the community unit is the WMA Chair, who oversees issues relating to the community in collaboration with the AA.

6.3 Ethnographic Context of Randilen WMA

For the most part, Randilen WMA's member villages are inhabited by WaArusha cultivators and Kisongo Maasai (hereafter Kisongo) pastoralists. The two exceptions are Nafco and Lolkisale villages, which include town-like sub-villages comprising mixed ethnicities.

The Kisongo likely arrived in the Tarangire Ecosystem a few hundred years ago, emerging from the "Maasai core" in southern Kenya and expanding southwards after defeating the Loogolala, a *loikop* sub-section that later fragmented into the Parakuyo (and perhaps the WaArusha) (Galaty 1993, 69; Spear and Nurse 1992). They encountered along the way Iraqw (near Engaruka), Barabaig (in the Ngorongoro highlands), and Wahehe (Wright 2019, 33; Spear 1997). The Kisongo are Nilotic pastoralists who view themselves as "people of cattle" in a cultural and economic sense, though they have diversified their livelihoods in recent years (Galaty 1982). Prior to colonialism, the areas which now form Randilen WMA were used for seasonal grazing as part of the pastoral mode of production, with sparse settlements throughout. Swahili-speaking in-migrants to Lolkisale and Nafco have roots in the colonial era and the recruitment of labour to work on the settler farms in the area. Oral life histories carried out in the member villages suggest that the WaArusha began settling the area between the 1950s-1970s, with encouragement from the government to out-migrate from the densely populated Meru area into the "underutilized" rangelands of Monduli (see also Igoe 2010; Bluwstein 2017). There are, however, some reports of WaArusha expansion into Monduli as early as the 1920s (see Hodgson 2001). While the WaArusha speak Maa (the language of the Maasai), and share many cultural institutions with the Kisongo (age-set system and rituals), they are primarily farmers, though they do also keep livestock. It is possible that the WaArusha descended from the Loogolala, having moved into the Meru highlands to specialize as "mountain farmers" after being displaced from the Pangani Valley by the Kisongo in the early 19th century (Spear 1993, 1997; Spear and Nurse 1992).

During the initial implementation phase of the WMA, conflicts emerged in Naitolia and Mswakini villages, as outlined by Loveless (2014). This was later re-emphasized by Brehony et al. (2018), but without new empirical evidence. In Loveless' (2014) view, villagers were discontent with the exclusionary model of WMA planning and decision-making. Her interlocutors did not feel that their villages were adequately consulted when the WMA was first created. People were afraid of losing their resource rights and facing potential displacement and dispossession. This led to open protest, including blockade of the A104 highway, and the closing of village government offices during a particularly heated period of social unrest in 2014 (see Loveless 2014, 46–47). Loveless' (2014) work provides an important backdrop of conflict and community-level opposition to the WMA planning process that should not be overlooked in any contemporary analysis of community attitudes towards the WMA. However, it is important to note that her study was based on 31 qualitative interviews and a survey of 63 respondents, administered during a month and a half of fieldwork in Mswakini Chini, Mswakini Juu, and Naitolia (see Loveless 2014, 34–36). Fieldwork was not conducted on the Lolkisale side of the WMA. By acknowledging these methodological limitations (as she does on pages 43–44), my intention is not to be critical, but to point out that her study may not have been representative of the views of all community members. People's attitudes may also have changed over time.

6.4 Methods

This chapter is based on a survey administered to 678 individuals across Randilen WMA's eight member villages from April to July 2020. The main objective of the survey was to provide an overview of community attitudes towards the WMA. The survey instrument was designed midway through a year of ethnographic fieldwork in the study villages based on qualitative themes that had begun to emerge in the context of participant observation and conversations with villagers. The questions were conceived to elicit community members' perspectives on the WMA, especially in the context of conservation governance, management, trade-offs, effectiveness, and equity. The survey instrument was close-ended, with coded numerical responses to facilitate data entry and analysis. The survey was designed to be administered in about half an hour. It covered a range of socioeconomic demographic questions (length of residence, ethnicity, education, gender, age, primary source of income, livestock assets, and farming practices). It then examined key metrics as either yes-no responses, three point ordinal items, or 5 point Likert-adapted ordinal items. Respondents chose from options ranging from 'strongly dislike' to 'strongly like' when asked directly about their attitudes towards the WMA, rather than indicating their degree of agreement with a statement. The survey questions assessed general attitudes towards Randilen WMA (5 point Likert item; a sixth option was also included for "I do not know what Randilen WMA is"); support for Randilen WMA (5 point Likert item); memory of general attitude towards Randilen

WMA five years ago (5 point Likert item); change in attitude towards Randilen WMA over the past five years (5 point Likert item); trust in authorities of Randilen WMA to act in the community's interest (yes or no); perceptions of costs and benefits (more costs; equal number of costs and benefits; more benefits); perceptions of inclusion in WMA governance (yes or no); perceptions of inclusion in WMA management (yes or no); perceptions of Randilen WMA as fortress conservation or community-based conservation (one or other); classification of Randilen WMA as successful or unsuccessful (one or other). The survey instrument was translated into written kiSwahili and administered across the member villages with the help of seven field assistants who were fluent in both kiSwahili and Maa. Survey questions were asked in kiSwahili or Maa depending on the linguistic profile of the respondent. This chapter presents relative frequencies of these responses (percentages) to provide a general quantitative overview of the current state of community attitudes towards Randilen WMA.

6.4.1 Sampling

To establish sampling frames, I sought the support of sub-village chairs to travel boma-to-boma on motorbike to compile a list of all household heads in each sub-village. I defined the first frame (male household heads) as those males who were married, irrespective of whether they lived in a single enclosed boma, or together with other married males. For the sake of establishing a non-redundant sampling frame, each married male was only listed once. The second sampling frame comprised female household heads, which I defined as women whose husbands had died, women who were divorced or separated, or women who simply lived in their own personal boma for various reasons. I included women as female heads even if they lived with married sons in a shared boma, and I included these married sons in the 'male household head' sampling frame.

Based on the inclusion criteria for determining household heads, 2037 male heads and 352 female heads were listed across the 26 sub-villages (Table 6.1). Appropriate sample sizes from these total frames were calculated using Cochran's (1963) sample size formula for finite populations, with a 95% confidence interval, and a p-value of 0.05. Using this formula, it was determined that 323 male heads and 184 female heads were needed for the samples of household heads to be representative. To take into account differences in population sizes across the study villages, I employed a stratified random sampling method involving a proportionately weighted random sample of each sub-village based on their sizes relative to the total frames (using sub-villages as strata). To determine the sample ratios, the total number of household heads in each sub-village was divided by the total number of heads in each sample frame (male and female) and multiplied by 100. These ratios were then used to calculate a proportionately-weighted sample from each sub-village. The total number of household heads sampled in relation to sub-village population size is shown in Table 6.1. To select participants, the lists of household heads from

Table 6.1 Total sampling frames and respondents sampled by sub-village during surveys administered in April–July 2020 across the Randilen WMA member villages

Sub-village	Total # of male household heads	# of male heads sampled	% of male heads sampled	Total # of female household heads	# of female heads sampled	% of female heads sampled	# of wives sampled
Oldonyo							
Nyorit A	54	8	15	5	4	80	3
Lengijape	44	7	16	3	3	100	3
Oldonyo	90	15	17	12	6	50	7
Loosikitok	63	10	16	0	0	0	4
Nafco							
Lengoolwa C	197	31	16	45	21	47	16
Lengoolwa B	66	10	15	21	13	62	4
Osilaley	60	9	15	8	6	75	4
Mswakini Chini							
Shuleni	91	14	15	12	6	50	8
Kanisani	44	7	16	7	7	100	4
Engasiti	51	8	16	15	8	53	4
Lolkisale							
Lolkisale B	113	17	15	37	20	54	9
Makao Mapya	87	13	15	21	12	57	7
Lolkisale A	67	12	18	25	11	44	5
Endarpoi	98	15	15	7	4	57	8
Lengoolwa							
Lengoolwa	57	9	16	6	4	67	6
Engosipa	93	15	16	12	4	33	8
Donyon	109	17	16	22	12	55	8
Orkisima	63	10	16	8	5	63	4
Naitolia							
Engusero	127	20	16	10	5	50	10
Ormang'way	107	17	16	9	5	56	9
Mswakini Juu							
Shimamo	68	11	16	18	8	44	6
Randilen	84	13	15	15	8	53	6
Orbukoi	99	16	16	21	10	48	8
Lemooti							
Olorisyo	54	8	15	5	4	80	5
Lesiday	16	3	19	1	1	100	1
Lemooti	35	11	31	7	4	57	4
Total	**2037**	**326**	**16**	**352**	**191**	**54**	**161**

Percentages were rounded to the nearest number each sub-village were numbered and entered into a spreadsheet. A random number generator was then used to select the numbered household heads from the compiled lists from each sub-village, until the designated quota for each stratum was reached.

As a consequence of the cultural context, male household heads outnumbered female household heads by more than 5:1. To avoid gender bias during data collection, I established a third sampling frame, 'females in male-headed households.' Within this frame, the first wife of every second surveyed male head was also recruited for participation. The total number of wives sampled was 161. In some cases, the proportionate weighting was not exact due to the real world practicalities of field research. For the male heads, between 14.8% and 16.7% of the total number of household heads were sampled. The exceptions were Lesiday (18.8%) and Lemooti (31.4%) sub-villages, which had small total population sizes, and thus had higher recruitment percentages. Given the limited number of female household heads that fit the inclusion criteria, the recruitment percentages for female heads were significantly higher and more variable, typically ranging from 40% to 80% with some exceptions. Loosikitok sub-village had 0 female household heads that fit the inclusion criteria, and Lesiday, Kanisani, and Lengijape had 100% recruitment rates owing to their small total numbers of female household heads.

6.5 Results

6.5.1 Demographics of Respondents

The majority of respondents were either WaArusha (61.1%), or Kisongo (26.4%). The remaining were Nyaturu (2.2%), Iraqw (2.4%), Mrangi (2.1%), Nyiramba (1.6%), Chagga (1.5%), Kamba (0.9%), Sandawe (0.7%), Pare (0.4%), Meru (0.3%), Sukuma (0.2%), Mfyomi (0.2%), and Gogo (0.2%). Age categories were determined based on the Kisongo age-set system (see **McCabe and Woodhouse** Chap. 4), which is used by both Kisongo and WaArusha. Non-Maasai respondents were assigned an equivalent age number and grouped into the respective category. Korianga (ages 28–44) was the dominant age group at 40.9%, followed by Landiis (ages 45–55) at 31%, Makaa (ages 56–70) at 11.8%, Seuri (ages 70–85) at 8.1%, Nyangulu (ages 18-28) at 5.6%, and Nyangusi (age 85+) at 2.7%. The majority of respondents had attended primary school (62.1%), while 31% had no education. A small minority (5%) had been to secondary school (Form 4 or 6), and 1% had attended university. The vast majority of respondents (84.7%) reported mixed livestock production and agriculture as their primary source of income and a few derived their income from business ventures (1.8%). Some exclusively farmed (11.9%) and a small number only kept livestock (1.6%). The primary crops grown in the area

Table 6.2 Livestock assets and land holdings (in acres) of survey respondents in Randilen Wildlife Management Area

	Cattle	Donkeys	Goats/Sheep	Chicken	TLU	Farm size
Mean ± SD	15 ± 22.9	1.7 ± 2.3	34.18 ± 44.6	15.2 ± 18.5	15 ± 20.3	12.98 ± 16.9

Tropical livestock units (TLU) were calculated following Jahnke (1982, 10) and Mkonyi et al. (2017, 252). The following TLU conversion factors were used: Cattle = 0.7, sheep and goats = 0.1, donkey = 0.5, chicken = 0.01

were maize and beans, followed by peas. A small number of cultivators grew sunflowers, fruit, and other vegetables. Table 6.2 presents a basic overview of livestock assets and land holdings of respondents.

6.5.2 General Attitudes Towards the WMA

Findings from the first two questions served as a barometer of general community sentiment towards the WMA. These questions revealed that the majority of respondents had positive attitudes towards the WMA (Fig. 6.2). In total, 36.4% of respondents liked the WMA, and 38.5% strongly liked it (74.9% with positive attitudes; Fig. 6.2a). This finding is encouraging, as it suggests that from the perspectives of local communities, the WMA is well received. This was further elucidated by respondents' expressed levels of support for the WMA: 64.7% of respondents supported the WMA and 10.9% strongly supported it (Fig. 6.2b). These responses indicate that local communities are no longer opposed to Randilen WMA, and are instead generally appreciative of its presence in their lives.

6.5.3 Change in Attitudes Over the Past Five Years

People's memories of the past are sometimes unreliable. Anthropological scholarship shows that people generally reconstruct narratives of past events in ways that are congruent with current understandings, and in ways that help derive meaning from experience (Garro and Mattingly 2000). Nonetheless, respondents were asked to recall their feelings towards the WMA five years ago (Fig. 6.3a), and the extent to which their attitude had changed since then (Fig. 6.3b). Most people reported feeling neutral towards the WMA five years ago (52.2%), and stated that their attitude had become more positive towards the WMA over the past five years (58.6%). Forty percent reported liking the WMA more now, while 18.6% reported liking it much more now than then. While these self-reported recollections of change in sentiment towards the WMA are not particularly marked, they do suggest increasing positivity towards the WMA.

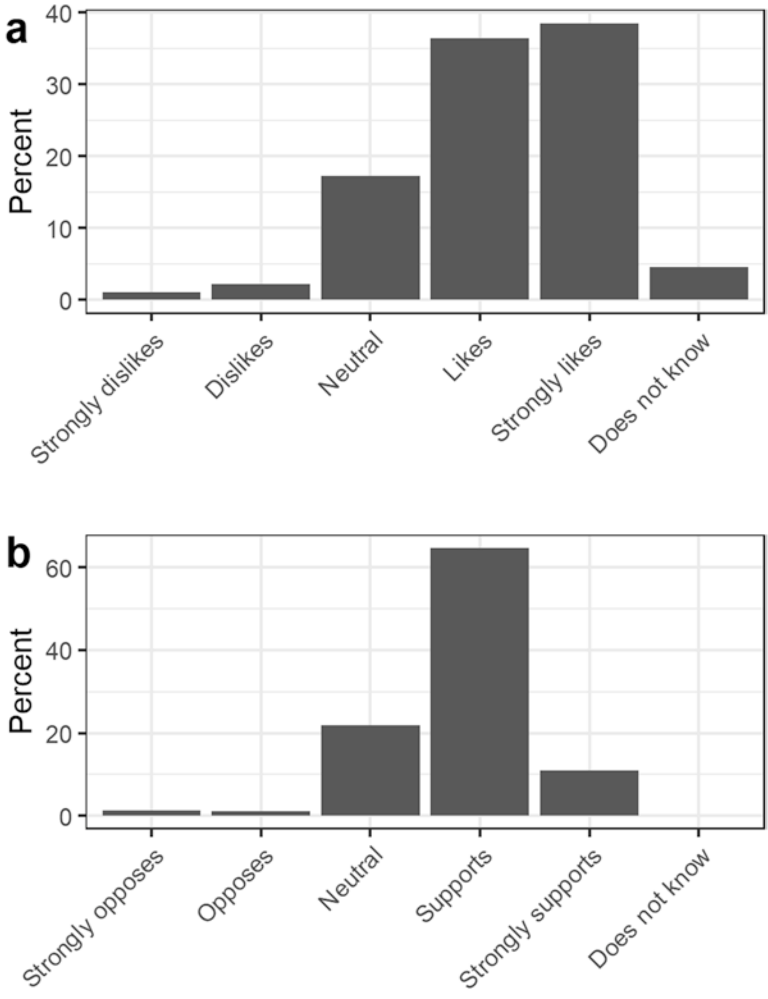


Fig. 6.2 Community attitudes towards Randilen WMA based on surveys administered in April – July 2020 to a proportionately weighted sample of 678 respondents from all 8 member villages, using sub-villages as sampling strata (26 sub-villages); (**a**) shows general sentiment towards the WMA based on response percentages; (**b**) illustrates respondents' stated level of support for the WMA

6.5.4 Lived Experiences of WMA Governance and Management

As a dimension of governance, respondents were asked the degree to which they trusted WMA authorities to act in their community's interests (Fig. 6.4a). Rather strikingly, 87.6% reported that they trusted WMA authorities to act in their interests.

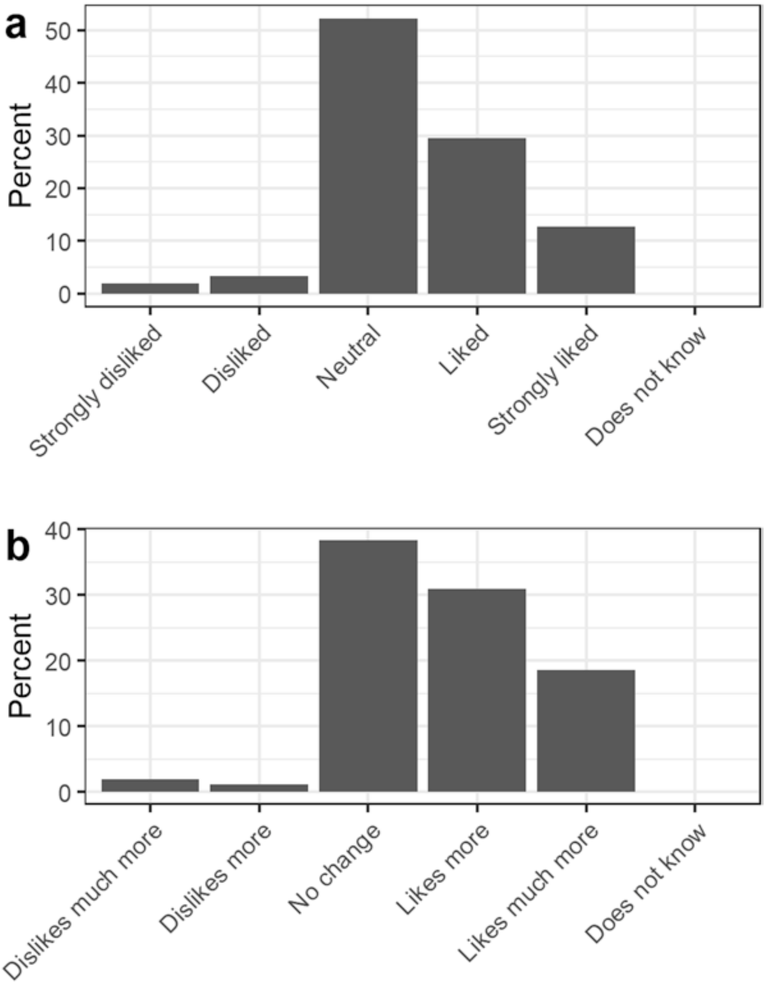


Fig. 6.3 Change in community attitudes towards Randilen WMA over past five years based on surveys administered in April – July 2020 to a proportionately weighted sample of 678 respondents from all 8 member villages, using sub-villages as sampling strata (26 sub-villages); (**a**) displays general sentiment towards the WMA five years ago based on response percentages; (**b**) shows respondents' current sentiment compared to five years ago

Perhaps most significantly, 92% of respondents felt that their community was included in WMA governance (Fig. 6.4b), and 91.4% thought that their community was included in WMA management (Fig. 6.4c). When asked about conservation trade-offs, and people's perceptions of the distributions of costs and benefits associated with the WMA, the majority of respondents reported that Randilen WMA had more benefits than costs (75.4%) (Fig. 6.5).

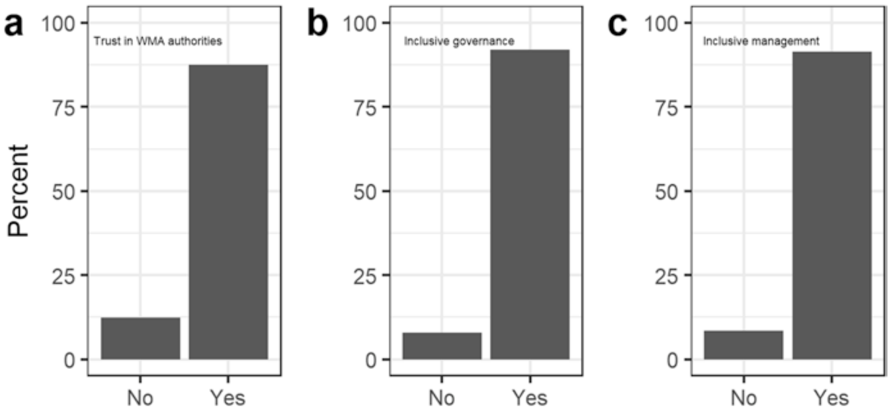
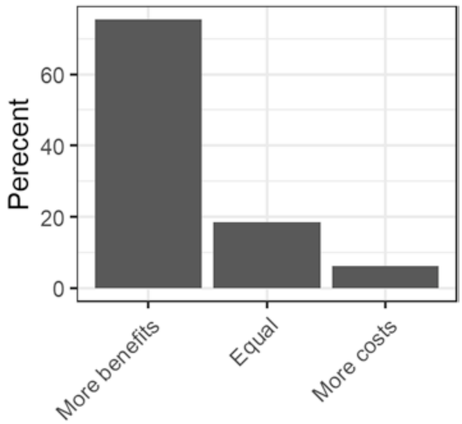


Fig. 6.4 Community perceptions of WMA governance and management measures based on surveys administered in April – July 2020 to a proportionately weighted sample of 678 respondents from all 8 member villages, using sub-villages as sampling strata (26 sub-villages). The graphs show percentages of responses to the questions: (**a**) "Do you trust WMA authorities to act in your community's interest?" (**b**) "Do you think people from your community are included in WMA governance (decision-making processes)?" (**c**) "Do you think people from your community are included in WMA management (enforcement of rules and regulations)?"

Fig. 6.5 Community perceptions of Randilen WMA costs and benefits based on surveys administered in April – July 2020 to a proportionately weighted sample of 678 respondents from all 8 member villages, using sub-villages as sampling strata (26 sub-villages). Equal refers to a perceived equal distribution of WMA costs and benefits



6.5.5 *Interpretations of the WMA as a Success or Failure*

Respondents were asked to report whether they viewed the WMA as a success or a failure, and whether they felt that the WMA represented a top-down strategy for securing resource control at the expense of local communities, or whether it constituted a community-based conservation area that distributed benefits to community

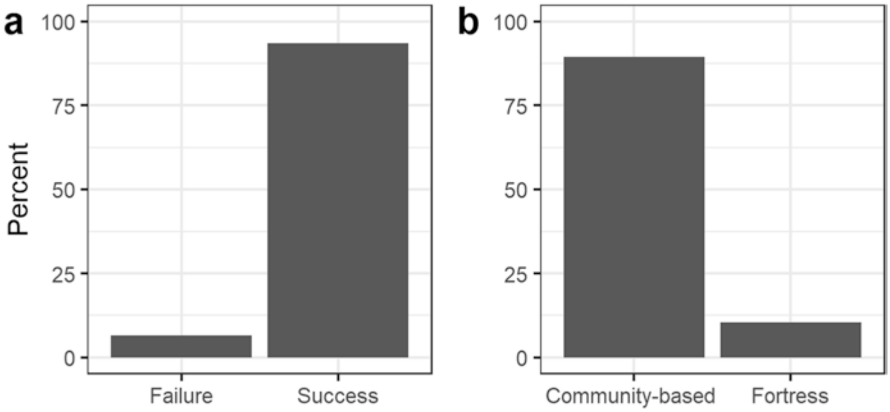


Fig. 6.6 Community evaluations of Randilen WMA as socially successful or exclusionary based on surveys administered in April – July 2020 to a proportionately weighted sample of 678 respondents from all 8 member villages, using sub-villages as sampling strata (26 sub-villages); (**a**) shows the percentage of survey respondents who considered the WMA to be a failure or success; (**b**) displays the percentage of survey respondents who interpreted the WMA as a community-based or fortress conservation model

members (Fig. 6.6). Results showed 93.5% viewed it as a success, rather than a failure (Fig. 6.6a), and that 89.5% of respondents felt that Randilen WMA represents a community-based conservation area, rather than a fortress model (Fig. 6.6b).

6.6 Discussion and Conclusion

The data in this chapter have provided a cross-section overview of community attitudes towards Randilen WMA. By taking community attitudes as the research lens, the study has provided key insights that bear on the future of community-based conservation in the Tarangire Ecosystem. The survey results suggest that over the past five years, there has been a change in how community members view the WMA since Loveless (2014) first documented community-level discontent during the WMA planning process. Villagers have since come to appreciate Randilen WMA and the benefits it brings. This important finding flies in the face of much of the existing literature on the social impacts of WMAs, in that it reflects lived experiences of inclusion and participation, rather than marginalization (Homewood et al. 2020; Kaswamila 2012; Keane et al. 2020).

It is still important to bear in mind the fact that WMAs are not always synonymous with community-based conservation (Igoe and Croucher 2007). Indeed, the wider literature on community relations with WMAs in Tanzania speaks to this important consideration (Francis 2019; Bluwstein et al. 2016; Kicheleri et al. 2018b, 2021). In many cases, there does appear to be valid reason for social scientists to be critical of WMAs, which may undermine the resource rights of local communities.

From a conservation standpoint, exclusionary models of governance coupled with strong-handed management can breed discontent from local communities and incite resistance (Raycraft 2019, 2020; Bennett and Dearden 2014). This may take the form of noncompliance and even disregard for environmental regulations, potentially undermining the central aims of conservation (Western 1994; Holmes 2007; Hoffman 2014).

The story that I have started to unfold in this chapter, however, is that communities can also embrace WMAs and come to view them as centrally important to their livelihoods. Ethnographic factors specific to each case likely play a crucial role including, but not limited to, ethnicity, history, livelihoods, cultural beliefs, land use practices, inter-village dynamics, community-investor relations, ecology, seasonality, and so on. In short, WMAs should not be implemented uncritically as 'community-based' conservation interventions given their troubled histories and potential to generate conflict among local stakeholders (Kicheleri et al. 2018a; Kicheleri 2018; Green and Adams 2015; Moyo et al. 2017). At the same time, it is equally important to note that WMAs can also come to represent community-based forms of conservation if the communities view them as valuable. When asked specifically about Brehony et al.'s (2018) description of Randilen WMA as fortress conservation (*Uhifadhi wa ngome/haijumuishi jamii*), the vast majority of respondents disagreed with this label, preferring to classify it as a community-based conservation area that is implemented in a way that benefits local communities (*uhifadhi wa msingi wa jamii ambao unajumuisha jamii*). Presenting this finding is not meant to be combative, but rather to show that communities themselves have come to view Randilen WMA as community-based. This should be celebrated.

Looking to the future, equity must continue to be a central component of any community-based conservation initiative in the Tarangire Ecosystem, where the migratory routes of wildlife intersect village land. In the case of Randilen WMA, positive attitudes towards the WMA at the community level suggest that conservation is being implemented in an equitable fashion. Based on these quantitative findings, I have forwarded an alternative anthropological perspective on WMAs in this chapter, suggesting that while they are often characterized by conflict, they can also show promise as institutional mechanisms for securing wildlife habitat outside of national parks and garnering support for conservation from resource-dependent rural communities.

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Part III
The Wildlife Dimension

Chapter 7
The Ecohistory of Tanzania's Northern Rift Valley – Can One Establish an Objective Baseline as an Endpoint for Ecosystem Restoration?

Herbert H. T. Prins and Joost F. de Jong

Abstract Often conservationists suffer from the 'shifting base line syndrome'. We illustrate this by elucidating the natural history of Tanzania's northern Rift Valley over the past centuries. White rhinoceros and possibly the sable antelope went extinct five centuries ago. Two centuries ago Maasai cattle started competing with plains wildlife, but a reset took place through diseases. Wildlife's zenith was around 1935, before commercial agriculture arose and before people and livestock had recovered from devastating epidemics. Elephant populations recovered from the ivory trade; wildlife benefitted from the expanding range of the tsetse fly. From the 1920s until the 1980s, cattle numbers soared and most fresh water became monopolized by farmers or pastoralists. Unlike in the Serengeti grasslands, the great herbivore migrations that could have developed after the rinderpest eradication were not attained in the grasslands of the northern Rift Valley: in fact, the wildebeest and zebra migrations to a large extent disappeared. It appears that conservationists who have fallen victim to the shifting baseline syndrome are content with the current impoverished natural state. Consequently, with the memory gone and baselines shifted, it is likely that the true natural state of the ecosystem of the northern Rift Valley will not be restored.

Keywords White rhinoceros *Ceratotherium simum* · Political ecology · Ecosystem restoration · Conservation success · Shifting baseline · Maasai/Masai

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7.1 Introduction

Pauly (1995) called attention that each generation of fisheries scientists accepts as a baseline the stock size and species composition that occurred at the beginning of their careers, and uses these to evaluate changes. Pauly called it a ‘shifting baseline syndrome’, because as a result lower targets for ‘normality’ are consistently set. This shifting baseline syndrome bedevils ecologists and conservationists because they suffer from a paucity of early studies to compare the present state of an ecosystem to earlier, less impacted states (*cf.* Knowlton and Jackson 2008) and hence they cannot know how (a) natural state(s) of an ecosystem should or could look like.

Pauly’s observation explains the worrying accommodation of conservationists to the present states of ecosystems that undergo and have undergone human impacts (e.g., Prins 1992; Prins et al. submitted; *pace* Rohde and Hilhorst 2001). If one would desire to restore areas, a fundamental problem is then knowing the appropriate baseline for restoration or for rewilding attempts (Jepson 2016). Often, it is difficult to establish which species have been lost (Venter et al. 2014); even more arduous is to assess how numerous populations were in the past. Yet, the desired endpoint for restoration critically hinges on knowing both the original (i.e., prior to significant human impact) flora and fauna plus their densities so that the desired trophic structure can be determined (Sinclair et al. 2018).

In the African conservation context, most attention is given to the battle to prevent the (local) extinction of species, but much less to the restoration of lost splendour. Partly, that is because too many ecologists apparently do not recognize what has been lost already – not only in areas subject to agricultural conversion, but also in the protected areas. For many years, researchers warned that because parks were becoming “islands in a sea of cultivation” they lose species (*pace* Prugh et al. 2008). History speaks for itself: the reality of eastern and southern African parks is a loss of mammal species during the last century (Prins and Olff 1998). Often, historical sources are insufficient or biased (e.g., Venter et al. 2014). Reconstructing population sizes is even more operose but also more perilous than determining which species were lost. A case in point is the Greenland – Spitsbergen Bowhead whale (*Balaena mysticetus*) population that currently stands at a few hundred individuals but was historically about 50,000 animals (Hacquebord and Leinenga 1994; Allen and Keay 2006). We do not know with much precision how many American bison (*Bos bison*) there were prior to their wholesale slaughter first by Native Americans and then by European settlers, but they were more numerous than perhaps could be imagined today. This shifting baseline syndrome demonstrates the importance of long time series of population counts (e.g., Prins and Douglas-Hamilton 1990; Kiffner et al. 2017; Dornelas et al. 2018). For the northern Gregory Rift of Tanzania (hereafter “Tanzania’s Rift Valley”), we can, however, guestimate what was lost because of the high quality of the historical sources.

The natural history of wildlife of Tanzania’s Rift Valley is entangled in the history of the local inhabitants and the western colonizers. For millennia this area has been inhabited by groups of people of various ethnicities and by cultural entities.

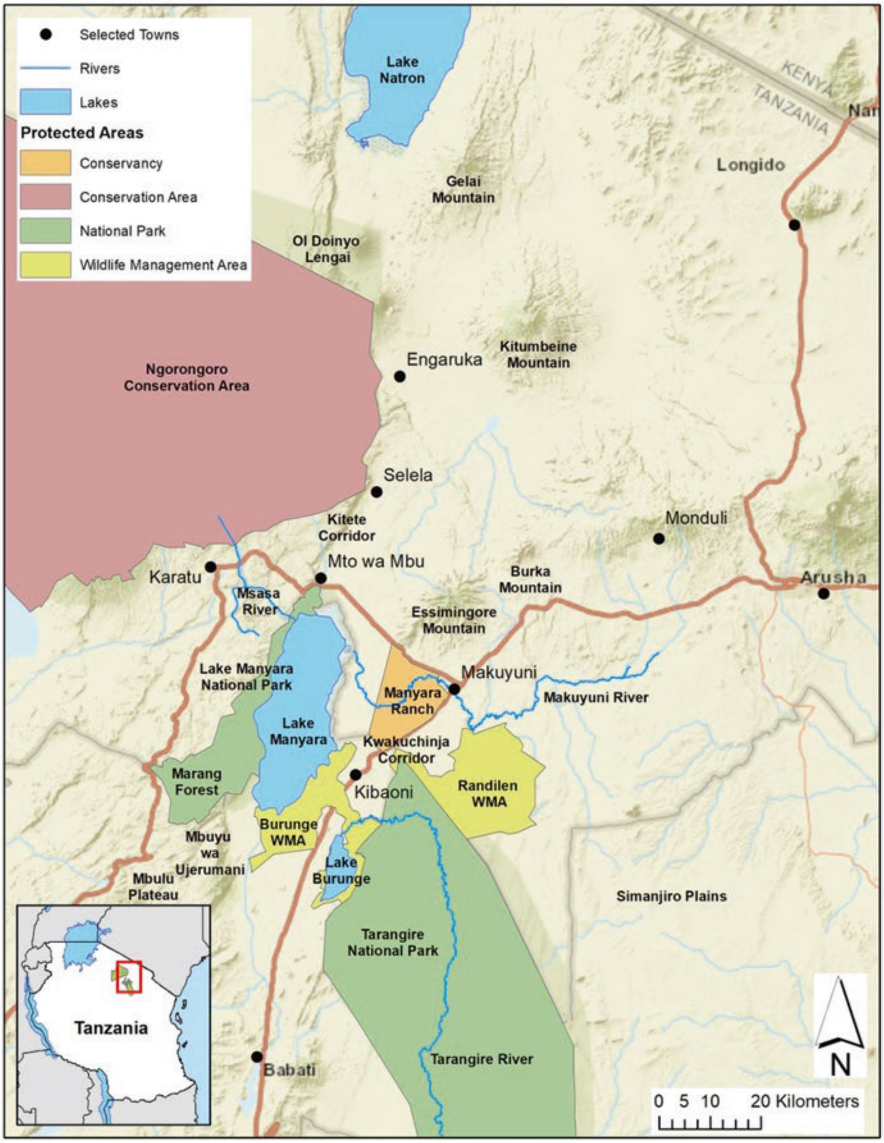

Fig. 7.1 Map of Tanzania's northern Rift Valley with locations of names mentioned in the text. (Map created by Jason Riggio)

Here, we reconstructed the natural history of wildlife of Tanzania's Rift Valley, which includes the Tarangire Ecosystem that is the focus of this book (Fig. 7.1), and discuss current conservation efforts and prospects.

In so doing, we (i) show how drastically species, communities and population abundances can change over the course of few human generations; (ii) show how complex the history of wildlife populations is, even in the last few centuries; (iii) illustrate how sparsely knowledge that exceeds two generations is passed on, and hence how easily the previous state (or even a natural state) of an area is forgotten; (iv) illustrate how cumbersome it is to reconstruct the previous state; (v) discuss how the aforementioned points lead to shifting baselines among conservationists; and (vi) argue that the amalgamation of shifting baselines with a well-intended appreciation of the needs of local people leads to celebration of an impoverished natural state as a conservation success.

7.2 The End of Prehistory in the Rift Valley

What did Tanzania's northern Rift Valley look like before settlement by modern pastoralists and agriculturalists? Rock art and archaeology provide a clue. An insightful archaeological site is Engaruka (Fig. 7.1) (and an unnamed site closer to Mto wa Mbu [Gillman 1944] which has not been investigated yet). At Engaruka, irrigated agriculture was carried out until about 1670 CE at a (for Tanzania and Kenya) unprecedented scale of 20 km^2, but then decreasing water yields from the Mbulu Plateau diminished agricultural output. Extensive dry-stone walling was employed to create channels and fields (Sutton 1984, 1990). For its demise, failing agriculture at that time can be ruled out (Lang and Stump 2017), so perhaps it was warfare with the very first of the nomadic pastoralists (possibly Maasai) coming into the Rift Valley that led to its final abandonment (Sutton 1984, 1998). This large settlement of perhaps 5000 people (Sutton 1984) likely had a negative impact on especially black rhinoceros (*Diceros bicornis*) and African elephant (*Loxodonta africana*) in the area of the Escarpment (note that in northern Tanzania, the Rift Valley is bounded by one escarpment, at the West only).

Approximately contemporary with Engaruka culture is the unique collection of rock art from Kondoa and Singida (at the southern edge of the area under scrutiny in this chapter) represented by some 500 sites with about 5000 paintings. This naturalistic art tradition stretched towards Manyara and Arusha (Bwasiri and Smith 2015). The most common depicted animals were cattle (*Bos* spp.), giraffe (*Giraffa camelopardalis*), greater kudu (*Tragelaphus strepsiceros*), common eland (*Taurotragus oryx*), and 'indeterminate antelope'; less so African elephant, black rhinoceros, plains zebra (*Equus quagga* a.k.a. *E. burchellii*), blue wildebeest (*Connochaetes taurinus*), African buffalo (*Syncerus caffer*), hartebeest (kongoni; *Alcelaphus buselaphus*), waterbuck (*Kobus ellipsiprymnus*), reedbuck (*Redunca arundinum* or *R. redunca*), roan (*Hippotragus equinus*), sable (*H. niger*), and Suids; further felids, hyena (apparently *Crocuta crocuta*), baboon (*Papio cynocephalus*) and ostrich (*Struthio camelus*) (Masao 1976; Bwasiri and Smith 2015). Probably, they were made by local hunter-gatherers (like today's Hadza or Sandawe: Bwasiri and Smith 2015). In those recorded by Leaky (1983), it appears that not only black but also white rhinoceroses (*Ceratotherium simum*) were depicted. White rhinoceroses were recorded in recent fossil sites of northern Tanzania (Geraads 2010), suggesting that this grazer had rather recently been lost from this ecosystem. Its extinction may have coincided with the expansion of Bantu-speaking people into the areas east of the River Nile about 2000 years ago (Moodley et al. 2018). It also matches the arrival of pastoralism in the Rift Valley (Prins 2000 and references therein) and the establishment of the current climate in this region. Indeed, white rhinoceros remains have been found near Lake Nakuru in Kenya just north of the Tanzania's Rift Valley (Gifford-Gonzalez 1998). The Grévy's zebra (*Equus grevyii*) had disappeared from northern Tanzanian grasslands earlier (Faith et al. 2013) even though one vagrant individual was seen in the Rift Valley between Manyara and Essimingore in 1960 (Warden's Reports 1960). Sable (also depicted in the rock art) nowadays occur further south although the habitat at some places in the northern Rift Valley still may be suitable (Boitani et al. 1999, pp. A-956 ff.).

Of course, East Africa being the cradle of mankind, hunter-gatherers had been present in the watered parts of northern Tanzania for a very long time. The last stage was a microlithic culture lasting some 40,000 years (see, e.g., Diez-Martín et al. 2009). Microlithic tools were used by hunter-gatherers but, later, also by pastoralists (Goldstein and Shaffer 2017). Before the eighteenth century, the foot of the escarpment was settled by the "il Datwa lol Orokishu" (likely Iraqw), the "il Datwa lol Kuroto" (likely Mbugwe) and the Sonjo (Fosbrooke 1948). They were agropastoralists practicing some irrigation. Around 1850, Maasai invaded the Crater Highlands and eliminated the Barabaig from the Ngorongoro Crater (Fosbrooke 1972). Around 1806, the Maasai defeated an ethnicity named "Il-Adoru" in the Rift Valley; 'Manyara' was probably an "Il-Adoru" name Fosbrooke (1948). Local oral history also recalls that ancestors of the Iraqw lived on the northern shores of Lake Manyara in the early 1800s. Their leader was Moya (a son of Chief Tipe), after whom Manyara was named. An Iraqw elder in 1982 equated those "Il-Adoru" with the "Hhay Lori", a clan of the Iraqw, now living around Mbulu town (Mr. Tseama Pissa pers. comm.), as eponymy of Lori, who was the brother of Chief Tipe.

Fosbrooke (1972) suggests that these "Il-Adoru" were either Datooga or Barabaig (note that the "Hhay Jorojik" – one of the clans of the Iraqw – is an incorporated group of Barabaig). Chief Tipe's people at the time lived in the Rift Valley, from Magara to Kondoa (pers. comm. Messrs. Tseama Pissa and Sjabaan Swalleh 1984; for caveat see Rekdal 1998). According to them, the Mbugwe (a Bantu-speaking tribe) came from Kisange and pushed northward. Hence these Iraqw settled on the plateau to the west of the escarpment above Magara (where many lived already before; see Widgren and Sutton 2004) (Fig. 7.1). The lineage of these chiefs is Tipe – Jandu – Bea – Banga – Isara – Nade – Shauri. Shauri was chief from 1961 to 1964 and then became the local Chairman of the ruling political party. Assuming an average generation duration of 25–30 years, Tipe became chief sometime between 1774 and 1809, agreeing with Fosbrooke's (1948) reconstruction of the 1806 Maasai victory. The oldest stand of *Vachellia* (formerly *Acacia*) *tortilis* in the area dates from about 1780 (Prins and Van der Jeugd 1993). These trees may have germinated when the Iraqw withdrew their livestock from the Rift Valley floor. Knowledge about a changing species composition or numbers of large herbivores is lost: conclusions about a shifting baseline (apart from the local extinction of Grévy's zebra, white rhinoceros and possibly sable in addition to some modification of the landscape by cattle and small stock) cannot be drawn from prior to 1800 yet.

In the arch between Lake Manyara and Mount Monduli, another pastoral ethnicity was supplanted by Masai coming from the north at the beginning of the nineteenth century (Fosbrooke 1948). It appears Maasai did not penetrate much further south than the present-day boundary between the Acacia steppe and the Miombo woodlands. Those who penetrated further south were recalled by their "spirito-political leader" (the Laibon) (Fosbrooke 1948). This other agropastoral ethnicity may have been the Kavim or Lumbwa; yet it is also possible that they were a section of the Maasai that had been defeated in one of the internecine wars of the early nineteenth century (see Fosbrooke 1948). The name "Lumbwa" is perhaps merely a derogative name for people of Maasai affiliation who practice agriculture (Barton 1923) and thus not an ethnicity. They dug deep wells (at the southern end of the Simanjiro Plains), smelted iron and made pottery. As compared with earlier times, it is now much more appreciated that pastoralism cannot develop without barter with agriculturalists who produce surplus. We thus would not be surprised if the "purely pastoral lifestyle" of the nineteenth century Maasai developed because agriculturalists increased their production, perhaps stemming from the adoption of maize from the New World. Indeed, before the building of railroads (1893 in Tanganyika; 1896 in Kenya) and roads for motor cars (between 1890 and 1910: for cars in Tanganyika see Grace 2013), caravans of (male and female) porters brought maize from the coast, and it became a staple only around 1880 in the interior of East Africa (Miracle 1965).

Little is known about the impact on wildlife populations of feeding the caravans carrying ivory and transporting slaves from the interior to the coast. Fouquer (1966) estimates that annually 50,000 wild animals had to be shot to provide food for the caravans that carried the ivory for *just the London market alone* (Fouquer 1966). He quotes 500,000 porters every year passing through Tabora (Fouquer 1966). On average, a porter carried 25 kg (Beachey 1967). Considering that other important markets were Antwerp and Bombay but also towns in China and Japan, the slaughter of wildlife must have been enormous. Therefore, the second half of the nineteenth century experienced appreciable reductions in wildlife numbers in the Rift Valley. While published knowledge of the interior of East Africa was still scant before the explorers of the 1880s (see Goodrich 1849; in contrast to West Africa: Levtzion and Hopkins 2000), Arab traders went far into the interior, and often stayed there for years on one trading-cum-slave buying expedition (Osgood 1854). Already then it was predicted that the elephant would go extinct from indiscriminate killing (op. cit. pp. 55). Some information was published about the abundance of wildlife species, e.g., African buffalo (Goodrich 1875, pp. 490). When the elephant numbers diminished at the coast, the first ivory traders went into Masailand in 1840s; Maasai at that time were heavily engaged in the trade with about 1000–1500 tusks a year (Beachey 1967). The export though Zanzibar was ~220 tons in 1859 (reflecting c. 4000 shot animals), and stayed at about that level until 1890 after which it declined to about 65 tons (reflecting c. 1500 shot animals) per year (Beachey 1967). The killing moved farther and farther into the interior (see Steinhart 2000), leaving depleted African elephant populations behind. This must have happened in Masailand too.

7.3 The Great Rinderpest of 1887, Smallpox of 1889 Followed by Cholera

With outside traders, colonizers and invading armies opening the interior, the Rift Valley became exposed to virulent diseases from overseas. The outbreak of the rinderpest epidemic (killing specifically ruminants) was associated with a particular lunar eclipse thus allowing it to be dated to 1887. The disease arrived with a British invasion army from India (Prins and Van der Jeugd 1993; Roeder et al. 2013) or an Italian invasion army (Spinage 2017). It nearly coincided with or was followed by a smallpox epidemic that killed numerous Maasai (Paterson 1909; Marieni Ole Kertella in Hanley 1971). Smallpox epidemics had been occurring regularly (Imperato and Imperato 2014; *cf*. Marieni Ole Kertella *ibid*.). The early nineteenth century form was mild, but aggressive strains were imported through Muscat in 1857 (Issa 2006). The smallpox epidemic of 1890 raged at least as far as Uganda (Peters 1891). Also a virulent cholera strain, which came from India through the dhow trading network and pilgrimage to Mecca, spread along the caravan networks (see Christie 1876; see more on the caravan network of the Arabs for instance Peters 1891, pp. 364; Unangst 2015; Beachey 1967). The northern Rift Valley, however, lay far to the north of the major trade link (between Bagamoyo and Tabora and later Ujiji: *ibid*. pp. 52 ff.) and it took years before cholera reached northern Tanzania in 1870; people died within hours after the first symptoms developed (Issa 2006). The extent of human mortality and resulting depopulation of northern Tanzania will never be known, but must have been vast (see Kjekshus 1977).

The famine of the 1890s decimated the Maasai. Some sections could survive because they turned to (irrigated) agriculture, e.g., the Engaruku section of the Maasai (Fosbrooke 1972). Peters (1891, pp. 143 ff) still encountered them "insolently" in central Kenya, as did Thomson a bit earlier (according to Peters 1891, pp. 222) and in December 1889 they still had large herds of cattle (Peters 1891, pp. 229 ff.). However, when Thomson reached Engaruka, he described the same skull-littered landscape in 1895 (cited in Mack 1970) as Merker encountered in 1895 (1910, pp. 348). Baumann (1894) reckoned that two-thirds of all Maasai died, while Unangst (2015, pp. 40) even reckons three-fourths. Many survivors were sold into slavery by neighbouring ethnicities (Merker 1910, pp. 349). Then internecine war between two factions of the Maasai broke out. It is not unlikely that Waller (1978, pp. 77, 1990, pp. 93) correctly concluded that the Masailand was mostly depopulated at the beginning of the twentieth century. Although tragic from a human perspective, this must have had a positive effect on wildlife recovery after the rinderpest and on the spread of the tsetse fly (*Glossinia* spp.) from the pockets of no-man's land described below. The Rift Valley of 1900 was not a 'Paradise Lost' but more a charnel house for man and beast.

7.4 Recovery After the Devastations of the Nineteenth Century

Many wild animal species had become rare at the end of the nineteenth century. The Germans were the first to set up game laws and reserves in East Africa in 1896, limiting the shooting of elephants to bulls carrying sufficiently large tusks (Beachey 1967). Such regulations are typically promulgated when huntable wildlife populations ("game") had declined. During the very early 1900s, the Rift Valley experienced favourable conditions. The valley was free of epidemics, there were no droughts, the tsetse fly had not expanded its range yet, wildlife protection laws came into force, and low densities of humans and their livestock allowed for fast recovery of wildlife. Empty lands soon became filled with wildlife, showing power of population growth.

At the same time however, livestock recovered as well. Patterson (1909, pp. 179) described "*where ten years ago only a very few cattle, sheep and goats were to be seen, now there are thousands*". Of course such a fast rebound is not possible through natural fecundity and undoubtedly livestock raiding played a role too. Yet it is important to realize that the void of the Rift Valley grasslands was more-or-less simultaneously filled by the expansion of wild and domestic herbivores: if the growth of both would go unabated, competition would set in.

There is a hint in the older literature that zebra (being non-ruminants and unaffected by the rinderpest) may have been released from competition during those post-rinderpest years, because Patterson (1909, pp. 168) stated that zebra "*are, alas, now looked upon as little better than vermin and … sportsmen are permitted to shoot them by the score*". According to reports by hunters, black rhinoceroses were very common all over East Africa (e.g., Harrison 1901; Roosevelt 1909; Stigand 1909; Radclyffe Dugmore 1910; Meintertzhagen 1957). Stigand (1909) did not even consider rhinoceros a suitable species for "gentlemen" to hunt: they were too docile and he compared it to as if one were shooting a cow! Or as Ahlefeldt Bille (1948, pp. 246) observed "*they are all too easy a mark … and about as exciting to shoot as the lock of a barn door*". There had been a good market for rhinoceros horn (for the manufacturing of cups and boxes) and rhinoceros skins; the latter was made into shields for Bedouin warriors because these were "impervious to the stroke of a sword" (Osgood 1854, pp. 179). The Baluchi soldiers of the Sultan of Zanzibar also used these shields (pers. comm. A.H.J. Prins; HP found one in 1982 in Mto wa Mbu).

African buffalo started rebounding in the early years of the 1900s, even though their behaviour was nearly exclusively nocturnal at the time (Radclyffe Dugmore 1910, pp. 14). Common eland also rebounded (Radclyffe Dugmore 1925, pp. 257). Other antelope species had recovered even faster (Patterson 1909; Stigand 1909). Johnson (1928, pp. 267) describes the Serengeti: "*tens of thousands of* [wildebeest]; *Thompson's gazelle more numerous than I had ever seen them before; hundreds of Grant's gazelle* [Nanger grantii], *topi* [Damaliscus lunatus] *and kongoni* [= hartebeest]; *ostriches*; *innumerable zebra …*". By the end of the 1930s, many of the wild herbivores were not considered to be threatened with extinction anymore. For the

Convention for the Protection of the Fauna and Flora of Africa (promulgated in 1936: Anon. 1936) some species were considered to be very vulnerable (Class A). Impala (*Aepyceros melampus*), Hunter's antelope (*Beatragus hunterii*), greater kudu and lesser kudu (*Tragelaphus imberbis*) were (still) considered in need of some protection (Class C) but buffalo, plains zebra, white-eared kob (*Kobus kob leucot*is), lechwe (*K. leche*), puku (*K. vardoni*) and indeed white-bearded (= blue) wildebeest were not (CCTA 1953, pp. 43 ff, pp. 128 ff). We think it is safe to deduce from this that these species had recovered well after the rinderpest devastation. In the first decades of the twentieth century, the recovering wildlife populations must have been free from pastoral competition (Prins 1992; Voeten and Prins 1999) because Maasai and livestock numbers were so depressed.

However, Maasai also recovered in numbers starting in the first decades of the twentieth century. Prior to the First World War, the Germans considered them so marginal, that they set aside a tribal reserve for them only to the south of the Moshi-Arusha-Mbugwe line (Fosbrooke 1948). The British removed them from much of their previously occupied lands in the Kenyan Rift Valley far to the north where Samburu now live, far to the northeast where now is Amboseli NP, and from the Ngong Hills into the "Masai Reserve". According to the Treaty between the Maasai and the British government of Kenya Colony (see for the Treaty Text Appendix 2 in Hanley 1971 or the Agreement of 1904 and Appendix 3 for the Agreement of 1911), the Maasai did this "voluntarily" (but see Hughes 2003). They were settled to the north of the border between Deutsch-Ostafrika and Kenya Colony, to the north of the Ngorongoro Highlands and Loliondo (so, some 500 km to the south from where they came from), resulting in much suffering (Hughes 2003); this area was then named the "Masai Reserve" in which people with other ethnicities were not allowed to settle or graze their livestock. Yet, tsetse encroached on this Maasai Reserve too making a sizable part of the Reserve inhospitable to the livestock of the Maasai but offering very good land for wildlife (Ahlefeldt Bille 1948, pp. 70, 104). After the First World War, when Deutsch-Ostafrika was taken over by the British, there was Maasai resettlement across the border but also within present-day Tanzania, and people left their "*native reserves*" (see Fosbrooke 1948). For instance, the Purko section of the Maasai, which at the end of the nineteenth century herded their livestock in the Lake Baringo area, now live between the Serengeti NP and the Crater Highlands (Homewood and Rodgers 1991, pp. 46) (again, some 500 km south from where they lived a few generations ago). They had penetrated even further south into the Ngorongoro area but were removed by the British in the 1920s again (Fosbrooke 1948). At the end of the 1930s, the Monduli, Longido and Ngorongoro areas became practically vacant, after which the Moibo sub-section of the Kisongo proceeded to occupy them (Fosbrooke 1948; *pace* Homewood and Rodgers 1991 (pp. 44 ff)).

It is likely that the Kisongo section of the Maasai, which now claims the lands between Lake Natron, Monduli, the Pare Mountains all the way south to the Nguru Mountains and from there back to Lake Manyara (Homewood and Rodgers 1991, pp. 47) did not have a continuous presence in that area since the end of the nineteenth century (*pace* Homewood and Rodgers 1991) but moved into a vacuum that had been created with the (near) disappearance of another group of Maasai; they may have persisted along the Pangani River further to the east (see Fosbrooke 1948). It is thus not inconceivable that the "unknown pastoral people" that had occupied large parts of the present-day Rift Valley (Fosbrooke 1948) were simply Maasai from prior to the rinderpest, the small-pox epidemic, cholera and the Maasai internecine war.

The cattle numbers of the Maasai could recover fast because of cattle raiding from Iraqw agropastoralists on, for instance, the Mbulu Plateau (Fig. 7.1). Apparently, these people were less severely hit by the rinderpest and subsequent epidemics than the Maasai. Indeed, when the Roman Catholic 'Society of the Missionaries of Africa' started their post in about 1906, there were about 200,000 Iraqw and some 350,000 cattle (Fouquer 1955; before that time next to nothing was known about this area: Rekdal 1998). Maasai raided to a great extent (pers. comm. Messrs. Sjabaan Swalleh and Tseama Pissa) into the 1980s (pers. comm. Mrs. Margaret Gibb; the 1st author encountered a raiding *moran* party armed with spears in 1983 outside of Karatu; *cf*. Marieni Ole Kertella in Hanley 1971, pp. 286) and may even have continued until the 2010s (pers. comm. C. Kiffner).

For wildlife, the recovery must have depended on their natural rate of increase. In the absence of large predators, one would expect for a Thomson's gazelle (*Eudorcas thomsonii*) a rate of increase of some 50% per year, but for an African buffalo only 8% per year (based on the relationship between body mass and intrinsic rate of increase for mammalian herbivores). Supposing that the rinderpest in 1889 had caused a mortality of 90%, then it would take only 12 years for the Thomson's gazelle to have recovered but about 30 years for the buffalo. Recovery must have been even slower because predators were not susceptible to the rinderpest (Stevenson-Hamilton 1974). It is likely that by 1930 wildlife had recovered well, thus explaining the lack of concern by the delegates of the Convention for the Protection of the Fauna and Flora of Africa in 1953 (NCCA 1953) referred to above. Indeed, on a single hunting permit one could at the time in Tanganyika shoot an unlimited number of lions (*Panthera leo*), leopards (*P. pardus*), hyenas (*C. crocuta* and *Hyaena hyaena*) or zebras. For other species one needed itemized hunting permits (Johnson 1928, pp. 16; pers. comm. Mr. Adam Seif). Arbuthnot (1954, pp. 51) counted wildlife when he was between Lake Manyara and Mt. Essimingore (on a hill close to Makuyuni – perhaps where Manyara Ranch is presently) in 1927: "*On this particular morning, we differed in our guesses by several thousand* [after counting in clusters of 50 animals]*, for animals were grazing over the veldt in all directions. We stood for a long time. Figuratively knocked off our feet in amazement of the staggering amount of game, perhaps hundreds of record specimens that were grazing somewhere on this limitless plain*". Our experience with counting yields an inter-observer difference of some 15%, which in Arbuthnot's words is "*several thousand*". In other words, Arbuthnot and his party are likely to have counted some 40,000 grazing animals in 1927 at this single place.

We posit that the best benchmark for setting a 'baseline' to compare with the present-day wildlife would be 1935. Since that year the ever-growing human population and its associated livestock started negatively impacting wildlife populations, as did the ever-increasing hunting pressure. Later, the War Effort did much damage to wildlife and wilderness too; the War Effort was a concerted action by the Administration of Tanganyika to increase local food production during the Second

World War. Talbot and Talbot (1963) stated that already in the early 1960s the Rift Valley was severely modified since the 1910s through overgrazing and desiccation, and that movements of wild animals were getting severely restricted. They also stated that the area to the southeast of Lake Natron, which had been a game reserve in the German days, was by the early 1960s overutilized by Maasai. Grassland was converted into thorny bush. The overstocking by Maasai cattle was surmised to have led to differential mortality in wildebeest, resulting in a sex ratio of 1 wildebeest male per 2 females and a low calf survival at the end of the 1950s (Talbot and Talbot 1963). The zenith of the Rift Valley's wildlife was definitely over in the 1960s.

7.5 Tribal No-Man's Land and Sleeping Sickness

The term "*tribal no-man's land*" is an important, yet little known concept. These lands frequently formed the nuclei on which during the colonial era forest and game reserves could 'crystallize'. These are literal no-man's lands, on the border between the military, or cultural or economic sphere of influence of some society or other (e.g., Gupta 1971; Coates 2014). Now and then, a raiding party may have entered but cultivation did not take place, and rarely livestock herding. Frequently, memories of the former existence of these tribal no-man's lands have gone with increased pacification. However, at the beginning of the previous century, many of these tribal no-man's lands were still very present in the landscape and important to understanding the spread of trypanosomiasis by tsetse flies that lived in these no-man's lands but not elsewhere (see Ford 1971; Kjekshus 1977).

In the Rift Valley, two of these tribal no-man's lands are known to us from oral history. The first lay to the east-southeast of Lake Manyara, from the Rift Valley escarpment to approximately Lake Burungi. To the north roamed Maasai, and to the south Mbugwe. The Mbugwe (a Bantu-speaking group) were no match for the warmongering Maasai but were renowned for their effective witchcraft and throwing of spells (pers. comm. Mhoja Burengo; see also Gray 1963; Mesaki 1995). Maasai are not known for using witchcraft but force; for more on their culture see Merker (1910) and Spear and Waller (1993). The tribal no-man's land was penetrated sometimes when Maasai went cattle raiding but there were neither Maasai *manyattas* (semi-permanent multi-family fenced traditional Maasai homesteads) nor Mbugwe huts in the area. This tribal no-man's land extended further east from Lake Burungi into the direction of the Simanjiro plains. The old caravan route towards Mbuyu wa Ujeromani ("the German baobab") on the Rift Wall ran through this no-man's land (Wakefield and Johnston 1870).

Another old trade route from the coast to the interior ran to the north of Lake Manyara (as reported by Wakefield and Johnston 1870; see J.L. Krapf's 1849 map: Beard 1988) and was pointed out to the 1st author in 1982 as running up the escarpment near to Msasa River. Both routes were frequented by "Arabs" from the coast. Their previous campsites are marked by *Tamarindus indica* trees (pers. comm. Mohenjo Burengo; Sjabaan Swalleh). Possibly, large baobab (*Adansonia digitata*) trees are indicative of Arab staging sites like they are on the coast (pers. comm.

A.H.J. Prins) but suggesting that they would be indicative of ancient settlements in Tarangire is without base for the interior (*pace* Årlin 2011, pp. 63). This trade route passed through the second tribal no-man's land, which was between the plateau on top of the escarpment and Lake Manyara itself. To the north and east of the lake lived pastoral Maasai, but on the plateau above lived Iraqw (a.k.a Mbulu) people. Old Iraqw men told us how they as young men would hunt black rhinoceros in what is now Lake Manyara National Park (further abbreviated as LMNP). A group of about 20 men would descend the escarpment, and set themselves up in a double picket line at two sides of a game trail leading upslope. The swiftest runner would stalk a black rhinoceros and entice the rhinoceros to chase him up the slope towards his mates. The rhinoceros would then be stabbed from both sides (pers. comm. Sjabaan Swalleh). The meat of the calves was especially delicious (pers. comm. Tseama Pissa). Further to the south, Barabaig would sometimes descend into the area at the foot of the escarpment (pers. comm. Tseama Pissa). There was no recall of any Maasai *manyatta* in the present day LMNP to the south of where now is Mto wa Mbu. A group of Maasai elders in 1982, who were then about 60 years old, recalled hunting for small animals and birds to the north of Ndala River but no *manyattas*. They also recalled women collecting raffia palm (*Phoenix reclinata*) fonds and palm sap for making wine but no herding.

A third tribal no-man's land may have been between the Pangani river, Naberera and Tarangire. This is the area that we believe Hemmingway (1936) hunted in (Fig. 7.2).

Fig. 7.2 Ernest Hemmingway with a buffalo shot in 1933. (Ernest Hemingway Photograph Collection/John F. Kennedy Presidential Library: public domain)

These tribal no-man's lands became foci for tsetse fly infestation and sometimes sleeping sickness and trypanosomiases (Ford 1971). Tsetse then kept livestock out, and so did sleeping sickness: trypanosomiases (a.k.a. nagana) prevented ox ploughing, or donkey populations to build up and sleeping sickness reduced labour available for hoeing. Details are in the chapter of **Bluwstein** (Chap. 2). Nagana killed for instance about 800,000 pack animals during World War I in Tanganyika (Ahlefeldt Bille 1948, pp. 145). To the north of Lake Manyara, tsetse flies made their appearance sometime at the beginning of the twentieth century. The Maasai who had by now many thousands of cattle there lost more and more grazing opportunities "*due to the insidious spread of the tsetse fly. No less than three species of this malevolent insect now* [1940] *occupy land which was formerly the home of thousands of head of cattle, and today only three Masai bomas remain. Unless the fly can be eradicated it seems likely that once again 4/5th of Lake Manyara will be tenanted only by waterfowl and big game*" (Watermeyer and Elliott 1943). Near Mto wa Mbu the fly was observed in 1921 (Radclyffe Dugmore 1925, pp. 30). Also in the Mbugwe area to the south of Lake Burungi, sleeping sickness arose, after tsetse moved into the area around 1925 (Årlin 2011). From 1942 onward there were outbreaks of sleeping sickness, and the government initiated clearing campaigns and forced people to move (Årlin 2011). In 1948, the Mbugwe were even reported to be dying out (Harris 1951) perhaps from sleeping sickness. In 1966, most of this resettlement out of the area was fulfilled (Årlin 2011, pp. 97). Sleeping sickness foci proved to be persistent in the 1950s (Glover 1967). The disease is often fatal (Baldry 1972), and new medicine has not been developed (Brun et al. 2010). It still has not been eradicated (Jelinek et al. 2002) despite attempts by Tanzania National Parks (Muse et al. 2015) The human trypanosomiasis was not found to the West of Tarangire anymore (Salekwa et al. 2014).

7.6 Establishing the Baseline for Lake Manyara National Park

In the northern part of Tanzania's Rift Valley (Prins 1987), there are presently two protected areas, namely LMNP and Tarangire National Park (hereafter TNP). IUCN (2021) defines protected areas as "*Large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area*", and gives as two of the objectives (i) "*to manage the area in order to perpetuate, in as natural a state as possible, representative examples of physiographic regions, biotic communities, genetic resources and unimpaired natural processes*", and (ii) "*to maintain viable and ecologically functional populations and assemblages of native species at densities sufficient to conserve ecosystem integrity and resilience in the long term*". Here the baseline discussion, as flagged by Pauly (1995), becomes pertinent. How close is LMNP still to its baseline? The IUCN definition suggests that one ought to take as baseline the natural state of an ecosystem, or some state closely resembling it. However, can one know such a state in the East African environment where people have been operating since the dawn of time?

Actually, LMNP is at present far from "the natural state" and perhaps there is even more than one "state". When Otto Baumann traversed what is now LMNP in 1892, his map was so reliable that we could locate the three mapped hot springs, although only two were known by the park management in 1981. Intriguingly, while Baumann was charged to make an inventory of marketable products and timber, he failed to signpost the groundwater forest directly to the south of Mto wa Mbu despite taking the caravan route up the escarpment towards his discovery of the Ngorongoro Crater a few weeks later (Watermeyer and Elliott 1943). Presently, the forest holds good stands of *Celtis africana* (white stinkwood) much of which was felled for timber in the 1950s (see Loth and Prins 1986) as was *Milicia (Chlorophora) excelsa* (African teak; Greenway and Vesey-Fitzgerald 1969). Baumann came from the southern end of the lake through Mbugwe land fully occupied by people then (Watermeyer and Elliott 1943). Yet, Baumann mapped the present groundwater forest as "umbrella acacias". We offer three explanations for this discrepancy. First, he suffered from malaria those weeks (*dixit* his diary). Second, he was distracted by raiding Maasai (Watermeyer and Elliott 1943). Or third, there was indeed no forest. Perhaps some tectonic movement brought water-bearing layers to the surface, thus irrigating a few square kilometres and creating a new groundwater forest. Actually, the underground hydrology of this area is quite complicated (Loth and Prins 1986), and sudden cracks in the rift valley floor arise occasionally (pers. obs.). This option is supported by the description of "*abundant evidence of sudden and local terrestrial movement*" (Watermeyer and Elliott 1943). Forest in this wider area may have been lacking in temporal permanence (Butynski and De Jong 2020). The finding that Lake Manyara's gentle monkeys (blue monkey: *Cercopithecus mitis*) are morphologically undistinguishable from those of the Crater Highlands to the west and quite distinct from those further east (Butynski and De Jong 2020) provides more evidence that the forest may be young and that the monkeys expanded their range recently.

Unfortunately, Baumann (1894) did not give much information on the large mammal fauna. Importantly, though, he refers to many black rhinoceroses in the area but not to African elephant. Yet, at the end of the 1920s, elephants were present in the forest, where it was then too dangerous to hunt them (pers. comm. Adam Seif, who had been professional hunter's guide before WWII; Arbuthnot 1954). When the British started governing Tanganyika as mandated territory, they had closed hunting to foreigners at least until 1921 (Radclyffe Dugmore 1921). When the area became a Game Controlled Area in 1930, wildlife was so plentiful in what is thus presently LMNP, but also in other GCAs in northern Tanganyika, that one only needed a license for shooting lion, black rhinoceros, leopard, buffalo and elephant, but all other game was free till 1945 (pers. comm. A. Seif). Drinking water was plentiful, and rivers carried crystal clear water all year round while there were no signs of erosion (pers. com. Adam Seif; Radclyffe Dugmore 1925). This is in stark contrast with the situation since the 1980s, thus proving Rohde and Hilhorst (2001) erroneous: photographs from 50 years ago are not sufficient to prove that there was little environmental change.

The reduced discharge of river water into LMNP as a result of increasing human pressure on the Mbulu Plateau may have decreased the grass production in the park too. For instance, the *Sporobolus consimilis* vegetation of the 1960s (Greenway and

Vesey-Fitzgerald 1969) had nearly completely disappeared in 1982 (Loth and Prins 1986). Likewise *Odyssea jaegeri* had almost vanished between these two surveys, also indicative of less flooding, and *Chloris gayana* was much less prevalent. Even the papyrus swamp, which had been so important for common reedbuck (*R. arundinum*) in the past, was negatively affected (Warden's Reports January 1961). This was confirmed by the Maasai elders in 1982 referred to above.

The area became a Game Reserve in 1955 even though the Colonial Administration had considered turning it into agriculture; indeed, the area to the south of the Endabash River had been a ranch (until about 1960), and further south there had been coffee and papaya plantations (pers. comm. HRH Prince Bernhard, Mr. E.H. van Eeghen; who were landowners until about 1970). The proclamation of Game Reserve was done very carefully (pers. comm. Sir Hugh Elliott, former Permanent Secretary of Tanganyika Territory; Warden's Reports 1958). It comprised an inventory of usage and native rights, then an assessment of the annual value which was then multiplied by ten, a series of community meetings to find agreement with the local community, and thence a plan to pay out the capitalized value of those rights after which the extinguishment of native rights was promulgated. This was also done with the community of Mto wa Mbu, which, in exchange for this extinguishment, received a dispensary and a school. In 1960, the Game Reserve became a national park. Yet, the area of the park had lost large mammalian species between the 1930s and 1970s, and the vegetation changed. The protected area then extended further to the northeast then presently, all along the northern shore of the Lake. There, in 1959 and again in March 1963, a herd of 30 fringe-eared oryx (*Oryx beisa*) was seen. Black rhinoceroses were still common. In September 1959, blue wildebeest walked through the lake and were seen in their thousands. In November 1959 there were 3500, in July 1960 their number swelled to some 4500 and even up to 7000 were counted (Warden's Reports: July and September 1961; October 1962). Between the lake and Mt. Essimingore there were still numerous Grant's gazelle and Thomson's gazelle. A group of 150 elephants moved down from the Marang Forest on the Mbulu Plateau in March 1960, indicating there was still a seasonal movement of elephants at the time (Warden's Report April 1961).

In the 1960s, African wild dogs (*Lycaon pictus*) in LMNP and cheetahs (*Acinonyx jubatus*) were reported in LMNP. Hartebeest and eland moved down from the Karatu area (June 1961; August 1964) and also zebra still occurred on the Mbulu plateau (October 1964). Thomson's gazelle could be numerous (1000 counted in February 1962), and numerous zebra, eland and hartebeest were reported from the area immediately to the north of the Park (June 1965). In the 1930s, large groups of eland frequented what is now LMNP, and lesser kudu still occurred here (pers. comm. Mr. Adam Seif). This shows how LMNP was linked to the area South of the Ngorongoro Crater which was still full with wildlife even though agriculture was expanding fast since the War Effort and subsequent *ujamaa* (i.e., the cooperative economics that resulted in 'socialization' of agriculture in the 1970s and 1980s).

In 1970, the elephant population of LMNP still could range from the Mbulu Highlands above the escarpment, move up and down to the Marang Forest and range to the Magara Farms to the south of the park; however, they were believed to have been compressed later (Lake Manyara Elephant Meeting 1970) and it was concluded that LMNP was too small to be ecologically viable for elephants unless it was managed. From Iain Douglas-Hamilton's (1972) dissertation research, it is clear that by then the elephants had become resident in LMNP. At the time, it was concluded that the high elephant density (some 5 per km^2) was the result of "*rapid contraction of the elephants' range over the preceding 50 years in competition with human settlement*" (Douglas-Hamilton 1973). With hindsight, we do not think there is any evidence for that conclusion. Indeed, the high density was maintained and even further increased in the years until poaching started in 1983 (Prins and Douglas-Hamilton 1990). We think the correct conclusion is that LMNP became a new ecological optimum for elephants, and because of the then-raging debate about overpopulation and overbrowsing by elephants as result of 'the Tsavo Drought', Douglas-Hamilton was coerced into this conclusion at the time (as stated by Owen 1970). Watson and Turner (1965) similarly referred to the compression of wildlife, especially of elephant and buffalo, but also without evidence and warned of the animals exceeding the 'carrying capacity' (*the* issue of those days) of LMNP. A thorough survey led to the conclusion that actually the park was in a healthy state and that there was no concern for overgrazing or overbrowsing (Vesey-FitzGerald 1973). The Warden's Reports of the 1960s did not provide any evidence of this 'compression' and animals moved in and out freely. Mwalyosi (1990) concluded that even if the compression idea held, it was of no serious consequence for the trees because the regeneration was high enough, which was confirmed by Prins and Van der Jeugd (1992, 1993).

In the early 1980s, elephants still moved up and down the escarpment to forage at night on the plateau (pers. obs.; Kalemera 1987) and movements (also of buffalo and lion) were still taking place via Mbulumbulu to and from the Crater Highlands (Prins 1987). Even though the Upper Kitete – Selela Corridor (now about 1 km wide only) is encroached upon and shrinking, it is still used by elephant and buffalo (Mangewa et al. 2009). Yet, reaching LMNP from the Crater Highlands gets increasingly difficult through the expanding town of Mto wa Mbu. Caro et al.'s (2009) assessment that this corridor was critically threatened must be right, but the predicted disappearance by 2018 has not come true (yet). Lohay et al. (2020) found that the Rift wall has also had a negligible influence on genetic differentiation of elephants between Lake Manyara and the Ngorongoro Conservation Area, indicating they are not only moving between these areas but likely still interbreeding as well.

Buffalo from LMNP still went to Mt. Essimingore (and were driven back) (Warden's Report March 1960) and then moved to Tarangire (*ibid*. May 1960) but then returned (*ibid*. July 1960). That interchange came to a stop somewhere in the 1960s (Prins 1996, pp. 72 ff.).

The sheer existence of LMNP was challenged from the first days. The northeast corner of LMNP was excised and handed over to Maasai in 1960; later, in the 1980s, it was converted to irrigated agriculture (financed by the World Bank). When a severe drought hit, the District Officer of Monduli asked for access to LMNP for Maasai to graze their cattle (Warden's Reports January 1961): access was denied. Intriguingly, poaching of black rhinoceros was substantial at the time (Warden's Reports 1960–1961). A senior ranger (Mr. Mhoja Burengo, pers. comm 1982) told us that this was mainly for meat, which was distributed amongst the villagers. In later years, when horn harvesting became the sole reason for poaching of black rhinoceroses, only the kidneys were collected for consumption. All spearman were Iraqw from the Mbulu District. At the end of the 1960s, the total number of black rhinoceroses in LMNP had decreased from about a hundred to about 50 (Vesey-FitzGerald 1973) or fewer (Prins and Douglas-Hamilton 1990 and references therein). Black rhinoceros went extinct in 1986 in LMNP.

However, not only the black rhinoceros went extinct in LMNP: common eland, hartebeest, oryx, and Grant's gazelle are gone, and there is no substantial lesser kudu population anymore. It appears that the mountain reedbuck and common reedbuck are functionally extinct too while they were very common in the 1980s. It is likely that bat-eared fox (*Otocyon megalotis*) are extinct. In the early 1980s it still denned there and raised pups (pers. obs.) but a year-long camera-trapping survey in 2016 did not detect them (Steinbeiser et al. 2019). Perhaps one may add the African golden cat (*Caracal aurata*) to this list of species that went extinct. Of this species very little is known, and recently it has not been camera-trapped in LMNP; it was observed by the first author in 1983 on the lower edge of the Marang Forest (grey phase). Local farmers along the edge of the Ngorongoro forest near Karatu also identified it after viewing pictures, not confusing it with African wild cat (*Felis sylvestris*). The elusive African golden cat has quite a wide distribution in similar habitats in Kenya (Butynski et al. 2012).

The important message is that in LMNP all these species went locally or at least functionally extinct; but also that the vegetation changed, permanent rivers became ephemeral, and water quality decreased. The annual migration of blue wildebeest stopped in effect and the population became mostly resident the resident population still may still have some genetic exchange (Morrison and Bolger 2014); Grant's gazelles and oryx do not annually visit the park anymore; there is no more buffalo movement between Manyara and Tarangire and finally the elephants became to all extent resident. In other words, 70 years of preservation could not prevent or halt pervasive changes to species composition and ecosystem functioning, and LMNP is now far removed from its 1935 baseline.

Objections against these stark conclusions may stem from the fact that vagrants are still observed now and then, reminding one of the once-extensive movements of individuals. As such they actually serve as an archive of conservation failure. Yet at the same time, these vagrants may enable the revival of historical migratory routes (*cf.* Mooij et al. 2008) because if this knowledge is lost such revival may take perhaps a hundred-odd years (*cf.* Jesmer et al. 2018; Merkle et al. 2019).

7.7 The Shifting Baseline of Tarangire National Park

The other national park in the ecosystem is Tarangire National Park (TNP: ~2600 km^2). It is about twenty times larger than LMNP (of which the present-day Rift Valley bottomland is ~168 km^2). Did this park change as dramatically as

LMNP? It is worth quoting Vesey-FitzGerald (1972) at length "*Tarangire was a game reserve until recently, 1969. It has always been a dry season concentration area for animals moving in from a wide area in Masailand. Increasingly so in recent years the harassment of animals in Masailand has been mechanised; yearly wild fires from all directions sweep across the park. The existing situation is therefore one in which there is too much grass (of the wrong sort at the wrong time) for too few animals.....* [thus maintaining] *fire subclimax grasslands …* [I]*f the prevailing fire impact is prevented, the course of succession will proceed through wooded to a woodland or forest formation. The animal impact is seldom evident on secondary grasslands; a grazing mosaic is seldom well developed. This is because at the onset of the rains when the new grass grows and is palatable, there is likely to be too much of it for too few animals. When the grasses mature they usually become fibrous and unpalatable and so are neglected, and other parts of the range will then be frequented…. The course of succession is truncated and maintained as a grassland formation by annual dry season fires. Frequently there is no animal community available to utilize the extensive fire sub climax grasslands that have been caused by overburning*". This shows that Vesey-FitzGerald (1972) was aware of the issues at stake in Tarangire. In other words, 'compression of wildlife' would have led to different vegetation rather than the dwindling of animal populations that in reality took place. But this insight does not help in formulating the baseline.

Lamprey et al. (1962) and Lamprey (1964) arrived at a total number of large wildlife for the resident game and migratory game combined that was anchored on the Tarangire River during the dry season of only about 20,000 large mammals, most of which were wildebeest and plains zebra; he estimated the density to be 60x lower than in the Serengeti at that time. He pointed out that the water of Mt. Meru was used for irrigated agriculture and the water of Mts. Gelai and Kitumbeini (Fig. 7.1) were depleted by Maasai cattle. Near European farms, there was heavy hunting, and livestock of pastoralists competed with the wildlife. His estimate did not include the resident Kirk's dikdik (*Madoqua kirkii*), steenbok (*Raphicerus campestris*), warthog (*Phacochoerus africanus*), waterbuck, impala or lesser kudu. Lamprey (1964) observed that the northeast corner of LMNP was an important cornerstone also for Tarangire, as was confirmed by the Warden's Reports from the time for LMNP. Yet, that part of LMNP had been excised and handed over to the Maasai for grazing in 1960. It is possible that Lamprey was underestimating, because Ecosystems Ltd. (1980) estimated a total number of wild large mammals of about 120,000, with the number of species as reported in Table 7.1. These animals were using some 20,000 km^2. The total number of large mammals in the Serengeti (about equal in size) was at that time about 2.2 million (Houston 1979) (Table 7.1). Yet, even with much better aerial survey estimates, the density of large mammals in Tanzania's northern Rift Valley was about 20x lower than in the Serengeti. Perhaps the vaccination of cattle against rinderpest (thus protecting wildlife: Sinclair 1979) had had a faster effect in the Serengeti where the interface between wildlife and livestock was much more limited, because it is noticeable how relatively more depressed the wildebeest and buffalo numbers were in the northern Rift valley (Table 7.1).

Table 7.1 Large mammals of Tanzania's northern Rift Valley and the Serengeti Ecosystem (both about 20,000 km^2) at the end of the 1970s

	Inside Tarangire N.P.	North of Tarangire N.P. (Lolkisale)	East of Tarangire N.P. (Simanjiro)	Total for the northern Rift Valley	Total for the Serengeti Ecosystem	Ratio between Serengeti and Northern Rift Valley
Zebra	0	1005	30,839	31,844	240,000	7.5
Impala	6422	7869	16,445	30,736	119,100	3.9
Wildebeest	397	0	24,066	24,463	720,000	29.4
Hartebeest	1092	2242	3359	6693	20,700	3.1
Topi	0	0	0	-	55,500	
Buffalo	101	450	5477	6028	108,000	17.9
Eland	936	193	4378	5507	24,000	4.4
Thomson's gazelle	593	0	3035	3628	981,000	270.4
Grant's gazelle	936	579	1736	3251	6000	1.8
Elephant	2891	0	0	2891	4500	1.6
Giraffe	819	115	1736	2670	17,400	6.5
Warthog	1014	309	603	1926	34,200	17.8
Oryx	40	443	551	1034	n.c	
Lesser kudu	162	0	157	319	0	
Common reedbuck	0	0	157	157	n.c.	
Greater kudu	0	0	39	39	0	
Gerenuk	0	0	39	39	0	
Total	15,403	13,205	92,617	121,225	2,330,400	19.2

Topi do not occur to the east of the Rift, and oryx are very rare to the west of the Rift. *nc* not counted. Waterbuck, greater kudu and black rhinoceros were already too rare to count. The data of Tarangire and the northern Rift Valley are from EcoSystems (1980); those of the Serengeti from Houston (1979). The overall ungulate density in the northern Rift Valley was about 20 times lower than that of the Serengeti

In their much-quoted book, Homewood and Rodgers (1984) optimistically and without quoting sources stated that there are no known cases of extermination of wildlife by pastoralists, and also that there was no evidence that livestock increased in numbers to the detriment of wildlife (*pace* Prins 1992; Scholte et al. 2021). That may be true, but we do assert that the one million odd cattle in the northern Rift Valley (Msoffe et al. 2011) must consume grass that then is no longer available for wildebeest, eland or buffalo (see Voeten and Prins 1999). The cattle numbers that Msoffe et al. (2011) refer to may be too high: for the Monduli District (which covers about 80% of the ecosystem, we think) the 1984 census reports 325,000 cattle, 223,000 goats, 165,000 sheep and 21,000 donkeys (Zwart 1995, pp. 16). In other words, if Maasai population numbers had continued to be as depressed as they were in the early 1900s, the livestock numbers now would have been so low that in the grasslands of Tanzania's northern Rift Valley some one million head of wildebeest and zebra could have lived, bringing it on a par with the Serengeti. The ratio of metabolic weights of wildlife was only about 15% of that of livestock in the 1980s (Prins 1992), perhaps yielding an even higher estimate. Indeed, Homewood and Rodgers' (1984) argument that there was no sign of overstocking or land degradation is a red herring in a debate about competition between wildlife and livestock: wildlife is not outcompeted by livestock because the range is degraded – wildlife is outcompeted primarily because livestock, protected by herders and dogs, has access to water and grass and eat it.

In the very early 1970s, Tarangire was still a stronghold for black rhinoceros, with 250 reported (Borner 1981). Indeed, Jonathan Simonson (pers. comm.) recalled in 1981 that 15 years earlier one could see some fifty rhinoceroses between the entry gate and the Tarangire Safari Lodge (c. 9.5 km), but none could be seen by the 1980s. Yet, no one ever suggested that this poaching was done by Maasai. Apart from black rhinoceros, local extinctions have not been reported from TNP itself, but from the southern wildlife corridor between LMNP and TNP: eland, hartebeest, buffalo, oryx, lesser kudu, cheetah and leopard (Hassan 2007). In TNP itself, the migratory Grant's gazelle, Thomson's gazelle, zebra, eland, hartebeest and oryx are declining, and so are the resident bushbuck (*Tragelaphus scriptus*), reedbuck and waterbuck; yet the more or less resident African buffalo, African elephant, giraffe, and warthog are not (notwithstanding possible omission of individuals of the smaller species) (Stoner et al. 2007; *cf.* **Lee and Bond** Chap. 9; **Bond et al.** Chap. 8; **Foley and Foley** Chap. 10). Between the 1970s and 1990, there may have been an increase of zebra (from 16,000 to 22,000) and a decrease of wildebeest (from 14,000 to 11,500), but the other species were decreasing (Kahurananga and Silkiluwasha 1997). At the end of the 1990s, these large numbers of wildlife had declined to one to two thousand zebra, and a similar number of wildebeest, plus a couple of hundred buffalo, elephants and eland (Msoffe et al. 2007). Yet, they still refer to this ecosystem as "… *among the richest areas in East Africa regarding wildlife diversity and abundance, hosting large populations of wild herbivores including the largest population of elephants. During the dry season, huge herds of migratory species, mainly elephants, buffalo, wildebeest, zebras, and eland migrate to the permanent waters of the Tarangire River*" demonstrating the shifting baseline syndrome in all its glory. Nelson (2012) even suggests that the high migratory wildlife numbers (which in reality are a shadow of what they were and even more so of what they could have attained) are the positive effect of pastoralist grazing and burning practices (see **Bluwstein** Chap. 2; **Brehony et al.** Chap. 5).

7.8 The Best Benchmark for Nature Conservation in the Rift Valley Is 1935

Perceiving the afore-described complex history, what then should be considered to be the 'natural state' of Tanzania's Rift Valley? Around 1935, wildlife had recovered from the onslaughts of the rinderpest, and from the devastating impacts of feeding the trading caravans. The Maasai had not recovered fully yet from the smallpox and cholera of the 1890s. But the build-up of their numbers started and was augmented by the forced settlement of the Purko Maasai from the Baringo area. Around 1920, the settlement of Mto wa Mbu began (Watermeyer and Elliott 1943). White farmers started carving out their farms in well-watered places on Mt. Oldeani, along the southern Ngorongoro Crater, towards Mt. Essimingore and from there to Mts. Monduli and Meru (Fig. 7.1). With World War II, many of these farms were taken over by the Custodian of Enemy Property (e.g., Fuggles-Couchman 1944; Redfearn and Fuggles-Couchman 1945) through "war legislation in the colonial empire" (Dale 1940; yet Tanganyika was not a part of the colonies). In the Karatu area, large-scale wheat farming took place while wildlife was eradicated through shooting (pers. comm. Mr. Sjabaan Swalleh). Even though the war-time managers did extensive contouring of the farms, erosion started leading to clogged-up river courses (pers. comm. Mr. Sjabaan Swalleh, Mrs. Margaret Gibb). Indeed, the escarpment of the Rift Valley, once covered in heavy forest before the road was built up from Mto wa Mbu in 1933 (Radclyffe Dugmore 1925; Watermeyer and Elliott 1943; pers. comm. Adam Seif) became increasingly denuded of woody cover (pers. obs.). Many rivulets reported by Watermeyer and Elliott (1943) or Harris (1951) had dried up in the 1980s (pers. obs.). The village of Mto wa Mbu continued to increase in size, and many trees of the groundwater forest were ringbarked for charcoaling (Warden's Reports August 1961).

Yet, in contrast to Kenya, we have not been able to find published accounts of wholesale slaughter of wildlife in northern Tanzania. It may have been as bad as in Kenya, where Ahlefeldt Bille (1948) describes "*killing for fun had been carried out along the [asphalt] roads to a horrifying and almost unbelievable extent" … "[I]n the [Naivasha] Valley bottom … just like the Athi Plains … everything had been shot down during the war. Many districts all over Kenya had been used as training-camps and rifle-ranges for the armed forces, and thousands of head of game were killed to provide leather and biltong, the need for which was understandable*" … "*The South African troops in particular were the chief offenders. They could not bear to see a living animal anywhere without letting off any weapon handy, from pistol or rifle to machine gun*"… "*There were cases where exercises were held out on the plains with tanks and machine-guns, with zebras, wildebeeste, and gazelles playing part of the 'enemy'- the only difference being that the corpses would rot and the wounded were left to the hyaenas and vultures. Herds of giraffe and elephants were machine-gunned from the air. Comment is superfluous – one can only state the facts with a painful feeling of shame for one's own race*" (*op. cit.* pp. 49; pp. 71). Just to the North of Lake Natron, in Kenya, for the sake of creating grazing lands for

Maasai five thousand zebra and the same number of wildebeest were killed on contract (op. cit. pp. 218). Elsewhere, Boer settlers were actively eradicating game (op. cit. pp. 77), and large numbers of wildlife were killed for tsetse control. About one thousand black rhinoceroses (and other animals) were killed in what is now Makueni County (near Machakos, Kenya) "*just to procure land for the Wakamba – who always have lived in the same area on the most easy terms with the rhinoceros*" (*op. cit*. pp. 201–2). One must realize, we think, that in many countries during many times, armies feed off the land. In Tanzania, soldiers in the army camp near Monduli and the one on the southern slope of Mt. Essimingore were provided with zebra and antelope meat by the truck load at least in the 1980s and early 1990s (pers. obs. first author). Perhaps we will never know what the impact of this has been on the wild herbivores of the grasslands of the northern Rift Valley of Tanzania.

At the end of the 1930s, some mammal species were already declining in the northern Rift Valley of Tanzania, namely black rhinoceros, hippopotamus (*Hippopotamus amphibius*) and possibly common eland and African buffalo. This was caused in LMNP by hunting and the increasing occupancy of their habitat by cultivators from Mto wa Mbu. The area was still rich in buffalo, common reedbuck, warthogs, baboons, and vervet monkeys. Wildebeest ranged in their scores of thousands; giraffe, oryx, Grant's gazelles, and Thomson's gazelles in their hundreds on the plains at the northeast corner of the lake. Elephants were not permanent residents (yet). They used the Mto wa Mbu area in the wet season and ranged to the uplands (i.e., what is now Karatu District and the forests east of Ngorongoro Crater) in the dry season (Watermeyer and Elliott 1943). We are not aware of resettlement by elephants after the massive hunting for ivory at the end of the nineteenth century of the forest relics on Mts. Essimingore, Burka, Monduli, Gelai, and Kitumbeine (Fig. 7.1) during the 1930s or later. Yet, black rhinoceroses occurred here in the 1950s (Warden's Reports 1959, 1960).

At the end of the 1970s, no Thomson's gazelles and few Grant's gazelles had been observed (Ecosystems Ltd. 1980) and the very large herds of eland ("as numerous as cattle" pers. com. Mhoja Burengo; the same had been observed by Radclyffe Dugmore 1925) on the slopes of Essimingore and in the northeast corner of the lake were gone (Ecosystems Ltd. 1980). In 1980, no wildebeest were observed by EcoSystems to the north of the lake anymore, where the Warden's Reports (January 1961) had mentioned 3000 at that corner which grew in numbers to 7000 in September 1961. In 1983, the number of wildebeest were still estimated to be 43,000 in the Rift Valley but in 2001 only 5000 were counted by TAWIRI (Msoffe et al. 2011). This was either an undercount, or the figure of some 12,000 for 2011 (Morrison et al. 2016) was an overcount (confidence intervals are large with their type of surveys). The area to the north of TNP and LMNP became clogged up with agriculture; in 1984 there was 170 km^2 of agriculture in the Tarangire Ecosystem and in 2000 this had increased to 881 km^2 (Msoffe et al. 2011). Mwalyosi (1992) already warned that this unplanned agriculture negatively impacts the resources for cattle. Anyhow, we would not be amazed if the few animals that survived became resident in LMNP instead of migratory as they had been before (Warden's Report April 1961) even though "a few are remaining" in the Game Controlled Area in between the two national parks (Lee 2018) and connectivity is severely compromised (Morrison and Bolger 2014). The erstwhile migratory population of wildebeest of LMNP became to a large extent resident, and then started increasing to about 1300 in 2011 (Morrison et al. 2016). This reminds one of the sedentary populations of the Western Corridor of Serengeti and the Ngorongoro Crater. Yet, the LMNP's wildebeest population (presently resident but 50 ears ago migratory) is much more vulnerable because the alkaline grasslands on which these animals depend (de Boer and Prins 1990) are as prone to flooding as before (Prins and Douglas-Hamilton 1990). They thus may easily go extinct like the wildebeest population in southwest Kenya (Ottichilo et al. 2000, 2001) and as they did, locally, in LMNP before but where they could recover because of good connectivity with the Rift Valley grasslands at that time (Prins and Douglas-Hamilton 1990).

So, in the smallest of the two parks, LMNP, a number of species went locally extinct: black rhinoceros, common eland, hartebeest, lesser kudu, wild dog, bat-eared fox and cheetah, Similar losses as those reported from LMNP have not been reported from the larger TNP.

In the 1970s, poaching of black rhinoceros started in Tarangire; where there had been some 250 in 1974, there were about 55 in 1977 and just 20 in 1980 (Borner 1981), after which they disappeared by 1985. In LMNP, poaching led to the same local extermination (Prins and Douglas-Hamilton 1990). The elephant numbers declined severely in the 1980s and 1990s: in Lake Manyara from about 500 to 60 (Prins et al. 1994); in Tarangire from 2334 in 1995 (Galanti et al. 2006) to 1938 in 2006 (Foley and Faust 2010). Both populations recovered to some extent (**Foley and Foley** Chap. 10), but elephants are rarely seen outside protected areas anymore (Kioko et al. 2013) while earlier at least some were migratory and many spent much of their time outside the park (Galanti et al. 2006; Pittiglio et al. 2014).

7.9 A Shifting Baseline in the Rift Valley

When Pauly (1995) called attention to the issue of shifting baselines in conservation, he offered a mirror to conservationists and went to the core of the issue: what do we want to conserve? In the Netherlands, a magistrate helped open the eyes of politicians and ornithologists to the fact that serially comparing the number of lapwings to that of the previous year, is not a good strategy for maintaining a viable population of lapwings, thus contravening the European Bird Directive (Council Directive 1979). The message is, of course, that for the lapwing population a proper baseline (although available) was ignored. In Tanzania, it apparently is hardly recognised that the shifting baseline syndrome is fully operational. LMNP and the area between LMNP and TNP lost several species. The corridor between the Ngorongoro Highlands and LMNP dwindled to a width of one kilometre. Large mammal populations of Tarangire are at approximately 20% of 40 years ago, the migration the Simanjiro Plains and the Gelai Plains is hampered, and large mammals to the north of Lake Manyara are an apparition of what they once were, with the numbers of eland, oryx, Grant's gazelle, Thomson's gazelle, wildebeest and zebra a faint facsimile of the past. And the migratory herds probably never reached their full natural potential size in the last one and half century. We think we can reconstruct the wildlife status of Tanzania's northern Rift Valley between Lake Natron, the Simanjiro Plains and the Rift Escarpment as it was in about 1935. Presently, the possibility of using fossil material to assist in such a reconstruction is quickly improving, as was shown for the Laetoli Beds north of Lake Eyasi (Louys et al. 2015) and this would bring the benchmark to even surer footing. Yet it is important to realize the speed at which large mammal species are being locally lost, as was also demonstrated for the Lake Turkana area (Prins et al. submitted) and southeast Kenya (Butynski et al. 2015). Great efforts are carried out to incorporate local people in conservation, and perhaps now there is still time to maintain, reopen and restore corridors for wildlife. Appreciation of what has been lost is a double-edged sword, because on the one hand it may decrease the danger of setting the targets for conservation ever lower. Indeed, false baselines badly compromise conservation (e.g., Ekblom 2015; Didham et al. 2020; Saunders et al. 2020 vs. Hallman et al. 2021). Yet, on the other hand this knowledge of what has been lost may lead to defeatism and under-appreciation of what has been achieved and preserved, because we may live in the best of all possible worlds.

7.10 Concluding Remarks

The ideal state of the northern Rift Valley of Tanzania for wild animals would have been a false reality, namely, (a) no pastoralists competing for grass and water with wildlife, (b) no agriculture in the Crater Highlands or on the mountains thus plenty of water in the many rivulets along the rift wall, (c) no agriculture in the Simanjiro Plains, (d) no poaching or hunting, and (e) no alien diseases. This would have been represented by a combination of pre-Engaruka culture (so a maximum number of black rhinoceros – in the order of thousands), no pastoralists (so still white rhinoceroses and perhaps some 400,000 large migratory wildlife), no modern agriculture (so another 400,000 large migratory wildlife and 2000 hippopotamuses), no hunting and poaching (so with some 30,000 African elephants). Is any of this likely? Probably not (as foreseen by Radclyffe Dugmore in 1925), unless some major human disasters would take place which no one hopes for. That is why we call this a 'false reality'.

Is it possible to expand LMNP and TNP beyond what has been achieved? Again, very unlikely. All area on the Mbulu Plateau, and the lands to the north of LMNP and to the south are now taken by agriculturalists while further east lands are needed for pastoralism. Would it be possible to rewild the swamps to the north or south of Lake Manyara and destroy a town like Mto wa Mbu to re-create dry season havens? Again, utterly unlikely. To remove agriculture from around Mt. Kitumbeini (Schüßler et al. 2018), to allow elephants moving again to Amboseli? The resettlement of people in the Loliondo area to protect the watershed for the Serengeti

(Kihwele et al. 2021)? The British and the Dutch governments removed people from the Isle of Rum and the Eiland Schokland, respectively, some 150 years ago: that still rankles with the descendants even if it was done for health and prosperity at the time. The newly established Randilen Wildlife Community Area only saw after a few years some increase of giraffe and Kirk's dikdik, but no decrease of goats and sheep and no increase in zebra (Lee and Bond 2018). Likewise, the Manyara Ranch Conservancy, a failed government ranch that had been lying idle for decades from which pastoralists were excluded but with good wildlife in the 1980s (pers. obs.; pers. comm. Mr. P. Byrne then safari operator in that area) is now a multi-use area (Kiffner et al. 2016): it has a high density of cattle, sheep and goats but no settlements (yet). The simple reality is that with a poorly developed economy, people need land for their sustenance and for their livestock. That does not call for a moral judgement beyond the moral justness of looking after kith and kin. If one wants to conserve nature for the benefit of future generations, it would be preposterous to suggest doing that to the detriment of the current one. However, three issues have to be flagged, namely, (a) the shifting baseline syndrome, (b) people of good will barking up the wrong tree, and (c) a wish of excellent ecologists to publish excellent papers in very reputable scientific journals based on sophisticated models.

The shifting baseline syndrome operates when it is has been forgotten what has been lost (Pauly 1995). Excellent cases in point are, e.g., Rohde and Hilhorst (2001) but also Årlin (2011). Equally, Lee and Bond (2018) compare the situation in 2015 with that of 2012, and not with, say, 1950 thus illustrating the shifting base-line syndrome. Likewise, Kiffner et al. (2016) use a short baseline, and the success of the multi-use of the Manyara Ranch Conservancy is measured not against a long-term baseline but against adjacent areas that are even more heavily used. The ecosystem of the northern Rift Valley has changed beyond belief, and is far removed from a desired end point for restoration (Sinclair et al. 2018). We commend people of good will and what they stand for (e.g., Lee and Bond 2018; Lee 2018; Kiffner et al. 2020a, b). Yet, we do not agree that by comparing a depauperized area outside a park with a depauperized park (resp. Manyara Ranch Conservancy and Burunge Wildlife Management Area or Randilen WMA) it can be concluded that something good was achieved. Keeping people happy and wildlife protected in the same area simply may be too big a challenge (see Igoe and Croucher 2007; Moyo et al. 2016). Finally, top-notch papers are published on ecosystem restoration (e.g., Sinclair et al. 2018) and corridor design (e.g., Bond et al. 2017). Yet, such enchanting science on its own does not result in the reversal of political or economic trends. Land hunger goes unabated in agriculture-based economies like Tanzania's with its exponential increase of human population. Indeed, in the 1940s only about 5% of all Maasai-headed households in the Arusha Region were engaged in agriculture, but this became 100% in 1990 already (McCabe et al. 2010; **McCabe and Woodhouse** Chap. 4; *cf.* Yanda and William 2010 for socio-economic drivers). Hence, we posit, the conclusion must be that nature conservation must be achieved **within** the two current national parks. One must assume that migratory wildlife will be going extinct or must stop migrating. Again, this is no moral judgement: the once vast migratory systems of North America or Central Asia and Siberia are gone too, while it is now predicted that the famous Serengeti migration system may collapse due to the offtake of nearly all water in the Mara River by Kenyan agriculturalists (Kihwele et al. 2021).

Conservation organizations and ecologists have talked too long about maintaining corridors (e.g., Borner 1985; Prins 1987; Mwalyosi 1992; Debonnet and Nindi 2017) but government fundamentally has not acted (see Riggio and Caro 2017) or was not transparent in its dealing with local people (Kicheleri et al. 2021). The Upper Kitete Wildlife Corridor (see Fig. 7.1) is now less than one kilometre wide (see above), the Kwakuchinja Wildlife Corridor is seriously encroached by farming (Martin et al. 2019), and the once open lands between the northern side of Lake Manyara and Tarangire is now a narrow corridor along Minjingu. Indeed, wildebeest do not vote but Maasai and other Tanzanian citizens do. To maintain some "rump populations" of wild large animals (see for the use of this term Dudley and Stolton 2020), some might conclude that the solution is to fence TNP and LMNP to prevent livestock coming in and wildlife going out, adding areas if and when becoming available as park (e.g., Manyara Ranch) and enabling local people to live without the burden of wildlife on their lands (Prins et al. 2021). Undeniably, this is land sparing instead of land sharing, a conclusion that was reached in, for example, South Africa long ago. The beautiful and adaptive migration strategies of large mammals will sadly disappear. So, then the hard work of restocking, predator control, disease control and vegetation management will start. Much later, perhaps, the piecemeal extension of the protected areas could continue (as in LMNP) or start (in TNP). A century of laissez faire led to extinction and ruin of a piece of paradise, but on these ruins a garden can be built. Many years from now, these could be the nuclei for rewilding.

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Chapter 8
Ungulate Populations in the Tarangire Ecosystem

Monica L. Bond, Christian Kiffner, and Derek E. Lee

Abstract Savanna ecosystems support the highest diversities of hoofed mammal (ungulate) species in the world. Ungulates provide critical ecosystem services such as nutrient cycling and redistribution and play a key role in the food web, yet many species of ungulates are in decline due to anthropogenic activities. The fragmented Tarangire Ecosystem supports at least 25 wild ungulate species, yet few studies have been conducted on population status and habitat use in this region compared to the better-known Serengeti Ecosystem. In this chapter we review and discuss historical and current research on population trends of eight commonly detected species of ungulates in the Tarangire Ecosystem, and provide recommendations for long-term conservation of these culturally, economically, and ecologically important taxa.

Keywords Hoofed mammal · Tarangire National Park · Lake Manyara National Park · Manyara Ranch · Wildlife Management Areas · Distance sampling · Aerial surveys · Population ecology

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8.1 Evolution and Diversification of Ungulates

The first ungulate ancestors appear in the fossil record about 55 million years ago in the Northern Hemisphere in North America and Eurasia after a global extinction, diversification, and dispersal event from an intense and rapid period of global warming (Gingerich 2006). Even-toed and odd-toed ungulates (Box 8.1) show parallel evolution, in that they both evolved high-crowned cheek teeth and standing on the tips of phalanges. After the Paleocene-Eocene thermal maximum, the global climate cooled. Grasslands evolved and began spreading about 26 million years ago (Strömberg 2011), and the huge variety of biomes that exist now began to differentiate during the middle Miocene, providing new ecological opportunities for ungulates to exploit (Fernández and Vrba 1999; Lorenzen et al. 2012). Ruminants spread from Eurasia and became the dominant herbivores, with craniodental adaptations for grazing (the first true grazers) appearing about 10 million years ago. Faunal communities typical of modern savanna ecosystems did not become established in East Africa until the later Miocene (~5 million years ago), through migration and in situ diversification of taxa such as elephants, giraffes, and antelopes (Leakey and Harris 2003). Africa currently supports more ungulates than any other continent (Sinclair 1983; Anderson et al. 2016).

The species richness and abundance of ungulates in East Africa is particularly high: there are more than 40 species of ungulates in the country of Tanzania alone compared to just 12 ungulate species native to the entire continent of North America. By sheer numbers, migratory species tend to dominate ungulate communities in grassland ecosystems, sometimes exceeding the abundance of resident species by an order of magnitude (Fryxell et al. 1988). For example, the number of migratory western white-bearded wildebeests (*Connochaetes taurinus mearnsi*) in the Serengeti-Ngorongoro Ecosystem tops one million individuals (Hopcraft et al. 2015). Medium-sized grazing ungulates tend to have the highest population densities in African savannas (Kiffner and Lee 2019). It may be that small ungulates are constrained by availability of quality forage, and very large ungulates are constrained by overall forage quantity, whereas medium-sized ungulates are less constrained by these requirements, thus explaining observed patterns of population densities (Kiffner and Lee 2019).

Why is there such an extraordinary diversity of ungulates, and particularly antelopes, in Africa? To start, the African continent is large and geographically diverse and is the largest tropical land mass (Mayaux et al. 2004). Furthermore, the African fauna has been repeatedly enriched by immigrations from Eurasia when land bridges connected the two continents. A massive invasion of Asian genera occurred in the early Pliocene about 5 million years ago which resulted in a major faunal revolution, and as late as the early Pleistocene about 2.8 million years ago new genera continued to appear in the fossil record due to Asian immigration and in situ evolution (Lorenzen et al. 2012). Most of the existing forms of bovids, giraffes, and zebras appeared in the late Pliocene and early Pleistocene, a couple of million years ago. Since the second half of the Pleistocene, the Sahara Desert imposed a barrier to intercontinental movement of all but the most desert-adapted forms. Most of the Eurasian tropical savanna fauna became extinct during the last Ice Age, leaving Africa as the final refuge of the Plio-Pleistocene mammals that had formerly moved between Eurasia and Africa. Speciation within Africa was enhanced by the expansion and contraction of the Equatorial Rainforest during wet (pluvial) and dry (interpluvial) periods of the Ice Ace (Dupont 2011). Finally, sub-Saharan Africa has optimal plant-available moisture levels and high soil fertility to support a relatively high diversity of large herbivores (Olff et al. 2002).

8.2 Why Are Ungulates Important?

Hoofed mammals occupy nearly every corner of the globe, from forests and grasslands to deserts and even the ocean (cetaceans are technically ungulates). Ungulates are critical in shaping and transforming the world around them: in African savannas, for example, browsers and grazers interact with fire to modify savanna ecosystems (Augustine and McNaughton 1998; Kimuyu et al. 2014). Browsers drive evolution of trees (Hanley et al. 2007; Mithöfer and Boland 2012) and influence structure and dynamics of forests (Weisberg and Bugmann 2003) and woodlands (Prins and van der Jeugd 1993; Palmer et al. 2008). Ungulates mediate key ecosystem processes such as nutrient cycles, net primary productivity, and fire regimes (Hobbs 1996).

In the Tarangire Ecosystem (TE) of northern Tanzania, species richness and population sizes of ungulates are, despite some local extinctions and population declines in the past (**Prins and de Jong** Chap. 7), still noteworthy. Ungulate populations in the TE provide both ecological and environmental services:

- Ungulates are important grazers (of grasses and herbs) and browsers (of trees and shrubs). By feeding on vegetation and processing it through their digestive systems they redistribute nutrients. Because they are important prey for a whole host of predators and scavengers, ungulates are key animals in shaping and maintaining the ecosystems where they live (McNaughton 1983; Olff and Richie 1998; Riginos and Grace 2008; Staver and Bond 2014).
- Wild ungulates increase herbaceous plant diversity (Riginos and Grace 2008), and less selective feeders like buffaloes and zebras reduce tall grasses to the heights preferred by selective feeders, including domestic livestock (Odadi et al. 2011). Wild ungulates also are the main prey for predators (Estes 1992) such as lions (*Panthera leo*), leopards (*P. pardus*), cheetahs (*Acinonyx jubatus*), and spotted hyenas (*Crocuta crocuta*), and scavengers such as vultures and marabu storks (*Leptoptilos crumenifer*).
- The region's ungulates are not only ecologically important but contribute substantially to Tanzania's economy (URT 2010). Ungulates figure prominently in photographic tourism as icons of wild nature and—for migratory species—as symbols of a nomadic existence that has been lost in much of the rest of the world. Wildlife-based tourism represents an important long-term source of income (Honey 2008; Sachedina 2008).
- Wild ungulates are often hunted by humans (mostly illegally) and consumed and thus provide an important source of protein for people (Kiffner et al. 2015).
- Several ungulates, especially large-bodied species, are strongly associated with bird species such as cattle egrets (*Bubulcus ibis*) or oxpeckers (*Buphagus africanus* and *B. erythrorhynchus*). By facilitating bird predation on insects (cattle egrets) and on ticks and other ectoprasites (oxpeckers), ungulates can contribute

indirectly to population control of parasites and potential insect pests (Kioko et al. 2016; Diplock et al. 2018).

- Several ungulate species are culturally important. For example, the giraffe is Tanzania's national animal.

In this chapter, we summarize available data about spatial and temporal population trends of the most commonly detected ungulates in the Tarangire Ecosystem. We also discuss some of the problems that arise when comparing results from different studies based upon dissimilar methods of data collection and analysis. We consider evidence that patterns of use of areas by ungulates may be changing due to environmental and anthropogenic influences, and finally we review current knowledge about human-ungulate coexistence in the Tarangire Ecosystem, and what this means for the future of hoofed mammals and humans there.

Box 8.1: What Are Ungulates?

Ungulates include hoofed mammals that walk on tiptoe and have high-crowned cheek teeth specialized for grinding vegetation cell walls. The hoof is keratinous tissue at the tip of a phalange (toe) and consists of a hard or rubbery sole and a wall of thick nail which supports the weight of the animal. Since only the hooves touch the ground, the rest of the foot has essentially become part of the leg, substantially increasing the length of stride. Raising the heel and digits off the ground increases the number of joints which move the legs, which increases the rate of stride.

Two orders of ungulates are classified according to the distribution of weight on their toes—the odd-toed perissodactyls and even-toed artiodactyls. Perissodactyls are specialized hindgut fermenters while artiodactyls are foregut fermenters (Gentry 1978). The digestive system of hindgut fermenters is less efficient at digesting plant fibers than foregut fermenters, and they compensate by eating more, including vegetation too fibrous and low in protein for foregut fermenters, and they digest much quicker, but as a consequence they have to spend more time eating (Clauss 2013).

Most foregut fermenters are ruminants, which have a superior ability to convert cellulose into digestible carbohydrates. This is accomplished by symbiotic microorganisms (bacteria, protozoans, yeast, and fungi) that digest cellulose by fermentation in the four-chambered stomach. In nonruminants fermentation occurs in the large intestine and an adjacent pouch, the cecum, after the food has passed through the stomach. In ruminants, fermentation occurs before gastric digestion, mostly in the rumen. Much more fiber is left undigested in the nonruminants, visible in the coarse dung of a zebra or rhinoceros versus the fine-grained pellet-like dung of a ruminant like a giraffe, wildebeest, or impala. Ruminants also re-chew the coarsest plant pieces ("cud").

In East African savannas, the bovids and the giraffes are ruminant artiodactyls, whereas hippopotamuses are pseudo-ruminant artiodactyls as they have foregut fermentation in a three-chambered stomach but do not chew cud. Zebras and rhinoceroses are nonruminant hindgut fermenting perissodactyls, and warthogs and bushpigs are nonruminant hindgut fermenting artiodactyls.

8.3 Ungulate Population Trends in the Tarangire Ecosystem: A Review of Available Data

Standardized, robust population monitoring of ungulates allows authorities to scientifically manage habitats, identify causes of population declines, establish proper hunting quotas, and determine whether conservation efforts are succeeding (Nichols and Williams 2006; Likens and Lindenmeyer 2010). In the sections below, we summarize historical and current population monitoring of the Tarangire Ecosystem's ungulate populations.

The Tarangire Ecosystem—here defined as an approximately 30,000 km^2 area between Lake Natron to the north, Simanjiro Plains to the southeast, and Irangi Hills to the southwest and the escarpment to the west (Fig. 8.1)—is home to at least 25 species of native ungulates,[1] categorized in four families of artiodactyls: the Bovidae (buffaloes and antelopes), Giraffidae (giraffes), Hippopotamidae (hippopotamuses), and Suidae (warthogs and bushpigs), as well as two families of perissodactyls: the Equidae (zebras) and—until its recent local extirpation—the Rhinocerotidae (Lamprey 1963). In the TE rain occurs almost exclusively from October–May, with a mean total annual rainfall of 650 mm for the years 1980–2009 (coefficient of variation = 42.6%, range = 312–1398 mm; Foley and Faust 2010). During the dry season wildlife aggregate where drinking water is available. These areas include the Tarangire River and the Silale wetland in Tarangire National Park, a few scattered waterholes in Tarangire National Park and Manyara Ranch, and rivers and springs flowing from the highlands such as those found in Mto wa mbu town, Lake Manyara National Park, Selela and Engaruka villages (Gereta et al. 2004; Morrison et al. 2016). There are also some small springs around Lake Natron that provide perennial freshwater sources (Morrison et al. 2016), but the supply is insufficient to support the populations of migratory ungulates during the dry season, so these animals mostly spend the dry season in the national parks and Manyara Ranch. Several ungulate species in the TE undergo seasonal migrations that track rainfall, plant phenology, and nutrient concentrations (Fryxell et al. 1988; Morrison et al. 2016; Voeten et al. 2010; **Lohay et al.** Chap. 12). The longest-distance and most widely dispersing species include plains zebras (*Equus quagga*; hereafter zebra), eastern white-bearded wildebeests (*Connochaetes taurinus albojubatus*, hereafter wildebeest), and common elands (*Taurotragus oryx*; hereafter eland) while African buffaloes (*Syncerus caffer*) migrate shorter distances.

In this chapter we focus on populations of antelopes and zebra. The species for which monitoring data are most widely available in space and over time include the following: zebra, wildebeest, eland, common waterbuck (*Kobus ellipsiprymnus*, hereafter waterbuck), impala (*Aepyceros melampus*), Grant's gazelle (*Nanger*

[1] Elephant, hippopotamus, black rhinoceros, common warthog, bushpig, plains zebra, giraffe, Kirk's dik-dik, steenbok, klipspringer, bush duiker, Thomson's gazelle, Grant's gazelle, gerenuk, Bohor reedbuck, common waterbuck, fringe-eared oryx, eastern white-bearded wildebeest, Coke's hartebeest, impala, lesser kudu, greater kudu, common eland, bushbuck, African buffalo.

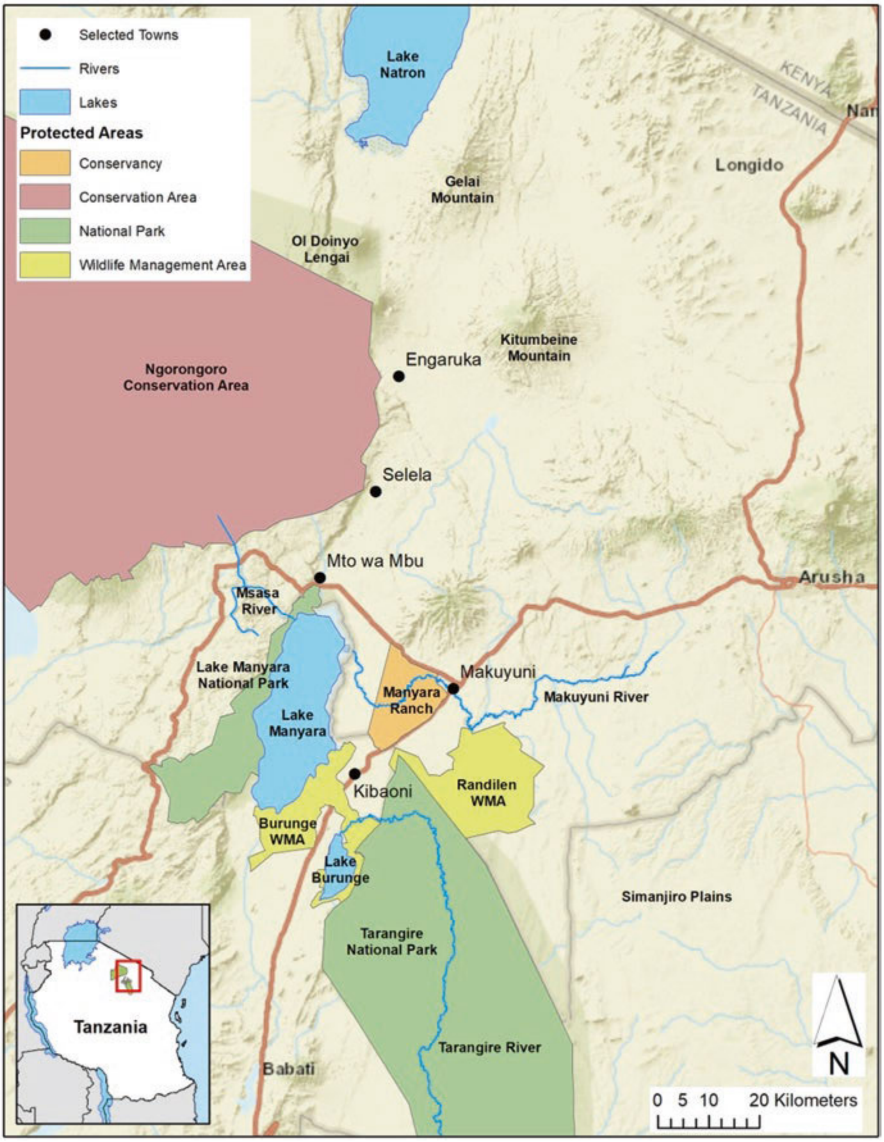

Fig. 8.1 Protected areas and key landmarks in the northern and central parts of the Tarangire Ecosystem. (Map created by Jason Riggio)

granti), Thomson's gazelle (*Eudorcas thomsonii*), and Kirk's dik-dik (*Madoqua kirkii*, hereafter dik-dik): see Fig. 8.2. Masai giraffes (*Giraffa camelopardalis tippelskirchi*) are also common but are covered in **Lee and Bond** Chap. 9. Other cryptic or rare species were occasionally documented in the literature but are not reported here.

Fig. 8.2 Eight commonly detected hoofed mammal species monitored in the Tarangire Ecosystem. Zebra (**a**), wildebeest (**b**), eland (**c**), waterbuck (**d**), impala (**e**), Grant's gazelle (**f**), Thomson's gazelle (**g**), and dik-dik (**h**). (Photographs by Derek Lee/©Wild Nature Institute)

Some of the earliest published studies of ungulate populations in the Tarangire Ecosystem were conducted in the 1950s and 60s (e.g., Lamprey 1963, 1964). These studies provide a baseline with which to compare population trends of ungulates in more recent decades, although even by the middle of last century it is likely wildlife numbers had already been severely depleted due to human activities (Estes and East 2009; **Prins and de Jong** Chap. 7).

Comparisons of population densities among studies are challenging if transects were not placed systematically across the landscape, analyses did not account for differences in habitat types, or if data were analyzed using different methodologies. For example, two density estimates for impalas in Tarangire National Park, both based on distance sampling from driving transects, ranged from an annual mean of 9.0 per km^2 from 2016 to 2018 (Foley et al. 2018) to 22.4 per km^2 from 2012 to 2018 (Kiffner et al. 2020a). Impala habitat selection in the Tarangire Ecosystem is strongly associated with both proximity to rivers and cover of the *Vachellia-Maerua* vegetation assemblage (James 2019). Therefore if placement of survey transects is biased towards areas close to rivers and in *Vachellia* woodlands, then density estimates extrapolated to the entire park are inflated. This makes comparisons among studies that do not have random placement of transects problematic. Stratifying the analysis by habitat type helps to reduce bias in density estimates: for a good example of this, see Table 8.4 below, adapted from Peterson (1978). Unfortunately, the practice of stratifying estimates by habitat type is rare.

Furthermore, different data collection methodologies can result in different population estimates in the same area covered. As an example, ground counts from distance sampling yielded an estimated abundance of 32,582 impalas in the Tarangire Ecosystem in dry season 2016, while an aerial count of the same area at the same time yielded a much lower estimated abundance of 5721 impalas (Foley et al. 2018). Undercounting bias in aerial surveys is well documented (Lee and Bond 2016; Greene et al. 2017). Therefore, comparing population estimates between studies that used different survey methodologies can be problematic.

On the other hand, robust inferences about trends (rather than assuming estimated densities along transects are representative of the wider study area) can be made based on longitudinal data from the same survey transects repeated over time using the same methods (e.g., Lee 2018; Lee and Bond 2018a).

Here we collated available data from published studies, dissertations, and unpublished reports, to elucidate spatial and temporal trends in ungulate populations in key regions of the Tarangire Ecosystem: (1) Tarangire National Park, (2) Simanjiro, (3) Manyara Ranch, (4) Lake Manyara National Park, and (5) Burunge and Randilen community Wildlife Management Areas (Fig. 8.1).

8.3.1 Tarangire National Park

Tarangire National Park (TNP) covers approximately 2850 km^2 of savanna and swamp habitat and is world-famous for its high density of African elephants (*Loxodonta africana*) and baobab trees (*Adansonia* spp.). The migration of wildebeests in and out of TNP is one of only three long-distance migrations of these animals remaining in Africa (Estes 2014; Morrison et al. 2016).

1958 to 1972—Lamprey (1964) quantified populations of ungulates in what at that time was the Tarangire Game Reserve and is now TNP. An aerial strip transect was conducted over a 52-km^2 area of the park in October of 1960, which yields density estimates that might be comparable to other studies. It must be noted that these estimates are not corrected for detectability and therefore should be interpreted with caution. We present these estimates in Table 8.3.

Twelve years later, in 1972, aerial counts were conducted in a 1350-km^2 area of northern TNP, and results were reported by Kahurananga and Silkiluwasha (1997). The authors estimated 4957 ± 909 (SD) zebras and 6255 ± 187 wildebeests during that count, representing a density of 3.67 zebras/km^2 and 4.63 wildebeests/km^2. Zebra density estimates were drastically lower in 1972 compared with Lamprey's estimates from 1960, whereas wildebeest density estimates had approximately doubled.

1987 to 2012—Kahurananga and Silkiluwasha (1997) analyzed wildlife populations in 1987 and 1990 during the dry season from aerial counts in the same area as in 1972 and compared results. Numbers were derived from Tanzania Wildlife Research Institute's (TAWIRI) systematic reconnaissance flights (SRF). These flights are conducted about every 5 years in the Tarangire Ecosystem. The authors reported 15,977 ± 3019 zebra and 14,006 ± 728 wildebeests in TNP in 1987. In 1990 the zebra population was slightly higher at 22,751 ± 4065 and wildebeest slightly lower at 11,429 ± 7566. Overall, in the early 1990s, the numbers of both of these migratory ungulates had increased substantially compared to the early 1970s.

Mtui et al. (2016) analyzed wildlife population density trends from 1994 to 2012 in TNP plus a 10-km buffer around the park, also from TAWIRI's SRF data. Data analyses indicated declining population densities over the time period for medium and large-sized antelopes as well as zebra, although the decline was significant only for zebra ($P = 0.04$). Densities of zebras declined by a rate of approximately 0.117 ind./km^2 per year (SE ± 0.058).

2012 to 2018—More recently, Kiffner et al. conducted seasonal ground-based distance sampling driving surveys in central and northwestern TNP from 2011 to 2019 (Kiffner et al. 2020a). For this chapter we used these data to estimate average annual population densities from three seasonal surveys per year, conducted from 2012 to 2018, for two migratory species (zebra and wildebeest) and for three resident ungulate species (waterbuck, impala, dik-dik). Overlap of confidence intervals for mean annual density estimates for the five species indicate no substantial upwards or downwards trends (Fig. 8.3). Further, Kiffner et al. (2020a) reported that a general linear mixed model indicated no significant decline in wildlife populations in TNP during the period of study.

Foley et al. (2018) conducted 5 ground-based distance sampling driving surveys throughout a large area of the Tarangire Ecosystem: during the long rains and dry seasons in 2016 and 2017 and one more during the long rains in 2018. The surveyors recorded detections of impala, Grant's gazelle, wildebeest, and zebra. Table 8.1 shows that density estimates for all four species increased in TNP over their survey period.

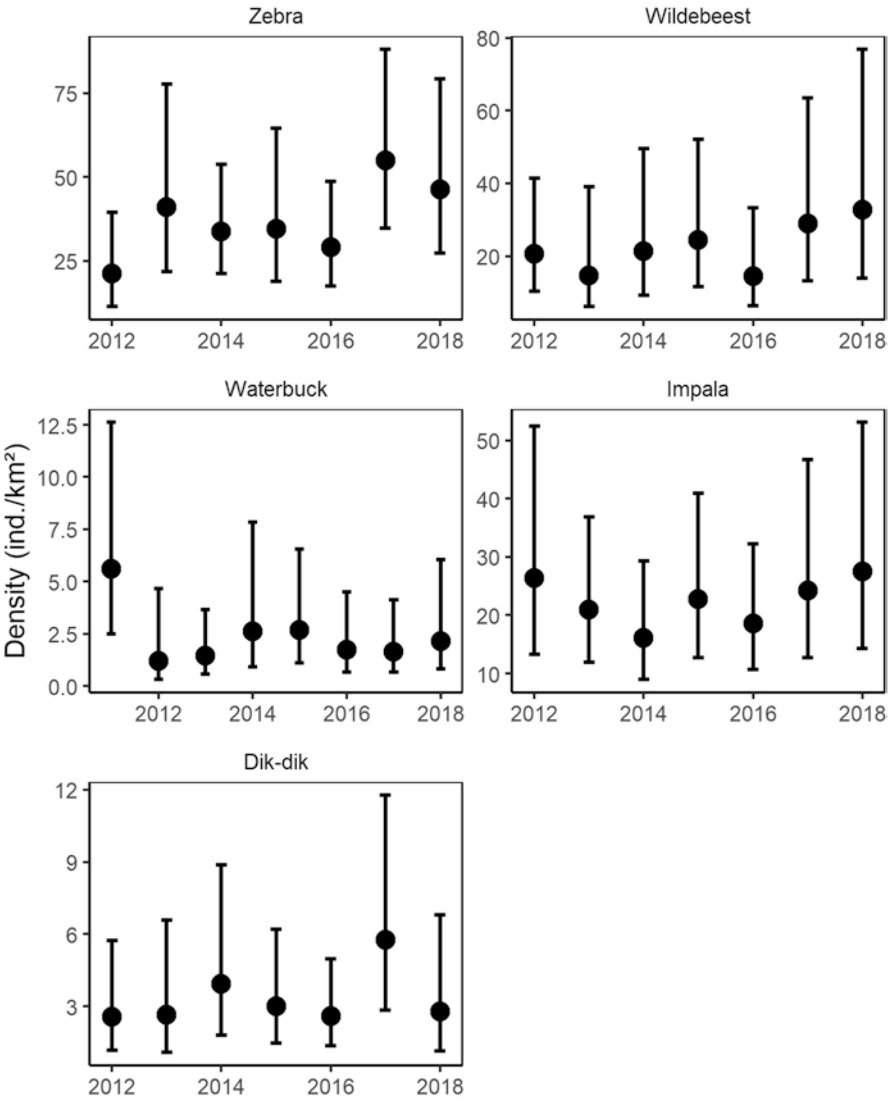

Fig. 8.3 Density estimates (ind./km^2 and 95% confidence intervals) for two migratory (zebra and wildebeest) and three resident (waterbuck, impala, and dik-dik) ungulate species in Tarangire National Park from 2012 to 2018. Data are from distance sampling along road transects (Kiffner et al. 2020a)

Challenges with comparisons among studies—It is noteworthy from the data we present above that population and density estimates are highly variable among surveys. It is possible that these numbers are relatively accurate and reflect actual fluctuations in numbers between years. However, it is also likely that differences in methodologies result in estimates that cannot be compared among studies. Below we provide density data for impalas, wildebeests, and zebras from three studies as

Table 8.1 Estimated densities (ind./km^2) for two resident (impala, Grant's gazelle) and two migratory (wildebeest and zebra) ungulate species in Tarangire National Park during wet (May/June) and dry (October) seasons from 2016 to 2018

Season	Impala	Grant's gazelle	Wildebeest	Zebra
Wet 2016	3.53	0.03	0	0.24
Dry 2016	7.44	0.04	0.09	5.03
Wet 2017	6.4	0.05	0	3.28
Dry 2017	16.56	0.14	13.82	38.57
Wet 2018	10.95	0.33	0	0

Data from Foley et al. (2018)

Table 8.2 Comparison of density estimates for wildebeest, zebra, and impala from an aerial count in 1960 (Lamprey 1964), and from distance-sampling driving transects from 2016 to 2017 (Kiffner et al. 2020a; Foley et al. 2018) during the dry season in Tarangire National Park, Tanzania

Species	Dry season density (ind./km^2)		
	1960 Lamprey	2016–2017 Kiffner et al.	2016–2017 Foley et al.
Zebra	29.0	74.7 (44.6–125.4)	21.8 (0–45.0)
Wildebeest	2.2	39.1 (18.2–84.1)	7.0 (0–16.5)
Impala	5.9	18.8 (10.1–35.0)	9.0 (5.06–12.9)

For 2016–2017, number is mean estimate for both years (95% confidence interval). Lamprey's (1964: Table 7) estimates are derived from a single aerial count on 4th October 1960 of a 52-km^2 area in the Tarangire Ecosystem, including Tarangire National Park, converted from densities reported in animals per square mile to animals per square kilometer

an illustration of the issues that arise when sampling design and analyses vary. Table 8.2 suggests that wildebeest and impala densities in TNP nearly 60 years later had increased, and that zebra densities were either comparable (Foley et al. 2018) or greatly increased (Kiffner et al. 2020a). However, the two density estimates for TNP during the same years (2016–2017) produced strikingly different results, with only slightly overlapping 95% confidence intervals.

The two later studies both used distance sampling from driving transects in Tarangire National Park during the same time period, and the same computer software program Distance (Thomas et al. 2010) yet yielded substantially different density estimates. Foley et al. analyzed one long survey transect per season, whereas Kiffner and colleagues placed multiple shorter transects along the roads and thus had spatial replication. On the other hand, Foley et al. (2018) accounted for the influence of group size, habitat type, and visibility on detectability which Kiffner et al. (2020a) did not, although due to the pooling robustness of the detection function, this difference is unlikely to explain the discrepancy in density estimates. The impala density estimates for wet season 2018 from both studies were even more different than the migratory wildebeests and zebras, with Kiffner et al. estimating 44.01 impalas/km^2 (95% CI = 23.22–83.42) and Foley et al. estimating 10.95 impalas/km^2 during that same time period—thus Kiffner et al.'s confidence interval did not overlap Foley et al.'s estimate.

These factors underscore the challenges of comparing results among studies with different methodologies. Long-term data from closely replicated studies using the same analytical methods should be used to reliably indicate population *trends*, but absolute abundances and density estimates must always be viewed with caution.

8.3.2 Simanjiro

The Simanjiro Plains region east of Tarangire National Park and is important wet season range for migratory ungulates, as well as critical habitat for resident ungulates such as impalas.

1970s—Kahurananga (1976) conducted 24 aerial counts in a 570 km^2 area in the Simanjiro Plains approximately 28 km east of TNP using stratified 300-m wide strip transects from January–July 1971 and October–December 1972. Kahurananga's original dissertation provided average biomass per species and average biomass densities. Here we converted the results into population densities that are presented below in Table 8.3, for comparison with other studies.

Peterson (1978) estimated densities of ungulates in the Simanjiro region during the wet season by driving a set of transects in 1975 and 1976. Survey areas included transects adjacent to TNP as well as in the plains farther east. Peterson calculated ungulate densities as the total number of a species recorded for a habitat, divided by the distance (km) traversed in that habitat, and then multiplied by an adjustment factor based on mean visibility in the habitat. Results by habitat are presented in Table 8.4. Note the substantially higher density of wildebeests and zebras in habitats characterized by the short grass *Panicum coloratum*. Wildebeests, zebras, and elands also favored seasonally waterlogged areas with *Pennisetum mexlanum*, *Duosperma kilimandscharicum*, *Cyathula erinacea*, and *Vachellia mellifera*, with localized thickets of *V. drepanolobium*. Estimating densities in different vegetation types provides important guidance into where habitat conservation measures could be directed.

2000s—More recently, two surveys were conducted in 2007 and 2017–2018 in the Simanjiro region. Msoffe et al. (2010) used distance sampling techniques to survey a 589-km^2 area in the Simanjiro Plains in May 2007. The surveyed area corresponds to approximately the same as Kahurananga surveyed in 1972, according to their maps, although Msoffe et al. (2010) describe the survey area as 40 km east of

Table 8.3 Density estimates (ind./km^2) of seven ungulate species from aerial counts during wet and dry seasons, 1971–1972, in a 570 km^2 area in the Simanjiro region of the Tarangire Ecosystem

	1971		1972	
Species	Wet season	Dry season	Wet season	Dry season
Zebra	10.81	0.04	10.94	0.08
Wildebeest	6.21	0.17	8.68	0.33
Grant's gazelle	1.07	0.63	1.16	1.20
Common eland	0.57		0.32	0.08
Thomson's gazelle	0.40	0.32	0.26	0.27
Impala	0.26		0.31	0.32

Data from Kahurananga (1976)

Table 8.4 Ungulate densities by species and habitat (ind./km^2) during wet seasons 1975–1976 in the Simanjiro Plains

Habitat type	Zebra	Wildebeest	Impala	Eland	Grant's gazelle
Short grass[a]	5.4	18.6	0	0.3	0.8
Wooded short grass[b]	11.5	7.2	1.3	0.2	1.2
Wooded tall grass[c]	7.6	1	2.9	0.5	0.4
Wooded tall grass[c] (modified)	2.1	0.2	1.2	0.2	0.9
Woodland (denuded)[d]	0.9	0.1	0.4	0	0.3
Woodland[d]	1.4	1.9	2.4	0	0
Woodland[e]	0.1	0	1.7	0	0
Bush grass[f]	4.5	0	1.9	1.3	0
Bush[g]	0	0	1.9	0	0
Bush grass[h]	0.1	0	3	0.2	0
Bush grass[i]	0	0	1.8	0	0
Bush grass[j]	1.4	0.7	1.7	1.3	0.4
Bush grass[k]	0	0	0	0	0
Medium grass/forb[l]	5.5	5	0.05	1.7	0.9
Medium grass/forb[l]	0.8	0.7	0	5.1	0.9
Tall coarse grass[m]	5.4	2.3	0	0.8	0.4
Medium grass[n]	7.2	9.7	0	0.8	1.1
Mean density	**3.17**	**2.79**	**1.19**	**0.73**	**0.43**

From Peterson (1978)
[a]Panicum-Digitaria; [b]Panicum-Digitaria-Commiphora; [c]Themeda-Panicum-Commiphora; [d]Vachellia-Commiphora; [e]*V. nilotica*-Azanza-Lanna; [f]Digitaria-Themeda-Commiphora; [g]Lannea-Croton-Grewia; [h]Grewia-Commiphora-*V. tortilis*; [i]*V. tortilis-Cynodon nlemfuensis*; [j]Urchloa-Cordia-*V. tortilis*; [k]Chloris-Sporobolus-Vernonia-Lannea; [l]Seasonally waterlogged Pennisetum-Duosperma-Cyathula-*V. mellifera*; [m]Seasonally waterlogged *Pennisetum mezianum*; [n]Seasonally waterlogged Sporobolus helvolus

TNP and Kahurananga describes it as 28 km east. We are uncertain about this discrepancy, as both Figure 1 in Msoffe et al. (2010) and Figure 1a in Kahurananga (1976) show Terat in the northeast of their respective, similarly sized survey areas.

Msoffe et al. (2010) systematically placed 25 five-km long transects throughout the survey area, spaced ≥1.5 km apart. Each transect was sampled twice by walking. Thus the data collection methods were different from the aerial surveys conducted in 1972, but both surveys corrected for detectability and were restricted to the shortgrass plains. Figure 8.4 below shows that from 1972 to 2007, in Simanjiro Plains, densities of zebras declined, wildebeest densities were stable, and Grant's gazelle densities slightly increased.

In 2016–2018, Foley et al. (2018) conducted distance sampling along roads in the Simanjiro Plains. Figure 8.4 shows that mean densities of zebras increased slightly since 2007, while Grant's gazelles decreased slightly but wildebeests declined substantially. However, zebra and wildebeest densities appeared similar to densities reported by Peterson (1978) in his survey of the Simanjiro Plains plus the area adjacent to TNP.

If these reported wildlife densities are reliable, it appears that densities of zebras in the Simanjiro area have quite drastically decreased between the 1970s and 2000s,

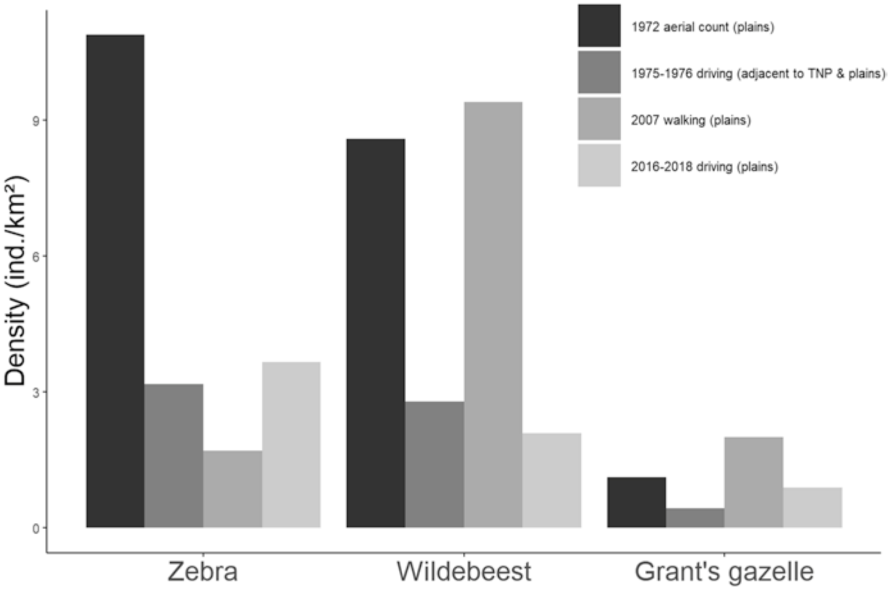

Fig. 8.4 Comparison of density estimates for three ungulate species over 4 time periods in the Simanjiro region east of Tarangire National Park, Tanzania. (Data from Kahurananga 1976; Peterson 1978; Msoffe et al. 2010; Foley et al. 2018)

whereas densities of Grant's gazelles slightly increased. Wildebeest density estimates appear to be highly variable and may reflect changing use patterns (see below).

8.3.3 Manyara Ranch

Manyara Ranch is an interesting example of conservation in Tanzania that is based on the 'land trust' concept, which provides a mechanism to acquire lands that are recognized as important for conservation but are outside the protected area system. The ranch was established as a cattle ranch during Tanzania's colonial period and was operated for agricultural and livestock production from 1956 to 1971 by a German cattle farmer, and from 1971 until 2000 by the National Ranching Company (Kiffner et al. 2020b). In 2000, the African Wildlife Foundation obtained a 99-year lease for management and conservation rights to the ranch, and in 2001 the Tanzanian Land Conservation Trust received title to the land. The Trust was managed by representatives from non-governmental conservation organizations, Tanzania National Parks, the local Maasai communities, and the private sector. In 2017, management of Manyara Ranch was handed over to the Monduli District and its operations have been subsidized financially by the African Wildlife Foundation.

According to the African Wildlife Foundation, the primary objective of the ranch is to "promote nature preservation and conservation and economic activities compatible with conservation for the benefit of present and future generations throughout Tanzania." The ranch remains an operating cattle ranch with small herds of livestock but also includes a tourism development program which offers facilities for visitors (African Wildlife Foundation 2005; Sachedina 2008).

The 17,807-ha property is located within the critical wildlife migration corridor connecting the Tarangire River in the south with the plains near Mounts Gelai and Kitumbeine and Lake Natron to the north (Morrison et al. 2016; Bond et al. 2017). At the onset of the wet season, migratory animals leave TNP and move north through the ranch, returning again at the beginning of the dry season.

There are no available data for historical ungulate populations or densities for Manyara Ranch prior to 2000, except for anecdotes of exceptionally high densities in the area (**Prins and de Jong** Chap. 7). From 2003 to 2008 ranch management sporadically counted wildlife by driving along four road transects. Starting in 2011, Kiffner and colleagues conducted regular distance sampling surveys by driving along roads on the ranch during each of the three precipitation seasons (short rains, long rains, dry). Estimating animal densities included testing for seasonal effects on species-specific detection functions (see methods in Kiffner et al. 2020b). Further, Kiffner et al. validated their road-based density estimates by counting ungulates along systematically placed transects during the same time frame as road transects during 2018 and 2019 short rains. Results of the validation indicated that density estimates from road surveys may yield conservative measures of animal densities but differences were not significant.

The long-term trends (2003–2019) suggested that densities of elands, wildebeests, waterbucks, Grant's and Thomson's gazelles, and dik-diks increased on Manyara Ranch, whereas zebra and impala densities were stable (Kiffner et al. 2020b). The more recent trends using statistically robust distance sampling data indicate that zebra, wildebeest, waterbuck, impala, and dik-dik densities fluctuated seasonally but have not showed notable consistent downwards or upwards trends over the entire study period (Fig. 8.5).

Foley et al. (2018) also conducted distance sampling surveys on Manyara Ranch during the long rains and dry season from 2016 to 2018; their impala density estimates were similar in the 2016 long rains (4.2/km^2) to estimates from Kiffner et al. during the same time (5.5/km^2) but all subsequent impala density estimates from Foley et al. (2018) were much lower—even for a commonly detected resident species (Kiffner et al.: 5.3, 5.5, 5.1, 7.3 ind./km^2 versus Foley et al.: 0.1, 1.9, 1.0, 2.5 ind./km^2). Again, comparisons among studies are usually difficult and therefore it is more useful to examine trends over time from studies that use the same data collection methods and statistical analyses.

Overall, however, it appears that Manyara Ranch supports substantial ungulate population densities within the Tarangire Ecosystem, and that wildlife populations have been relatively stable there over the past decade. Moreover, the seasonal surveys indicate that wildlife densities in Manyara Ranch are particularly high during the dry season, suggesting that the ranch is permanent habitat for multiple wildlife species *as well as* a key dispersal area and stepping stone for the annual wildebeest and zebra migration in the TE.

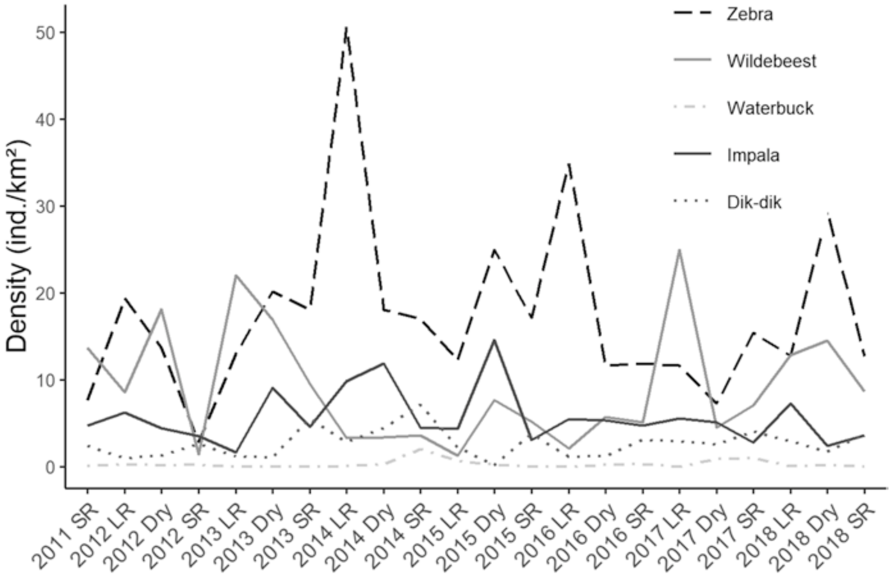

Fig. 8.5 Mean seasonal (SR: Short rains; LR: Long rains; dry: Dry season) density estimates of five ungulate species from 2011 to 2018 in Manyara Ranch, Tarangire Ecosystem, Tanzania. (Data from Kiffner et al. 2020b)

8.3.4 Lake Manyara National Park

Established in 1960, Lake Manyara National Park (LMNP) is the oldest national park in the TE (**Prins and de Jong** Chap. 7). The core area of the park is located between the alkaline Lake Manyara and the escarpment. Though relatively small in size (the core area comprises c. 168 km^2 terrestrial habitats), LMNP boasts a variety of habitats, including short alkaline grasslands on lacustrine plains near the shore, swamps, *Vachellia* woodlands, bushlands, and evergreen groundwater forests (Greenway and Vesey-Fitzgerald 1969; Loth and Prins 1986).

Occasional total counts of wildlife were conducted by the management authorities between 1959 (when the area was still a Game Reserve) and 2010 (Prins and Douglas-Hamilton 1990), and seasonal road counts were carried out from 2011 to 2019 by the School For Field Studies. Thus ungulate populations in this management unit of the TE can be described over a long time scale (Kiffner et al. 2017). In the "early" years (1959-mid 1980s), the ungulate community in the park was characterized by exceptionally high densities of African buffalo (Prins 1996).

Historically, LMNP supported a particularly high herbivore biomass density of around 16,000 kg/km^2 (Prins and Douglas Hamilton 1990). From 1959 to the mid-1980s, ungulate populations were subject to a variety of perturbations: multiple disease outbreaks (e.g. a rinderpest epidemic that reduced the buffalo population in 1959; an anthrax outbreak in 1983/1984 that reduced the density of impalas), and variation in rainfall (which affected lake levels and caused temporary emigration of the wildebeest population during the 1970s) substantially affected population dynamics of ungulate species. However, as depicted in Fig. 8.6 wildlife populations quickly recovered from these perturbations or wildlife declines in one species were compensated by population increases in other species (Prins and Weyerhaeuser 1987; Prins and Douglas Hamilton 1990).

More recently, the biomass density of herbivores ranges around 9000 kg km^{-2}, and has thus declined considerably over time (Kiffner et al. 2017). Although the underlying reasons for these declines are not exactly established, the timing of wildlife population declines provides circumstantial evidence for likely causes of the observed population dynamics. The substantial decline of the buffalo population occurred during the mid-1980s, a time when wildlife in the park was subject to high

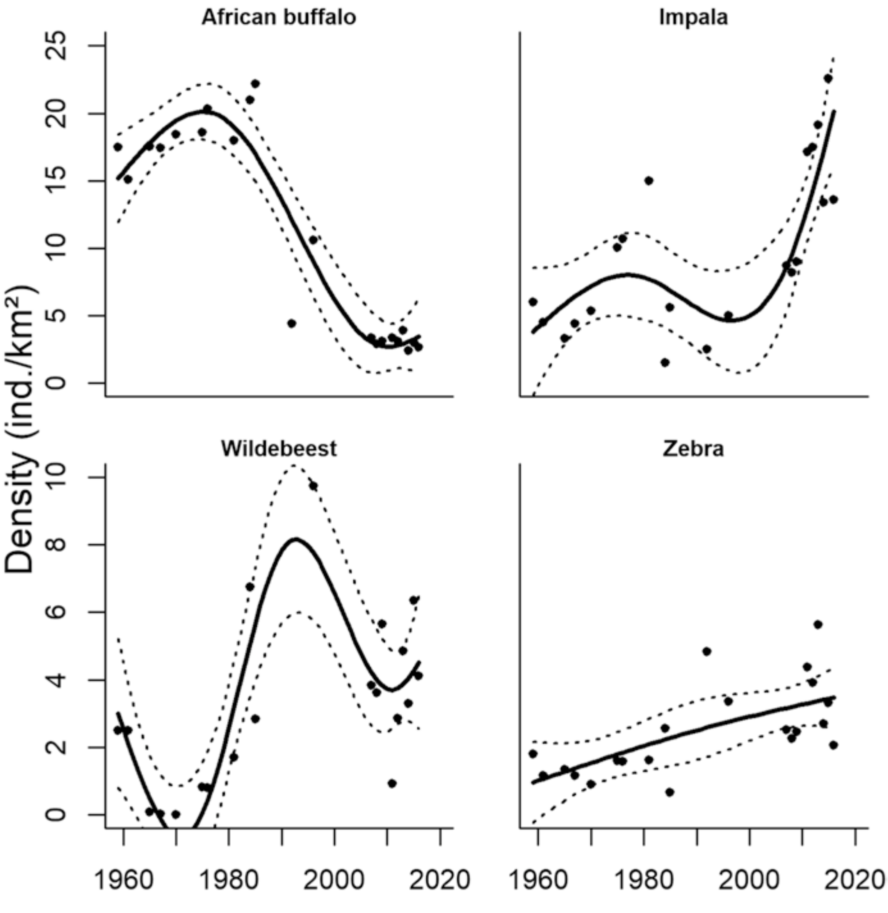

Fig. 8.6 Mean annual population density estimates (solid points) of African buffalo, zebra, wildebeest and impala in Lake Manyara National Park from 1959 to 2016. The trend line is based on a general additive model that describes the time series; dashed lines indicate the 95% confidence interval of the trend line. (Data are from Kiffner et al. 2017)

levels of poaching that also caused the local extinction of the black rhinoceros (*Diceros bicornis*) population (Borner 1981) and decimated the elephant population (Prins et al. 1994). Although unquantified due to its clandestine nature, there is anecdotal evidence for substantial buffalo poaching during this time (Prins 1996), making poaching the most likely underlying reason for the observed buffalo population decline. While wildebeest temporarily emigrated from the park when the grassland was flooded by the alkaline water of the lake, the wildebeest population rapidly increased during the 1980s and 1990s and nowadays seems to have stabilized at around 5 wildebeest km^{-2}. The zebra population shows a linear and positive population trend over time. It is plausible that the population increases in both wildebeest and zebra were facilitated by release from food competition with buffalo. However, it is noteworthy that the population densities of both wildebeests and zebras in LMNP are c. one order of magnitude lower than corresponding densities in TNP. Possibly these density differences can be explained by the different movement strategies of the subpopulations. Due to the increasing insularization of LMNP (caused by rapid human population growth and agricultural development projects along the northern border of LMNP), the wildebeest (Morrison et al. 2016) and zebra populations are mostly resident in LMNP, whereas the TNP subpopulations use the national park only during the dry season.

The poaching-induced decline of the elephant and black rhinoceros populations as well as the anthrax outbreak related to the decline of the impala population likely facilitated regeneration and recruitment of woody plants. The understory of many parts of LMNP are nowadays much more densely vegetated than in the late 1980s (Kiffner et al. 2017). In turn, this vegetation increase in the shrub layer likely facilitated the quick recovery of the impala (a species feeding on both grass and browse material) population after the anthrax outbreak in 1983/1984 and impala now exceed densities observed during earlier decades (Fig. 8.6). Indeed, the dense understory in most parts of LMNP (and the concurrent decrease of grass patches in the savanna habitats) caused an increase in species that thrive on woody vegetation such as browsing ungulates (e.g. bushbuck, *Tragelaphus sylvaticus*) and primate species (e.g. olive baboon, *Papio anubis*) (Kiffner et al. 2017).

These examples highlight one key lesson. While wildlife populations can recover from natural perturbations such as disease outbreaks, the additive effects of geographic isolation and excessive poaching can trigger multiple cascading effects that have repercussions on other species and vegetation structure even many decades after the initial perturbations occurred.

8.3.5 Wildlife Management Areas

Wildlife Management Areas (WMAs) are a form of community-based natural resource management implemented in Tanzania. In creating a WMA, several villages set aside land for wildlife conservation and implement management activities in the WMA, in return for most of the tourism revenues from those areas (Nelson

2010; URT 2012). Each WMA is independently managed and is spatially structured by land-use plans that regulate human activities in specific zones.

Three WMAs are currently operating in the Tarangire Ecosystem: Burunge (one of the first WMAs operational in the country, first gazetted in 2006); Randilen; and Makame (**Baker et al.** Chap. 15). The ecological effectiveness (indicated by stable or increasing wildlife population trends and by comparing densities in WMAs with corresponding densities in adjacent areas) of the Tarangire Ecosystem's WMAs in terms of ungulate densities has been quantified in three published studies, which we summarize here.

Randilen WMA—Lee and Bond (2018a) conducted 24 distance sampling surveys to estimate ungulate densities from 2012 to 2015 in Randilen WMA and the adjacent control site in Lolkisale Game Controlled Area, both along the eastern border of TNP. Density estimates for dik-diks, impalas, and zebras were compared in this before-after control-impact study design. Prior to establishment of the Randilen WMA, annual densities of all species were similar in the control and impact site. After the implementation of WMA management activities, Randilen WMA had significantly greater densities of dik-diks (Fig. 8.7) and significantly lower densities of cattle, relative to the adjacent area. Densities of impalas and zebras in the WMA did not differ from the control site throughout the survey period.

Burunge WMA—Lee (2018) used a before-after control-impact design to quantify the ecological effectiveness of the Burunge WMA, on the western border of TNP. From 2012 to 2017, transects were distance-sampled 36 times within a portion of the WMA. In 2016, transects were surveyed 6 times throughout the entire WMA and areas outside the WMA. The author compared relative densities of ungulates

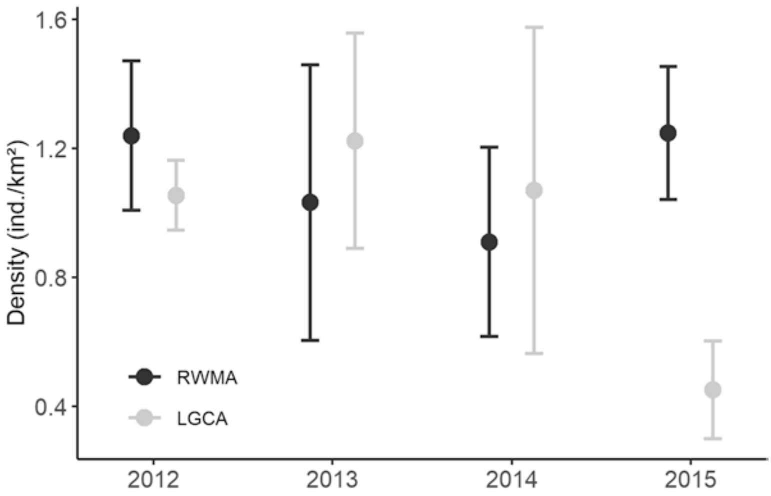

Fig. 8.7 Mean annual densities (ind./km^2 ± SE) of dik-diks in the impact site Randilen Wildlife Management Area (RWMA) and adjacent control site Lolkisale Game-Controlled Area (LGCA) in the Tarangire Ecosystem, Tanzania, from 2012 to 2015. Randilen WMA was established and management activities began in May 2014. (Data from Lee and Bond 2018a)

before and after the establishment of management activities by Chem Chem Safaris and PAMS Foundation in 2014–2015, and compared densities inside and outside the WMA during 2016.

The study found 11 of the 16 species of wildlife had significantly greater densities inside the WMA relative to outside. In comparing before and after WMA establishment, impala densities increased significantly (as did densities of buffalo), while dik-diks and waterbucks increased but not significantly so (Fig. 8.8). No species decreased.

Kiffner et al. (2020a) compared ungulate densities in Burunge WMA with densities in TNP during varying seasons from 2011 to 2018. Ungulate densities were not significantly different during most seasons, although on several surveys zebra densities were significantly higher in TNP than the WMA, with the exception of the 2016 long rains when densities were significantly higher in the WMA. Over the intermittent time series, impala populations showed significant population increases, similar to the time series presented by Lee (2018). Over the study period, wildebeest populations also increased in the WMA.

These studies of WMA ecological effectiveness in the Tarangire Ecosystem show that (1) ungulate population densities often increase after WMA management activities are initiated, compared with nearby control sites; (2) ungulate densities can be comparable to the adjacent, fully protected TNP; and (3) WMAs provide considerable conservation value indicated by high species richness and relatively high wildlife densities.

While the evidence strongly suggests that WMAs can be ecologically effective, controversy over the creation of these WMAs and their ecological effectiveness has

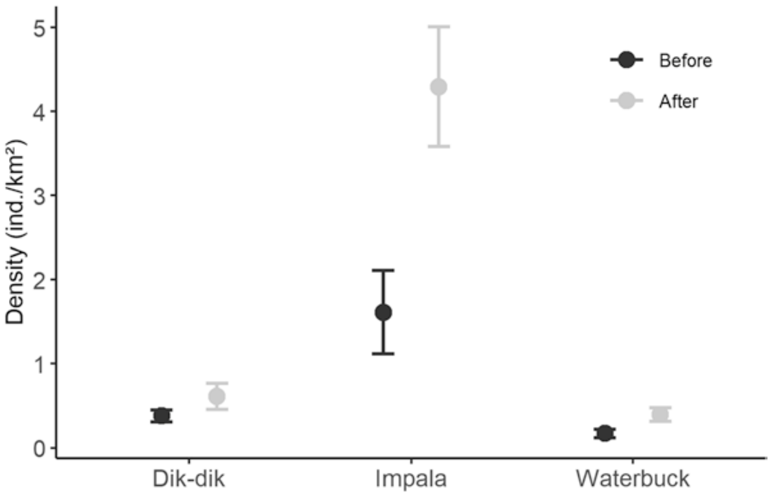

Fig. 8.8 Mean annual densities (ind./km^2 ± SE) of dik-diks, impalas, and waterbucks in the Burunge Wildlife Management Area in the Tarangire Ecosystem, Tanzania, from 2012 to 2015. Burunge WMA was established and management activities began in May 2014. (Data from Lee 2018)

been reported by some (Brehony et al. 2018, but see Lee and Bond 2018b, **Raycraft** Chap. 6). Recent research discussed in **Raycraft** (Chap. 6) suggests that under certain circumstances, local communities can grow to appreciate and support the WMAs.

8.4 Changing Patterns of Use in the Tarangire Ecosystem: Wildebeest Example

Examining population trends in different areas of the Tarangire Ecosystem over time can provide important information about patterns of use of different areas. The wildebeests provides an illustrative example: in contrast to the declines noted by Mtui et al. (2016) in TNP, wildebeest abundance estimates from 1988 to 2011 throughout the entire ecosystem—based on the same systematic reconnaissance flights—ranged from a high of 48,783 animals in the early 1990s to a low of 2916 in 1997, but then subsequently increased to 11,934 animals in 2011 (Fig. 2 in Morrison et al. 2016). Morrison et al.'s wildebeest abundance estimate in the entire ecosystem from the final survey in 2011 is similar to Foley et al.'s (2018) estimate of 11,588–15,835 wildebeests between 2016 and 2018, from the Lake Natron area in the north to West Kilimanjaro in the northeast to Makame WMA in the south, lending credence to this population estimate and suggesting the population has stabilized at this number.

If these numbers are correct, it is possible that the increasing abundance of wildebeests from 1997 to 2011 throughout the TE, as presented by Morrison et al. (2016), with the possibly decreasing densities of wildebeests in TNP over approximately the same time frame and using the same aerial count survey data, as presented by Mtui et al. (2016), may indicate changing patterns of use of different areas in the ecosystem. Wildebeests may be shifting their use to areas outside the park such as Manyara Ranch. Indeed, Lee et al. (2013) estimated a mean annual abundance of 881 wildebeests in Manyara Ranch in 2012–2013.

Interestingly, Foley et al. (2018) also described a change in spatial patterning of wildebeests in the TE over several years. They noted that in the dry season of 2016, more wildebeests were observed on the shore of Lake Manyara in Burunge WMA and in Manyara Ranch than in TNP itself. The authors suggested that some wildebeest that had formerly migrated to either Lake Natron or Simanjiro may have altered their migration patterns and are now calving in Burunge WMA. Kiffner et al. (2020a) also reported increasing densities of wildebeests in Burunge WMA over their study period.

Therefore it is likely that overall patterns of use of areas by wildebeests and other wildlife have shifted throughout the Tarangire Ecosystem, compared with earlier decades, possibly due to anthropogenic influences (see also **Igoe** Chap. 3).

8.5 Human-Ungulate Coexistence in the Tarangire Ecosystem

Available evidence suggests that ungulate populations in the Tarangire Ecosystem were drastically reduced between the turn of the nineteenth century to the middle of the twentieth century (**Prins and de Jong** Chap. 7). There remain concerns about the viability of migration routes in the region (Bond et al. 2017). The Tarangire-Manyara-Natron migration corridor was identified as one of the top five priority corridors for conservation in the Priority Corridor Action Plan for Tanzania.

However, results from recent longitudinal wildlife population monitoring studies in TNP and on Manyara Ranch (Morrison et al. 2016; Kiffner et al. 2020a, b) as well as in the Wildlife Management Areas bordering Tarangire and Lake Manyara national parks (Lee and Bond 2018a; Lee 2018; Kiffner et al. 2020a) found that densities of ungulates may fluctuate but have been overall relatively stable over recent years in these protected areas. Furthermore, ungulates continue to use the Simanjiro Plains, where several non-governmental organizations established conservation easements (**McCabe and Woodhouse** Chap. 4) and Certificates of Customary Rights of Occupancy (**Brehony et al.** Chap. 5). These conservation efforts appear to be successfully maintaining ungulate populations although their numbers are likely well below their historical baselines (**Prins and de Jong** Chap. 7).

Even more heartening is the demonstrated ecological success of community-based natural resource management in the form of the WMAs: the studies we report here from the Tarangire Ecosystem are some of the first investigations of the ecological success of community-based natural resource management in the scientific literature. We recommend these community conservation areas, as well as conservation easements, Certificates of Customary Rights of Occupancy, and the pastoral Manyara Ranch, continue to be fully supported, and the northern migration route to the Natron area be protected via similar community conservation efforts that equally address the needs of wildlife as well as those of the pastoralist communities in the TE.

The seasonal movement of the ungulate populations throughout the ecosystem is key for sustaining relatively high densities of the migratory subpopulation of wildebeests and other migratory ungulates. From an anthropocentric perspective, the key beneficiaries are Tarangire National Park and associated tourism enterprises. While communities in Burunge and Randilen WMAs and the Simanjiro Plains now also benefit economically from wildlife-based tourism, the seasonal movement of ungulates across the ecosystem may impose substantial costs for some people. During the wet season, zebras may feed on agricultural crops (Bencin et al. 2016); the presence of wild ungulate populations on communal lands makes those areas attractive for large carnivores which may in turn kill livestock (**Kiffner et al.** Chap. 11; **Kissui et al.** Chap. 14); and calving wildebeests shed a virus that causes malignant catarrhal fever in cattle, which causes direct (the disease is often fatal in cattle) and indirect (pastoralists typically avoid wildebeest calving areas and therefore have to shift grazing areas elsewhere) costs to pastoralists (Lankester et al. 2015). Acknowledging these ecological interdependencies and economic inequalities of costs and benefits associated with wildlife populations may be a first step towards designing a more just, equitable, and sustainable conservation approach for both people and wildlife in the TE.

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Chapter 9
Giraffe Metapopulation Demography

Derek E. Lee and Monica L. Bond

Abstract The Masai giraffe is the national animal of Tanzania and a globally iconic megaherbivore, but numbers have declined precipitously and the subspecies is now listed as endangered on the IUCN Red List. We studied the Masai giraffe population in the Tarangire Ecosystem over nine years to quantify population structure and demography of a large, free-living, wild megaherbivore population inhabiting a coupled human-natural system. This system supports a high density of giraffes and is representative of the current diversity of threats and conservation opportunities across the range of the species. We describe population structure (subpopulations within a metapopulation) and demographic structure (age and sex distributions) among subpopulations defined three ways: geographically discrete areas defined by human administrative boundaries; and subpopulation units derived from two types of social relationships among giraffes. The Tarangire giraffe metapopulation still functions via natural movements among subpopulations. Demographic variation exists among subpopulations, so maintaining habitat connectivity to ensure giraffe movements across the greater Tarangire Ecosystem is essential to long-term population viability.

Keywords Masai giraffe · *Giraffa camelopardalis* · Tarangire ecosystem · Capture-mark-recapture · Social network analysis · Population ecology

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9.1 Introduction

Giraffes (*Giraffa camelopardalis*: Box 9.1) are endemic African browsing ruminants, and one of only a few extant terrestrial megaherbivore species.[1] The Masai giraffe (*G. c. tippelskirchi*; Fig. 9.1) is the national animal of Tanzania and the most numerous of the nine recognized subspecies, but the global Masai giraffe population declined by approximately 50% recently, leading to the subspecies being classified as endangered on the IUCN Red List (Bolger et al. 2019).

The giraffe's primary natural predators are African lions (*Panthera leo*), leopards (*Panthera pardus*), and spotted hyenas (*Crocuta crocuta*) (Dagg and Foster 1976).

Fig. 9.1 Male Masai giraffe in Lake Manyara National Park. (Photo by Derek Lee)

[1] Defined as animals reaching up to 1000 kg in mass (Owen-Smith 1988).

Predation is an important factor affecting juvenile survival (Strauss et al. 2015; Lee et al. 2016a) but only a minor source of adult giraffe mortality (Schaller 1972; Strauss and Packer 2013). Adult Masai giraffes are poached by humans for meat and products such as hide, bones, and tail hairs (Bolger et al. 2019) including in the Tarangire Ecosystem (Kiffner et al. 2015).

The social structure of giraffes is described as a fission-fusion process wherein herd composition and size changes frequently over time but is structured by non-random social associations between adult females that reflect kinship (Bercovitch and Berry 2012; Carter et al. 2013a, b). Giraffes are long-lived (over 30 years) and slow breeding (Bingaman Lackey 2009; Dagg 2014). They can become sexually mature as early as age 2–3 years (Bingaman Lackey 2009), but wild females typically mature at a mean of 4.8 years of age (Bercovitch and Berry 2009). Giraffes have a mean gestation period of 14.7 months (del Castillo et al. 2005), with a mean interbirth interval of 20 months (Lee and Strauss 2016). Females reproduce throughout the year, with estrous cycling approximately every 15 days, and can become pregnant while still nursing their previous offspring (Dagg and Foster 1976; Bercovitch et al. 2006; Lee et al. 2017). Female giraffe home ranges are large and overlapping (Knüsel et al. 2019; Deacon and Bercovitch 2018). Female giraffes in estrous are dispersed over space and time, so reproductive adult males adopt a strategy of roaming constantly among female groups to seek mating opportunities, with periodic hormone-induced rutting behavior approximately every 2 weeks, a temporal scale that would overlap with local cycling females (Pratt and Anderson 1985; Bercovitch et al. 2006; Seeber et al. 2013).

Box 9.1: Giraffes: A Closer Look

Giraffes are the tallest living land mammals, with males reaching a maximum height of 5.5 m and females 4.5 m (Owen-Smith 1988). They have a long, muscular tongue which gathers leaves into the mouth, and as ruminants they are efficient at extracting nutrients from leaves. Giraffes help shape the vegetation in African savannas (Strauss et al. 2015) and maintain complex ant-plant mutualisms that improve the health of *Vachellia* [formerly *Acacia*] woodlands (Palmer et al. 2008).

Prehistoric peoples in Africa created rock artwork featuring giraffes an estimated 10,000 years before present. Giraffes drawings grace the tombs of Egyptian kings (Shorrocks 2016). The Romans named the giraffe “cameleopard” because its head and tail looked like a camel and the coat like a leopard (according to the writings of Pliny the Elder). The origin of the giraffe’s long neck and long legs has fascinated mankind throughout recorded history, and became a focal point of debate on evolutionary theory between Lamarck and Darwin in the nineteenth century (Agaba et al. 2016). Regardless of origin, the giraffe’s great height places substantial burdens on its cardiovascular, musculoskeletal, and nervous systems that are accommodated by unique adaptations such as a ‘turbocharged’ heart, thickened blood vessel walls in the legs, and an enlarged nuchal ligament along the neck (Agaba et al. 2016).

The family Giraffidae comprises giraffes (*Giraffa camelopardalis*) and their closest relative okapis (*Okapia johnstoni*), which diverged from a common ancestor about 11.5 million years ago (Agaba et al. 2016). Understanding giraffe taxonomy is hampered by the extinction of several populations as late as the nineteenth century (Petzold et al. 2020). Genetic analyses have suggested three (Petzold and Hassanin 2020; Petzold et al. 2020), four (Fennessey et al. 2016), and six (Brown et al. 2007) species of giraffes.

The study of demography (births, deaths, and movements) elucidates mechanisms that drive increases or decreases in populations and provides the scientific basis for data-driven population management (Frederiksen et al. 2014). Few studies have investigated the demography of large tropical herbivores (Owen-Smith and Marshall 2010), and giraffes provide a tropical, asynchronously breeding, fission-fusion species with which to test hypotheses derived from research on ecology and behavior of temperate ungulates (Lee et al. 2016a, 2017). Giraffes are particularly suitable for demographic research as their unique and unchanging coat patterns (Foster 1966) allow researchers to individually identify them using digital photography and computer vision technologies (Bolger et al. 2012). These advances have facilitated collection and analyses of unprecedentedly large sample sizes of individually monitored giraffes, which has expanded knowledge of the species' demography (Lee and Strauss 2016; Lee et al. 2016a, b, 2017; Lee and Bolger 2017) and sociality (Carter et al. 2013a, b; VanderWaal et al. 2014; Bond et al. 2020, 2021a, b).

Here we focus on describing the population and social structure and demography of a large, free-living, wild giraffe population inhabiting the coupled human-natural Tarangire Ecosystem (TE) using nearly a decade of longitudinal data. We describe population structure (subpopulations within a metapopulation) and demographic structure (age and sex distributions) among areas of the TE defined by human administrative boundaries as well as subpopulation units derived from social relationships among giraffes. Socially defined subpopulations include adult female 'communities', which show substantial spatial overlap but are socially discrete (Bond et al. 2020, 2021a), as well as mixed-sex 'supercommunities' that are more spatially discrete than the communities (Lavista-Ferres et al. 2021). Defining subpopulations based on administrative boundaries helps to evaluate the effectiveness of management strategies in improving vital rates (Lee 2018; Lee and Bond 2018), while comparing demography among distinct socially defined subpopulations facilitates an understanding of environmental versus social influences on vital rates (Bond et al. 2021a, b).

9.2 Study Area

Our core study area for the Masai Giraffe Project in the Tarangire Ecosystem (TE) of northern Tanzania is a coupled natural-human system that supports a high density of giraffes and is representative of the current diversity of threats and conservation opportunities across the range of the species (Fig. 9.2). Giraffe habitat outside the

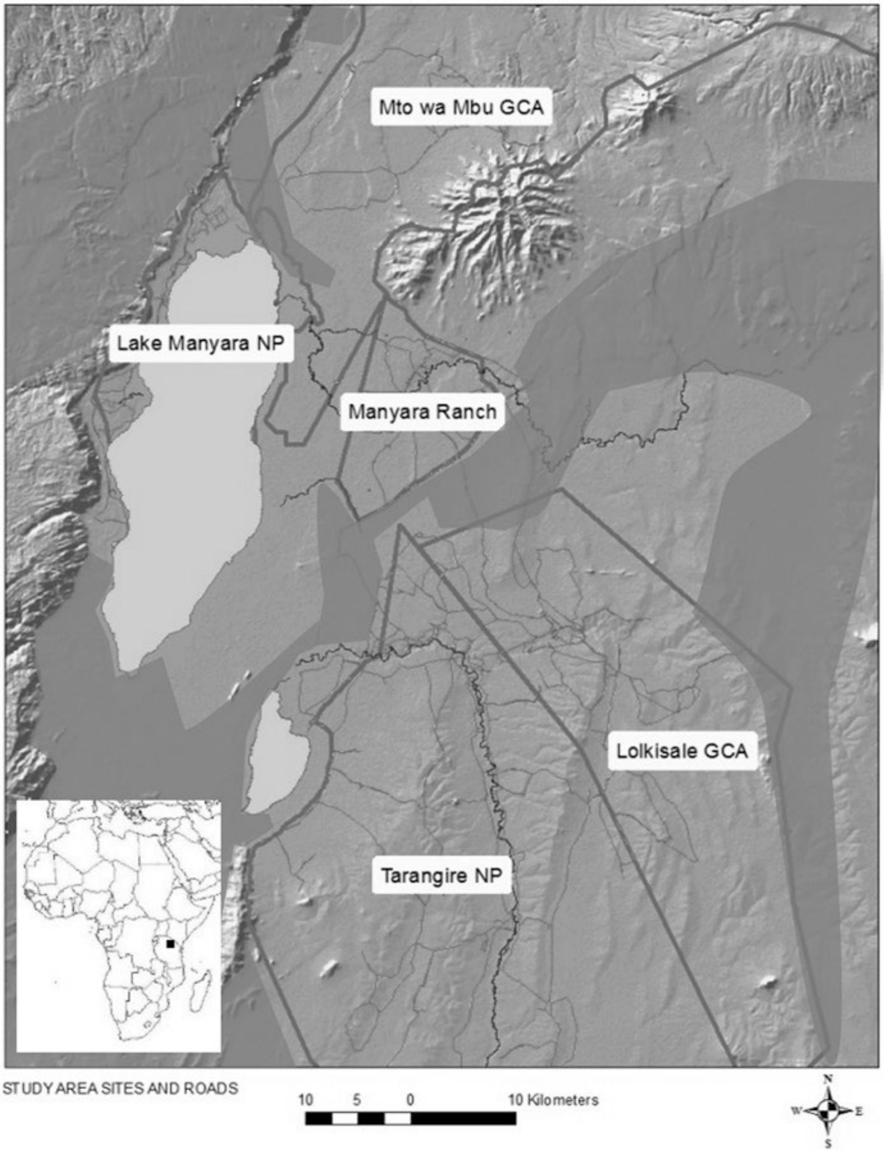

Fig. 9.2 Tarangire Ecosystem giraffe study area in northern Tanzania. Thick gray lines delineate some human-defined administrative areas. Dark shaded areas are dominated by human agriculture uses, and light gray shaded areas are used by pastoralists. National parks (NPs) are areas where human use is restricted to ecotourism, Manyara Ranch has ecotourism but allows livestock, game controlled areas (GCAs) allow livestock and settlements, and thin lines are dirt roads we used for giraffe photographic encounter surveys. Inset shows location of study area within Africa

TE's two national parks has been either conserved by traditional pastoralists (livestock keeping people) and ecotourism operations, or degraded by agriculture, charcoal making, and other human activities (Newmark 2008; Msoffe et al. 2011; Morrison et al. 2016). Giraffe habitat throughout Africa has become similarly fragmented, thus the TE is illustrative of much of the existing landscape for these megaherbivores. The observed variation in natural, degraded, and fragmented habitat in the TE makes the ecosystem an ideal study area to assess the relative effects of these anthropogenic impacts on giraffe populations.

The TE is a savanna biome with heterogeneous vegetation types ranging from open grasslands to dense deciduous bushlands and thickets, supporting one of the most diverse large-mammal communities in the world (Lamprey 1963). We sample for giraffes in a 1500 km^2 area along dirt road transects in four administrative areas: Tarangire National Park, Lake Manyara National Park, Manyara Ranch Conservancy, and Mto wa Mbu and Lolkisale Game Controlled Areas (Fig. 9.2). The entire study area is unfenced and all administrative areas are connected by movements of adult female giraffes, making this a metapopulation (Lee and Bolger 2017).

9.3 Data Collection and Demographic Analyses

Since January 2012, we have been conducting active encounter photographic surveys for giraffes 3 times per year near the end of every precipitation season by driving the same network of fixed-route dirt road transects in our study area. We survey according to a robust design sampling framework with 3 sampling occasions per year, where each sampling occasion is composed of 2 consecutive sampling events where we survey all road transects in the study area (3 occasions per year × 2 events per occasion = 6 independent, complete survey events per year). Road density throughout our study area is high relative to giraffe home range size (Knüsel et al. 2019). Survey teams maintain a driving speed between 15 and 20 km/h on all transects, and all teams include the same trained observers and a driver. We sample each road segment only 1 time in a given event. We systematically shift the order and direction in which we sample sites and road transects similar to a Latin Square design to reduce sampling biases.

During photographic capture-mark-recapture sampling events, when we encounter any giraffes we 'mark' newly observed individuals or 'recapture' previously observed animals by slowly approaching and photographing the giraffe's right side. We make efforts to standardize image collection such that every subject's identifiable region of interest is perpendicular to the camera, in focus, well lighted, and high resolution. We attempt to photograph every giraffe encountered for individual identification from within a distance of approximately 100 m ($\bar{x} = 90 \pm 39$ m) at an angle that is as close to perpendicular (90°) as possible. For every photograph, we record sex (male, female), GPS location, and age class. We categorize giraffes into 4 age classes: newborn calf (0–3 months old), older calf (4–11 months old), subadult (1–3 years old), or adult (≥4 years) using a suite of physical characteristics. Sex, age class, and location are useful gradients for stratifying and grouping individuals for analyses to ensure that assumptions are not violated regarding equal detectability and survival within groups. We obtain one GPS location for the group, which is defined as one or more giraffes that were foraging or moving together and were >500 m from the next nearest giraffe (Bond et al. 2019). We use group membership to conduct social network analysis for quantifying social structure.

We developed an automated procedure to crop photos to the giraffe torso (Buehler et al. 2019), our area of interest for individual identification and matching. We match giraffe torso identification images using WildID (http://software.dartmouth.edu/Macintosh/Academic/Wild-ID_1.0.0.zip), a computer program that matches large datasets of giraffe images collected using our protocols with low error rates (Bolger et al. 2012). We summarize photographic capture-mark-recapture data into individual encounter histories, and analyze the encounter histories using recapture statistics (Burnham and Anderson 2002; Williams et al. 2002; Amstrup et al. 2006; Cooch and White 2019).

We analyze our encounter histories using Pollock's Robust Design models in program MARK (Cooch and White 2019) to estimate age-specific apparent survival (*S*; Pollock 1982; Kendall et al. 1995), as well as capture (*p*), recapture (*c*), and temporary emigration (γ' and γ'') rates. We test goodness-of-fit of encounter histories using programs MARK or U-CARE (Choquet et al. 2009), and adjust for lack of fit if necessary by adjusting c-hat (c-hat = χ^2/df; Choquet et al. 2009; Cooch and White 2019). We rank and select models using AIC_c (or $qAIC_c$ if c-hat is adjusted) and use model weights (W) as a metric for strength of evidence supporting a given model as the best description of the data (Burnham and Anderson 2002). During survival model selection, we begin with the most fully parameterized model in our set with all relevant age and time effects in capture, recapture, and temporary emigration rates. We then rank competing models of temporary emigration and detectability parameters with reduced temporal complexity or linear trends of age. Once the most parsimonious form of temporary emigration and detectability parameters is obtained, we rank models of survival, including age trends as linear (A), quadratic (A^2), and cubic (A^3) models, as well as a constant (.) or null model, and age-specific survival (age). Seasonal probabilities of apparent survival are converted to annual rates by multiplying the appropriate seasonal rates.

We calculate sex ratios (males/females) and an index of reproduction (calves/adult female/year) using our enumerated individuals within the total metapopulation and subpopulations.

9.4 Metapopulation Demography

Overall, ignoring subpopulation structure, the metapopulation of giraffes in our core study area in the TE from 2012 to 2018 ($n = 42$ independent surveys) included 2891 uniquely identified individuals. In any given year, the metapopulation within our study area was composed of approximately 1520 giraffes with an age class distribution of approximately 60% adults, 15% subadults, and 25% calves. Sex ratios (males/females) of calves and subadults were slightly male biased (m/f = 1.1), but

Table 9.1 Age-specific annual survival probabilities of Masai giraffe in the Tarangire Ecosystem

Age (years)	Annual survival
0	0.691
1	0.849
2	0.920
3+ males	0.911
3+ females	0.945

adult sex ratios were female biased (m/f = 0.68). Annual survival probability was similar for both sexes in calf and subadult age classes, but by age 3 year, female survival probability was significantly higher than that of males (Table 9.1). Season of birth also affects first-year survival (Lee et al. 2017), but here we present the mean probability of survival for all calves. Adult survival is best modeled as constant with age, but there is some evidence of survival rates decreasing slightly with age among adult giraffes, possibly indicating somatic senescence. Presenting these data in another manner, including age-related senescence: 50% of all giraffe calves born in the TE die before they attain sexual maturity at age 4. Giraffe calves born in the TE have approximately a 25% chance of reaching age 11 if male, and age 14 if female, and a 10% chance of reaching age 17 if male, and age 22 if female.

9.5 Defining Subpopulations

The structure of populations is a central concern to biologists (Thomas and Kunin 1999), and a number of techniques have been used to identify subpopulations of animals for research and management. Our early work in the TE found the landscape still functioned as a metapopulation (defined as a regional set of subpopulations that exchange individuals) with connectivity movements among subpopulations in human-defined administrative areas (Lee and Bolger 2017), and significant demographic variation among the administrative units (Lee et al. 2016a). We also documented the ecological success of community-based natural resource management in the human-defined giraffe subpopulations in Wildlife Management Areas (WMAs) in the TE, with WMAs successfully increasing giraffe density (Lee and Bond 2018) and survival (Lee 2018).

For social, sexually reproducing animals a better definition of a subpopulation might be interactions among members of the subpopulation, so subpopulations are determined by the relationships between individuals and not by an externally imposed geographic classification (Harwood 2009). Social structure within a metapopulation or superpopulation can be determined by patterns of relationships which are then used to define subpopulations and management units.

To better understand the TE giraffe metapopulation structure and dynamics, we used network analysis of giraffe social associations among the nearly 3000 individually identified giraffes in our study area to define socially discrete subpopulations (see Bond et al. 2020, 2021a; Lavista-Ferres et al. 2021 for methodology). We identified 2 levels of social subpopulation organization within our TE giraffe population (Fig. 9.3):

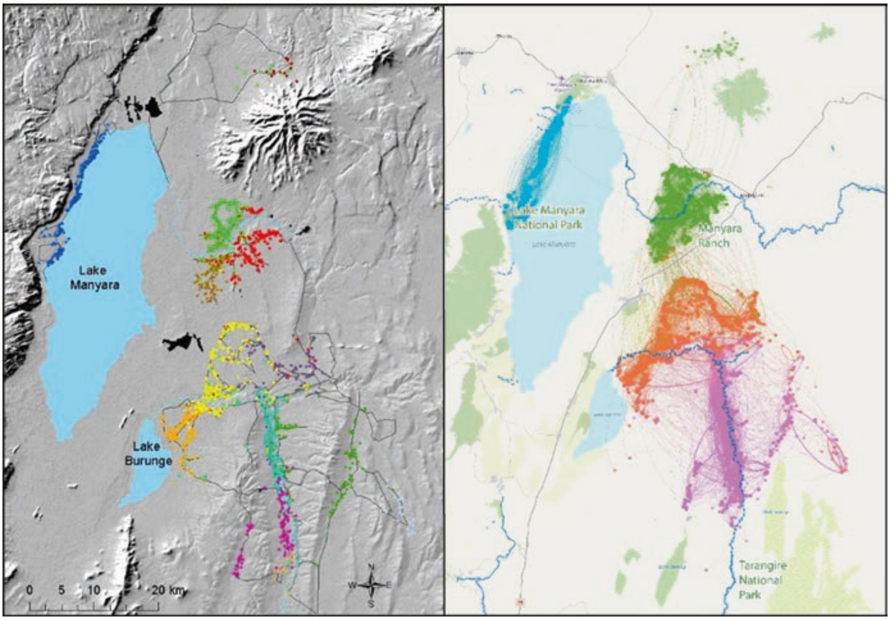

Fig. 9.3 Maps of the Tarangire Ecosystem giraffe population where colored dots indicate individual giraffes in 14 female-only communities (left) and 4 all-age and both-sex supercommunities (right). (Communities map reproduced with permission from John Wiley and Sons. Supercommunities map reproduced with permission from Elsevier)

the first was 14 communities defined by social interactions among adult females (Bond et al. 2020, 2021a), and the second was 4 supercommunities defined by social interactions among all ages and sexes (Lavista-Ferres et al. 2021). Both communities and supercommunities are subgroups of animals that associate more with each other than with the rest of the network (Girvan and Newman 2002).

We assigned males and non-adults to female-only communities based on the dominant community membership of the females they were observed interacting with, although some animals could not be assigned to a community because no majority existed. Community subpopulations were discrete in terms of social associations despite substantial overlap in space use, and the presence of individual movements among subpopulations and supercommunities indicated they met our definition of a metapopulation.

We then computed summaries of the demographic structure of the subpopulations defined by the three different methods: (1) human administrative site; (2) adult female community; and (3) all ages and sexes supercommunity. We used slightly different administrative boundaries here than in previous work, to reflect Wildlife Management Areas (Burunge WMA and Randilen WMA) established west and east of the northern portion of Tarangire NP (TNP). TNP, Lake Manyara NP (LMNP) and Manyara Ranch Conservancy (MRC) were defined the same as in previous work, but here we excised the remote, far northern observations in the Mto wa Mbu Game Controlled Area. Sites were entirely spatially discrete, whereas supercommunities were slightly overlapping (Lavista-Ferres et al. 2021), and adult female communities were substantially overlapping (Bond et al. 2021a). We calculated population densities of each of the subpopulations by either dividing the number of individuals in a site or supercommunity by the home range area of all combined individuals in the site or supercommunity, or dividing the number of all adult females present (regardless of community membership) in the home range of all combined females in a community, by the area of the home range. This method of estimating population density accounts for the spatial overlap of communities.

9.6 Subpopulation Demography

Our 5 administratively defined subpopulations revealed very different population sizes among the sites, which is largely a function of the spatial area each occupies, but there was also significant variation in giraffe population density (N/area [km^2]) among sites (Table 9.2, top). We documented significant movements of giraffes

Table 9.2 Summary statistics for subpopulations of giraffes in the Tarangire Ecosystem. 'Site' subpopulations were defined by administrative boundaries, 'Supercommunities' were defined by social relations among all giraffes, 'Communities' were defined by social relations among adult females, with other age and sex classes being assigned to a community based on co-occurrence in groups with females

		All	C/AF/Year	Adult	Calves
Site	N	Density	Repro	M:F	M:F
LMNP	157	1.40	0.14	0.55	0.68
MRC	918	5.50	0.28	0.53	1.05
TNP	1453	2.57	0.17	0.81	1.09
BWMA	138	4.60	0.24	0.36	2.63
RWMA	170	1.72	0.41	0.40	1.26
Mean	567	3.16	0.25	0.53	1.34
Community					
LMNP	146	0.57	0.14	0.54	0.64
North MRC	171	2.13	0.13	0.48	0.87
East MRC	225	2.25	0.21	0.68	1.14
SW MRC	176	1.6	0.16	0.27	1.10
North TNP	232	1.37	0.23	0.52	0.96
Central TNP	151	1.03	0.12	0.65	0.79
South TNP	161	3.38	0.11	0.88	1.07
Lemioni RWMA	79	1.33	0.25	0.39	1.43
West TNP BWMA	218	0.64	0.11	0.58	1.32
LGCA	148	1.17	0.10	0.83	1.20
Mean	171	1.55	0.16	0.58	1.05
Supercommunity					
LMNP	155	1.38	0.13	0.55	0.64
MRC	919	5.50	0.26	0.52	1.11
TNP north	814	2.95	0.23	0.47	1.16
TNP south	792	1.76	0.14	1.00	1.08
Mean	670	2.90	0.19	0.64	1.00

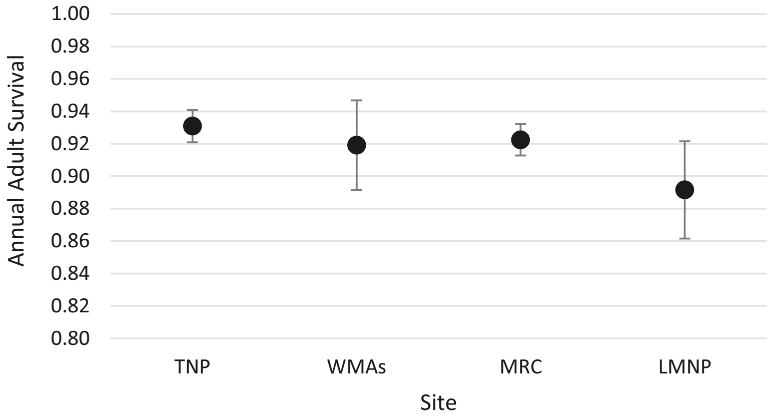

Fig. 9.4 Giraffe mean adult apparent annual survival probabilities among human-delineated administrative sites in the Tarangire Ecosystem 2012–2018. Sites are Tarangire National Park (TNP), Burunge and Randilen Wildlife Management Areas (WMAs), Manyara Ranch Conservancy (MRC), and Lake Manyara National Park (LMNP). Error bars are ±1 SE

among all sites, with the exception of LMNP where in 7 years we witnessed only 1 movement of a giraffe in or out of LMNP: a female adult who went from MRC to LMNP, then back to MRC (Lavista-Ferres et al. 2021).

Demography varied among sites (Table 9.2), with highest reproduction in MRC and RWMA, more adult males in TNP, and more calf males in WMAs. Adult survival was statistically similar among sites (Fig. 9.4), but LMNP had lower mean adult survival. MRC had the highest calf survival and LMNP the lowest.

LMNP is a demographically isolated subpopulation of approximately 100 living giraffes. It is consistently identified as a subpopulation under all three methods used to subdivide the TE giraffe metapopulation. LMNP is unusual in that it has the lowest population density of any subpopulation, a very low index of reproduction (# of calves / # of adult females / year), and a very female-biased calf sex ratio (Table 9.2). We believe the natural predators of LMNP, or some other factor, greatly elevates calf mortality there such that few calves recruit into the breeding population (Figs. 9.5 and 9.6). Interestingly, the LMNP subpopulation has been highly stable in number over the decades since 1960 (van der Jeugd and Prins 2000; Fig. 2 in Kiffner et al. 2017), so it could be that the LMNP subpopulation is at local carrying capacity, and the low recruitment and female-biased sex ratio of offspring are density-dependent effects.

9.6.1 Adult Female Communities

Communities defined by adult female associations were similarly sized, but when we computed population density of all adult females (regardless of community membership) within the home range of each community, we found significant variation in density (Table 9.2). Female community membership is highly stable over

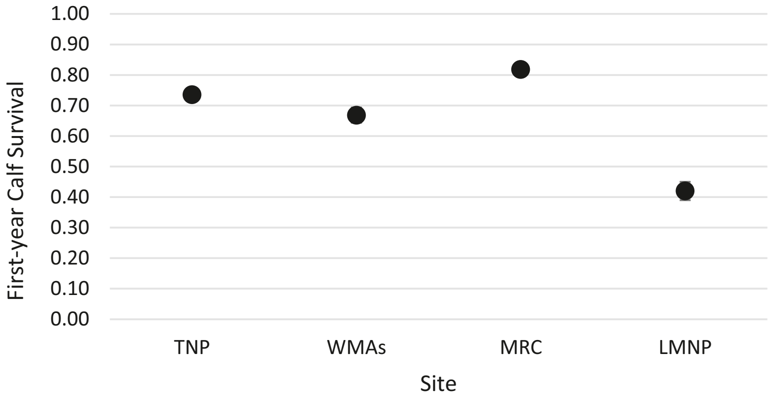

Fig. 9.5 First-year calf survival of giraffes among human-defined administrative sites in the Tarangire Ecosystem 2012–2018. Sites are Tarangire National Park (TNP), Burunge and Randilen Wildlife Management Areas (WMAs), Manyara Ranch Conservancy (MRC), and Lake Manyara National Park (LMNP). Error bars are ±1 SE

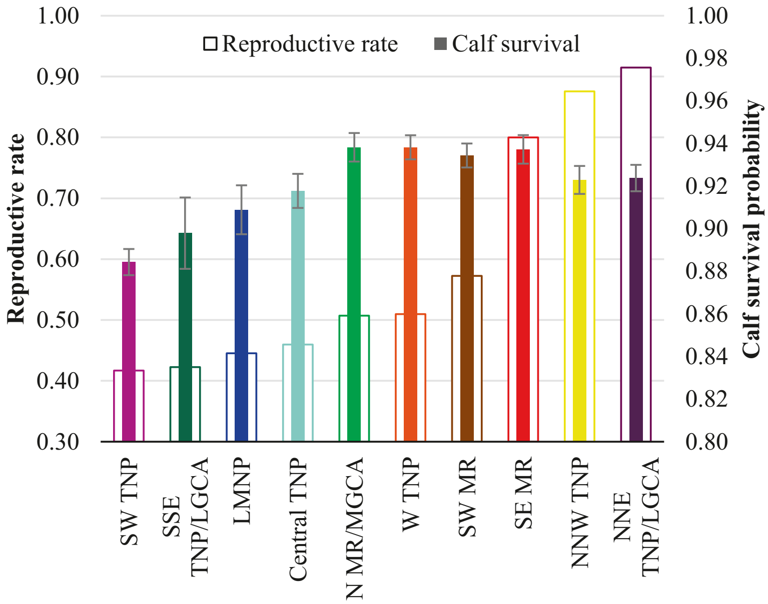

Fig. 9.6 Community subpopulations exhibited variation in demography, demonstrating our ability to detect fine-scale population dynamics associated with socially mediated population structure within this large metapopulation. (Reproduced with permission from John Wiley and Sons)

time, and movements among communities are largely made by dispersing subadult males (Bond et al. 2021c).

Demographic rates of reproduction and calf survival varied among communities (Fig. 9.6; Bond et al. 2021a). We suspect that pastoralists disrupting natural predation along with favorable forage conditions are possible mechanisms for some of our observed spatial demographic patterns (Bond et al. 2021a). Reproductive rates were highest in subpopulations with more volcanic soils in the home range (Lemioni RWMA, North TNP, and East MRC communities; Table 9.2), which likely creates higher quality of forage (Hansen et al. 1985). Proportion of bushlands was correlated with lower calf survival (Bond et al. 2021a), possibly because giraffe calves are vulnerable to lion predation in such dense vegetation. Subpopulations closer to human settlements had higher reproductive rates, and the highest calf survival probabilities were found in the subpopulations that included MRC with its high levels of pastoralists and their livestock, as well as in the subpopulation on the western edge of TNP where pastoralists also are common (Bond et al. 2021a). Females with calves were more likely to be found closer to bomas, likely because of reduced predator densities there (Bond et al. 2019). We believe adult female giraffes seek to lower predation risk to their calves by aggregating closer to pastoralist human settlements. Adult female survival was high and nearly constant across all subpopulations, but there is evidence that sociability of individual females is the main driver of individual variation in adult female survival (Bond et al. 2021b).

Variation in reproductive index and sex ratios among communities other than LMNP partially reflect patterns of spatial segregation among different age and sex classes (Table 9.2, Fig. 9.7). Calf spatial utilization distribution (which partially reflects higher indices of reproduction) was highest in northwest TNP and BWMA, as well as eastern MRC (Fig. 9.7a). Adult and subadult males are concentrated in TNP, along the Tarangire River where it flows south to north (Fig. 9.7b, c). The spatial utilization distribution of bachelor herds (groups comprised of a majority of adult and subadult males) is greatest along the river southeast of Tarangire Hill, as well as small hotspots in BWMA and western MRC (Fig. 9.7b). The distribution of single adult males, likely dominant breeders, is greatest north and south of the Tarangire Hill bachelor herd concentration, but also extends along the east-west section of Tarangire River into BWMA, out east into RWMA, and into northwestern MRC (Fig. 9.7c). These male giraffe hotspots are reflected in the greater adult male

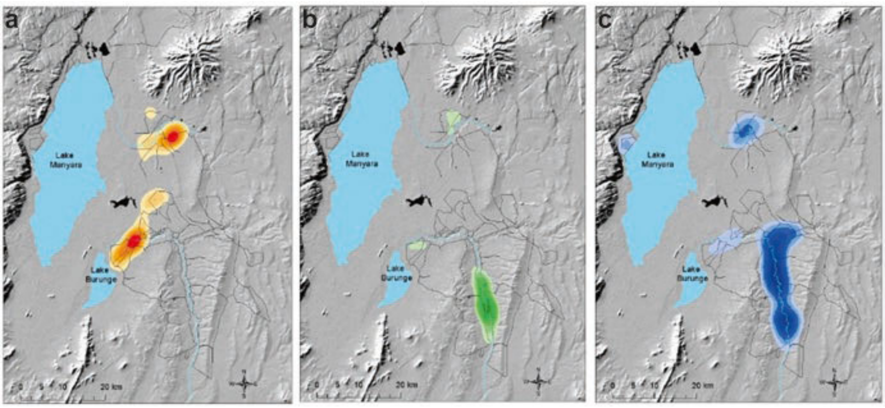

Fig. 9.7 Utilization distribution heat maps of observations of: giraffe calves (**a**), bachelor herds (**b**), and single adult males (**c**)

to female ratios and lower reproduction index values in the TNP site, TNP South supercommunity, and South TNP and LGCA communities (Table 9.2).

9.6.2 All Ages and Sexes Supercommunities

Subpopulation sizes were similar among three of the supercommunities but significantly lower in the LMNP supercommunity, yet there was significant variation in giraffe population density among supercommunities (Table 9.2, bottom) due to differences in spatial area. Most individuals that transitioned among supercommunities returned to their original supercommunity, with only 10% transitioning permanently, indicating the stability and interconnectedness of these groupings (Lavista-Ferres et al. 2021). Adult survival varied among supercommunities and by sex, with female survival generally higher than male survival (Fig. 9.8). MRC and LMNP had the greatest disparity in adult survival between the sexes. TNP North and MRC had the highest adult survival probabilities of the supercommunities. First-year survival of calves also varied among supercommunities in a pattern similar to that seen in adult survival, except MRC had higher first-year survival than any other supercommunity (Fig. 9.9).

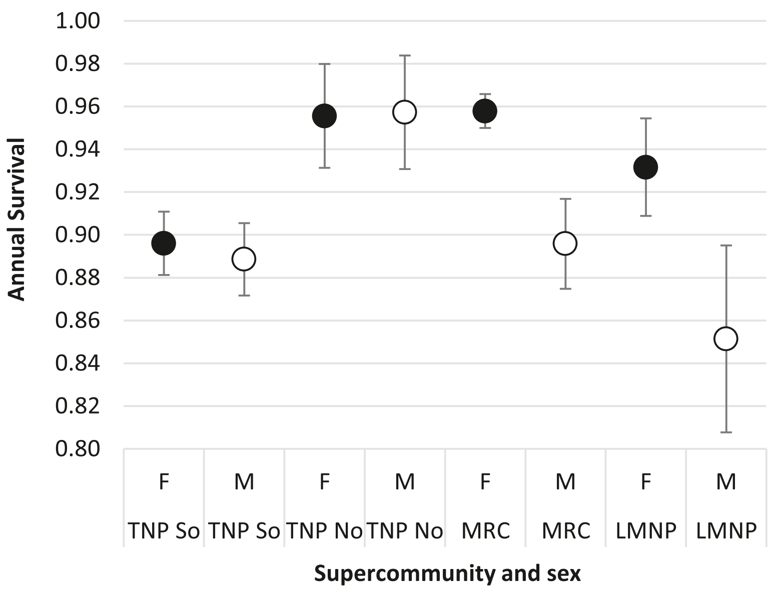

Fig. 9.8 Adult female (F, closed circles) and male (M, open circles) giraffe annual apparent survival probabilities among supercommunities in the Tarangire Ecosystem 2012–2018. Error bars are ±1 SE

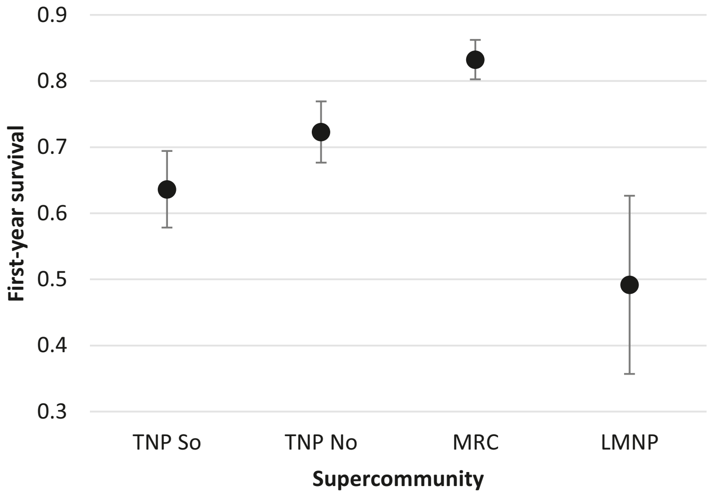

Fig. 9.9 Giraffe calf first-year apparent survival probabilities among supercommunities in the Tarangire Ecosystem 2012–2018. Error bars are ±1 SE

9.7 Conclusions: Human-Giraffe Coexistence in the TE

Management and conservation of species, subspecies, or populations can be facilitated by understanding how and why populations are structured, and how and why demographic vital rates vary among (sub)populations. This is especially important for declining species that are hunted or that inhabit fragmented landscapes subject to human activities, because specific anthropogenic factors implicated in population declines can be identified and potentially ameliorated. Learning where subpopulations are doing well and where they are not—and why—can help pinpoint the best conditions for improving population growth rates and help identify problem areas and effective solutions.

We identified what are likely high-quality habitats for giraffes on volcanic soils in Northern TNP and MRC with high calf and adult survival, and we documented substantial movements between those two areas despite a tarmac road between them (Fig. 9.3, right). Preserving movement opportunities between these supercommunities will help maintain the metapopulation in the TE, as well as all the ancillary economic and ecological benefits to local communities. LMNP is an isolated and potentially vulnerable subpopulation with low calf and adult survival, but interestingly the subpopulation size has remained remarkably stable over many decades. This indicates a carrying capacity of about 100 individual giraffes in LMNP at any given time. Future research should concentrate on whether inbreeding is occurring there, as this population is insular but still not completely cut off from the rest of the metapopulation.

Sparsely distributed human settlements such as Maasai bomas appear to be compatible with giraffe population persistence, and data suggest adult females actually aggregate closer to Maasai bomas, possibly to reduce predation risk on calves despite the impacts on the females' social relationships (Bond et al. 2019, 2020, 2021b). Earlier work in this metapopulation showed lower adult female survival in the Mto wa mbu Game Controlled Area north of Manyara Ranch (Lee et al. 2016a; Lee and Bolger 2017), which is likely due to poaching. We found anti-poaching efforts in WMAs surrounding TNP and LMNP are successfully conserving giraffes, with increased density and survival after establishment and compared with adjacent areas outside the WMAs (Lee and Bond 2018; Lee 2018). This confirms the importance and effectiveness of community conservation for the future of giraffes in the TE and throughout Tanzania.

Investigating potential links among the environment, sociality, and demography requires long-term, large-scale studies, because such studies are more likely to include contrasting ecological and social conditions (Clutton-Brock and Sheldon 2010). These multi-scaled demographic analyses which revealed multi-level social and population structure in giraffes in the TE were possible thanks to longitudinal data collection across a vast, heterogeneous landscape over nearly a decade.

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Chapter 10
The History, Status, and Conservation of the Elephant Population in the Tarangire Ecosystem

Charles A. H. Foley and Lara S. Foley

Abstract The Tarangire Ecosystem is well known for its elephant population and Tarangire National Park is marketed as one of the best places in East Africa to see large herds of elephants. In the past century the elephant population in the ecosystem has undergone significant changes in response to poaching and land use changes. These have impacted elephant ranging and migration patterns and have led to dramatic alterations in the demographic structure of the population. This chapter provides a historical account of the elephant population in the Tarangire Ecosystem from the mid-1900s until the present day. Included in this chapter is how the demographic structure of the population was impacted by and has recovered from heavy elephant poaching, how the elephant range has contracted and expanded over time in response to poaching and changing land-use patterns, and how connectivity with other elephant populations has been impacted by human land use. It also describes some of the main problems that are likely to affect the elephant population in the future.

Keywords Elephants · Tarangire National Park · Poaching · Dispersal

10.1 A History of Tarangire's Elephants

Little is known about the elephant population in the Tarangire Ecosystem prior to the 1900s. Elephant populations in East Africa have been long affected by the trade in ivory, which has occurred in the region since at least the first century CE. During the nineteenth and twentieth centuries, the trade expanded considerably (Spinage 1994), fueled in part by slave traders who introduced firearms that allowed them to kill large numbers of elephants (Parker and Amin 1983). In 1859 over 221 tons of ivory were exported from Zanzibar, which was the main trading hub (Spinage

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1994), and by 1900 hunting had devastated elephant populations across much of the region (Grogan and Sharp 1900). However, it is unclear how much hunting took place within the Tarangire Ecosystem. Although the Maasai were known to trade in ivory (Spinage 1994), they were reportedly extremely hostile to invaders, causing the large ivory and slave caravans to divert southwards and travel through central Tanzania (Croze and Lindsay 2011). Game laws passed at the beginning of the 1900s led to a cessation of much of the large-scale hunting, and elephant populations once again began to increase (Chadwick 1992).

10.2 1920s to 1970s – Early History of the Tarangire Elephants

The Tarangire Ecosystem has been known for its elephant population since the 1920s, when the area that is now Tarangire National Park (Tarangire NP) was a hunting area and formed part of the 'large tusker' belt in East Africa that extended from Mount Kilimanjaro down to Ruaha, where many of the largest tusked bulls had been shot (H. Lamprey pers. comm.). Indeed, during the 1950s, the Tarangire Ecosystem was well known as a reliable area to shoot bulls with 80–100-pound tusks. The northern section that is now Tarangire NP was made a game reserve in 1957 and elephant hunting continued in the reserve until 1970, when it was upgraded to a national park (Borner 1989).

The first detailed description of the elephant population in the Tarangire Game Reserve came from Hugh Lamprey, the reserve's first game warden. During the 1960s, elephants were widely dispersed across the ecosystem (Lamprey 1963), and large groups of elephants were known to exist outside the reserve, particularly to the east. Based on Lamprey's observations from 1958 to 1961, elephants used the reserve predominantly during the very dry months of September and October when they concentrated along the Tarangire River (Fig. 10.1). For the remainder of the year, the elephants used the reserve only as a temporary migratory stopover (Lamprey 1963). Estimates of elephant numbers, acquired using a combination of aerial and ground counts, suggested that some 420 elephants (both family groups and bulls) utilized Tarangire Game Reserve at the height of the dry season, while during the wet months only 20 bulls remained in the northern area of the reserve (Lamprey 1964). It should be noted that Tarangire Game Reserve covered only approximately 65% of the current Tarangire NP, as the southern section (formerly part of the Mkungunero Game Controlled Area) was not incorporated until the reserve was expanded and upgraded to a national park in 1970 (Borner 1989). Extrapolating the 1964 population estimate to the current national park area would suggest some 550 elephants may have used what is now Tarangire NP during that time.

Lamprey reported significant movement between the Tarangire Game Reserve and Lake Manyara National Park (Lake Manyara NP) to the northwest: 'The elephant herds which move at this time are those migrating from the Lake Manyara concentration area over to the…east'. This migration occurred predominantly in January, a relatively dry month that separates the short and the long rains. In 1969, Iain

Fig. 10.1 Elephants in the Tarangire River in Tarangire National Park. (Photo by Charles Foley)

Douglas-Hamilton, who was carrying out his seminal study of the elephant population in Lake Manyara NP, put a VHF radio-collar on a female from a family unit in the north of Tarangire Game Reserve. During the 3 months (May–July) that the female was collared, the group circulated across much of the northern and southwestern part of the reserve, as far as Gursi Swamp (I. Douglas-Hamilton pers. comm.). This range was much larger than current ranges used by family groups in Tarangire NP, supporting the idea that elephant groups moved widely during that period.

10.3 1970 to 1990 – The Poaching Years

The Tarangire elephant population experienced significant upheaval during the 1970s, when poaching for ivory increased dramatically, altering the range use and movement patterns of elephants in Tarangire NP and the greater ecosystem. In the early 1970s, Tanzania had one of the largest elephant populations in Africa, estimated at approximately 110,000 individuals (UNEP 1989). From the 1970s until 1989, the year an international ban was placed on all trade in ivory, poaching had decimated elephants across Tanzania, leading to the estimated loss of 55,000 elephants (TAWIRI 2010). While the heaviest poaching concentrated in the large elephant populations in central and southern Tanzania, by 1979 the poaching was also severe in the northern part of the country (Ecosystems Ltd 1980).

Aerial censuses of elephants in the Tarangire Ecosystem conducted in 1977–78 (Douglas-Hamilton 1978) and February 1980 (Ecosystems Ltd 1980) showed a large increase in elephant numbers using Tarangire NP since the estimate of 420 in 1964. The 1977–78 counts found 1342 (±484 SE) elephants in Tarangire NP, while the 1980 census estimated a figure of 2891 elephants (standard errors not provided). The likely reason for this apparent increase was high levels of immigration by elephants escaping heavy poaching outside the park (H. Lamprey pers. comm.). The 1980 aerial survey conducted by Ecosystems Ltd (1980) revealed extensive poaching across the entire Arusha Region; they reported an estimated 10,102 (±262 SE) live elephants and a staggering 5645 (±790 SE) dead elephants – a 56% carcass ratio. Elephant carcasses were distributed throughout the region, including around Tarangire and Lake Manyara NPs, and, as the document describes, "very impressively throughout Kiteto District" – an area to the southeast of Tarangire NP and now recognized as Makame Wildlife Management Area. The best information available for poaching records in the Tarangire Ecosystem comes from professional hunters (G. Hoops pers. comm. and G. Angelides pers. comm.) and safari operators (D. Peterson pers. comm.), who had concessions in this area. The general agreement is that there were constant levels of poaching throughout the 1970s and into the 1980s, and then a big upsurge in the years between 1982 and 1985 as heavily armed bands moved into the area with automatic weapons. During the 1980s, the elephant herds remained predominantly within the park boundaries, and the regular dispersal patterns to contiguous areas outside the park effectively ceased (D. Peterson pers. comm.). The elephant population in the park surged during these years, while groups living outside the park either disappeared or had drastically reduced numbers.

While the elephants were safer within the confines of the Tarangire NP boundary, they still did not enjoy complete protection and continued to be killed. Data from tusks collected within the park by rangers suggest that the heaviest poaching occurred between 1974 and 1977, when 558 tusks were collected or confiscated, with a peak of 213 tusks collected in 1974. However, these figures refer only to numbers of tusks recovered, not to how many elephants were found dead with their tusks removed, so they do not provide a complete picture of the intensity of poaching. Douglas-Hamilton (1977) reported a 46% carcass ratio in Tarangire NP and the 1980 aerial survey (Ecosystems Ltd 1980) recorded a 12% carcass ratio. Aerial surveys in 1977–78 (Douglas-Hamilton 1978) found all the live elephants were concentrated around the only operating tourist lodge (Tarangire Safari Lodge) and the park headquarters in the north of Tarangire NP, which were probably the safest areas in the park at the time. Away from these two areas, elephant skeletons were evenly distributed in the park; in 1983, carcasses were found scattered throughout the park, particularly in the more remote areas further from the Tarangire Safari Lodge and park headquarters (J. Simonson pers. comm.). In the mid to late 1980s, poaching continued at a reduced level with an estimated 20+ elephants being shot each year in the park. This decline in poaching in the park is also supported by aerial counts in the park in October 1987 (TAWIRI 1987) and May 1988 (TAWIRI 1988), which reported low numbers of carcasses detected in Tarangire NP. These counts reported concomitantly high numbers of carcasses outside the park boundary, suggesting that poaching was still occurring in the surrounding areas.

The 1989 CITES international trade ban on ivory led to a large-scale reduction in elephant poaching across Tanzania and much of East Africa (Blanc et al. 2007). This coincided with the launch in 1989 of Operation Uhai by the Tanzania Wildlife Division, during which suspected poachers were arrested and thousands of weapons confiscated across the country (TCP 1997). The combination of these events served to eliminate the majority of the elephant poaching both in and around Tarangire NP.

10.4 1990 to 2020 – A Period of Expansion

During these three decades, the Tarangire elephant population recovered from the intense 1970–1980s poaching and experienced rapid population growth and a gradual expansion of their range that incorporated newly established community protected areas outside Tarangire NP. Much of what is known about the Tarangire elephants from this period is a result of the research of the Tarangire Elephant Project, which commenced in 1993 (Foley 2002a). The project focused on the northern subpopulation of elephants (see Population Structure section, below) and examined aspects of demography (Foley 2002a; Foley et al. 2008; Foley and Faust 2010), endocrinology (Foley et al. 2001; Wasser et al. 1996), genetics (Ishengoma et al. 2008) and movement patterns (Foley 2002a; Galanti et al. 2000) of the elephant population. Since 1993, all of the female elephants and their infants in the northern subpopulation have been individually identified using photographic files of ear markings and monitored intensively (Foley and Faust 2010). Adult bulls in the northern subpopulation were first photo-identified in 1998 and monitored regularly until 2004 and sporadically after that. Information on the central subpopulation was gathered during regular assessments of the family groups, while data on the southern subpopulation were gathered on an *ad lib* basis when members of the subpopulation were seen.

Box 10.1: Elephants: A Closer Look

The African savanna elephant (*Loxodonta africana*) is the largest land animal, weighing up 3.2 tons for females and 6 tons for males (Poole et al. 2013). Elephants are easily recognizable by their long prehensile trunk, large ears, and modified upper incisors that form tusks (Poole et al. 2013).

Elephants live in a multi-tiered social system: the core is a basic mother–offspring unit that expands outwards through family groups, bond groups, clans and subpopulations (Moss 1981; McComb et al. 2001; Moss and Lee 2011a, b). The standard female social unit is a relatively stable family group of between two and 50 individuals (Moss and Lee 2011a, b) composed of related adult females and their immature offspring (age < 10 year) (Douglas-Hamilton 1972). They are led by a matriarch (Laws 1969), generally the oldest and largest female within a family group, that dictates the group's movements and activities (Douglas-Hamilton 1972; Moss 1988). Elephants do not exhibit a restricted breeding season, and estrus females may be

Box 10.1 (continued)

observed in any month of the year. There are, however, peaks of frequency of estrus during and shortly after the rains, and in areas of highly seasonal rainfall the great majority of breeding takes place at this time (Laws 1969; Moss 2001; Foley and Faust 2010). Female elephants in the wild live to over 60 years of age (Moss 2001), while reproduction generally ceases or declines markedly in their 50s (Moss and Lee 2011a).

Females rarely disperse from their natal group and have strong, long-lasting affiliative relationships, particularly between mother–daughter pairs and sisters (Douglas-Hamilton 1972). Family groups may fission to form bond groups of two or three family groups (Douglas-Hamilton 1972; Moss and Poole 1983), which display particular behavioral patterns towards each other such as strong greeting ceremonies and protective behavior (Moss and Poole 1983). Family groups sharing predictable home ranges during the dry season are called clans (Douglas-Hamilton 1972; Moss 1988; Foley 2002a), while several clans form a subpopulation of family groups that share broad wet season ranges (Moss and Poole 1983). Different subpopulations may overlap, particularly in the wet season, but have different core home range areas and often have separate migration patterns and dispersal areas. Two or more subpopulations, and the adult males, comprise the full population. Other studies have described only four social tiers (Wittemyer et al. 2005; Wittemyer and Getz 2007). The term ‘subpopulation’ used in this chapter is most consistent with the fourth social tier described in (Wittemyer et al. 2005).

Male elephants typically leave their family group between the ages 9–14, staying close to their family group at first, and then gaining independence as they reach sexual maturity around age 28 (range 17–34 years) (Lee and Moss 1999; Poole et al. 2011). During periods of heightened sexual activity known as ‘musth’, mature bulls move large distances in search of females in estrus (Poole 1987). Musth typically first occurs between the age of 25–30 (Poole et al. 2011). When they are not in musth, males typically aggregate in ‘bull areas’ (Moss and Poole 1983). Independent males are either solitary, or they associate with consistent groupings of 2–20 other males, and occasionally associate temporarily with female groups (Poole 1987).

Elephants of different subpopulations may aggregate into large herds (300–500 individuals) during the rainy season, and then separate into smaller social units as the dry season progresses and food supplies diminish (Poole and Moss 1989). These large social congregations help facilitate the location of estrus females by sexually active bulls, and are also thought to lead to the re-establishment of inter-group dominance hierarchies and social bonds (Western and Lindsay 1984; Moss 1988).

Family group home ranges vary considerably depending upon the quality of forage available, varying in size from less than 50 km^2 (Douglas-Hamilton 1972) and up to 8700 km^2 (Lindeque and Lindeque 1991).

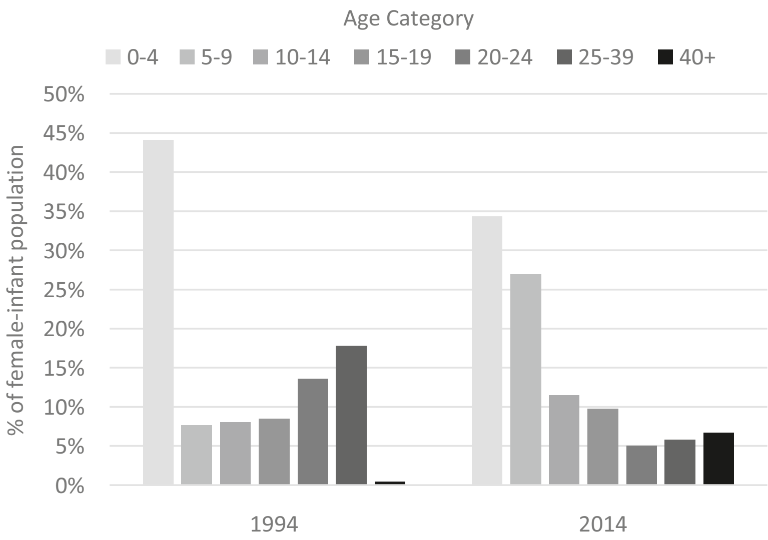

Fig. 10.2 Age structure in Tarangire NP's northern subpopulation (adult females and calves) compared in 1994 and 2014. The age structure in 2014 shows a more mature and normal age structure after 20 years of growth and protection

In the early 1990s, the demographic structure of the population showed clear signs of the heavy impact of poaching during the previous two decades. Poaching is a selective process that targets bulls and older females for their larger tusks, with poached populations showing skewed age and sex structures in favor of younger females (Poole 1989b; Jones et al. 2018). In 1994, less than 2% of the adult females (age >10 year) in the population were over 40 years old. Of the 32 family groups monitored at that the time, just 46% had matriarchs over the age of 30. Seven family groups were comprised of only a single adult female, and six of those had only one adult female over 30 years of age. While 44% of the population in 1994 was under 5 years of age, indicating a recent birth surge, only 8% of the population was in the 5–9.9 year age category (Fig. 10.2), suggesting there had been either low recruitment, high mortality, or both, in that age group, which would have been born in 1985–1989. With poaching pressure essentially eliminated for this subpopulation in the late 1980s, there may have been a delayed effect of several years before reproductive effort increased, accounting for the low number of animals in the 5–9.9 year age class. The Tarangire bull elephants had been similarly affected: in 1994, a demographic census from the north of Tarangire NP found that only 17% of adult bulls (age ≥15 year) were over age 25, which is typically the earliest breeding age for bulls (Poole 1989a). By contrast, in Amboseli NP (Kenya), where little poaching occurred, 50% of adult males were over 25 years in 1989 (Poole 1989a).

During the next decade elephant reproduction surged. By 2005 the northern subpopulation was increasing at close to maximal rates, with interbirth intervals of 3.3 years, age of first reproduction at 11.2 years, and mortality levels extremely low at 1% and 2% for adult females and calves, respectively (Foley and Faust 2010).

The average annual growth rate was 7.1% (Foley and Faust 2010). In the first decade of the study, 5% of infants born were twins (Foley 2002b). Four years of very high rainfall between 1998 and 2001 undoubtedly contributed to the rapid population expansion of the Tarangire elephants. Vegetation growth during this period was intense, providing optimal feeding conditions for elephants, which allowed them to feed on grasses throughout the year. This in turn meant that they were in peak physical condition, allowing extremely rapid postpartum recovery and consequently very short interbirth intervals. Since 2010 the growth rate has reduced, probably due to density dependent effects impacting the population (Foley et al. in prep).

With poaching at negligible levels and a high growth rate, the age structure of the population has changed substantially over the past 25 years. In 2014, 17% of adult females were over age 40, and the age structure (Fig. 10.2) now resembles that of other mature populations (Laws et al. 1975; Moss 2001). All family groups now have matriarchs aged over 30 years, and there are no longer any single adult female groups in the subpopulation. Mean group size (defined as adult females and offspring age <10 year) increased from 8.7 (±0.7 SE, range 3–17) in 1994, to 29.9 (±2.6 SE, range 10–65) by 2014. With the increase in older animals, mortality from natural senescence is now being increasingly recorded in the population. The number of older bulls over age 25 in the population also increased from 17% in 1994 to 41% in 2004, demonstrating the gradual aging of the bull population.

Aerial total counts of elephants in the Tarangire Ecosystem show that the total population expanded from approximately 2300 individuals in 1995 to 4200 in 2014, the year of the most recent aerial total count (TAWIRI 2015). The survey area includes Tarangire NP and Lake Manyara NP (which has a small population of approximately 250–270 individuals, Kiffner et al. 2017), as well as the surrounding community Wildlife Management Areas (WMAs) and Manyara Ranch. The elephant population of the Tarangire Ecosystem in 2020 is now likely to be over 5000 individuals.

Underpinning this expansion and growth is the high level of security experienced by the elephants in the Tarangire Ecosystem. Since the 1989 international ivory trade ban, there has been very little elephant poaching in the Tarangire Ecosystem and generally in northern Tanzania (TAWIRI 2010; TAWIRI 2015). This contrasts with the heavy poaching that inflicted parts of central and southern Tanzania starting in approximately 2007 (TAWIRI 2010). Between 2009 and 2014 Tanzania's elephant population declined from 109,051 (±5899 SE) to 43,521 (±3078 SE) – a 60% decline (TAWIRI 2015), despite population increases in the north of the country.

The Tarangire Ecosystem has several advantages for elephant protection when compared to protected areas in the central and southern parts of Tanzania: (a) the two national parks (Tarangire and Lake Manyara) are relatively small and easily patrolled by park rangers, (b) the area is on the popular northern safari circuit and attracts many tourists with tourism-related infrastructure, which makes it difficult for poachers to operate unnoticed, and (c) a large network of community scouts patrol WMAs and community lands adjacent to Tarangire NP. These factors all contribute to Tarangire having one of the best protected elephant populations in Tanzania.

In the Tarangire Ecosystem, recent poaching incidences have been restricted to small, sporadic outbreaks on community lands adjacent to the national parks, with elephants either being shot or poisoned. The Tarangire Ecosystem (including

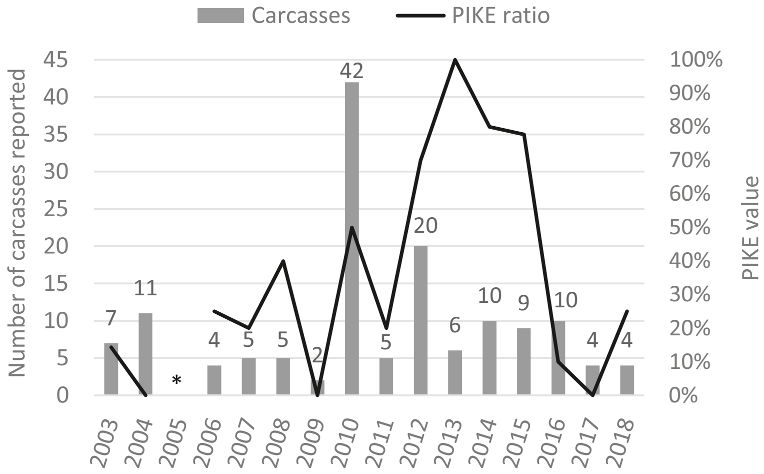

Fig. 10.3 Total number of elephant carcasses found in the Tarangire MIKE monitoring site between 2003 and 2018 and the corresponding PIKE (proportional of illegally killed elephants) values. A PIKE value of 1.0 (100%) indicates all carcasses found were illegally killed. *No data were available for 2005 (CITES 2018)

Tarangire and Lake Manyara NPs) is part of the CITES Monitoring of Illegal Killing of Elephants (MIKE) program, which monitors the elephant mortality and the illegal killing of elephants in range states (CITES 2020). This program uses a ratio called PIKE (Proportion of Illegally Killed Elephants) to evaluate trends in poaching at each monitoring site. A PIKE ratio divides the number of illegally killed elephants found by the total number of carcasses found (including natural mortality and legal killing). While overall PIKE data from the Tarangire Ecosystem between 2003 and 2018 are low, there was an increase between 2010 and 2015, with 21 elephants poached in 2010 and 14 in 2012, and a total of 57 elephant poached between those 5 years (Fig. 10.3). As not all poaching carcasses are discovered, these numbers must be viewed as minimum figures, and carcasses in dense and isolated parts of the ecosystem, such as Makame WMA, are likely to have been missed. After 2015, PIKE figures again declined to very low levels. The number of elephants lost each year to poaching is greatly outweighed by the number of annual births, and the impact on the overall population size is therefore small. Remarkably, there are no records of any individuals from family groups in the northern subpopulation, which have been the focus of the Tarangire Elephant Project, having been poached in the past 25 years.

10.5 The Population Structure of Tarangire's Elephants

The elephant family groups (defined as adult females and calves <10 year, see Box 10.1) in the Tarangire Ecosystem can be divided into three distinct subpopulations (northern, central and southern), based on different ranging and association patterns during the wet season (Fig. 10.4, Foley 2002a). This most closely equates to

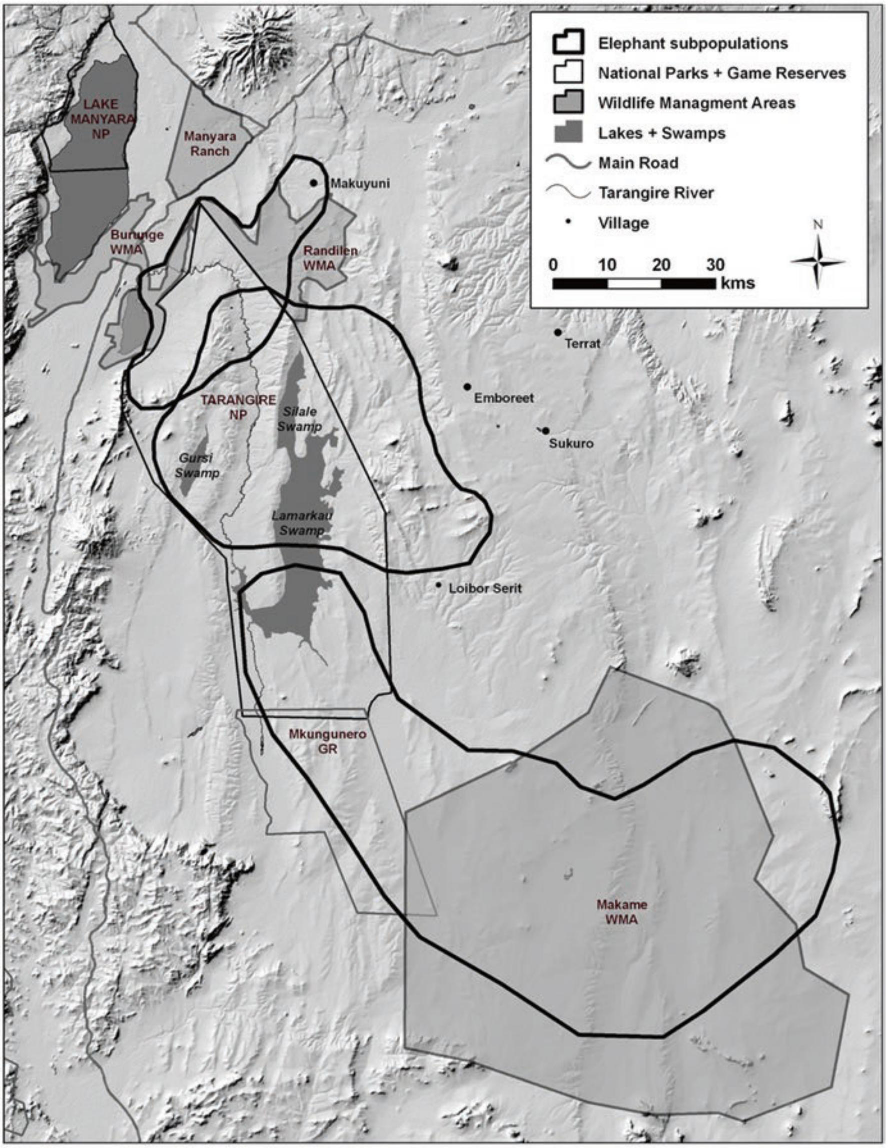

Fig. 10.4 The approximate core ranges of the three subpopulations of family groups (northern, central, and southern) in Tarangire National Park, based on GPS radio collar data, aerial counts and ground observations

the 'fourth social tier' described in Wittemyer et al. 2005, where groups may only consistently aggregate in their entirety during the wet season. The northern and central subpopulations may overlap during the wet season, particularly along the boundaries of their range (Foley 2002a), and family groups from these two subpopulations have been recorded on several occasions moving as one group of over

600 individuals in the wet season. Little is known about interactions between the central and southern subpopulations, although their dry season ranges overlap in Silale Swamp.

Direct interactions between family group members of different subpopulations are frequently antagonistic, particularly during the dry season, and often involve high levels of aggression and prolonged stand-offs (Foley 2002a). Evidence of coalitionary support between fellow subpopulation family group members has been observed, where dominant females were seen to retaliate to protect fellow subpopulation members that had been supplanted by females from a different subpopulation. These reprisal attacks could be severe in their intensity, leading to chases of over 80 m. Females in family groups led by younger, less dominant matriarchs thereby receive a level of protection from groups with more dominant matriarchs within their subpopulation. Episodes such as this, characterized by intense aggression between family groups, likely structure movement patterns of family groups over extended periods of time.

10.6 Northern Subpopulation

The core range of the northern subpopulation covers approximately the northern 20% of Tarangire NP (Fig. 10.4). During the dry season, their movements are confined mostly to a 700 km^2 area within the park and the Burunge WMA, predominantly remaining southeast of the Arusha–Dodoma highway. In the wet season, they disperse eastwards outside Tarangire NP to the Randilen WMA and concentrate in a communal grazing area on Makuyuni village land. At this time of the year the total range of the northern subpopulation increases to approximately 900 km^2. During this period, the community members remove their livestock from the communal grazing areas to avoid the risk of running into elephants, and they return with their livestock during the dry season when the elephants have returned to the park. The elephant family groups often spend 1–2 months outside the park at the height of the wet season (March–May), although some may return periodically to the park. This seasonal movement onto community land mirrors that of other large ungulates in the Tarangire Ecosystem, which migrate to dispersal areas with high mineral and crude protein during the wet season and concentrate around the Tarangire River within the national park during the dry season (Voeten 1999; **Bond et al.** Chap. 8; **Lohay et al.** Chap. 12). The causal factors underpinning the seasonal elephant migration have not been studied but it is likely to be driven by similar ecological influences.

Elephants from the northern subpopulation, which has been the focus of the Tarangire Elephant Project since 1993, have experienced high frequencies of interactions with tour vehicles and low levels of human harassment and are typically relaxed and exhibit normal grouping behavior. While there has been some group fission, fusion, and emigration of the family groups over time, no new family groups have immigrated into the northern subpopulation since the start of the study. The social dynamics in Tarangire NP mirror those described by Moss (1988) in Amboseli National Park, Kenya, as being relatively stable, but not unchangeable, with family groups able to join other subpopulations, although generally suffering high levels of aggression before being accepted. It is unclear why some family groups, or parts of family groups, have emigrated, although it is possible that this has been driven by increased competition for resources as group sizes have increased.

10.7 Central Subpopulation

The family groups of the central subpopulation range between Silale Swamp in the north, Lamarkau Swamp in the south, and Gursi Swamp in the west (Fig. 10.4). During the dry season they concentrate around the Tarangire River and Silale Swamp. In the wet season they disperse onto village land in Lolkisale, Emboreet, Loibor Siret, and more recently to Sukuro and Terrat villages, which are approximately 40 km east of the park. During this time of the year they also disperse south to Lamarkau Swamp, creating a total range of approximately 2200 km^2. The behavior of family groups in this central subpopulation reflects that of the northern subpopulation, with groups moving in small herds that are familiar with and comfortable around vehicles. The family groups in the central subpopulation share similar population dynamics and demographic patterns with those in the northern subpopulation, with similar birth pulses and infant-to-mother ratios, and their population growth rate probably closely mirrors that of the northern subpopulation. While overlap between the central and northern subpopulations is frequent during the wet season, little is known about interaction between the central and southern subpopulations.

10.8 Southern Subpopulation

In contrast to the northern and central subpopulations, the southern subpopulation only utilizes Tarangire NP during the dry season when the waterholes outside the park have dried up. For the remainder of the year, they reside in thick bushland in what is now Makame WMA, about 75 km southeast of Tarangire NP (Fig. 10.4). The Makame WMA has a varied habitat, with open swampland, lightly wooded grassland, as well as very dense thicket. The thicket is characterized by tightly packed stands of *Grewia spp.*, *Vachellia* [formerly *Acacia*] *mellifera*, and *Commiphora spp.,* interspersed with *V. tortilis* and *Erythrina spp.,* parts of which are impenetrable to humans. This provides useful cover for the elephants and allows them to mostly avoid interactions with humans. During the dry season the elephants move to the Mkungunero area on the southern border of Tarangire NP, concentrating their activity around two large waterholes in this area. These waterholes would disappear during dry years, although they were re-excavated in 2017 and now typically retain water throughout the dry season. In former years, when the Mkungunero waterholes dried up, the family groups of this subpopulation would move to Silale Swamp in central Tarangire. These family groups have the largest range of the population, covering approximately 4500 km^2. The northern part of their range in Silale Swamp overlaps with that of the central subpopulation; there are no records of them overlapping with the northern subpopulation.

During the early 1990s the southern subpopulation experienced continuous low levels of poaching and harassment from humans, and their behavior was correspondingly different from that of the other two subpopulations (Foley 2002a). When the elephants were in the Tarangire NP they commonly formed large aggregations of several hundred individuals that congregated in Silale Swamp. The mean group size of the southern subpopulation in the swamp from August to November 1994–1996 was 217 individuals (±7.3 SE) (Foley 2002a). By contrast, mean group size in the northern subpopulation for the same months was only 15.3 animals (±1.75 SE; $t = 26.96$, $p < 0.0001$) (Foley 2002a). The aggregation of elephants remained in Silale Swamp throughout the day, either feeding on grass tubers or resting, and only leaving at sunset to feed and drink in the Tarangire River valley or on the ridges. Such aggregation behavior is a typical response to heavy poaching or harassment by humans (Poole 1989b), and has been recorded in several other parks, including Murchison Falls, Uganda (Laws et al. 1975), Queen Elizabeth National Park, Uganda (Poole 1989b) and the Serengeti (Dublin and Douglas-Hamilton 1987). In Tarangire, this behavior has been observed mostly in areas of open grassland or swamp, where it is easy for large herds to move synchronously, although aerial monitoring found that the animals also remained in large groups of up to 300 individuals in the thicket areas outside the park (Galanti et al. 2000). Data from three radio-collared elephants from November 1997–April 1998 support the theory that the elephants of this subpopulation adjusted their behavior to avoid interactions with humans. The elephants hid in denser vegetation, moving very little during the day, and were most active at night, with a peak at 2:00 am (Galanti et al. 2000). When the elephants traveled to and from the park, they moved rapidly at night covering almost 15 km in 10 h (Galanti et al. 2000) and avoided areas of human settlement and agriculture. Two of the radio-collared elephants were poached in the Makame area during the course of the study, indicating that the poaching threat was acute at that time.

The southern subpopulation formerly exhibited markedly lower infant to adult ratios than the northern subpopulation. In October 2000, two large aggregations of the southern subpopulation (of 103 and 41 animals) were seen closely enough so that all individuals could be accurately aged and sexed. Analysis of the age structure showed that the ratio of infants under five to adult females (1.25 ± 0.07 SE, $n = 52$ adult females) was significantly lower than that of the northern subpopulation (1.44 ± 0.03 SE, $n = 161$ adult females; $z = -2.44$, $p < 0.01$) (Foley 2002a). Additionally, the ratio of infants less than 10 years old to adult females was also lower in the southern subpopulation (1.57 ± 0.05 SE, $n = 52$) than in the northern subpopulation (1.86 ± 0.03 SE, $n = 161$; $z = -4.4$, $p < 0.0001$) (Foley 2002a). With no obvious differences in resource abundance and quality, the disparity in infant-to-mother ratio between the subpopulations was probably the result of the different levels of human-induced stress to which each subpopulation was exposed. There is good evidence that the stress that accompanies poaching suppresses recruitment; demographic surveys across elephant populations in East Africa have shown that breeding rates, indicated by the ratio of dependent calves to adult females, were inversely correlated with levels of poaching (Poole 1989b; Jones et al. 2018). Lower recruitment may be the result of fewer breeding opportunities as the number of breeding males is often low in heavily poached populations (Poole 1989b; Foley 2002a), or because disruption in group leadership and altered movement and feeding behavior increases calf mortality (Barnes and Kapela 1991; Foley 2002a; Gobush et al. 2008). When inside the park, the southern subpopulation would remain in the middle of the swamp throughout the day, potentially subjecting the infants to high levels of heat stress. These human-avoidance techniques, combined with the long distances that the southern family groups cover during their annual migration, may have put higher strain on infants leading to increased mortality, or prevented mothers from foraging optimally and thus delayed cyclicity.

Since 2002 there have been few sightings of large aggregations of the southern subpopulation, and evidence from group sightings suggest that they are reverting to more traditional grouping patterns of smaller family units. This could suggest that poaching levels of this subpopulation have declined. In 2003, the core area of the southern subpopulation was protected as a Wildlife Management Area (Makame WMA), and community members have been hired as game scouts to patrol the area. Evidence from a 2014 aerial count (Burgess et al. 2016) and a 2018 camera trap survey (Foley et al. 2018b) in the area did record elephant carcasses, particularly near waterholes, but most of the carcasses were old, suggesting that elephant poaching in the area was by then infrequent. An aerial reconnaissance flight over the Makame WMA in 2005 recorded over 500 elephants, suggesting that this was a minimum population estimate for the subpopulation at the time. This is significant as it is one of the largest resident elephant populations on community land in Tanzania. Nocturnal sightings of family groups drinking at the Mkungunero waterholes from 2015 onwards recorded substantial numbers of infants and juveniles, and while sightings have not been sufficiently reliable to determine accurate age structure, all evidence suggests that the population is growing. There is abundant forage for elephants in the Makame WMA area, and the principal factor limiting future elephant movement and growth is the extent of suitably dense stands of *Grewia–Commiphora* thicket, which likely provides a measure of protection against human interference.

10.9 Bull Elephants in Tarangire

There are two separate subgroups of bull elephants in Tarangire NP. The northern subgroup (comprised of approximately 223 bulls over age 15 in 2004) is commonly found in the north between the Tarangire River, the park headquarters, and, more recently, on Manyara Ranch and Burunge WMA (see Fig. 10.5). These bulls

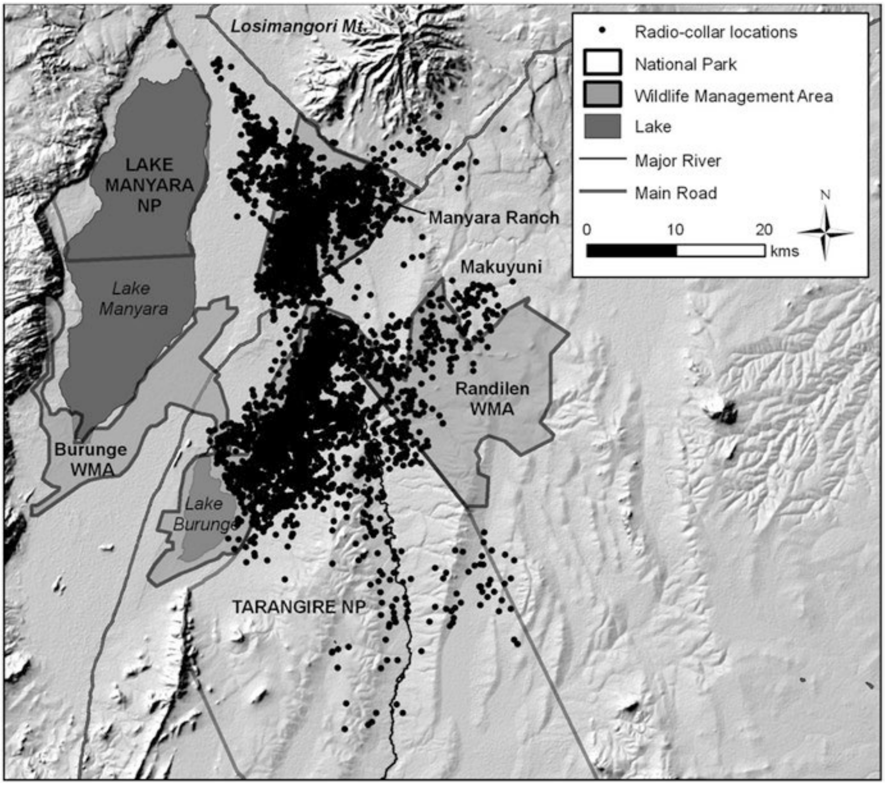

Fig. 10.5 Ranging patterns of three bulls in northern Tarangire, based on satellite collar data (2006–2008). Note the importance of Manyara Ranch, Randilen WMA, and Makuyuni village land in their ranging patterns

regularly interact with the northern and central female subpopulations. The northern bulls are increasingly mobile with some males spending 6 months or more outside Tarangire NP. The second bull subgroup appears to be restricted to the southern half of the park – south of Kuro ranger post and Gursi Swamp. The size of this subgroup is unknown. There are no records of these individuals mixing with the northern subgroup of bulls, although how separation is maintained is still unclear. It is likely that these bulls disperse and breed primarily with the southern subpopulation of family groups, either when they are in Makame WMA or in the south of the park, although they also overlap with and are likely to breed with females from the central subpopulation. Until the early 1980s, Tarangire's Mkungunero area was an important bull dispersal area, with large numbers of males using the thicket during the dry season (G. Hoops pers. comm.). This is no longer the case, with these bulls congregating either further north or in the Makame WMA area.

10.10 Elephant Migration and Conservation

During the early 1990s, following many years of poaching, elephant populations across the Tarangire Ecosystem were restricted to the areas where they felt safe: the two main national parks, Tarangire and Lake Manyara, and the Makame area. There was also a very small population resident in the forests on Losimangori Mountain. The large-scale migration of elephants in the 1960s described by Lamprey (1963) between Lake Manyara and the area to the east of Tarangire NP had ceased. When elephants moved to dispersal areas outside the national parks they would travel mostly at night and conceal themselves in thick bushland during the day. This situation started changing in the early 2000s as new protected areas were established on community land within the ecosystem, which led to elephants expanding their range.

The earliest range expansion was to Manyara Ranch, an area between Tarangire and Lake Manyara NPs that covers nearly 180 km^2 (Fig. 10.5). The area was formerly run as a government-owned cattle ranch, but in 2001 the ranch was converted into a community-owned land trust by the Tanzania Land Conservation Trust (TLCT) with the help of the African Wildlife Foundation (AWF 2003). Under this new management, livestock use on the ranch was reduced and an anti-poaching unit established. Prior to 1998, the ranch had been used only sporadically by elephants. The ranch manager (F. Shangama pers. comm.) had seldom seen elephants on the ranch but occasionally found elephant spoor and believed that elephants were coming down from the neighboring Losimangori Mountain for nocturnal visits. A PhD student conducting field research on Manyara Ranch between 1994 and 1996 never recorded any elephant sightings there during her research (M. Voeten pers. comm.). By 1998, bull elephants from northern Tarangire NP started using the Ranch and small groups of up to 15 bulls soon established semi-residency there. The ranch has increasingly become part of the core range of many northern Tarangire bulls (Fig. 10.5), and groups of 10–20 males are now frequently seen on the ranch (Foley et al. 2018a). Elephant family groups use the ranch less frequently, although one female and her offspring from one of the northern subpopulation family groups moved to the ranch and are now mostly full-time residents. Bulls that are now resident on Manyara Ranch have also started to venture further afield and have begun utilizing a large, unused farm (Stein's land) on the southeast slopes of Losimangori Mountain (see Fig. 10.5). The bulls remain on the upper slopes of the farm during the day and sometimes move at night to drink from a waterhole south of the Arusha–Dodoma road.

Movement between Tarangire and Lake Manyara NPs appears to be infrequent. In 2007, one radio-collared bull traveled from Manyara Ranch into Lake Manyara NP along the northern edge of the lake, although only stayed for 3 days in Lake Manyara NP (see Fig. 10.5). A Lake Manyara NP bull was photographed in Tarangire NP in 2000, indicating that the movement occurs in both directions. The bull was only seen once in Tarangire NP, suggesting it only remained there for a short time.

The establishment of the Burunge WMA in 2006 and Randilen WMA in 2012 led to greater elephant use of both of these areas, particularly by bulls. Sightings of bulls started in late 2014 when they were regularly seen during the day in the Lake Burunge area of Burunge WMA (R. Tosi pers. comm.). Prior to this, the area was a hunting block, which included hunting bull elephants, so elephants used this area sporadically. Initially it was only a few of the larger, older bulls that moved into the Lake Burunge area, although over time more bulls were encountered there (R. Tosi pers. comm.). Elephant family groups started using the Lake Burunge area in 2016, with sightings becoming frequent in 2018. In 2017, bull elephants were first sighted north of the Arusha–Dodoma road in the Burunge WMA, which was rarely used in the past; family groups are now also occasionally present in that area (R. Tosi pers. comm.). Similarly, elephants from the northern subpopulation can now be found year-round in Randilen WMA. Bulls started using the Randilen area in 2006 prior to it becoming a WMA, although they would typically move in at night and return to Tarangire NP during the day. Bulls started using the area regularly during the day in 2009 and family groups by 2012 (G. Dennis pers. comm.). In 2019, for the first time in decades, a small group of elephants was seen in Terrat village in the Simanjiro District, and small groups of elephants continued to return to Terrat and Sukuro villages regularly in 2020 (E. Loure pers. comm.). The general, relatively recent expansion of the Tarangire elephants' range onto community-protected land has been driven in part by increasing anti-poaching activities by community game scouts. Within the WMAs and village lands in the Simanjiro District, there are now approximately 100 community game scouts operating across the ecosystem, as well as units from the Tanzania Wildlife Authority (TAWA), which serves to greatly reduce the threat of poaching in the area.

Unlike other large mammals such as plains zebra (*Equus quagga*), wildebeest (*Connochaetes taurinus*) and common eland (*Taurotragus oryx*), that migrate large distances on a seasonal basis across the ecosystem (TCP 1997; Morrison et al. 2015; Foley et al. 2018a; **Bond et al.** Chap. 8; **Lohay et al.** Chap. 12), the northern and central elephant subpopulations typically disperse to village lands directly adjacent to the national park (see Fig. 10.4). Many of these seasonal dispersal areas have now been protected through the establishment of new WMAs and Manyara Ranch. Wildlife, including elephants, have also benefited from communities securing their own communal grazing areas for pastoralism and other traditional activities through the use of community CCROs (Certificates of Customary Right of Occupancy). CCROs typically restrict settlement and agriculture to prevent further loss of open rangeland, and while their principal purpose is to safeguard community livestock practices, they also serve to maintain connectivity within the ecosystem which benefits migratory wildlife.

Most of the migration of the southern elephant subpopulation is now protected by the Mkungunero Game Reserve (established in 1996) and the Makame WMA, and the dispersal areas of the central subpopulation are largely covered by two CCROs established adjacent to Tarangire NP in Loibor Siret and in Emboreet villages. The range of northern subpopulation is mostly protected by the Burunge and Randilen WMAs and a CCRO in Makuyuni village. The elephant corridor that is now most at risk is the 'Kwa Kuchinja' corridor, a narrow strip of land that connects Tarangire NP with Manyara Ranch. This corridor is currently predominantly used by bull elephants, but also extensively by wildebeest, zebra, and other large ungulates migrating northwards in the ecosystem (Morrison et al. 2015) as well as by resident Masai giraffes (*Giraffa camelopardalis tippelskirchi*) whose ranges incorporate both Manyara Ranch and TNP (Lee and Bolger 2017; **Lee and Bond** Chap. 9). There is now extensive agriculture within the corridor area, and while a small CCRO established in Mswakini Chini village provides some connectivity between the two areas, the window of opportunity for creating a sustainable functional corridor between these two areas is rapidly disappearing.

While most annual elephant migration occurs in areas adjacent to Tarangire NP, evidence from drought periods suggests that access to more distant parts of the ecosystem may be critical. During a severe drought between 1992 and 1994, northern elephant family groups that remained in Tarangire NP suffered significantly higher infant mortality than groups that migrated out of the park (Foley et al. 2008). Calf survival in the latter groups was no different to an ordinary year, indicating that the groups that migrated out of the national park were able to access vital resources of adequate food and permanent water. The destination of the migrating elephants was not determined at the time of the study, although it must have offered sufficient food and water to support large numbers of elephants for several months. Areas with permanent water in the ecosystem include Lake Manyara NP and Losimangori Mountain to the north and the Ruvu River to the east. The impact of rare events such as severe droughts, which are likely to increase with climate change, highlights the long-term risks to Tarangire's elephant population if the national park were to be isolated from surrounding community areas.

In addition to the local migrations described above, there are two other historical, long-range corridors that help maintain important genetic connections for Tarangire's elephants; these corridors connect the Tarangire Ecosystem to the Ngorongoro highlands and the Ruaha–Rungwa ecosystem. Genetic evidence suggests that there has been relatively little movement and genetic exchange between elephant populations in Tarangire NP and the Ngorongoro Conservation Area (NCA) (Lohay et al. 2020; **Lohay et al.** Chap. 12). Ground surveys conducted in 2015 suggest a corridor may still link the two protected areas via Losimangori Mountain and Selela Forest Reserve (Chlebek and Stalter 2015). Fresh elephant dung and spoor were found during the wet season (March–April) in the area between Manyara Ranch and Losimangori Mountain and also between Selela Forest Reserve and the NCA. Only one dung sample was found in the putative corridor between Losimangori Mountain and Selela Forest Reserve, but local people confirmed seeing elephants moving in that area towards Selela from the south (Chlebek and Stalter 2015). While it is possible for an individual elephant to move from Tarangire NP to the Ngorongoro Conservation Area, it is not known whether any animals complete this entire route or whether the Manyara Ranch–Losimangori–Selela corridor is used primarily by the small population of elephants on Losimangori Mountain. The Upper Kitete corridor linking Selela with Ngorongoro is surrounded by farms on each side and will require concerted protection to remain viable.

Elephants in the Tarangire Ecosystem share a high proportion of genetic material with those in the Ruaha Ecosystem (Epps et al. 2013; Lohay et al. 2020), suggesting that regular movement occurred between the two populations in recent times. There is now extensive agriculture and settlement between these two areas, although a potential corridor still exists from Tarangire via Swagaswaga Game reserve (east of Babati) and then on to Rungwa Game Reserve, in the northern part of the Ruaha-Rungwa Ecosystem. In 2010, a team from the Tanzania Wildlife Research Institute (TAWIRI) and the Wildlife Conservation Society (WCS) surveyed the area between Mkungunero Game Reserve (abutting the southern boundary of Tarangire NP) and Swagaswaga Game Reserve. The survey team collected information on elephant sign and conducted interviews with village leaders and local community members in the area. They determined the main corridor linked Mkungunero Game Reserve to the Chirimo Hills, where elephants would spend a few weeks a year in the dense forest of the hills, before moving to Kelema village and on to Swagaswaga Game Reserve. Villagers reported that most movement occurred during the dry season. Elephants still used the route every year, but parts of the corridor were partially or fully severed by agriculture or settlement, and the team concluded that the route would soon be closed due to high levels of human encroachment (TAWIRI 2010). It is unknown if the corridor is still functioning. The extending corridor between Swagaswaga and Rungwa Game Reserve was listed as being in ‘Critical’ condition in 2009 due to anthropogenic impacts (Jones et al. 2009). Given these circumstances, it is likely that migration between the northern and southern parts of Tanzania has now ceased or occurs rarely.

10.11 Human-Elephant Interactions

The recent increase in elephant movement outside Tarangire NP onto community lands has led to an increase in human-elephant conflict (HEC) across the ecosystem (Warlick 2006). Agriculture in the area is dominated by small-scale farmers producing crops for their own consumption or for local sale (Meing’ataki 2005), and the majority of farms are not fenced. HEC is most pronounced in areas directly adjacent to the park where farmers have cultivated up to the park boundary – this is most acute along the northern border, the southwestern boundary, and around Sangaiwe Hills on the western boundary. Conflict has been exacerbated by an increase in agriculture within elephant migration corridors, such as the Kwa Kuchinja corridor that links Tarangire NP with Manyara Ranch. While crop raiding represents the greatest source of conflict, one study reported that elephants destroying and eating the grain inside storage shelters accounted for 15% of HEC incidences around Tarangire NP (Meing’ataki 2005). Crop raiding by elephants peaks in July when crops are ready to be harvested (Warlick 2006). Anecdotal evidence suggests that bulls are more likely to crop raid than family groups, while ‘problem animals’ that specifically target grain stores are typically large males. Most of the crops grown are staple foods such as maize and beans, which may be the farmers’ primary source of food and sole source of income. Elephant raiding of crops or grain stores can therefore severely affect food security for farmers living adjacent to Tarangire NP. HEC also causes resentment towards the national park, which is often perceived as doing too little to ameliorate the situation. Because HEC occurs across a very wide area in communities both close and far from the park boundaries, it becomes very difficult for park authorities and wildlife agencies to respond to all HEC events. Several mitigation efforts have been trialed in the area, including the use chili fences (Warlick 2006; Chang'a et al. 2016), torches and firecrackers (Honeyguide 2021) and small drones (Hahn et al. 2017) to either prevent or repel raids by elephants. With Tarangire's elephant population still expanding both in population size and range, HEC is likely to continue to increase in the ecosystem and amelioration efforts will necessitate significant cooperation between the wildlife authorities and local communities (Niskanen 2009).

10.12 Conclusion

Formerly distributed widely across the Tarangire Ecosystem, heavy poaching of the elephant population in the 1970s and 1980s led to elephants seeking refuge in the safety of the national parks, greatly reducing their range in the ecosystem. Poaching also created a highly altered demographic structure, with a skewed sex ratio with few breeding bulls and low numbers of older animals. With elephant poaching virtually eliminated in the ecosystem since the international trade ban on ivory in 1989, the population has grown rapidly, group sizes have increased, and the population has aged, and the current demographic structure now reflects that of a mature, undisturbed population. Elephants have expanded their range to include several new community protected areas that have been established in the ecosystem since 2000, and they are now being observed in areas far outside Tarangire NP, where they have not been seen for many years. The core wet season dispersal areas of all three elephant subpopulations now have some level of community protection, either through WMAs or CCROs, which augurs well for the long-term sustainability of the population.

As the population continues to grow and to expand its range, conflict with humans is likely to become increasingly acute and will need to be addressed. Loss of connectivity with other elephant populations, particularly to the south, may also have long-term impacts on genetic variability within the Tarangire elephant population. Overall, however, the present status of the elephant population in the Tarangire Ecosystem represents a remarkable transformation from the late 1980s when the population had been decimated by poaching, and exemplifies how elephant populations can stage rapid recoveries when provided with adequate protection. This represents a true conservation success story.

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Chapter 11
Large Carnivores in the Tarangire Ecosystem

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Abstract We synthesize data on the ecology of large carnivores in the Tarangire Ecosystem (TE). Despite anthropogenic pressures, all large carnivore species (lions *Panthera leo*, spotted hyena *Crocuta crocuta*, striped hyena *Hyena hyena*, leopard *Panthera pardus*, cheetah *Acinonyx jubatus*, and wild dog *Lycaon pictus*) have persisted in this fragmented ecosystem consisting of multiple protected areas among a matrix of village lands. The focal species were widely distributed across land-use gradients. While the comparatively abundant spotted hyena permanently occupied village lands, other species only sporadically used these human-dominated areas. Across species, carnivores used village lands more frequently during the rainy season, possibly following seasonal shifts in the movement of prey species. These processes can increase human-carnivore interactions, expanding the potential for conflict. In some areas, leopards, lions, and striped hyenas reached high densities, whereas cheetahs and wild dogs occurred patchily and at low densities. Our review suggests that the existence of diverse protected areas contribute to the persistence of the large carnivore community. The persistence of lions, cheetahs, and wild dogs appears dependent on human-induced mortality and prey depletion. Conserving

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large carnivores in TE requires the application of interventions that reduce human-induced mortality while simultaneously conserving the spatio-temporal distributions of prey species.

Keywords Carnivora · Human-carnivore conflict · Livestock depredation · Conservation effectiveness · Population persistence

11.1 The Importance of Large Carnivores in Coupled Social-Ecological Systems

Across the globe, large carnivore species (here focusing on lion *Panthera leo*, spotted hyena *Crocuta crocuta*, striped hyena *Hyena hyena*, leopard *Panthera pardus*, cheetah *Acinonyx jubatus*, and African wild dog *Lycaon pictus*) are essential components of the ecological processes in ecosystems (Estes et al. 2011; Ripple et al. 2014; Newsome and Ripple 2015). However, due to their dual role of providing both costs and benefits to people, human-carnivore relations are very complex (Expósito-Granados et al. 2019; Lozano et al. 2019).

Few other taxonomic groups of animals evoke similarly controversial and strong emotions among humans. Depredation events on livestock and attacks on humans cause substantial direct and indirect costs to human safety, livelihoods, and wellbeing (Kissui et al. 2019b; Kushnir and Packer 2019) and such negative experiences can instill or reinforce a deep-rooted sense of fear of these species (Koziarski et al. 2016; Jacobsen et al. 2020; **Kissui et al.** Chap. 14). At the same time, species such as lions are often admired for their beauty and strength and are considered to be one of the most charismatic species in the world (Albert et al. 2018; Courchamp et al. 2018; Hoffmann and Montgomery 2021). These negative and positive non-material contributions of large carnivores to humans affect not only human health, rural livelihoods and wellbeing but also identity and spirituality. For example, ritual killing of lions is still a key part of traditional Maasai culture (Ikanda and Packer 2008; Hazzah et al. 2017).

Despite their generally low densities (Carbone and Gittleman 2002), large carnivores are often of high economic importance. For pastoralist communities, large carnivores primarily cause direct (e.g. loss of livestock) and indirect costs (e.g. costs associated with preventing livestock depredation). The presence of large carnivores requires vigilant herding practices (Ogada et al. 2003; Beck et al. 2021) as well as construction and maintenance of protective livestock pens, called "bomas" in the East African context (Kissui et al. 2019b; Chaka et al. 2020). These costs are coupled with social costs of livestock depredation associated with long-held cultural values of livestock by Maasai and other ethnicities (Kissui 2008; Kissui et al. 2019b). Other stakeholders, however, benefit substantially from the presence of large carnivores. For example, wildlife-based tourism – a major foreign revenue-generator in northern Tanzania – crucially depends on sightings of cheetahs, leopards, lions, and wild dogs (Arbieu et al. 2017). Similarly, trophy hunting of lions and leopards disproportionately contributes to revenue generation via consumptive wildlife utilization (Baldus and Cauldwell 2004; Lindsey et al. 2012).

Ecologically, large carnivores can also impact ecosystem structure. Large carnivore predation can limit prey populations directly and indirectly (Hopcraft et al. 2010). For example, increased predation rates on giraffe (*Giraffa camelopardalis*) calves during the wet season when zebra (*Equus quagga*) and wildebeest (*Connochaetes taurinus*) populations are outside Tarangire National Park (TNP), affects giraffe recruitment and population dynamics inside TNP (Lee et al. 2016). In addition, the risk of predation can affect grouping patterns, activity budgets, and the spatio-temporal distribution of prey populations which in turn can, potentially, have cascading effects on prey populations (Creel et al. 2014; Moll et al. 2016, 2017). Both direct and indirect effects of predation can have top-down impacts on the abundance and distribution of herbivores and these effects can have further cascading repercussions on vegetation structure and primary productivity (Ford and Goheen 2015; Atkins et al. 2019; Mwangi et al. 2019). Impacts of large carnivore species on prey populations and by extension on vegetation structure are, however, context dependent. First, these ecological systems are often highly complex, featuring multiple predators and multiple prey (Montgomery et al. 2019). Thus, tracking the mechanistic connections of the direct, and particularly the indirect, effects can be challenging. Second, direct and indirect anthropogenic effects on other trophic levels can cloud direct and indirect predation-related effects in human-dominated areas (Haswell et al. 2017).

The intrinsic, cultural, economic and ecological importance of large carnivores lies in stark contrast to widespread population declines, range contractions, and local extinctions of large carnivores across the globe and the African continent (Riggio et al. 2013; Ripple et al. 2014). Conserving large carnivores in fragmented landscapes such as the Tarangire Ecosystem (TE) is a major challenge (Kissui et al. 2019b; Abade et al. 2020). Due to their low population densities, large home ranges, and seasonal fluctuations in prey abundance, large carnivores frequently leave the relative safety of protected areas. As a result, they frequently come into contact with livestock and humans which may put them at risk of preemptive or retaliatory killing by humans (**Kissui et al.** Chap. 14). In the long-term, such sustained human-induced mortality can pose a serious threat to the persistence of carnivore populations in human-dominated landscapes (Woodroffe and Ginsberg 1998).

Beyond direct human-carnivore interactions, biotic factors shape the distribution and abundance of large carnivore populations. Prey density determines the overall carrying capacity of large carnivores such as lions, leopards, and spotted hyenas (Carbone and Gittleman 2002; Hayward et al. 2007) but interspecific interactions such as kleptoparasitism and interspecific killing by larger competitors (Caro and Stoner 2003) can cause cheetahs and wild dogs to primarily select habitats with low competitor densities (Durant 1998; Darnell et al. 2014). As anthropogenic perturbations can impact large carnivore populations directly (e.g. via increased mortality) and indirectly (e.g. by mediating prey densities or by affecting densities of competitors), one could thus expect that conservation management (or the lack thereof) affects the landscape-scale distribution and abundance of large carnivore populations. The TE is a fragmented ecosystem consisting of multiple protected areas among a matrix of village lands and thus features a gradient of human-natural land uses (Fig. 11.1). National parks, such as Lake Manyara (LMNP) and TNP, do not allow human activities besides photographic tourism and research. In recent years,

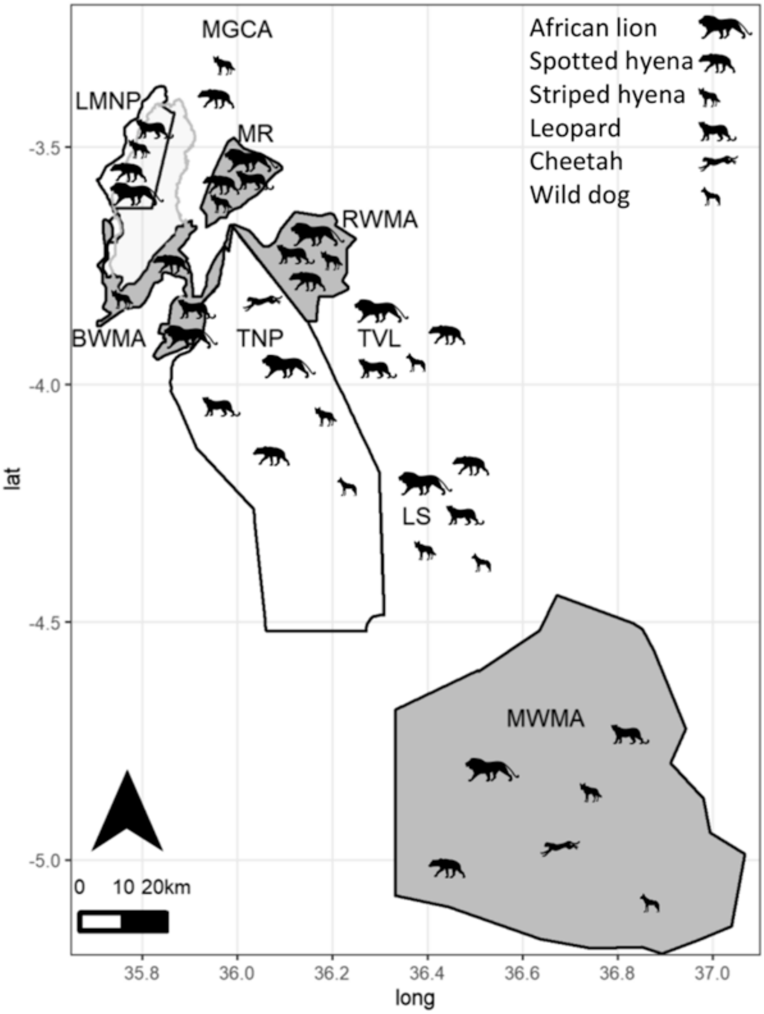

Fig. 11.1 Map of the Tarangire Ecosystem, showing Lake Manyara (LMNP) and Tarangire (TNP) national parks (black outline), and community-based conservation models (grey shaded) Burunge (BWMA), Makame (MWMA), and Randilen (RWMA) wildlife management areas and Manyara ranch (MR). In addition, the map shows the approximate location of the Mto wa Mbu Game Controlled Area (MGCA), Tarangire village lands (TVL) and the area around Loibor Siret (LS). Carnivore silhouettes indicate if the corresponding species had been detected by systematic surveys carried out in each of the management units (Table 11.1). As we did not have any information on large carnivores in the Mkungunero Game Reserve (located in between TNP and MWMA), we omitted this area. Animal silhouettes are not associated with copyrights (http://phylopic.org/)

multiple community-based conservation schemes have been initiated in the ecosystem to augment conservation efforts and to provide benefits to people. Land-use plans of the three wildlife management areas (Burunge WMA, Makame WMA, and Randilen WMA) entail that village land is set aside for wildlife, while other land is

Table 11.1 Confirmed presence of six large carnivore species in different management units of the Tarangire Ecosystem based on systematic ecological surveys. For each presence record, we include the survey methodology and refer to the original source

	Lion	Spotted hyena	Striped hyena	Leopard	Cheetah	Wild dog
Tarangire NP	Road transects (Foley et al. 2018b; Kiffner et al. 2020b); camera traps (Msuha et al. 2012), spoor survey (Foley et al. 2018b)	Road transects & spoor transects: (Foley et al. 2018b); camera traps (Msuha et al. 2012)	Spoor survey (Foley et al. 2018b)	Road transects & spoor transects: (Foley et al. 2018b); camera traps (Msuha et al. 2012)	Spoor transects: (Foley et al. 2018b)	Spoor transects: (Foley et al. 2018b)
Lake Manyara NP	Road transects (Kiffner et al. 2015); camera traps (Steinbeiser et al. 2019); spoor survey (Foley et al. 2018b)	Camera traps (Steinbeiser et al. 2019); spoor survey (Foley et al. 2018b)	Spoor survey (Foley et al. 2018b)	Road transects (Kiffner et al. 2015); camera traps (Steinbeiser et al. 2019); spoor survey (Foley et al. 2018b)		
Burunge WMA	Line transects (Kiffner et al. 2020b); camera traps (Kissui et al. 2019a)	Line transects (Kiffner et al. 2020b); camera traps (Kissui et al. 2019a)	Camera traps (Kissui et al. 2019a)	Line transects (Kiffner et al. 2020b); camera traps (Kissui et al. 2019a)		
Makame WMA	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)	Camera trap (Foley et al. 2018a); spoor survey (**Baker et al.** Chap. 15)
Randilen WMA	Line transects (Kissui and Kiffner 2013); spoor survey (Foley et al. 2018b)	Spoor survey (Foley et al. 2018b)	Spoor survey (Foley et al. 2018b)	Spoor survey (Foley et al. 2018b)		

(continued)

Table 11.1 (continued)

	Lion	Spotted hyena	Striped hyena	Leopard	Cheetah	Wild dog
Manyara Ranch	Road transects (Kiffner et al. 2016; Foley et al. 2018b); camera traps (Beattie et al. 2020); spoor survey (Foley et al. 2018b)	Road transects (Kiffner et al. 2016; Foley et al. 2018b); camera traps (Beattie et al. 2020); spoor survey (Foley et al. 2018b)	Camera traps (Beattie et al. 2020), spoor survey (Foley et al. 2018b)	Spoor survey (Foley et al. 2018b)		
Mto wa Mbu GCA		Road transect (Kiffner et al. 2016)	Road transect (Kiffner et al. 2016)			
Loibor Siret	Camera traps (African People & Wildlife, unpublished data)	Camera traps (African People & Wildlife, unpublished data)	Camera traps (African People & Wildlife, unpublished data)	Camera traps (African People & Wildlife, unpublished data)	Camera traps (African People & Wildlife, unpublished data)	Camera traps (African People & Wildlife, unpublished data)
Tarangire village lands	Camera traps (Msuha et al. 2012); spoor survey (Foley et al. 2018b)	Camera traps (Msuha et al. 2012); spoor survey (Foley et al. 2018b)		Spoor survey (Foley et al. 2018b)	Road transects and spoor survey (Foley et al. 2018b)	

set aside for livestock keeping or settlement and agriculture. In Makame WMA, trophy hunting is also permitted. Manyara Ranch is managed for livestock production, wildlife habitat, and to ensure dry season livestock grazing for two adjacent Maasai communities (Kiffner et al. 2020a). Although conversion to cropland in the surrounding village lands has increased in the last decades (Msoffe et al. 2011), a substantial portion of village lands is rangeland and is mainly used for livestock grazing.

To date, a systematic and ecosystem-wide large carnivore survey (e.g. Henschel et al. 2016) has not been carried out in the TE. Thus, it is not presently clear how large each of the carnivore populations is in each of the land-use zones and the potential types of competition that are evident. However, the distribution and abundance of large carnivores in the ecosystem has attracted substantial attention in the last decades. Drawing upon the available, yet scattered, scholarly work on this charismatic and functionally important group of species, we summarize the status of large carnivores in the TE as a baseline to discuss how conservation management and biotic factors hinder or facilitate their persistence in this ecosystem.

11.2 The Challenge of Monitoring Large Carnivores in Social-Ecological Systems

Monitoring large carnivore populations is typically complicated by their low population densities, wide ranging patterns, and often predominantly nocturnal activity patterns (Gese 2001). Thus, the standard monitoring techniques that are suitable for herbivores are largely unsuitable to detect carnivore occurrence in a given area, let alone to estimate their abundance (Steinbeiser et al. 2019). Indeed, large carnivores in LMNP show either almost exclusively (spotted hyena) or predominantly (lion, leopard) nocturnal behavior patterns (Fig. 11.2). At least among spotted hyenas, the activity patterns seem distinct from those detected in the neighboring Serengeti-Ngorongoro ecosystem, where spotted hyenas are frequently observed during daytime (Hofer and East 1993). Such seemingly altered activity patterns are possibly a behavioral response to avoid interactions with human activity (Boydston et al. 2003). If carnivore activity is concentrated during nighttime, interspecific temporal overlap increases and may in turn inflate the potential for interference competition within the large carnivore guild (Hayward and Slotow 2009). As nocturnal activity of species is typically positively associated with human disturbance (Gaynor et al. 2018), monitoring large carnivores in human-dominated landscapes requires methods that detect nocturnal species.

Methods suitable for monitoring large carnivores are diverse and each method is associated with its own set of advantages and challenges (Barea-Azcón et al. 2007; Thorn et al. 2010), making comparisons of results derived by different methods difficult. Therefore, we explicitly state the methods of data collection that were

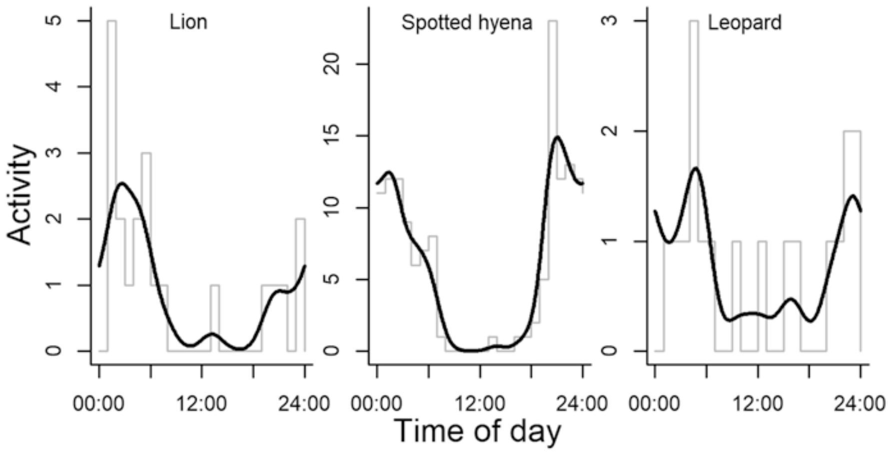

Fig. 11.2 Diel activity patterns of lion, spotted hyena, and leopard in Lake Manyara National Park. Activity patterns were estimated based on independent camera trap events (independence ensured by omitting subsequent pictures that were within 1 h of the initial camera event by the focal species) using the *activity* package (Rowcliffe et al. 2014) implemented in *R* 3.63 (R Core Team 2016)

inherent to all of our results. Primarily, data in this chapter were derived using the following methods:

Individual identification of lions has been carried out by the Tarangire Lion Project since 2003. Researchers of this project identify lions in the core study area of TNP, extending into Manyara Ranch, and since 2016 also including LMNP, Burunge, and Randilen WMAs based on whisker-spot patterns. A subset of lions are also tracked using VHF- or GPS-collars (Laizer et al. 2014). These methods provide detailed and fine-scale information on the distribution and abundance of lions, yet the information is largely confined to protected areas and to a single focal species.

Systematic camera trapping has been conducted in select study areas [e.g. LMNP (Steinbeiser et al. 2019), Manyara Ranch (Beattie et al. 2020), Burunge WMA (Kissui et al. 2019a), and Makame WMA (Foley et al. 2018a)] within the TE. These surveys have been done across a land-use gradient from within the protected areas into the human-dominated areas surrounding the protected areas (Msuha et al. 2012). This technique allows for unambiguous species identification, and, providing that the sampling effort is sufficient, allows for documenting most of the species that are present in a given area. Density estimation of target species has not been carried out in this ecosystem using this method. Although theoretically a promising monitoring tool, camera trapping is relatively expensive and vandalism and theft can be a limiting factor of sustained monitoring in human-dominated areas (Steinbeiser et al. 2019).

Spoor surveys, which involve driving slowly along minor tracks and detecting and identifying species via sign and pug marks, are another means of studying carnivores. In the TE, these surveys are regularly performed in collaboration with knowledgeable trackers, typically members of the Hadza ethnicity, hunter-gatherers living in northern Tanzania (**Baker et al.** Chap. 15). Once again, unbiased methods involve the application of these spoor surveys across land-use gradients in the TE (Foley et al. 2018b; Mkonyi et al. 2018). Spoor surveys are relatively cost-effective, allow detection of multiple species, and can be implemented across relatively large spatial scales (Karanth et al. 2011; Kiffner et al. 2019). Challenges include difficulties in definite identification of sister species such as spotted and striped hyenas (Kiffner et al. 2019). Although potentially biased (Dröge et al. 2020), it is possible to estimate carnivore density as a function of track density (Winterbach et al. 2016).

Transect counts and recording of direct sightings along roads (hereafter road transects) or systematically distributed transects (hereafter line transects) have been widely carried out in the TE, primarily to monitor herbivore populations over time and across management zones (Kiffner et al. 2020a). Though typically implemented during daytime, these surveys occasionally yield direct sightings of large carnivores.

Finally, to verify presence of large carnivores in human-dominated areas, we summarize data collected during interviews with residents of the TE. A substantial body of work on human-wildlife interactions has been carried out in multiple villages across the ecosystem (Bencin et al. 2016; Koziarski et al. 2016; Mkonyi et al. 2017a) and the local ecological knowledge gleaned from this wealth of data can serve as a suitable indicator of the distribution and relative abundance of large carnivores in human-dominated landscapes (Madsen et al. 2020; Mbise et al. 2020).

11.3 Distribution and Abundance of Large Carnivores Across a Conservation Gradient

A review of the available literature on large carnivore surveys suggests that most large carnivore species, particularly lions, spotted hyenas, and leopards are widely distributed across the TE (Table 11.1; Fig. 11.1). Most recent surveys confirmed their presence in both national parks, multiple-use areas (Burunge, Makame, and Randilen WMAs; Manyara Ranch), and in less-protected areas such as village lands (Table 11.1; Fig. 11.1). A notable spatial exception seems to be the Mto wa Mbu Game Controlled Area, the area in between LMNP and Manyara Ranch, and northwards towards Selela. In this human-dominated area, we detected only one sighting of striped and spotted hyena each, despite 1979 km of transects driven during 25 seasonal surveys from 2011 to 2020. However, as no additional survey methods were carried out in this area, we postulate that the target species use this area during nighttime but were missed by diurnal surveys. Indeed, livestock predation records and retaliatory killing incidences in these locations suggest that lions, leopards, and occasionally also cheetahs and wild dogs use the area between LMNP and MR as well as village lands around Selela village (**Kissui et al.** Chap. 14).

Interestingly, the striped hyena seems to be comparatively widely distributed across the ecosystem as well. However, as this species is mostly detected by indirect methods such as spoor and camera trap surveys, relatively little is known about this elusive carnivore in the TE.

Cheetahs and wild dogs were detected in relatively few of the surveyed management units (Table 11.1; Fig. 11.1). However, this does not necessarily mean that these species are truly absent in areas where surveys failed to detect them. For example, wild dogs and cheetahs are very rarely sighted in LMNP (Foley et al. 2014), and cheetahs are occasionally detected in Manyara Ranch. In summary, these results suggest few systematic differences in carnivore species across different protected areas, yet that carnivore species richness may be reduced in village lands (Msuha et al. 2012).

Comparing densities of large carnivores across protected areas may provide a more nuanced understanding of carnivore distribution in the TE. The most suitable and available dataset for such a comparison has been derived by a series of spoor surveys. We note, however, that due to the limited coverage, relatively small sample size (e.g. 52 km of transects repeated four times in Makame WMA, in an area that covers c. 4500 km^2), and lack of temporal repetition (e.g. leading to failed detection of species known to be permanently present in an area such as lions in LMNP) density estimates need to be treated cautiously (i.e. not to be extrapolated) and serve primarily as a metric to compare relative densities across management units.

Overall, spotted hyenas appear to reach the greatest densities of all the large carnivore species (Fig. 11.3). Leopards also occurred at relatively high densities in all surveyed study areas. Striped hyenas were detected in five out of six study areas and reached relatively higher densities in Manyara Ranch, the Simanjiro area and

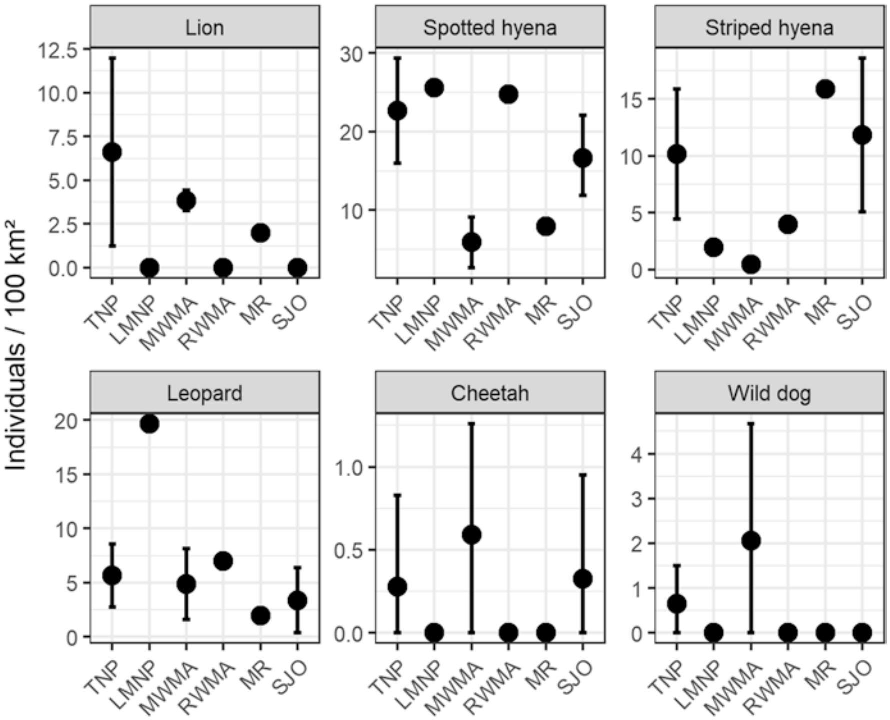

Fig. 11.3 Estimated densities of lions, spotted hyenas, striped hyenas, leopards, cheetahs, and wild dogs in six management units of the Tarangire Ecosystem (TNP: Tarangire National Park; LMNP: Lake Manyara National Park; MWMA: Makame Wildlife Management Area; RWMA: Randilen Wildlife Management Area; MR: Manyara Ranch; SJO: Simanjiro). Field work was carried out in the dry season 2016 (TNP; LMNP; RWMA; MR; SJO) and rainy season 2019 (MWMA). Field protocols are summarized in Foley et al. (2018a, b) and **Baker et al.** (Chap. 15). Track densities (unique fresh tracks 100 km^{-1}) were transformed to densities using the equation: track density = 3.26 x carnivore density (Winterbach et al. 2016). If possible, 95% confidence intervals were computed based on the spatial (TNP, RWMA, MR, SJO) or temporal (MWMA) replication of transects

TNP. Lion densities were greatest in TNP. Cheetah and wild dog were detected in few of the study areas and where they occurred, their densities were roughly one order of magnitude lower than those of other large carnivore species (Fig. 11.3). While there are distinct density differences across study areas, species-specific densities did not appear to be consistently correlated with conservation status of an area. As exemplified by lion, leopard, cheetah, and wild dog densities in Makame WMA or spotted and striped hyena, leopard, and cheetah densities in the Simanjiro area, carnivore species in multiple-use areas can occur at densities similar to those observed in adjacent national parks.

11.4 Determinants of Space Use of Large Carnivores

While the movement and spatial ecology of large carnivores in the TE has received relatively little attention relative to other study systems (**Lohay et al.** Chap. 12), the available studies depict a consistent picture. Analyses of VHF- and GPS-collared lions suggest that lions move widely outside of protected areas and expand their home ranges during the wet season (Laizer et al. 2014) and thus, at least partly, track the seasonal distribution shifts of their main prey species in this ecosystem (**Bond et al.** Chap. 8). Therefore, seasonally mediated expansion of space use brings lions closer to humans during the rainy season (Beattie et al. 2020) increasing the potential for interactions, many of which can be negative. Indeed, when asked about the times in which rural people encounter large carnivore species most frequently, interviewees mentioned months during the rainy season more frequently than dry season months (Fig. 11.4). Based on this local ecological knowledge, seasonal patterns of large carnivore presence in human-dominated areas are largely consistent across species (Fig. 11.4) indicating that human-carnivore interactions occur mainly during the rainy season (**Kissui et al.** Chap. 14).

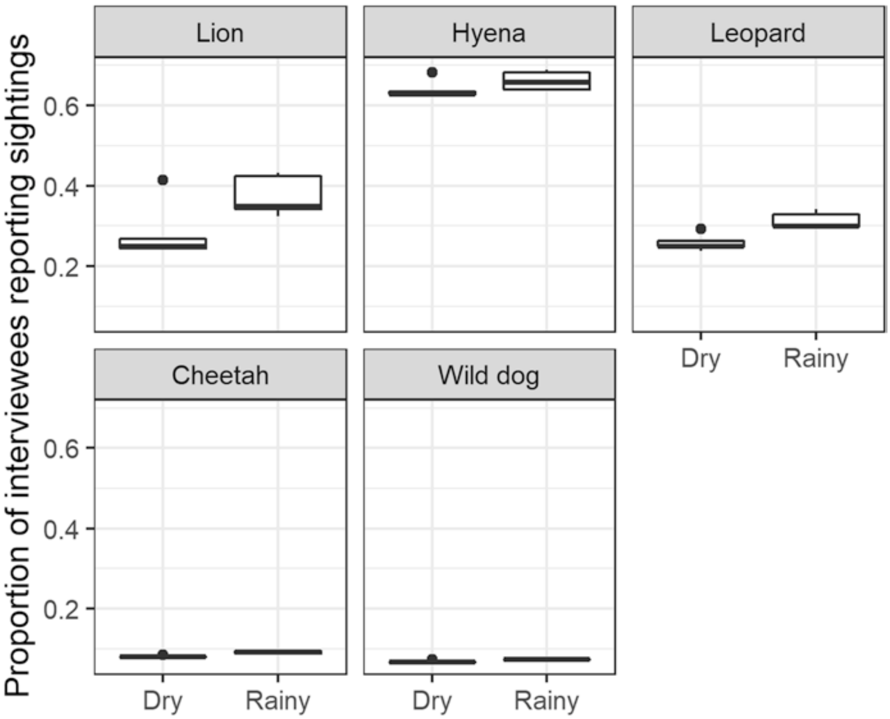

Fig. 11.4 Proportion of interviewees reporting sightings of lion, hyena (spotted and striped hyena combined as interviewees typically did not differentiate between the two species), leopard, cheetah, and wild dog in the Tarangire Ecosystem during a dry (June–October) or rainy season (November, December, January–May) month. The figure is based on data from Koziarski et al. (2016)

Beyond seasonal influences, space use of large carnivores is also informed by a suite of anthropogenic and environmental variables. For example, lion occupancy in the eastern part of the TE was negatively correlated with distance to protected area boundaries (Mkonyi et al. 2018) and bomas (Foley et al. 2018b), highlighting the vulnerability of lion populations to human impacts. Dry season lion predation risk (a proxy for space use) was positively associated with high primary productivity and prey density, and negatively associated with distance to surface water (Beattie et al. 2020), underscoring the importance of prey density and prey catchability as key environmental variables affecting lion space use (Hopcraft et al. 2005). In contrast, hyena occupancy appears to be positively associated with human population density (Mkonyi et al. 2018), a pattern consistent with high hyena densities in human-dominated areas (Fig. 11.3).

11.5 Persistence of Large Carnivores in Human-Dominated Landscapes

Hyenas were by far the most frequent large carnivore species reported to occur in human-dominated areas between LMNP and TNP (Fig. 11.5).

Lions, leopards, and particularly cheetah and wild dog, appear to be detected far less frequently in human-dominated areas. While the absolute differences across species can partly be explained by differences in conspicuousness (e.g. spotted hyenas are often very vocal during nighttime whereas leopards can be very secretive) and maybe also by the (perceived) conflicts caused by each species, the relative differences are generally in line with data from ecological surveys carried out in village lands (Msuha et al. 2012; Foley et al. 2018b).

In line with their relative high abundance in human-dominated areas, spotted hyenas are responsible for most livestock predation events in the TE (Kissui 2008; Mkonyi et al. 2017b; Kissui et al. 2019b; **Kissui et al.** Chap. 14). Spotted hyenas typically kill small stock such as goats, sheep, and juvenile cattle (**Kissui et al.** Chap. 14), which are less valuable to the herders than adult cattle, and consequently hyenas appear to be less persecuted. Spotted hyenas are susceptible to the occasional occurrence of carcass poisoning (Kissui 2008) but, likely due to their elusive (they typically run away if disturbed by people whereas lions typically defend carcasses) and nocturnal behavior, do not seem to be particularly susceptible to direct killing by humans (Kissui et al. 2019b). Occasionally, spotted hyenas die following accidental vehicle collisions (Kioko et al. 2015). Therefore, spotted hyenas are directly affected in human-dominated areas, yet in the absence of long-term monitoring data (Durant et al. 2011) or detailed demographic studies, it remains unclear how the current magnitude of such human-induced mortality affects their population persistence in the TE in the long term. Studying the foraging strategy and prey preferences of spotted hyenas may provide valuable insights how this carnivore copes in a human-dominated world.

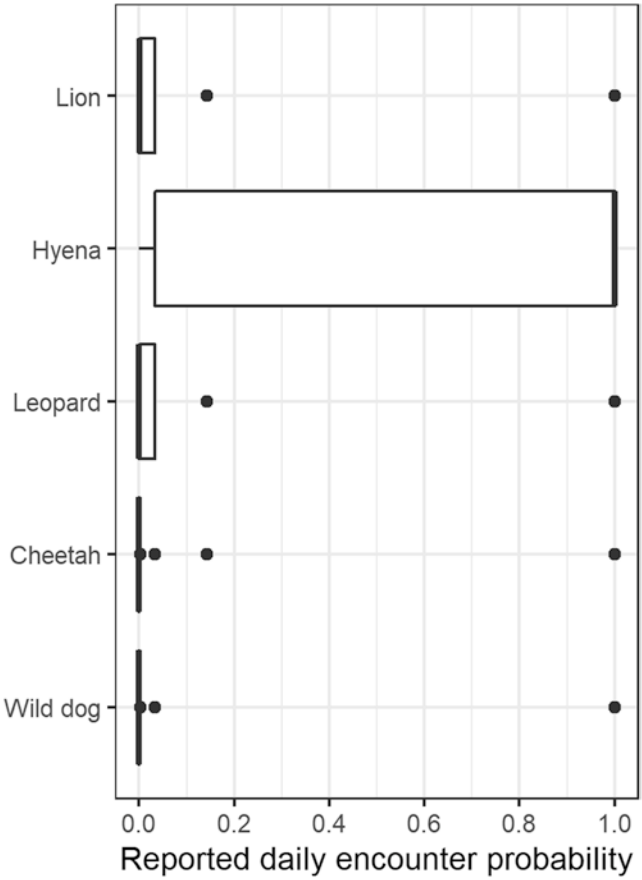

Fig. 11.5 Reported daily encounter likelihood of five large carnivore species (spotted and striped hyena combined) assessed via structured interviews with residents of the TE (Koziarski et al. 2016)

In contrast to spotted hyenas, lion population persistence appears to be more obviously dependent on conservation efforts. The available monitoring data suggest that the lion population in the TE experienced substantial declines in the last 25 years (Fig. 11.6a) and current evidence suggests that two key factors are likely causing this worrisome trend. First, rates of direct killing (primarily as a response to livestock depredation) have been relatively high across the ecosystem (Fig. 11.6b). Given the scale of the issue, direct killing of lions likely constitutes a form of additive mortality to the population that is unlikely to be compensated. On a positive note, the number of lions killed following livestock depredation has declined consistently over the last years (Fig. 11.6b) and indicates that past and current mitigation methods to reduce retaliatory killing of lions (**Kissui et al.** Chap. 14) have been effective. Second, herbivore populations have declined from historical baselines (Morrison et al. 2016; Kiffner et al. 2017; **Prins & de Jong** Chap. 7), lowering the available prey for lions and other large carnivores. Indeed, reduced prey biomass

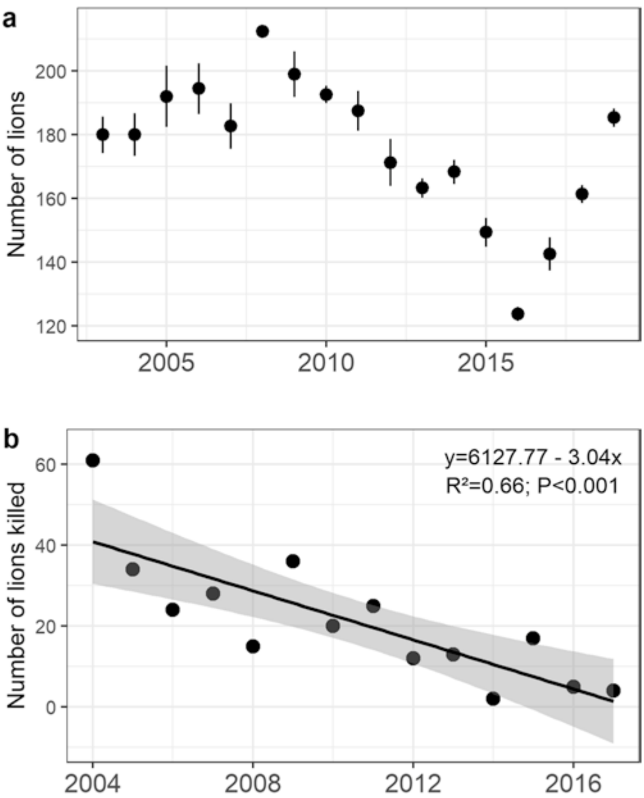

Fig. 11.6 (**a**) Annual population trends of lions in the research area of the Tarangire lion project from 2003 to 2019 based on monthly total counts of known individuals (error bars indicate 95% confidence intervals). Please note that the extent of the project area was extended in 2016 and may partially explain the greater number of recorded lions. (**b**) Minimum number of lions killed in retaliation by humans from 2004 to 2017 and the yearly trend of retaliatory killing as estimated by a linear model

which primarily reduces recruitment rates in lions (Vinks et al. 2021), and elevated human-induced mortality appear to be the two key proximate threats across the current geographic range of this species (Bauer et al. 2020). In combination, both of these proximate threats therefore contribute in tandem to the observed decline in the TE lion population.

As little is known about the ecology and population status of striped hyena and leopard in the TE, we can merely speculate about their population persistence. However, as their presence continues to be detected via indirect surveys across the TE, it seems that these two species can persist in this human-dominated ecosystem. People residing in rural communities also regularly report the presence of leopards in village lands (**Kissui et al.** Chap. 14). We hypothesize that the combined effects of their elusive and predominantly nocturnal behavior as well as a broad prey species and food niche (Hayward et al. 2006; Alam and Khan 2015) contribute to their apparent persistence.

While spotted and striped hyenas, lions, and leopards occur at relatively high densities in at least several management units (which may serve as source populations for the entire ecosystem), cheetahs and wild dogs appear to occur at low densities across the TE. Given their small population sizes, the long-term persistence of these species is subject to the small-population paradigm (Caughley 1994), and stochastic events may substantially affect the viability of these species. Compared to other carnivore species, livestock depredation by both cheetah and wild dog appears to be an infrequent occurrence (**Kissui et al.** Chap. 14). Nevertheless, given the already small population sizes, even the rare occasion of retaliatory (e.g. two cheetahs were killed outside TNP a few years ago) or accidental killing of individual cheetahs or wild dogs (e.g. through vehicle collisions) may substantially imperil the viability of these species (Durant et al. 2007; Parchizadeh et al. 2018).

11.6 Towards Human-Carnivore Coexistence

Coexistence of large carnivores and humans in shared landscapes is challenging and will require adaptive behavior in both carnivore species as well as in the human population (Carter and Linnell 2016). While the large carnivores can seemingly cope in the ecosystem, human activity will dictate whether carnivore populations will persist in the generations to come. Fundamentally, conservation of large carnivores in the TE needs to address two key elements. First, human-large carnivore interactions need to be managed by implementing sustainable methods that effectively reduce negative impacts on human and carnivore wellbeing (**Kissui et al.** Chap. 14). At the same time, conservation measures need to protect the seasonal migration and dispersal of major prey species that is so essential for the persistence of these ungulates (**Bond et al.** chap. 8; **Lohay et al.** Chap. 12). Although the movement of prey species likely increases the potential for (negative) interactions between large carnivores and humans, implementing these two measures simultaneously is not a paradox but a necessity if we want to conserve these species. As we have the tools for land-use planning that effectively considers wildlife movement (Bond et al. 2017), and for cost-effective, non-lethal mitigation methods to reduce livestock depredation (Lichtenfeld et al. 2015; Kissui et al. 2019b), we need to make sure that these tools are widely implemented. As none of these tools are 100% effective (Lichtenfeld et al. 2015; Eklund et al. 2017; Kissui et al. 2019b), the persistence of large carnivores in the TE will also depend on changing local attitudes, which could be facilitated by ensuring that local people accrue tangible and intangible benefits from the presence of large carnivores (Kansky et al. 2021).

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Part IV
Human-Wildlife Interactions

Chapter 12
Wildlife Movements and Landscape Connectivity in the Tarangire Ecosystem

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Abstract A fundamental condition for maintaining viable populations of wildlife is to ensure that animals can access resources. In landscapes where the boundaries of protected areas encompass only a fraction of annual home ranges, animal movement is often curtailed by human activities, often with negative population consequences. In the Tarangire Ecosystem (TE), wildlife generally aggregates in three main protected areas during the dry season (Tarangire and Lake Manyara National Parks, and Manyara Ranch Conservancy) and disperses to several other areas during the wet season. Connectivity between and within seasonal ranges in the ecosystem has generally become more restricted over time, though the apparent effects of these changes have been species-specific. Historical accounts of wildlife movement suggest that animals once moved over much larger areas than they do currently. In this chapter, we review historical information on wildlife movement and distributions in the TE and synthesize data on population genetic structure and individual movements from studies of elephants, giraffes, lions and wildebeests conducted over the past 25 years. Given the continued expansion of agricultural and urban areas, there

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is a need to coordinate efforts across land management agencies and local governments to ensure that wildlife can continue to move across the landscape.

Keywords Gene flow · Landscape connectivity · Least-cost path · Movement · Telemetry · Wildlife corridor

12.1 Introduction

One of the major concerns regarding the conservation of large mammal populations in Africa and elsewhere is the decline of functional connectivity within ecosystems (Caro et al. 2009; Fynn and Bonyongo 2011; Beale et al. 2013; Wegmann et al. 2014; Lee and Bolger 2017; Riggio and Caro 2017; Lobora et al. 2017; Lobora et al. 2018). Populations living in ecosystems that lack connectivity can suffer from (1) an inability to disperse between seasonal or protected areas, (2) compromised genetic variability within isolated populations due to lack of immigration, (3) an inability of dwindling populations to be rescued (i.e., recolonized) from extirpation, and (4) reduced opportunities for range shifts in response to global climate change (Rudnick et al. 2012). Indeed, some scholars argue that the long-term viability of wildlife species relies on maintaining connectivity between protected areas (e.g., Crooks and Sanjayan 2006; Nabe-Nielsen et al. 2010; Hilty et al. 2019), and the association between movement (or lack thereof) and wildlife abundance has widespread theoretical and empirical support. Consequently, wildlife corridors are an increasingly popular conservation tool used for promoting functional connectivity between protected areas (e.g., Roever et al. 2012; Silveira et al. 2014; Ramiadantsoa et al. 2015; Belote et al. 2016).

Tanzania is home to an impressive network of protected areas that safeguards a diverse wildlife assemblage from activities associated with a rapidly growing human population (Riggio et al. 2019). However, many of Tanzania's protected areas are small, isolated, and not always effective in addressing conservation goals (Newmark 2008; Kiffner et al. 2017; Mtui et al. 2017). As a result, wildlife populations have been declining in many areas across the country in recent decades (Stoner et al. 2007; Western et al. 2009; Craigie et al. 2010; Ogutu et al. 2016). Recently, wildlife corridors were recognized as a formal land-use entity through the 'Wildlife Corridors, Dispersal Areas, Buffer Zones and Migratory Routes' Regulation (Tanzania TUR 2018), a subsidiary of the Wildlife Act of 2009. Thus, there is considerable interest and legal scope for identifying, protecting, and restoring functional connectivity between Tanzanian protected areas (Caro et al. 2009; Jones et al. 2012; Bond et al. 2017).

The Tarangire Ecosystem (TE) includes a patchwork of protected areas, conservation easements, livestock-grazing areas and communal lands that together support seasonal movements of large mammals, including elephant, giraffe, lion and wildebeest (Table 12.1; Fig. 12.1). Longstanding research in the ecosystem provides a

Table 12.1 Protected area entities, associated managing authorities, and permitted human activities in the Tarangire Ecosystem

Entity	Managing authority	Role	Example in the Tarangire Ecosystem
National Park	Tanzania National Parks (TANAPA)	Photo tourism and management of wildlife	Tarangire (TNP), Lake Manyara (LMNP)
Game Reserve (GR)	Tanzania Wildlife Management Authority (TAWA)	Photo tourism and trophy hunting[a]	Mkungunero GR
Game Controlled Area (GCA)	TAWA	Trophy hunting, livestock grazing, settlement, agriculture	Mto wa Mbu GCA, Lolkisale GCA, Simanjiro GCA
Conservancy	District government, Non-governmental organizations	Photo tourism and livestock grazing	Manyara Ranch Conservancy (MRC)
Wildlife Management Areas	TAWA, Local community	Photo tourism, trophy hunting in designated wildlife areas; in delineated areas, livestock grazing, agriculture and settlements are permitted	Burunge WMA, Makame WMA, Randilen WMA
Conservation easement	Local community	Livestock grazing, photo tourism	Terat & Sukuro villages
Certificates of Customary Right of Occupancy (CCRO)	Local community	Livestock grazing	Naitolia & Selela villages
Corridor, dispersal buffer, migratory areas	Varied		None to date

[a] Trophy hunting of giraffes is legally prohibited in Tanzania

rich assortment of information on animal movement, from a variety of data sources (Table 12.2). In this chapter, we synthesize these datasets to provide an overview of the spatial dimensions of movement within and between the TE and other ecosystems. We focus on connectivity and genetic differentiation between the three main protected areas in the ecosystem – Tarangire National Park (TNP), Manyara Ranch Conservancy (MRC) and Lake Manyara National Park (LMNP) – but where possible, we reference evidence for connectivity with other geographic areas in or adjacent to the ecosystem (Fig. 12.1). Because elephant and wildebeest appear to move across larger spatial extents in TE than other large mammal species (Lamprey 1964; Kikoti 2009), our focus is on those two species.

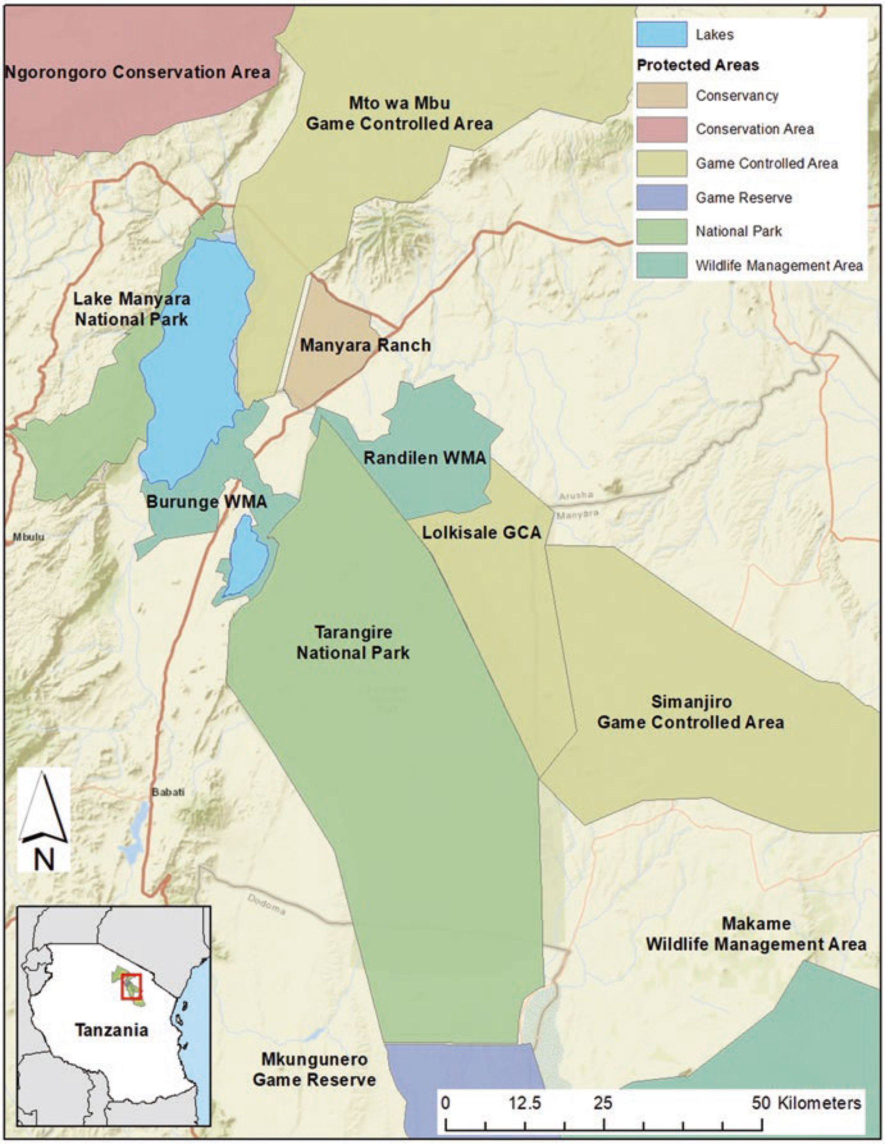

Fig. 12.1 Outline of the main protected areas in the central part of the Tarangire Ecosystem

12.2 Wildlife Corridor Mapping

Mapping wildlife corridors and migratory movements involves many different methodologies, each with its own advantages and spatial-accuracy trade-offs (Table 12.2). The relatively high visibility in the TE is ideal for *aerial and ground surveys* of wildlife. This method typically involves sample-based counts of animals

Table 12.2 Summary of sources and types of connectivity data used or referenced in this study

Data type	Species	Sample size	Timeframe	Sources
Aerial + ground transects	Large mammals		1958–1961	Lamprey (1964)
Aerial survey	Large mammals		1985	Borner (1985)
GPS telemetry	Wildebeest	6	2009–2011	Morrison and Bolger (2014)
	Elephant	14	2018–2020	Lobora (unpub)
	Elephant	4	2006–2008	Foley (2002)
	Lion	9	2012–2020	Kissui (unpub)
VHF telemetry	Wildebeest	10	1996–1997	TCP (1998)
	Elephant	7	1997–2000	Galanti et al. (2006)
Mark-recapture and individual identification	Wildebeest	5571 individuals, 1287 recaptures	2005–2009	Morrison and Bolger (2014) and Morrison et al. (2016)
	Giraffe	1094 individuals	2012–2014	Lee and Bolger (2017)
	Lion	799 individuals	2004–2021	Kissui (unpub)
	Elephant	1000+ individuals	1993–2021	Foley and Foley (2014)
Predictive functional connectivity	Wildebeest	–	2009–2015	Bond et al. (2017)
Predictive functional connectivity	Ungulates	–	2015	Riggio et al. (in review)
Predictive structural connectivity	Large mammals	–	~2010–2016	Riggio and Caro (2017)
Mitochondrial DNA	Elephant	126	2015–2017	Lohay et al. (2020)
Microsatellite	Elephant	169	2015–2017	Lohay et al. (2020)
Mitochondrial DNA	Wildebeest	144	Unspecified	Georgiadis (1995)
Microsatellite	Wildebeest	157	2007–2009	Ernest (2012)

along stratified transects and has been employed in the TE since the mid 1900s. Historical surveys help establish a baseline for spatial distributions but generally do not provide information on connectivity because of a lack of individual-level data on movement. *Population genetics* relies on obtaining genetic material from individuals (typically dung, hair or tissue biopsies) and examining rapidly evolving non-functional regions of the individuals' DNA to establish the degree of relatedness (or genetic separation) between animals in different geographic areas. Similar to aerial and ground surveys, population genetics provide important context for population structure and gene flow over evolutionary time but lacks geographic detail in pinpointing where on a landscape movement occurred. Multistate *mark-recapture* models generate unbiased estimates of population-level movement between discrete areas. Mark-recapture data can be extremely time consuming to collect as the method involves recapturing or resighting individuals across multiple sampling periods. While the resulting estimates of movement or transition between areas are unbiased because they account for detectability, the method again does not provide detailed information on the spatial routes that individuals taken between areas. *VHF and GPS telemetry*, in contrast, overcomes this lack of spatial accuracy

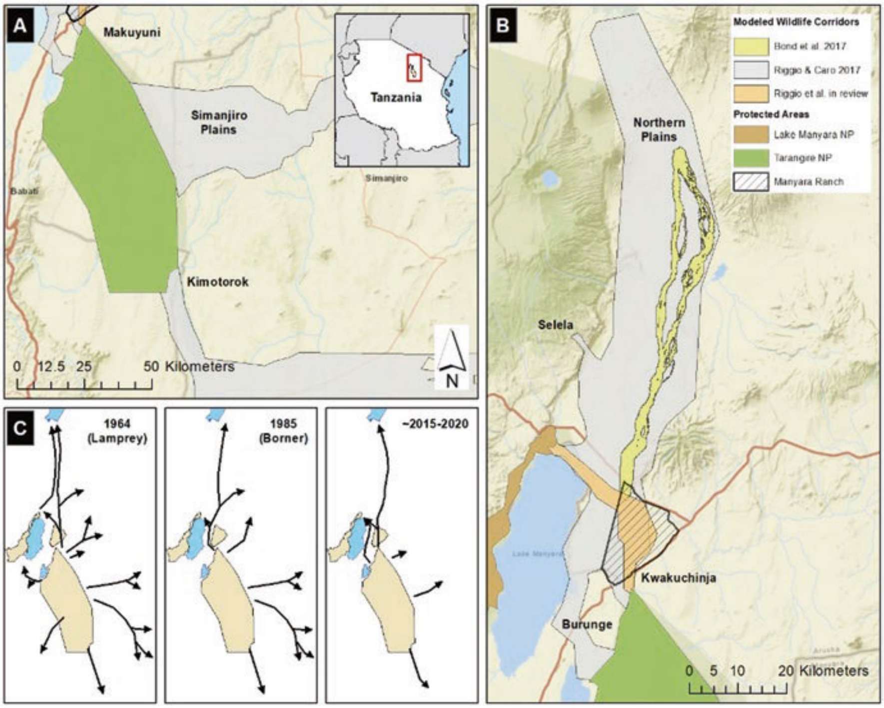

Fig. 12.2 Wildlife corridors in the Tarangire Ecosystem linking Tarangire National Park to (**a**) Makuyuni and the Simanjiro Plains to the east, Kimotorok in the south, and (**b**) Manyara Ranch Conservancy, Lake Manyara National Park and the Northern Plains via the Burunge and Kwakuchinja corridors and Ngorongoro Conservation Area via the Selela corridor based on predictive modelling studies (Bond et al. 2017; Riggio and Caro 2017; Riggio et al. in review). (**c**) Landscape connectivity across the greater ecosystem has declined since initial mapping efforts in the 1960s (Lamprey 1964; Borner 1985). [Modified from Morrison et al. 2016]

in connectivity but the method is often limited to small sample sizes of individuals collected over short time periods and often fails to provide a comprehensive characterization of movement routes (Hazen et al. 2021). Data from the above methods can also be modelled to provide predictions of connectivity within a landscape. *Structural connectivity* models characterize the physical features of a landscape that foster or inhibit movement, while *functional connectivity* quantifies the degree of actual movement of individuals within a landscape.

Early descriptions of wildlife movements across the TE indicated a vast wet season dispersal of ungulates in all directions from dry season food and water refuges in TNP and LMNP (Lamprey 1964; Fig. 12.2c). By the 1980s dispersal routes to the west of TNP had become blocked by expanding agricultural production, and the remaining routes to the north and east were increasingly threatened by an expansion of croplands surrounding the protected areas (Borner 1985; Fig. 12.2c). Indeed, the first call for the formal establishment of a wildlife corridor in Tanzania was to protect the shrinking linkage between TNP and areas to the north and west

(Kwakuchinja Wildlife Corridor[1]; Borner 1985). The erosion of connectivity across the TE continued (Mwalyosi 1991), and by the 2000s only seven wildlife corridors remained linking (from north to south) LMNP and MRC to the Northern Plains and Ngorongoro Conservation Area (Selela), and TNP to MRC (Burunge and Kwakuchinja corridors), Makuyuni, Simanjiro Plains, and Kimotorok (Caro et al. 2009). Five of the seven remaining wildlife corridors were in "extreme" or "critical condition" (Caro et al. 2009), with concerns they would rapidly disappear without immediate conservation attention. The establishment of Wildlife Management Areas in the TE slowed the threat of land-use change in the region, and recent modelling efforts have indicated that all these wildlife corridors remain likely structurally open to movement (Riggio and Caro 2017; Fig. 12.2a, b).

Recent efforts to map functional connectivity across these historical migration routes have provided evidence that wildlife still use the Northern Plains (Morrison and Bolger 2014; Morrison et al. 2016; Bond et al. 2017), Kwakuchinja (Morrison and Bolger 2014; Kiffner et al. 2016; Morrison et al. 2016; Lee and Bolger 2017; Lobora unpub.; Riggio et al. in review), Burunge (Lobora unpub.), Makuyuni (Morrison and Bolger 2014; Morrison et al. 2016; Lobora unpub), and Simanjiro Plains (Pittiglio et al. 2014; Morrison et al. 2016) corridors (Table 12.2).

To reconstruct *past connectivity* in the ecosystem, we rely on a combination of anecdotal accounts, colonial-era ground and aerial surveys, and population genetic analyses. To delineate r*ecently used movement paths*, we compile GPS and VHF radio-collar data collected from elephants, lions, and wildebeests with methods described previously (Table 12.2). Finally, for wildebeests and giraffes we highlight *movement probabilities* between protected areas using photo capture-recapture studies (Morrison et al. 2011; Lee and Bolger 2017).

12.3 Evidence for Functional Connectivity

12.3.1 Elephant

Elephants range over a large portion of the greater TE and have been relatively well-studied through a long-term individual-based project in TNP and recent telemetry and genetic studies (**Foley and Foley** Chap. 10). Protection seems to strongly affect elephant behavior: individuals tend to move between protected areas more at night and travel four times faster when they are outside than inside protected areas (Foley 2002; Douglas-Hamilton et al. 2005). In TNP, the population appears to have three main sub-populations (northern, central, and southern), with the southern

[1] Kwakuchinja corridor refers to an area that provides a link between Tarangire NP and Lake Manyara NP through Burunge WMA. Many researchers have used this term loosely to refer to any wildlife corridor connecting Tarangire and Lake Manyara NP (Goldman 2009). Corridors that connect Tarangire and Manyara Ranch are referred to as Makuyuni, Mswakini Chini and Mswakini Juu (Kissui 2008; Kikoti 2009).

subpopulation being a distinctive group that rarely overlaps spatially with the other two subpopulations (Foley and Foley 2014; **Foley and Foley** Chap. 10). The LMNP population remains relatively isolated and its abundance has fluctuated considerably over time because of poaching and habitat loss (Kiffner et al. 2017). Elephants in the northern part of the TE east and southeast of Lake Natron exhibit long-distance movements through the woodlands to Longido and possibly West Kilimanjaro (Kikoti 2009; Lobora unpub).

12.3.2 Elephant Movement Between Tarangire NP and Makame WMA

The southern TNP subpopulation numbers roughly 500 individuals and moves between the dense bushland of the Makame depression in the Makame WMA in the wet season and Mkungunero GR or southern TNP in the dry season (Galanti et al. 2000; **Foley and Foley** Chap. 10). VHF collar data from the mid-1990s (Galanti et al. 2000) suggest that individuals of this subpopulation have larger home ranges compared to other subpopulations, thus may require larger areas.

12.3.3 Elephant Movement Between Tarangire NP and Lolkisale GCA

Using a combination of aerial survey and GPS-collar data Foley (2002) identified several elephant corridors between TNP, Lolkisale GCA, and the recently established Randilen WMA. Elephants spend time in Lolkisale GCA and Randilen WMA during wet seasons and TNP during dry seasons. Restricting land uses in and around the Lolkisale area that conflict with wildlife conservation (agriculture and settlements) have helped support the conservation of elephants in the ecosystem (Foley and Foley 2014).

12.3.4 Elephant Movement Between Tarangire NP and Areas to the North and West

The linkage between TNP and areas north and west has been a concern for conservationists for many decades (Borner 1985; Mwalyosi 1991) because of the increasing extent of cultivated land in areas used historically by elephants and other species (Kideghesho 2000). Manyara Ranch Conservancy, established in 2001, has significantly contributed to conserving connectivity between TNP and LMNP. GPS-collaring studies on elephants in and near MRC (Foley 2002; Kikoti 2009; Lobora unpub), along with dung and track surveys (Foley and Foley 2014), provided

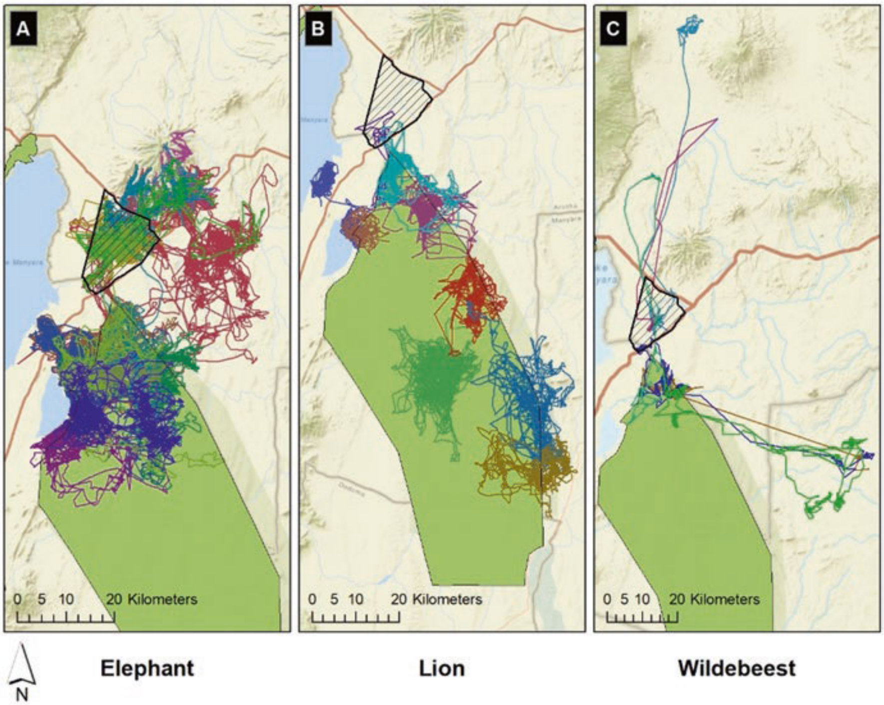

Fig. 12.3 Wildlife GPS telemetry tracks between Tarangire National Park (green polygon), the Manyara Ranch Conservancy (striped polygon) and other areas from (**a**) elephants (Lobora unpub), (**b**) lions (Brown et al. in review) and (**c**) wildebeest. (Morrison and Bolger 2014)

compelling evidence that elephants frequently travel between MRC and TNP, relying on multiple routes in Mswakini Juu and Mswakini Chini villages (Fig. 12.3a). Movements outside of protected areas occur mostly at night (Kikoti 2009). GPS data also suggested that elephants move widely outside the boundaries of the protected areas in the vicinity of Makuyuni village and Losimingori mountain and that they reached the eastern edge of Lake Manyara via Burunge WMA (Fig. 12.3a).

12.3.5 Elephant Gene Flow and Population Structure Within and Outside of Tarangire Ecosystem

For more than three decades genetic tools have been used to uncover population structure, gene flow within and between populations, inbreeding depression, effective population size, and genetic pedigrees (Freeland et al. 2011). The tools have been particularly important for answering crucial questions for conservation of biodiversity. In northern Tanzania, elephant population structure has been documented in several previous studies (Ahlering et al. 2012; Ishida et al. 2013; Lohay 2019; Lohay et al. 2020). Mitochondrial DNA analysis of 4258 base pairs (bp) revealed three subclades of elephants in Tanzania: savanna-wide (SW), southeast-savanna (SS) and east-central (EC) (See Figure 2, Ishida et al., 2013). Most TE elephants carry haplotypes in the SW subclade whereas most Serengeti elephants carry haplotypes in the EC subclade. Only two haplotypes were shared between TE and Serengeti and they were only carried by one elephant each. There was significant genetic differentiation between the two ecosystems. These results were consistent with Ahlering et al. (2012) which showed a clear separation of mtDNA haplotypes between the TE and Serengeti. A more extensive study with a greater spatial extent of sampling was recently conducted in the TE and Serengeti that sampled elephants from all protected areas within the TE including TNP, MRC and LMNP (Lohay et al. 2020). Elephants from Ruaha NP and Selous GR (now Nyerere NP) which are found hundreds of kilometers to the southeast of TNP, were also sampled.

The mtDNA analysis showed that LMNP and MRC elephants had 10 haplotypes whereas TNP had only 4 haplotypes. About 80% of TNP elephants were carrying haplotypes that were in the SW subclade (Fig. 12.5). Surprisingly, Ruaha elephants also shared 80% of haplotypes with TNP elephants. These data suggest that in the recent past Ruaha and TNP elephants were a single population. Movement among Ruaha, Muhezi, and Swaga-Swaga GRs is still thought to occur (Caro et al. 2009), though the area between Swaga-Swaga and TNP (roughly 40 km in a straight line) has extensive agricultural cultivation, so movements are now unlikely (Riggio and Caro 2017). Analysis of nuclear markers shows similar results of no significant genetic differentiation between Ruaha and TNP elephants (Lohay et al. 2020), further supporting the existence of a genetic connection between populations in these two protected areas (Caro et al. 2009). Mitochondrial DNA data show genetic similarity between elephants from MRC, LMNP and Ngorongoro suggesting historical gene flow between these areas (Table 12.3). These data are supported by early records of elephant movements between the Karatu highlands, Ngorongoro Conservation Area and LMNP (**Prins and de Jong** Chap. 7), and the existence of movement routes along the Upper Kitete corridor west of the Selela Forest Reserve.

A higher haplotype diversity of mtDNA was observed in LMNP and MRC than in TNP, suggesting the populations are different, despite the areas being within 20 km of one another (Lohay et al. 2020). Our genetic results may reflect sex-biased movements and a lack of interbreeding between the two areas, as most observed movements between MRC and TNP from telemetry data involved males and MRC is largely a bull area (Kioko et al. 2013). Mitochondrial DNA is inherited in a

Table 12.3 Genetic diversity indices for elephants in the Tarangire Ecosystem using 126 mitochondrial DNA sequences

	Populations			
Genetic diversity indices	TNP	MRC	LMNP	All samples
Number of sequences (N)	42	36	48	126
Number of haplotypes (h)	4	6	7	12
Haplotype diversity (Hd)	0.443	0.759	0.738	0.713
Nucleotide diversity (Pi)	0.141	0.145	0.240	0.187

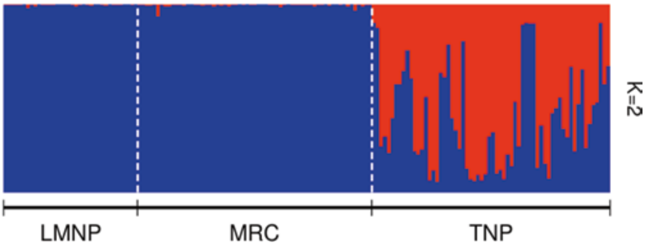

Fig. 12.4 STRUCTURE analysis for 169 African savanna elephants in the Tarangire Ecosystem using 10 microsatellite loci. Each elephant is represented by a thin vertical bar partitioned into colour segments representing the individuals' ancestry into subpopulations. Lake Manyara NP and Manyara Ranch Conservancy formed one genetic cluster whereas Tarangire NP formed another cluster. (Lohay et al. 2020)

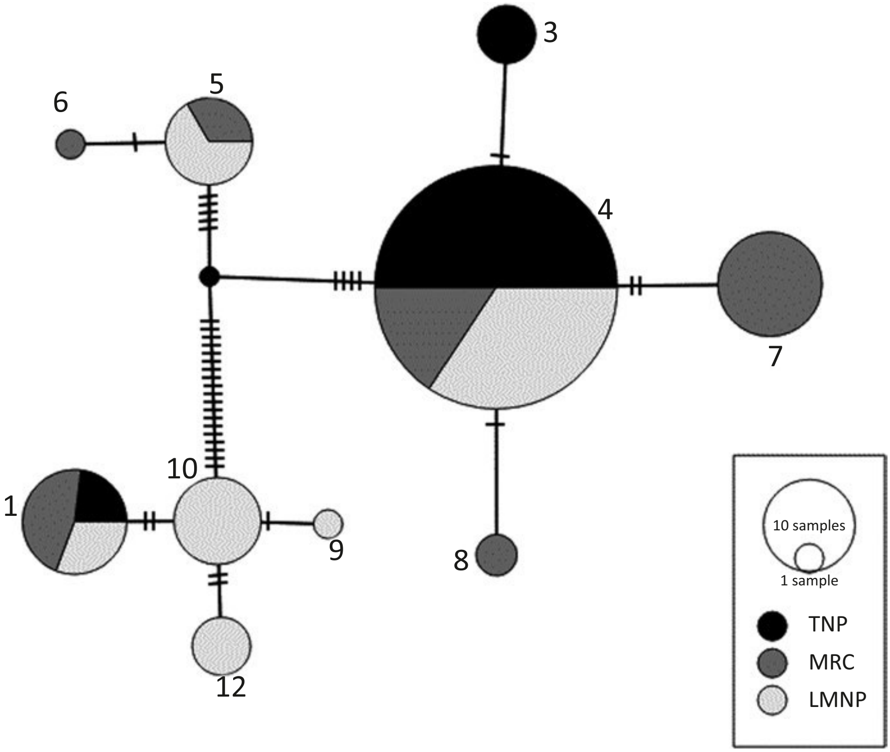

Fig. 12.5 A median joining network created from mtDNA haplotypes of elephants from the Tarangire Ecosystem (Lohay et al. 2020). Numbers on the network indicate haplotypes and the size of a circle reflects haplotype frequencies. Tarangire and Lake Manyara NPs shared only two haplotypes (Haplotype 4 and 1). Both Manyara Ranch Conservancy and Lake Manyara NP had higher numbers of haplotypes than Tarangire NP. The number of mutations between haplotypes is indicated with hatch marks

haploid fashion from mothers to calves and can only tell us about female-mediated gene flow. Nuclear loci analysis further reveals low but significant genetic differentiation between TNP and LMNP/MRC (Fig. 12.4). Nuclear loci reflect much less population differentiation than mitochondrial loci (Nyakaana and Arctander 1999). Female elephants are philopatric and remain with their natal herd for life whereas males disperse at sexual maturity and mediate gene flow between herds (Nyakaana and Arctander 1999). However, given the geographic proximity between the areas, we expected to see more genetic similarity using nuclear markers (Fig. 12.4). Genetic differentiation between TNP & MRC (F'st = 0.126), TNP & LMNP (F'st = 0.175), and LMNP & MRC (F'st = 0.077) was low but statistically significant.

Low genetic diversity among elephants in TNP could also reflect a bottleneck effect. Another explanation could be heavy poaching in the past that might have reduced genetic variability. Elephants experienced heavy poaching outside TNP during the 1970s and early 1980s, but their numbers increased from 440 in 1960 (Lamprey et al. 1964) to about 5000 individuals in 2020 (**Foley and Foley** Chap. 10) (Fig. 12.5).

12.3.6 Wildebeests

12.3.6.1 Past Connectivity

Wildebeests have been present in East Africa for more than 1.5 Myr as evidenced by fossil records found in Oldupai Gorge in Tanzania (Arctander et al. 1999). Out of five subspecies of blue wildebeests (*C. taurinus*), two are found in northern Tanzania and Kenya: eastern white-bearded wildebeest (*C. t. albojubatus*) are distributed east of the Gregory Rift wall from southern TNP to Nairobi National Park in the north, and the western white-bearded wildebeest (*C. t. mearnsi*) are found west of the rift wall in Serengeti, the Ngorongoro highlands, and the greater Mara ecosystem, stretching west to the Ukerere peninsula and Mau escarpment (Talbot and Talbot 1963). Museum collections from the early 1900s indicate that the western race also occurred at the bottom of the rift valley near Lake Naivasha (Talbot and Talbot 1963), with the possibility of interconnectivity with the eastern race north of Lake Natron. In the TE, wildebeests once moved throughout the grasslands in the region along several routes radiating from the area now designated as TNP (Lamprey et al. 1963). As late as the 1930s, wildebeests congregated around water sources at the base of Mt. Meru during the dry season; some movement between TE and the Amboseli basin likely occurred along routes to the east and west of Mt. Meru (Lamprey 1964).

Genetic data suggest significant differentiation between the TE and Serengeti wildebeests based on mtDNA (Georgiadis 1995). Wildebeests found east of the rift valley are morphologically different from wildebeests west of the rift valley (Talbot and Talbot 1963). The TNP and Nairobi wildebeests shared one haplotype but did not share any haplotype with the Serengeti population (Georgiadis 1995). The rift valley wall is an important physical barrier to movement and determines the distribution of mtDNA haplotypes (Georgiadis 1995). For example, wildebeests have not moved between Ngorongoro and Manyara for at least the last 1000 years largely due to the dense forest in Ngorongoro area (Georgiadis 1995).

Microsatellite loci data further revealed significant genetic differentiation between the TE and Serengeti wildebeests (Ernest 2012). The TNP population was found to have a reduced amount of genetic variation compared with the Serengeti, possibly because of large historical differences in relative population sizes. Tarangire wildebeests probably experienced a significant reduction in the effective population size in the past, or there was a small founder population during the colonization of the Tarangire area many years ago. While it is unlikely that the wildebeest population in the TE rivalled abundances in the Serengeti in recent times given the TE has limited dry-season water availability and a smaller and more arid expanse of nutrient-rich grasslands, the eastern white-bearded wildebeest may have numbered several hundred thousand in the region (Estes 2014). By the time of the first rigorous counts, the population had collapsed to only a few thousand animals due to habitat loss, restrictions in water access, and intensive hunting (Lamprey 1964). However, rinderpest likely played an important limiting role as well, as evidenced by the rapid increase in abundances in the 1970–80s, coincident with population growth in the Serengeti population. Extensive habitat loss near Athi-Kaputei and Nairobi National Park in Kenya due to fencing and cultivated farming has severely impacted the eastern race throughout its range (Estes 2014).

12.3.6.2 Recent Connectivity

Wildebeests in the TE currently migrate in three main directions: (1) between TNP and the Gelai Plains near Lake Natron (TCP 1998; Morrison and Bolger 2014; Morrison et al. 2016), (2) between TNP and the Simanjiro Plains (Kahuranganga and Silkiluwasha 1997), and (3) a small population (several hundred animals) moving between TNP and Kimotorok village in the southeast (Fig. 12.2; TCP 1998), though apparently not moving into the more densely wooded Makame WMA. These routes follow the structural connectivity identified in the landscape from satellite imagery (Fig. 12.2a) and are substantiated by GPS data (Morrison and Bolger 2014) and connectivity modelling based on circuit theory and least-cost path analysis (Bond et al. 2017). Several important movement bottlenecks have been identified (Morrison and Bolger 2014) and represent important conservation priorities for both wildebeests and other species (Fig. 12.3). Photo mark-recapture data show that, each year, 0–12% of wildebeests switch wet season ranges between the Gelai plains and Simanjiro, suggesting wildebeests may have relatively strong fidelity at the scale of ranges (Bolger and Morrison 2012). As landscapes change rapidly, site faithfulness poses an additional threat to the long-term viability of populations because it inhibits movements to alternative sites, assuming alternative sites are still available. Grasslands are relatively easy to convert to cultivated cropland, and accordingly grassland habitat used by wildebeests has been widely lost in many areas, particularly the Simanjiro Plains (Msoffe et al. 2011) which is rich in phosphorus relative to soils in TNP (TCP 1998; Voeten et al. 2010). Conservation easements in several villages in Simanjiro have recently been established to protect some grazing areas from cultivation (Nelson et al. 2010).

12.3.7 Lions

Across the entire TE, a relatively small proportion of the lion population occurs in formally protected areas (i.e., IUCN categories I-V; Riggio et al. 2013), suggesting *in situ* conservation efforts in community lands are needed to ensure populations remain viable (Lichtenfeld et al. 2015; Beattie et al. 2020). In the absence of such efforts, corridors may only serve to exacerbate existing conflict between lions, livestock, and people in the TE (Beattie et al. 2020).

Relative to elephants and wildebeests, lions in the TE have small home ranges (52–616 km^2 with an average of 209 km^2, Laizer et al. 2014), and GPS and VHF-collars have documented relatively limited movement of lions between different areas in TE (Fig. 12.3). Between TNP and MRC, lions appear to use similar routes to travel as elephants and wildebeests (Fig. 12.3). A single GPS-collared lion also traveled west of TNP to the Lake Manyara shoreline (Fig. 12.3), and VHF tracking suggests lions move between TNP, Lolkisale GCA, and the Simanjiro Plains. To date, no population genetic studies been conducted on TE lions.

12.3.8 Giraffes

Since 2011, giraffes have been intensely monitored in TNP, MRC, LMNP, and Mto wa Mbu and Lolkisale GCAs (the northern portion of which is now part of Randilen WMA) using computer-assisted photographic mark-recapture techniques (Lee and Bolger 2017; **Lee and Bond** Chap. 9). Giraffes in the TE have relatively small home ranges compared to elephants and wildebeests (adult females = 114.6 ± SD 49 km^{2,} n = 109 and adult males = 157.2 ± SD 44.9 km^{2,} n = 23; Knüsel et al. 2019), so movements between protected areas are not expected to occur frequently particularly in areas with high intensity of use by humans. Lee and Bolger (2017) found the highest levels of population connectivity between TNP and MRC, and between TNP and Lolkisale GCA. Importantly, TNP and MRC were the demographic engines driving population growth in the larger ecosystem. This suggests that maintaining connectivity with TNP and MRC will be important for giraffe population viability in other subpopulations, particularly the GCAs which act as demographic sinks (Lee and Bolger 2017).

12.4 Is Lake Manyara National Park Functionally Isolated?

There are incipient signs of isolation of LMNP wildlife populations. While movement between TNP and LMNP is physically possible through the recently established Burunge WMA, MRC (Fig. 12.2) and via unfenced community land west of MRC, empirical data suggest relatively little movement between these areas (e.g., Lee and Bolger 2017; Morrison and Bolger 2014; Morrison et al. 2016; Lohay et al. 2020). Genetic data from three species (elephants, giraffes, and wildebeests) suggest early signs of genetic isolation. LMNP elephants and giraffes show significant genetic differentiation from those in TNP, MRC, Ngorongoro and Serengeti NP (Lohay et al. 2020; Lohay et al. unpub). Over the past 40 years, the LMNP elephant population had declined (but not linearly) due to poaching and loss of habitat especially from the expansion of human settlements and farming (Prins et al. 1994; Kiffner et al. 2017). In 1973, LMNP was recorded to have the highest density and most rapid population growth rate of any elephant population (Douglas-Hamilton 1973). Despite high poaching intensities in the 1970s and 1980s, in recent years elephant poaching was significantly reduced in the park. Currently, the density of elephants is relatively high (Kiffner et al. 2017). There is evidence for genetic similarity between LMNP and Ngorongoro Conservation Area for elephants which suggests recent gene flow (Lohay et al. 2020) or reflects that elephants were once one population (**Prins and de Jong** Chap. **7**). The most likely travel route presently connecting these populations is via the corridor above the Selela Forest Reserve (Upper Kitete corridor), north of Mto wa Mbu (Fig. 12.2).

For giraffes, mtDNA data reveal significant genetic differentiation between LMNP and MRC (Lohay et al. unpub), suggesting interchange is relatively rare. Photo mark-recapture data collected from 2011 to 2014 show low movement rates between LMNP and MRC, and between Mto wa Mbu GCA and LMNP (Lee and Bolger 2017). Only one individual (female) has been observed moving between MRC and LMNP (Lavista-Ferres et al. 2021).

Photo mark-recapture data of wildebeests also showed a low degree of movement between LMNP and other parts of the TE. A total of 12 wildebeest were observed moving between LMNP and Mto wa Mbu GCA or MRC, but not TNP (Bolger and Morrison 2012), out of >700 recaptures. The routes of these movements are unclear, though we suspect they occur through village land between the lakeshore and MRC. When lake levels are high and grazing habitat is flooded, wildebeests tend to move away from LMNP (Prins and Douglas-Hamilton 1990). If LMNP becomes functionally isolated, flooding will severely threaten the wildebeest population in this area.

Lions have not been observed moving from LMNP and other areas, although there is considerable conflict with lions in community land between MRC and LMNP in Oltukai and Esilalei villages, and it is likely lions follow migratory prey at certain times (**Kiffner et al.** Chap. 11; **Kissui et al.** Chap. 14).

In the long run, wildlife in LMNP may be at risk of extinction or require intensive management (e.g., translocations) to avoid inbreeding if natural movement connectivity is lost. Small populations are prone to genetic and demographic stochasticity and may easily drift apart or suffer extirpation (Frankham et al. 2017).

12.5 Conclusion

The TE is a complex savanna landscape with at least 14 different protected areas which are managed under the authority of at least 9 different entities, not including forest reserves (Table 12.1). Movement data and historical accounts suggest the TE is one large, functionally connected ecosystem, with some connectivity to neighboring ecosystems in the west (Ngorongoro), south (Ruaha), and north (West Kilimanjaro). Wildlife moving within and between protected areas must navigate an increasingly fragmented landscape (Msoffe et al. 2011), with wildlife populations under pressure from illegal hunting for bushmeat (Rentsch and Damon 2013), vehicle collisions (Kioko et al. 2015), and conflict with humans (Shemweta and Kideghesho 2000). Balancing the needs of wildlife with those of the local human communities is a considerable challenge in land-use planning, cooperation, and communication between various stakeholders. In this regard, corridors could come at a relatively low cost, as they need not be particularly large geographically to be effective at maintaining functional connectivity, particularly if corridors are thoughtfully located with respect to historical wildlife movements (Morrison and Bolger 2014).

The ecological benefits of corridors, in terms of population persistence and outbreeding, are considerable. Our synthesis suggests that elephants, lions, and wildebeests and (likely) giraffes use some of the routes to travel between different parts of the ecosystem. In particular, the narrow strip of land between the northern tip of TNP and southern part of MRC was used as a travel route by all species and is only partially protected by the Randilen WMA (Fig. 12.3). Similarly, Burunge WMA serves as an important linkage between LMNP and TNP and may prevent the isolation of LMNP. While movement through Burunge WMA has yet to be documented empirically with individual-based data, the area already has higher species abundance and diversity compared to unprotected areas nearby (Lee 2018; Kiffner et al. 2020; **Bond et al.** Chap. 9). Nonetheless, parts of the Burunge corridor are relatively densely wooded which may be unsuitable for grazers such as wildebeests and zebras, so alternative routes are still needed. Indeed, research in the Okavango Delta in Southern Africa suggests that connectivity, based on circuit theory models, differs considerably between different wildlife species (Brennan et al. 2020), and species-specific planning of corridors may be necessary.

Challenges facing wildlife corridors have been well documented in Tanzania (Caro et al. 2009; Jones et al. 2012; Riggio and Caro 2017; Riggio et al. 2018). Yet, Tanzania is now amongst the first countries globally to enact legislation that affords legal protection for wildlife corridors as a unique form of land use (Kauffman et al. 2021). While this legislation will undoubtedly aid landscape-scale conservation planning, wildlife will continue relying on many areas that are outside of formal protection. Thus, community-driven conservation efforts will remain critical to maintaining human-wildlife coexistence in these areas. People must be involved in decision making and should realize benefits from such corridors. Such essential efforts include fortified bomas to protect against lion predation (Lichtenfeld et al. 2015; Kissui et al. 2019), village game scouts to monitor human and wildlife activities (Foley and Foley 2014), conservation easements and community designation of grazing areas (Nelson et al. 2010), and village land-use planning (Kaswamila and Songorwa 2009). The absence of impermeable man-made barriers such as fencing in the TE also remains hugely beneficial to wildlife because movement is still physically possible in many areas despite intensification of human activities. Looking forward, we envision a need for viewing the TE as a single, large, connected landscape that requires coordination amongst the various land-use managers, including local community leaders, to ensure functional connectivity is maintained.

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Chapter 13
Characterizing Elephant-Livestock Interactions Using a Social-Ecological Approach

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Abstract In the Tarangire Ecosystem, elephants frequently use pastoral areas, where they interact with people and livestock. To characterize the elephant-livestock interface in Manyara Ranch, we used a social-ecological approach to capture the herders' and the elephants' perspectives of these interactions. We interviewed cattle herders to assess their perceptions of elephants relative to other wildlife species (n = 117 interviews) and observed how elephants responded to sound playbacks associated with humans and cattle relative to sounds of wildlife species (n = 300 playbacks). Most herders (86%) supported elephant conservation, and reported spatial avoidance of elephants as the main strategy to avoid negative interactions. Among eleven large mammal wildlife species, herders ranked elephants as the fifth most problematic species to cattle. Elephants frequently reacted (e.g., bunching, fleeing, shaking the head and moving the trunk, or approaching) to human-related sound playbacks (79% of playbacks), and reacted less frequently when exposed to

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sounds of cattle (62%) or wildlife (34%). Playback experiments suggested that while elephants primarily reacted non-aggressively when faced with livestock, aggressive elephant behavior may be triggered by human behavior. Evidence from both the interview data and the behavioral experiments suggest that coexistence between elephants and pastoralists is mostly facilitated by mutual spatial avoidance.

Keywords Human-elephant conflict · Livestock-elephant conflict · Stakeholder participation · Animal behavior · Crop raiding

13.1 Introduction

In East Africa, the range of African elephants (*Loxodonta africana*) extends well beyond the boundaries of formally protected areas (Epps et al. 2011; Kioko et al. 2015b; Thouless et al. 2016; Osipova et al. 2019). Elephants use space outside formally protected areas as core habitat (Kioko et al. 2013), as seasonal dispersal areas (**Foley and Foley** Chap. 10), or as corridors to move between their core habitats **Lohay et al.** Chap. 12). From a biological perspective, such extensive use of human-dominated areas and connectivity across the landscape may ostensibly seem conducive for the long-term persistence of elephant populations (Caro et al. 2009). However, the use of human-dominated areas by elephants presents a formidable conservation challenge (Shaffer et al. 2019). On the one hand, with elephants being a high profile species for conservation and subject to substantial human-caused declines across the continent (Chase et al. 2016), landscape connectivity is a key goal for their conservation. On the other hand, when elephant and human space use overlaps, interactions can be detrimental for both elephant and human wellbeing (Songhurst et al. 2016). Human-wildlife interactions are thus often framed as "human-elephant conflicts" (HEC). While most fundamental and applied research on the patterns and mitigation efforts of HEC have focused on crop and property damage caused by elephants (Sitati et al. 2003; Hoare 2015; Denninger Snyder and Rentsch 2020), relatively little attention has been directed towards interactions between elephants, livestock, and pastoralists. This is surprising because a substantial share of the elephant range in East Africa overlaps with grazing lands of the Maasai (Kioko et al. 2015c) and this spatial intersection can be expected to result in frequent interactions between elephants, livestock, and humans. In pastoral areas, human-elephant interactions can be diverse in nature and can have desirable and undesirable consequences for both elephants and livestock alike.

Being large and powerful animals, elephants are capable of killing both humans and livestock (Rodriguez and Sampson 2019; Shaffer et al. 2019). To date, systematic assessments of the elephant-livestock interface have not been conducted in the Tarangire Ecosystem (TE), yet examples from multi-use areas in Kenya suggest that such interactions can have severe consequences for elephants, livestock and humans. For example, Thouless (1994) mentioned that in 1992, elephants killed six cows and one sheep in Laikipia, Kenya. During the same year, six people were injured and 13 were killed by elephants (Thouless 1994). People often respond to such traumatic events by killing elephants. Occasionally, rural people retaliate directly and kill elephants with spears, other weapons, or poison. In other cases, wildlife authorities conduct targeted "problem animal control" and kill elephants with rifles. For example, during 1992, 42 elephants were killed either illegally or legally as part of problem animal control in Laikipia (Thouless 1994).

Beyond such direct interactions, elephants may be involved in the transmission of zoonotic pathogens that can cause disease in livestock. Examples include blue tongue, rift valley fever, tuberculosis, trypanosomiasis, and foot-and-mouth disease (Howell et al. 1973; Pastoret et al. 1988). In addition, elephants consume large amounts of grass (primarily during the rainy season) and browse (primarily during the dry season) (Clauss et al. 2007; Owen-Smith and Chafota 2012). From an anthropogenic perspective their feeding ecology could be perceived as either ecosystem service or disservice. On the one hand, elephants and cattle may compete for grass (Young et al. 2005, 2018), while on the other hand, extensive browsing and destruction of woody vegetation may facilitate grass growth (Young et al. 2005; Augustine et al. 2011). In addition, in their role as charismatic megafauna, elephants are crucial for wildlife-based tourism and may thus be associated with substantial economic benefits (Naidoo et al. 2016). Elephants may also provide non-material contributions to human societies (Methorst et al. 2020). For example, elephants hold a prominent role in Maasai culture. Maasai people not only use elephant parts for medicinal, ceremonial, ritual, or consumptive purposes, they also view elephants to be similar to humans (Kioko et al. 2015c).

As human-elephant interactions are diverse, people may have different attitudes towards elephants. Understanding human perceptions of these interactions can greatly influence human tolerance towards wildlife and eventually determine how people comply with wildlife protection regulations, and how they respond to interactions with wildlife. Perceptions can explain the degree to which people are willing to coexist with wildlife (Kansky et al. 2016; Expósito-Granados et al. 2019). Thus, understanding how people perceive elephant-livestock interactions may determine whether they support or oppose elephant conservation (Kuriyan 2002; Struhsaker et al. 2005; Lee and Graham 2006; Kansky and Knight 2014).

In addition to human perceptions, it is equally important to understand how human activities (such as livestock keeping) affect elephant behavior. Because direct observations of interactions are rare, we employed an experimental approach to simulate elephant interactions with humans and cattle. To simulate such interactions we used sound playbacks in a landscape frequented by elephants, pastoralists, and livestock. Elephants are a very intelligent species, with remarkable sensory abilities to differentiate between a variety of threats (Kelley and Garstang 2013; McComb et al. 2014; Soltis et al. 2014) and their response towards broadcasted sounds serves as a suitable proxy for assessing the elephant-perceived threat associated with the sound (McComb et al. 2014).

In this chapter we present the results of this social-ecological assessment of human-elephant interactions in the Tarangire Ecosystem. Our study intends to analyze these interactions from both the elephant as well the human perspective as a basis for assessing possibilities and challenges of coexistence between livestock keeping and elephant conservation (Nyhus 2016; König et al. 2020).

The study area, Manyara Ranch (MR), functions as permanent habitat, seasonal range, and a stepping stone for movements of elephants and other wildlife species using Tarangire and Lake Manyara national parks (Kioko et al. 2013; Kiffner et al. 2016, 2020a; Bond et al. 2017; **Bond et al.** Chap. 8; **Lohay et al.** Chap. 12). In addition, MR is a dry season grazing area for Maasai pastoralists (Warwick et al. 2016). This dual function makes it an ideal system to conduct an interdisciplinary study on human-elephant interactions. We discuss the results of our study with respect to enhancing human-elephant coexistence in rangelands (Keesing et al. 2018; Young et al. 2018).

13.2 Manyara Ranch

Manyara Ranch (MR) is a 183 km^2 unfenced ranch located between Lake Manyara and Tarangire national parks (Fig. 13.1) that is managed for wildlife and livestock. The climate is semi-arid; annual precipitation ranges between 434 and 824 mm and occurs mainly during the long (March–May) and the short (November–December) rain seasons (Prins and Loth 1988). The vegetation is mainly dominated by *Acacia (Vachellia)-Commiphora* bushland and grassland. In the dry season, surface water in MR is limited to a few human-made dams and natural pools in the Makuyuni River.

Historically, pastoralists from two Maasai communities, Esilalei and Oltukai, utilized the area during the dry season for livestock grazing. From 1956 to 1971 the land was leased by a German farmer who operated a commercial cattle farm on the land. From 1971 to 2001, MR was part of a system of nationally owned cattle ranches in Tanzania but MR did not operate profitably. In 2001, its status was changed to a conservation trust model and more recently (2017) the area administration was handed over to Monduli District. Since 2001, the management objectives of MR are to ensure sustainable wildlife conservation and to improve livestock development. Currently, African Wildlife Foundation (AWF) is financially supporting most operations in MR. The two adjacent villages (Esilalei and Oltukai) hold grazing rights to MR, and receive income from wildlife tourism activities in MR (Sumba et al. 2005). MR itself also holds several hundred head of cattle and sheep (Grzeda et al. 2017) whose daily movements are managed by herders employed by MR and follow a rotational grazing scheme. In this study, we focused solely on livestock herders from Esilalei and Oltukai who graze their livestock in MR during the dry season only.

MR holds about 70–100 elephants (Kiffner et al. 2020b), and the sex ratio in the MR elephant population is skewed towards males (Kioko et al. 2013). The elephant population in MR has experienced some recent cases of poaching (Kioko et al. 2013; **Foley and Foley** Chap. 10). However, due to improved anti-poaching efforts and coinciding with a general decline in poaching in Tanzania, poaching rates in MR were minimal during the study period (2015–2017). Farms adjacent to MR are often subject to crop raiding by elephants (Bencin et al. 2016; Hahn et al. 2017).

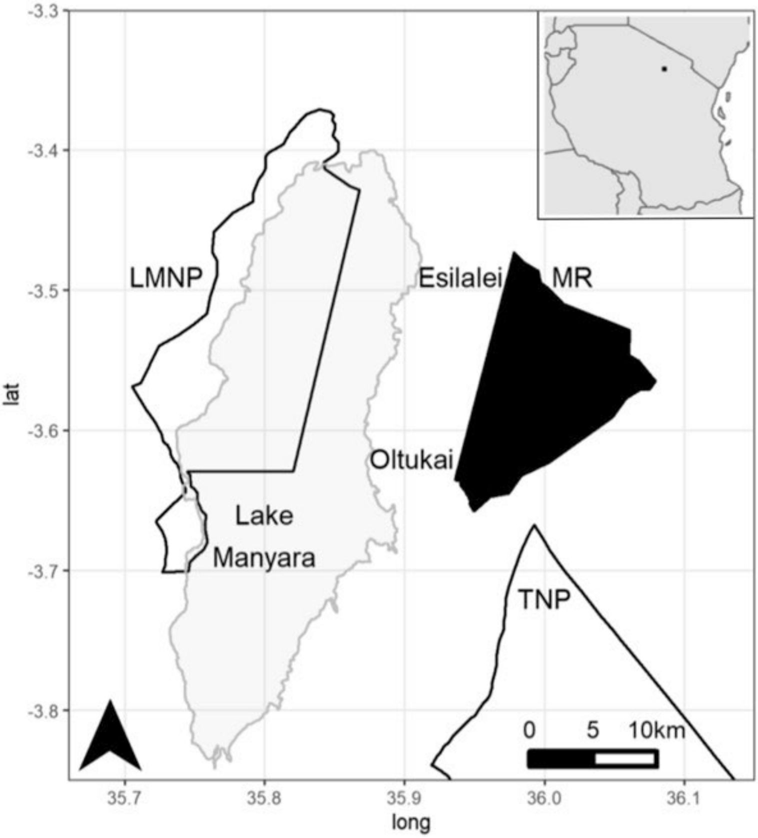

Fig. 13.1 Location of Manyara Ranch (MR) in relation to Lake Manyra National Park (LMNP), Lake Manyara and Tarangire National Park (TNP). Pastoralists of the two villages Esilalei and Oltukai possess dry season grazing rights in MR. The inset in the top right indicates the location of the study area within Tanzania

13.3 Assessing Attitudes and Perceptions of Herders Towards Elephants

To assess attitudes and perceptions towards elephants, we conducted interviews with 117 herders in MR during 3 sessions: November 2015, November 2016, and April 2017. Prior to the study, the interview protocol was reviewed and exempted from further Institutional Review Board review under the U.S. Code of Federal regulation title 45 public welfare part 46 protection of human subjects 46.101b (Type B, Category 2; IRB protocol number: TZ-05-15). We divided the ranch into four sections based on the cattle grazing sectors defined by the management of MR. In each sector, a research team (consisting of the authors, one MR ranger, and a translator) walked in the area in search of herders for 2 days. This was done repeatedly in each study session. Upon approaching herders, we asked for consent

Table 13.1 Questionnaire used to assess perceptions of livestock-elephant interactions and attitudes of pastoralists towards elephants in Manyara Ranch

Question/ranking	Answer/ranking options
Do you farm crops around Manyara Ranch?	Yes; No
How often do you encounter elephants in MR?	Often; Rarely; Never
Please rank the following animals from most problematic to least problematic to cattle in MR	Ranking of buffalo, elephant, lions, hyena, wildebeest, zebra, giraffe, eland, raptors, jackal, baboon based on pictures
Please rank the following animals from most problematic to least problematic to humans in MR	Ranking of buffalo, elephant, lions, hyena, wildebeest, zebra, giraffe, eland, raptors, jackal, baboon based on pictures
What do you do when you see elephants approaching your cattle?	Nothing; Chase the elephants away; Herd cattle away from elephants; Abandon cattle and run away
Do you think elephant presence in MR affects food availability for your cattle?	Yes; No. If yes: Grass; Bushes/Trees; Grass and Bushes/Trees
Do you avoid taking cattle to areas where you see elephants?	Yes; No
Do you avoid taking cattle to areas where you know elephants are likely to occur?	Yes; No
Do you support continued elephant presence in MR? Please explain your response.	Yes; No. Open-ended.
Do you believe that diseases can be transmitted by elephants to your cattle? If yes, please name the disease.	Yes; No. Open-ended.
Has any of your or others herder's livestock been injured or killed by elephants in Manyara Ranch?	Cattle: Yes; No. Goats: Yes; No. Sheep: Yes; No.
Has anyone you know been injured or killed by elephants in Manyara Ranch?	Yes; No.
Where did the injury occur?	Grazing area; Watering point /dam; Boma; Salt lick; Road

to take part in this study, ensured anonymity, and explained they had the right to stop the interview at any time. If herders verbally expressed their consent, we conducted the interview with the aid of field assistants who were residents in the study area and well versed in English, Swahili, and Maa (the language of Maasai people). We interviewed adult herders only (above 18 years of age), and if there was more than one herder with one livestock herd, we interviewed the most senior one. We attempted to interview individual herders only once, by asking interviewees whether they had been interviewed previously.

The questionnaire (Table 13.1) included closed- and open-ended questions to capture information on attitudes and beliefs related to elephants and other wildlife species. Open-ended questions helped to obtain more context and explanations for the closed questions. We designed the questions to include elements of triangulation (Heale and Forbes 2013) such as species ranking, timeline analysis, and open discussions in order to increase the robustness of the responses given.

We asked the cattle herders questions regarding their interactions with elephants when grazing their livestock in MR. The questions included whether they avoided taking cattle to areas known to be occupied by elephants, what they did if elephants approached their cattle, and whether they believed elephants in MR affected food availability for their cattle. To assess the scope and nature of elephant-cattle conflicts, we asked the herders about their perceptions regarding possible pathogen transmission between cattle and elephants, and whether elephants are a direct threat to their livestock.

We also assessed how the herders perceived the relative conflict severity of eleven wildlife species with respect to human and livestock wellbeing. For this ranking, we considered the following ten wildlife species: African buffalo (*Syncerus caffer*), elephant, lion (*Panthera leo*), hyena (*Crocuta crocuta* and *Hyena hyena*), wildebeest (*Connochaetes taurinus*), zebra (*Equus quagga*), giraffe (*Giraffa camelopardalis*), eland (*Taurotragus oryx*), brown snake eagle (*Circaetus cinereus*), black-backed jackal (*Canis mesomelas*), and olive baboon (*Papio anubis*). These species were chosen because they are common in the ranch (Kiffner et al. 2016) or known to cause conflict with livestock in the study area (Bencin et al. 2016; Koziarski et al. 2016). We presented pictures of the species and requested the herders to place them in order (1–11) of perceived severity as a "conflict" species to cattle and humans (rank 1: most problematic; rank 11: least problematic). "Conflict" was explained to involve any problem or threat they may have or foresee to arise when the herders (or their livestock) encountered the wildlife species. To assess herders' attitudes towards elephant conservation in MR, we asked herders if they did or did not support continued elephant presence in MR and requested them to provide reasons for their answers.

13.4 Playback Experiments to Study Elephant Behavior Towards Livestock Herding

We conducted playback experiments (n = 300 playbacks) to assess elephant behavioral responses to recorded sounds of cattle, wildlife, humans, and dogs during 3 sessions: November 2015, April 2016, and April 2017. We assumed that these distinct sessions limited the extent of elephant habituation to repeated sounds (Goodyear and Schulte 2015). Sounds included: zebra, cattle (cattle mooing, cattle bell sounds), hyena, lion, herding dogs barking, and human herders whistling to cattle). We converted the recordings from MP4 to wav format using the Zamzar software (http://www.zamzar.com), normalized the soundfiles using the Audacity 2.1.2 software (http://www.audacityteam.org), and then exported the soundfile as 16 bt. wav files into a Firestorm Foxpro speaker (Bushcraft USATM).

From the top of an open vehicle we broadcasted the sounds at distance of 50 m from elephants (measured using a laser range finder) using the same setting of the speaker system. A single playback involved playing all sounds to an individual elephant or a group of elephants. The order of the sounds was kept constant and started with what we assumed was the least threatening sound and ended with the most threatening. The order of playback sounds was: zebra, cattle, cattle bell, hyena, lion, dog, herders. However, not all sounds were played for each experiment because elephants occasionally disappeared before all sounds were played. We used the observed elephant responses to each sound to gauge the perceived level of threat that elephants associated with each sound (Thuppil and Coss 2013; McComb et al. 2014). We classified the observed elephants' responses to the sound playback as: (0) no response, (1) bunching (elephants gather tight, often positioning juvenile elephants in the center, (2) fleeing (walking or running away from the speaker), (3) approaching (elephants moving toward the speaker, often smelling the ground and air), (4) shaking the head and flapping ears, (5) freezing (stopping activity for an extended period of time), (6) giving an alarm call, and (7) backing up (following McComb et al. 2014). Elephants typically reacted as group and we thus recorded one response per elephant group.

Within each session, we conducted playback experiments at most twice (but never on the same day) to a specific elephant group, to maintain independence of observations and to reduce chances of habituation to the sounds. We identified elephant groups using characteristics of individuals such as ear shape, general morphology, and tusk length and shape and used digital photographs to support individual identification of elephants. However, due to the fission-fusion grouping behavior of elephants (Archie et al. 2006), cases of repeat playbacks to some of the same individuals may have occurred.

13.5 Statistical Analyses

We analyzed data using the Statistical Package for Social Science (SPSS) program (SPSS 2006) and R 3.6 (R Core Team 2016) for visualization of results. We used alpha = 0.05 for all statistical tests.

To analyze interview data, we extracted descriptive statistics (percentage of responses in each response category) and used chi-square tests to assess if responses were equally distributed. To establish the threat ranking of different wildlife species for human or livestock health, we estimated the mean rank (and associated standard errors) of each wildlife species.

To analyze the behavioral response of elephant to playbacks, we calculated percentages of elephant group responses per playback sound and used chi-square tests to assess if the elative frequency of elephant group responses differed by playback sound. To assess patterns in elephant responses to similar species, we combined sounds into 3 categories: humans (herders and dog), wild animals (lion, hyena, and zebra), or cattle (cattle and cattle bells).

13.6 Attitudes and Perceptions of Herders Towards Elephants

Most herders (66%) reported that they often encountered elephants in MR when grazing cattle. The majority of herders (71%) said that they would generally avoid areas occupied by elephants. Similarly, if herders encountered elephants, the majority of herders (78%) stated that they would herd their cattle away from the elephants. Herders' responses on how they would react if elephants approached their cattle were not equally distributed significantly ($\chi^2 = 99.65$, df = 3, $p \leq 0.01$). Most herders (63%) stated that they would leave the cattle and move away. About a fifth (22%) said they would drive their cattle away, 11% said they would try to scare off the elephants, and 4% said they would not do anything. In the hypothetical event an elephant walked towards their cattle, 65% of herders said that cattle would retreat from elephants by walking or running away, while 35% of herders said that cattle would be vigilant and then continue their initial activity ($\chi^2 = 99.65$, df = 3, $p \leq 0.01$).

Herders ranked elephants as the fifth most problematic wildlife species to cattle and as the third most problematic species with regards to humans (Fig. 13.2). The majority of interviewed herders (80%) believed that elephant presence in MR decreased food availability for their cattle, with 14% of herders believing that elephants specifically decreased grass availability for their cattle. About a third (31%) of the herders stated that elephants caused health problems to their cattle. Reported diseases included skin photosensitization, 3-day sickness, lumpy skin disease, and hemorrhagic septicemia. Some herders believed that elephants were hosts of tsetse flies and ticks and would amplify their abundance.

Most herders (87%) expressed their support for continued elephant presence in MR. Underlying reasons for this relatively high level of tolerance for elephants included both tangible and intangible benefits (Fig. 13.3). Elephants were considered important for tourism (42%), and employment (19%). Interviewees often elaborated on these tangible benefits and reported that elephants are conducive for wildlife-based tourism and that village members were employed as game rangers, herders, and workers in the tourism lodge of MR. Beyond these tangible benefits, intangible benefits and attributes such as the conservation value of elephants (18%), their peacefulness (9%), their role as national heritage (7%) and their perceived beauty (5%) were frequently mentioned (Fig. 13.3). Herders who did not support (13%) continued elephant occupancy in MR said it was because elephants are a potential threat to humans, that elephants destroyed crops and depleted forage for their cattle.

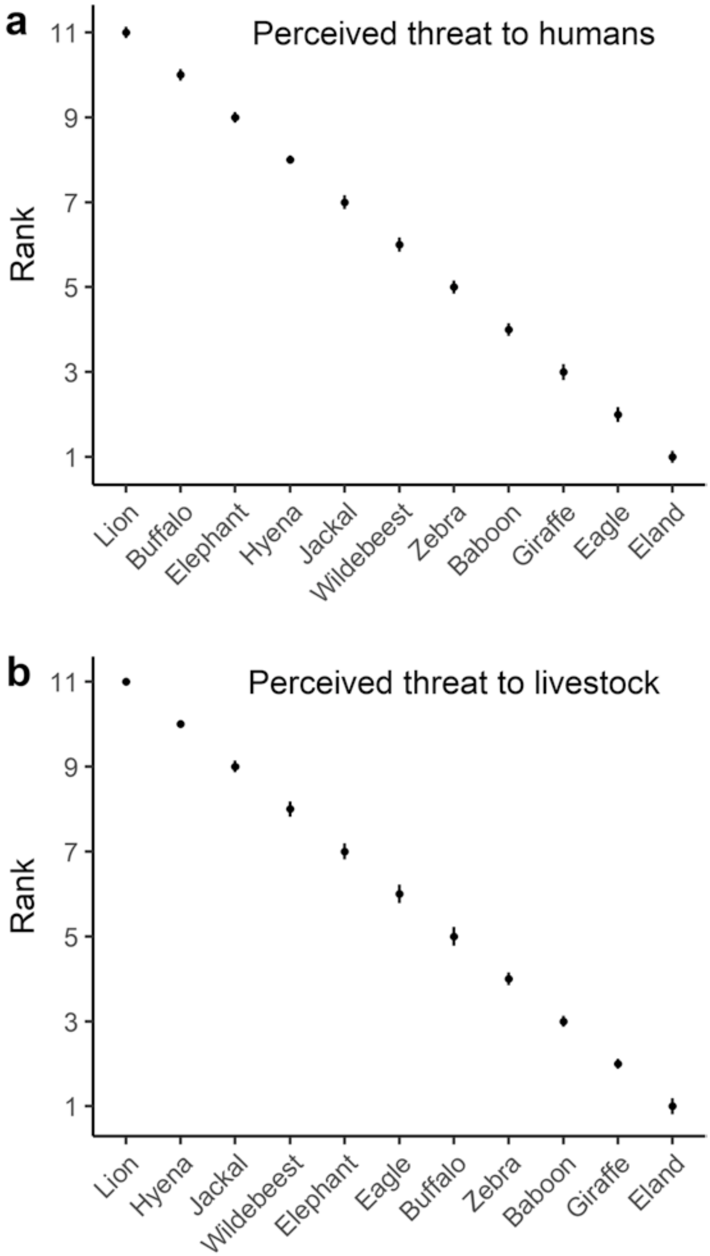

Fig. 13.2 Perceived conflict severity associated with eleven common wildlife species in Manyara Ranch. Livestock herders ranked the threat imposed to humans (**a**) and livestock (**b**) by the different species from most (rank 11) to least (rank 1) severe. Error bars indicate the associated standard errors of the mean ranks

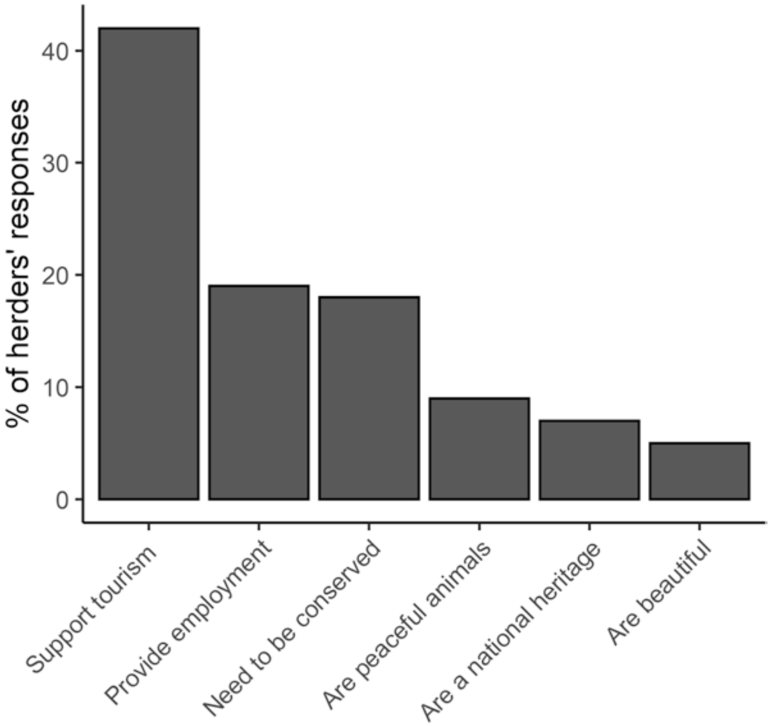

Fig. 13.3 Relative frequency of stated reasons for supporting continued elephant presence in Manyara Ranch. Percentages were based on interviews with 102 livestock herders who supported elephant conservation efforts in the ranch (15 interviewed herders did not support elephant conservation)

13.7 Behavioral Responses of Elephants to Sound Playbacks

Elephants' reactions to sound playbacks depended on whether the sound was related to humans (herders and dog), wild animals (lion, spotted hyena, and zebra), or cattle (cattle and cattle bells) ($\chi^2 = 106.66$, df = 2, $p \leq 0.01$). In general, elephants reacted to human-related sounds most often (80%), followed by cattle sounds (41%), and wild animal sounds (16%). Elephant responded most frequently to sounds of herders (86%) and dogs (73%), and were least responsive towards sounds of wildlife species such as zebra (6%), spotted hyenas (11%), and lions (30%) (Fig. 13.4; $\chi^2 = 145.91$, df = 6, $p \leq 0.01$). When subject to playback recordings of herders, domestic dogs, or cattle, elephant groups most frequently walked or ran away. We observed that elephant groups with calves tended to be more reactive to sound playbacks and typically fled into nearby closed habitats.

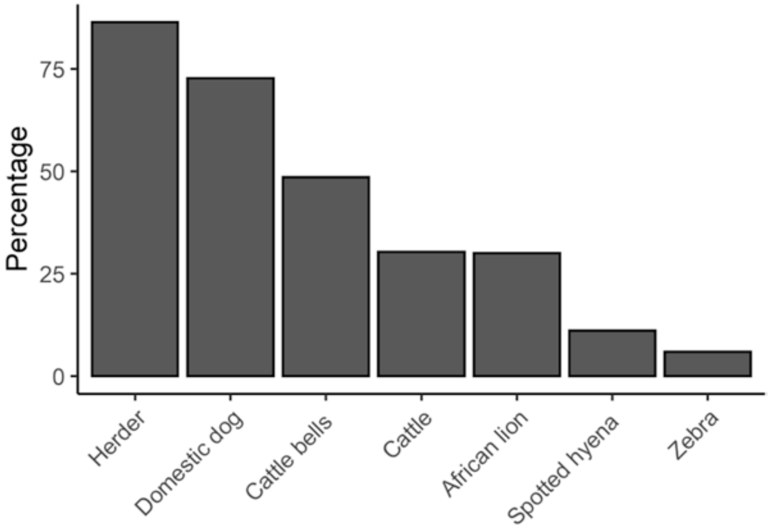

Fig. 13.4 Relative frequency of elephant group responses to playbacks of human-related (herder whistling and dog barking), cattle-related (cattle, cattle bells), and wildlife-related (zebra, spotted hyena, and African lion) sounds. Responses were coded as "response" if elephants showed any of the following responses: bunching; freezing; shaking the head; walking away; running away

13.8 The Human Dimension of Human-Elephant Interactions

Herders generally perceived elephants as a relatively minor threat to cattle, particularly compared to large carnivores (**Kissui et al.** Chap. 14).The interviewed herders reported that there were two cases of cattle injuries and two cattle deaths in MR from 2010–2015. Albeit rare, considering the high livestock densities on the ranch (Kiffner et al. 2016), these incidents can be a major source of animosity between livestock herders and elephants (Thouless 1994; Okello et al. 2014; Shaffer et al. 2019). Interestingly, some of the herders (13%) reported crop damage as a main form of conflict with elephants from MR. Profound land-use and livelihood changes have occurred in the area, resulting in a net increase in farming areas (Msoffe et al. 2011; Nkedianye et al. 2019). While this study primarily targeted pastoralists, most of the interviewed herders (96%) also practiced crop cultivation adjacent to MR and attributed crop losses on their farms to elephants from MR.

About a third of interviewed livestock herders reported that elephants were associated with the transmission of diseases. Some of the interviewed herders also suggested that oxpeckers (*Buphagus africanus* and *B. erythrrhyncus*) act as vectors and transmit pathogens from elephants to livestock. While elephants can be involved in the transmission of zoonotic pathogens if elephants share habitat with livestock (Pastoret et al. 1988), we found no evidence in the literature for the stated elephant-disease associations. While traditional veterinary knowledge is present in many pastoralist systems (Alhaji and Babalobi 2015; Dharani et al. 2015; Kioko et al. 2015a), such knowledge is largely shaped by cultural beliefs (White 2015), and may amplify local peoples' negative attitudes towards wildlife.

Overall, the interviewed livestock herders did not perceive competition for forage as a key conflict. Even though elephants may compete with livestock for forage, this is apparently not a major concern for livestock herders (Gadd 2005) and may be explained by the limited amount of overlap between elephant and cattle forage selection (Owen-Smith and Chafota 2012). While herders argued that elephants would reduce browse material and thus compete with goats (*Capra aegagrus*), only a few herders (14%) noted that elephants would reduce the amount of grass, the main food of cattle. Occasionally, interviewed herders believed that elephants mainly removed browse material that were beyond the reach of cattle. None of the interviewed herders mentioned that elephants possibly improved grazing conditions. This is surprising because the feeding behavior of elephants (i.e. pushing over or uprooting trees, breaking of trees and branches) can increase the availability of browse material to other browsing species (Kohi et al. 2011) and may also open up bushland, and thus facilitate grass growth (Young et al. 2005; Kohi et al. 2011).

Despite these experiences with and beliefs about elephants, most herders stated that the elephant population in MR should be conserved. The variety of reported tangible and intangible benefits of elephants illustrate the complex interplay between herders' values and elephant conservation and the importance of both tangible and intangible benefits for wildlife conservation (Kansky et al. 2021).

Nevertheless, several herders opposed elephant conservation in MR. Herders justified these negative attitudes by stating that elephants caused death and injuries to humans and livestock, damaged their crops, and depleted forage for their cattle. Clearly, such costly interactions with elephants have the potential to cause deep-rooted resentment and conflict among local communities and undermine elephant conservation efforts (Zimmermann et al. 2020). In this specific case, a conflict in agricultural fields (e.g. crop raiding) could possibly spill over to rangelands and, at least partially, affect attitudes of herders towards elephants. Thus, implementing locally feasible and effective solutions to prevent and mitigate elephant crop raiding (Chang'a et al. 2016; Kiffner et al. 2021) could be a potential pathway towards improving herder perceptions of elephants and relations with MR in general.

13.9 The Elephants' Perspective of Human-Elephant Interactions

In line with the overall positive herder perceptions, results of the playback experiments suggest that elephant and cattle interactions are predominantly not based in conflict. However, the presence of humans and/or dogs may provoke anti-predator behavior in elephants, suggesting that elephants perceive more danger when exposed to humans and dogs as compared to livestock or wildlife species. Interactions with dogs may aggravate elephants directly, but also through indirect pathways. For example, dogs are often used to protect farms and will bark when elephants approach the farm (JK; personal observations); often, farmers subsequently try to chase away elephants by making sounds, throwing objects such as stones and sticks, and lighting fire (Thouless 1994; Hoare 2012, 2015). Given the superb memories of elephants, they are likely to remember such antagonistic encounters with humans and may associate such interactions with the sounds of dogs (McComb et al. 2014).

The low response rate of elephants to lion roars was slightly surprising because young elephants can be subject to lion predation (Hayward and Kerley 2005; Loveridge et al. 2006). This results may be explained by the sex and age structure of MR elephants that is skewed towards adult males (Kioko et al. 2013) which are typically not susceptible to lion predation.

13.10 Human-Elephant Coexistence in Rangelands

This socio-ecological study demonstrates the substantial potential for coexistence between elephants and cattle in rangelands, mostly facilitated by mutual spatial avoidance (Valls-Fox et al. 2018). Cattle herders in MR generally perceived elephants as a minor threat to their cattle compared to other wildlife species and generally supported elephant conservation efforts. Elephants showed frequent behavioral responses to sounds of humans and dogs, suggesting that negative elephant-cattle interactions may be aggravated by herders and/or domestic dogs. Humans and elephants have historically shared social and ecological landscapes (Sukumar 2003; Kioko et al. 2015c). Encounters between livestock herder and elephants were reported to be common, yet herders mostly perceived interactions as unproblematic and supported elephant conservation. Most herders said they would let their cattle intermingle with elephants, yet personally keep a distance from elephants. To reduce potential tense interactions between livestock and wildlife, the ranch management currently dedicates one section of the major water dam for wildlife use only and a separate section for livestock which helps to reduce spatiotemporal overlap (and thus direct interactions) between elephants and livestock.

Our study demonstrates that elephants and cattle keeping are generally compatible. However, our results also offer pathways for improving coexistence between livestock and elephants in the rangelands of the TE. First, community-based participatory livestock disease education programs could clarify the role of elephants in disease transmission (de Garine-Wichatitsky et al. 2013). Such improved knowledge could contribute to removing false beliefs that underlie some negative attitudes towards elephants.

Possibly more important, our study suggests that herders' attitudes are substantially affected by human-elephant interactions in farmland. As human attitudes are central for livestock-elephant coexistence in rangelands, we recommend focusing on mitigating crop raiding by elephants. As pastoralists in the ecosystem continue to diversify their livelihoods by increasing crop cultivation (Nkedianye et al. 2019; **McCabe and Woodhouse** Chap. 4), it will be increasingly important to implement locally acceptable, effective, sustainable, and scalable intervention techniques to reduce crop damages (Denninger Snyder and Rentsch 2020).

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Chapter 14
Human-Carnivore Coexistence in the Tarangire Ecosystem

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Abstract Facilitating coexistence between humans and large carnivores is one of the most complex and pressing conservation issues globally. Large carnivores pose threats to human security and private property, and people may respond to those risks with retaliation which can jeopardize the persistence of carnivore populations. The nature of these interactions can be influenced by several variables including ecological, anthropogenic as well as political dimensions. The Tarangire Ecosystem (TE) of northern Tanzania is a stronghold for multiple large carnivore species. Despite multi-faceted and long-term carnivore conservation efforts being implemented in the ecosystem, the anthropogenic impacts on carnivore populations are pervasive. As only a portion of the TE is fully protected, the wide-ranging nature of carnivores brings them into close contact with people living among a matrix of village lands. Consequently, this ecosystem experiences high levels of human-carnivore conflicts. In this chapter, we synthesize the existing information to characterize the extent, impacts, and spatiotemporal patterns of human-carnivore interactions (which often result in severe conflicts, causing harm to people, livestock, and carnivores), examine the efficacy and challenges of implementing interventions designed

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to reduce human-carnivore conflict, and explore the socio-economic dimensions of these mitigation efforts.

Keywords Carnivora · Retaliatory killing · Livestock depredation · Impact assessment · Conservation intervention

14.1 Human-Carnivore Coexistence

Coexistence of humans and large carnivores is a veritable challenge (Carter and Linnell 2016; Lute et al. 2018; Linnell et al. 2020). From an anthropocentric perspective, carnivores provide both ecosystem services and disservices (Ceaușu et al. 2019; Lozano et al. 2019; König et al. 2020). The direct effects (mediating demography and numbers of prey species) and indirect effects (mediating multiple aspects of prey behavior and physiology) of predation by large carnivores shape trophic interactions and contribute to the functioning of ecosystems (Estes et al. 2011; Ripple et al. 2014). At the same time, large carnivores act as flagship species that can be used to support biodiversity conservation (Dalerum et al. 2008; Caro 2010). In numerous human cultures and traditions, large carnivores are considered to be charismatic animals and hold great appeal and fascination among people (Courchamp et al. 2018; Sommerville 2020). The African lion (*Panthera leo*), for example, plays a central role in the folklore of most cultures in Africa (Hazzah et al. 2009). Large carnivores also play a key role in economic development through their contribution to wildlife tourism enterprises (Lindsey et al. 2012; Stolton and Dudley 2019). On the other hand, interactions between carnivores and humans in shared landscapes can have negative consequences for carnivore populations as well as for people's livelihoods, health and overall wellbeing (Peterhans and Gnoske 2001; Treves and Karanth 2003; Packer et al. 2005; Thirgood et al. 2005; Carter et al. 2012; Carter and Linnell 2016). Within this context, large carnivores are often perceived as a severe threat to people as they may attack livestock, and attack, injure, or even kill humans. Thus, the interactions of humans and large carnivores are often framed as conflicts, and the threats to human security and private property may result in the retaliatory killing of carnivores (Woodroffe and Frank 2005; Kissui 2008; Carter et al. 2017; Hazzah et al. 2017). In turn, the consequences of human-carnivore conflicts are one of the leading causes of global decline in carnivore populations and represent the most persistent and complex aspect of carnivore conservation (Michalski et al. 2006; Winterbach et al. 2013; Ripple et al. 2014; van Eeden et al. 2018).

The drivers of human-carnivore conflicts are diverse. Based on a literature review, Montgomery et al. (2018) proposed that human-carnivore conflict is an inherently interdisciplinary problem with five dimensions that are important to consider in the design of any conflict mitigation strategy. These five dimensions include humans, carnivores, livestock, wild prey, and environmental factors. Drivers of human-carnivore conflict can be of a socio-cultural nature as, for example, level of education, belief systems, wealth, and political systems (including compensation for carnivore damages) can substantially influence different attitudes towards carnivores and people's perception of conflict (Treves and Bruskotter 2014; van Eeden et al. 2021). Such differences can affect levels of tolerance to carnivores, which in turn has implications for carnivore conservation (Hazzah et al. 2009, 2017; Dickman 2010; Rust et al. 2016). Availability of prey may influence the seasonal movements, abundance, and population dynamics of large carnivores (Hayward et al. 2007a, b). Changes in wild prey abundance due to seasonal distribution shifts (**Bond et al**. Chap. 8) can affect the frequency with which large carnivores depredate livestock (Fuller and Sievert 2001). In multi-use landscapes, wild prey movement outside protected areas may increase human-carnivore conflicts due to increased encounter rates between livestock and carnivores (Hemson 2004; Kissui 2008; Mponzi et al. 2014). Therefore, a thorough understanding of the interrelationships between environmental conditions, prey availability and their effect on the distribution of carnivores, and the patterns and magnitude of human-carnivore conflict is essential for conservation planning and prioritization of conflict mitigation strategies. Biological factors such as the social behavior of individual carnivore species as well as the behavior of prey species can also significantly influence the patterns and extent of human-carnivore conflicts (Valeix et al. 2012). Husbandry practices play a critical role in the management of human-carnivore conflicts as well. For example, depredation risk by lions was found to increase among livestock that were herded by young boys relative to livestock herds attended by adults (Ikanda and Packer 2008; Mkonyi et al. 2017c). Furthermore, the type, design, and physical characteristics of livestock enclosures determine levels of livestock depredation. The effectiveness of these enclosures can be increased when additional deterrent measures such as night-time guard dogs are used to prevent livestock depredation by carnivores (Ogada et al. 2003; Lesilau et al. 2018). Preventive measures applied by livestock keepers vary depending on livestock type. For example, in the Tarangire Ecosystem (TE) of northern Tanzania, it is common practice for pastoralists to keep juvenile livestock in small shelters constructed of poles with walls plastered using mud and cow dung and thatch grass, and additionally keep domestic dogs as an early warning sensor. In contrast, adult cattle, donkeys, goats, and sheep are commonly kept in enclosures constructed with thorn bushes (Ukio 2010). However, the effectiveness of husbandry practices varies across carnivore species and landscapes, and no single technique is 100% effective in preventing livestock depredation (Ogada et al. 2003; Lichtenfeld et al. 2015; Eklund et al. 2017; van Eeden et al. 2018; Kissui et al. 2019).

While husbandry practices, as well as the behaviors of both humans and carnivores, are important attributes mediating human-carnivore conflict, carnivore attack events on livestock are often not evenly distributed across landscapes (Baker et al. 2008; Mponzi et al. 2014; Kissui et al. 2019). Therefore, it is theoretically possible to design and parameterize spatially explicit models to identify and predict conflict hotspots where carnivores kill livestock. Such risk maps could be instrumental for spatial prioritization of interventions to reduce livestock depredation by large carnivores and human retaliation on large carnivores (Stahl et al. 2002; Treves et al. 2004, 2011; Marucco and McIntire 2010).

The hallmark of most African ecosystems has been that people living near protected areas often bear the costs of coexisting with wildlife, while receiving disproportionately little economic benefit. In ecosystems such as the TE where the majority of people are livestock keepers and subsistence farmers, human-carnivore conflicts are common (Kissui 2008; Koziarski et al. 2016). While various traditional and modern strategies exist to minimize the negative impacts of large carnivores on humans, persistence of large carnivores in human-dominated landscapes is only possible through facilitating coexistence by integrating peoples' socioeconomic traditions and ethical considerations into carnivore conservation efforts. For example, the engagement of stakeholders with diverse interests and viewpoints toward carnivores, that ensures the recognition of multiple viewpoints and widespread participation, could increase peoples' tolerance of carnivores (Carter and Linnell 2016; Lichtenfeld et al. 2019). Because a deep understanding of the ecological and social factors influencing human-carnivore interactions is critical for developing effective management and conservation strategies for carnivores (Bagchi and Mishra 2006; Mkonyi et al. 2017a; Eshete et al. 2018; Lute et al. 2018), we review and synthesize the challenges and opportunities for human-carnivore coexistence in the TE.

14.2 The Tarangire Ecosystem

The Tarangire Ecosystem (TE) is comprised of a matrix of protected areas, including national parks (Tarangire and Lake Manyara National Parks), a game reserve (Mkungunero), three wildlife management areas (Burunge, Makame, and Randilen), multiple game-controlled areas (Mto wa mbu GCA, Lolkisale GCA, Lake Natron and Simanjiro NGA), and communal village lands – all with different levels of regulations regarding human activities and protection of wildlife and the environment (Fig. 14.1). The protected areas are not fenced, allowing unrestricted movement of wildlife from core protected areas out into dispersal areas with other forms of land uses (Lamprey 1964; Borner 1985; Kiffner et al. 2016).

To the west, the TE is bordered by the Rift Valley escarpment. The elevation ranges from 1000 to 2600 m above sea level and the annual rainfall varies between 500 and 650 mm. October to December is a short rainy season with a long rainy season from February to May (Prins and Loth 1988). The ecosystem is renowned for its migratory ungulates including wildebeest (*Connochaetes taurinus*), zebra (*Equus quagga*), buffalo (*Syncerus caffer*), and African savanna elephant (*Loxodonta africana*) which typically concentrate in the protected areas during the dry season yet occupy communal land during the rainy season (Kahurananga & Silkiluwasha 1997; Bond et al. 2017; Kiffner et al. 2017; **Bond et al.** Chap. 8; **Foley and Foley** Chap. 10). The ecosystem is also home to multiple medium- to large-sized carnivore species including African lions, spotted hyenas (*Crocuta crocuta*), striped hyenas (*Hyena hyena*), leopards (*Panthera pardus*), cheetahs (*Acinonyx jubatus*), African wild dogs (*Lycaon pictus*), and black-backed jackals (*Canis mesomelas*) (**Kiffner et al**. Chap. 11).

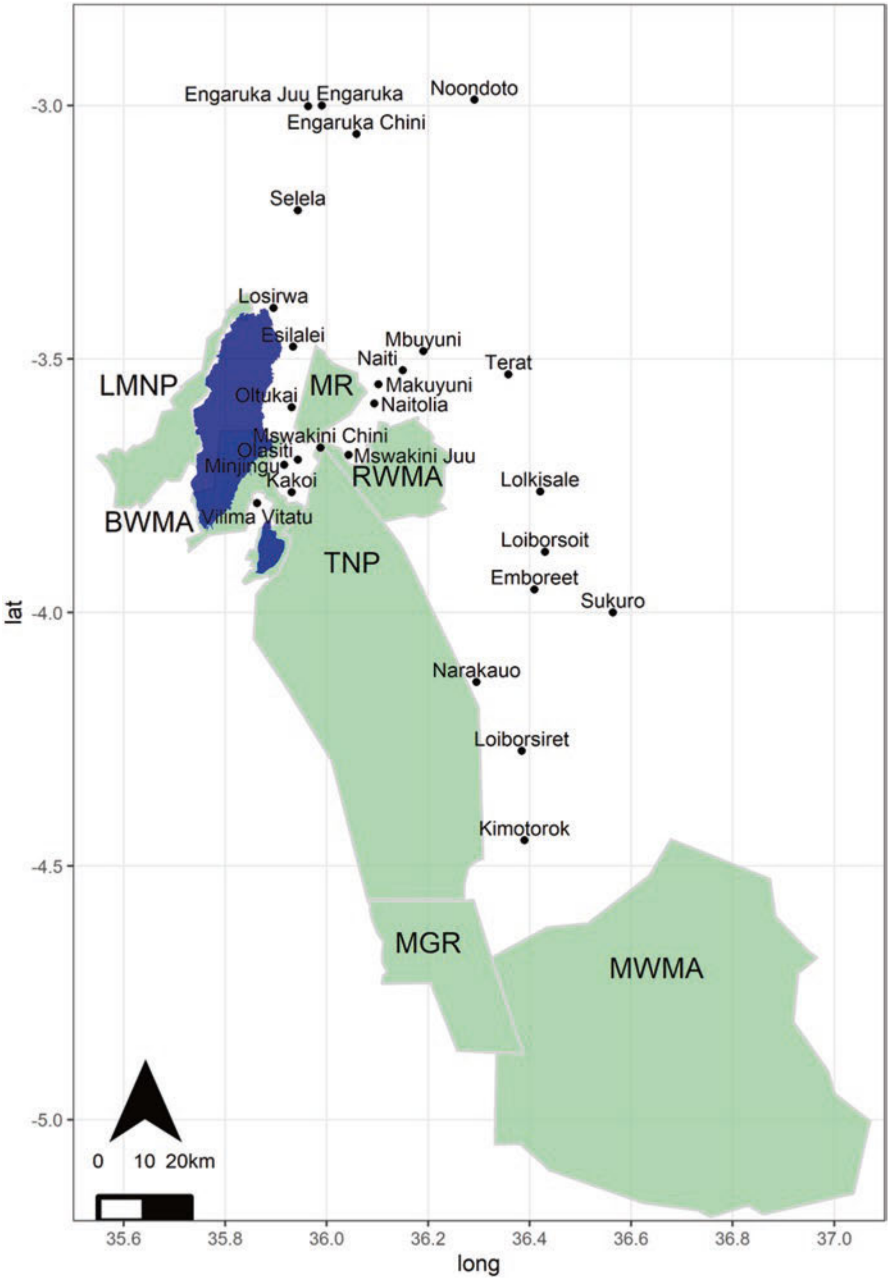

Fig. 14.1 Map of the Tarangire Ecosystem, showing the main protected areas (*LMNP*, Lake Manyara National Park, *TNP* Tarangire National Park, *MGR* Mkungunero Game Reserve, *BWMA* Burunge Wildlife Management Area, *RWMA* Randilen Wildlife Management Area, *MWMA* Makame Wildlife Management Area, *MR* Manyara Ranch), the locations of villages referred to in Tables 14.1 and 14.2, as well as Lake Manyara and Burunge (blue polygons)

Table 14.1 Reported number of attacks on humans by large carnivores (lion, leopard, and hyena) in 18 villages in the TE from 1943 to 2010

Village	Hyena	Leopard	Lion	Total
Emboret	0	0	17	17
Engaruka Chini	1	5	50	56
Engaruka Juu	3	6	11	20
Esilalei	4	3	2	9
Kimotorok	0	4	11	15
Loiborsiret	0	2	11	13
Loiborsoit	3	0	13	16
Lolkisale	0	0	2	2
Makuyuni	0	1	5	6
Mbuyuni	0	1	1	2
Minjingu	0	0	2	2
Mswakini Chini	0	0	10	10
Mswakini Juu	0	7	1	8
Narakauo	1	0	4	5
Oltukai	0	0	15	15
Selela	1	15	36	52
Sukuro	0	0	1	1
Total	**13**	**44**	**192**	**249**

In the communal areas of the TE, the main land-use activities are livestock keeping and agriculture. Livestock (cattle, goats and sheep, donkeys) densities usually exceed those of wildlife species in areas outside fully protected areas (Kiffner et al. 2016). Agricultural expansion (small-scale subsistence farming to large-scale commercial farming) is a major driver of land-use changes and the loss of rangelands in the TE, exacerbated by rapid growth of human settlements, infrastructure, and economic development activities offered by immigration and tourism (Msoffe et al. 2011; Hariohay 2013). The Maasai, who mostly practice pastoralist lifestyles, is the dominant ethnic group in the area (**Igoe** Chap. 3; **McCabe and Woodhouse** Chap. 4; **Brehony et al.** Chap. 5). After the Maasai, the most common ethnic groups are Waarusha and Barbaig, tribes that are mostly engaged in livestock keeping and small-scale agriculture. The rapidly growing town of Mto wa Mbu attracts a substantial number of more than 120 ethnicities immigrating from different parts of Tanzania following small agricultural and tourism business opportunities (Msoffe et al. 2011).

14.3 Living with Large Carnivores in TE: Carnivore Attacks on Humans

Encounters between people and large carnivores are commonplace in the TE (Koziarski et al. 2016). Several species of medium- and large-sized carnivores are involved in one way or another on human attacks, causing injuries or even death. Regarding carnivore attacks on humans, we focus our analyses on the three most

Table 14.2 Number of livestock attack events by lions, leopards, and hyenas recorded in 29 villages of the TE between 2004 and 2017 (see Kissui 2008 for data collection methods)

Village	Hyena	Leopard	Lion	Total
Emboreet	187	41	65	293
Engaruka	48	40	11	99
Engaruka Chini	8	4	0	12
Engaruka Juu	4	2	0	6
Esilalei	31	11	36	78
Kakoi	2	0	6	8
Kimotorok	6	3	2	11
Loiborsiret	45	35	52	132
Loiborsoit	95	85	85	265
Lolkisale	8	2	8	18
Losirwa	0	0	2	2
Makuyuni	25	3	15	43
Mbuyuni	0	0	1	1
Minjingu	2	1	15	18
Mswakini Chini	7	0	4	11
Mswakini Juu	5	3	11	19
Naiti	0	0	2	2
Naitolia	0	0	2	2
Narakauo	1	1	1	3
Noondoto	1	0	0	1
Olasiti	15	0	5	20
Oltukai	60	4	12	76
Manyara Ranch	0	0	1	1
Selela	173	52	89	314
Sukuro	0	0	3	3
Terat	0	1	0	1
Vilima vitatu	1	2	5	8
Total	**724**	**290**	**433**	**1447**

common large predators: African lions, leopards, and hyenas. People residing in the TE often do not differentiate between spotted and striped hyenas and we thus simply refer to hyena; conflict with hyenas is however likely to be driven mostly by spotted hyenas. In the TE, only two studies have systematically quantified carnivore attacks on humans and these studies covered only a few villages or were conducted over a short time span. Skuja (2000) studied carnivore attacks on human with special reference to lion attacks in selected villages on the western boundary of Tarangire National Park. Packer et al. (2005) analyzed lion attacks on humans across Tanzania including TE. This study indicated that on a regional scale, most of the lion attacks were found in the southeastern part of Tanzania, suggesting that despite being a human-dominated socioecological system, the TE experienced moderate levels of human attacks by lions compared to other regions in Tanzania. However, because of the importance of this topic, we report on additional research here. Most of the carnivore attacks on humans (fatalities or injuries) are reported to the district game offices in Monduli and Babati districts. In a survey conducted by the Tarangire Lion Project (TLP) from 2003 to 2010, information on carnivore attacks on humans was collected from records at district game offices. Because of incompleteness in records at the district game offices, additional information was collected from focus group discussions, key informants, and household residents by means of formal and informal interviews. In total, TLP recorded 249 historical incidences of human attacks by large carnivores spanning a period from 1943 to 2009 in 18 villages (Table 14.1). Figure 14.2 shows the number of attacks on people across

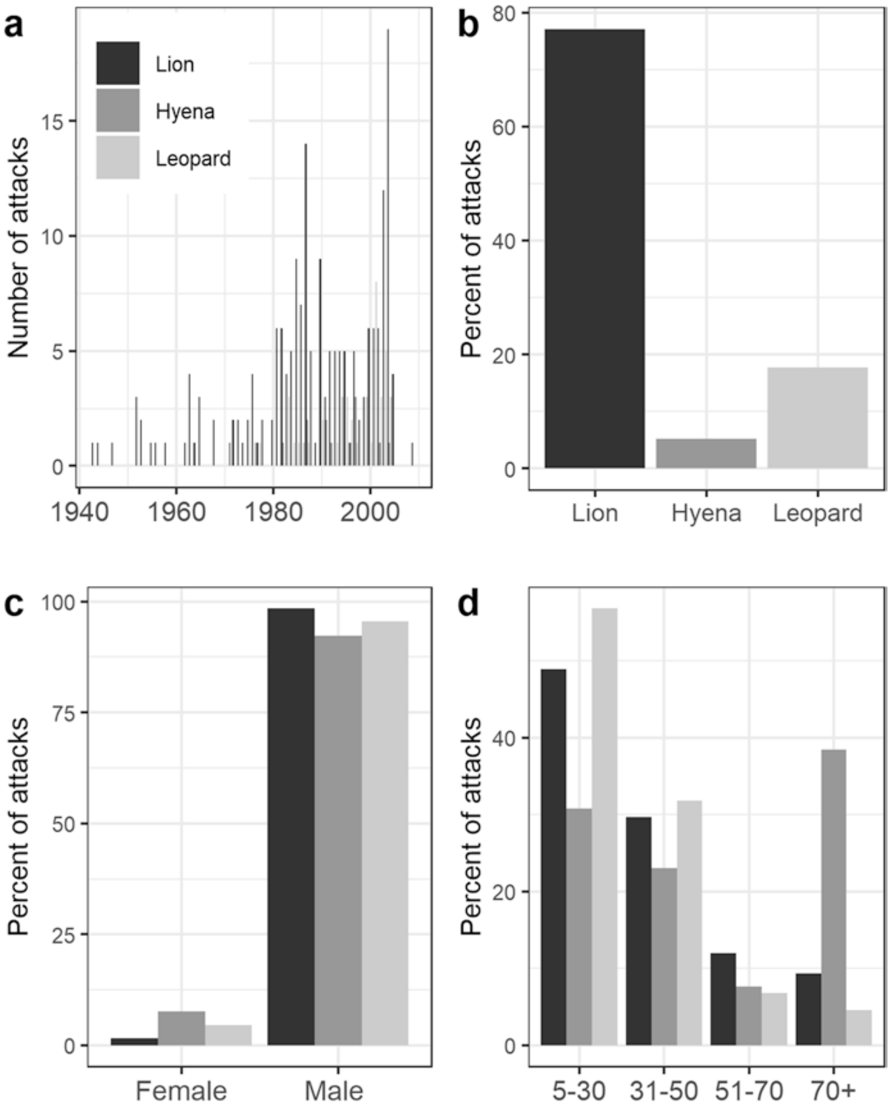

Fig. 14.2 (**a**) Reported number of carnivore attacks on humans in TE between 1943 and 2009; (**b**) comparison of human attacks by the three carnivore species; (**c**) carnivore attacks by gender of the human victims, (**d**) carnivore attacks by age group of the human victims

the study villages. The number of reported cases appeared to have slightly increased from the early 1980s to 2000s (Fig. 14.2a). This temporal trend could have multiple underlying reasons: it is possible that this trend is associated with the increase in human population in the TE (Msoffe et al. 2011) or by the employed method because events that occurred relatively recently are typically more easily memorized and recalled compared to events that occurred multiple decades ago.

Large carnivore species that typically pose a threat to human safety in the TE are primarily lions (77% of attacks); leopards (18%), and hyenas (5%). Hyenas were much less frequently involved in attacks on humans (Fig 14.2b). More recent data collection (2018 to mid-2021) by Tanzania People & Wildlife (TPW) includes 9 incidents of human attacks by carnivores in the TE and suggests a relatively high proportion of hyena attacks on people (five attacks by hyena, three by lion, one by wild dog).

The long-term dataset of TLP suggests that human males are particularly susceptible to be attacked by large carnivores (Fig. 14.2c; 98% of attacks were on males) presumably because they are more likely to be involved in activities such as herding cattle, participating in retaliatory carnivore hunts following livestock depredation events, and walking alone at night (Fig. 14.3). The distribution of carnivore attacks by age indicated that 49% of people attacked were younger than 31 years old; 30% were between 31 and 50 years old, while 10% were victims older than 70 years (Fig. 14.2d). Older people were particularly susceptible to attacks by hyenas.

Most of the carnivore attacks on humans were concentrated in a few villages (Table 14.1). On average TLP recorded 13.83 ± 3.57 (SE; range 1–56) carnivore attacks per village over the 66-year timespan. The spatial variation in incidences suggests that carnivore attacks were not evenly distributed across the TE, with some

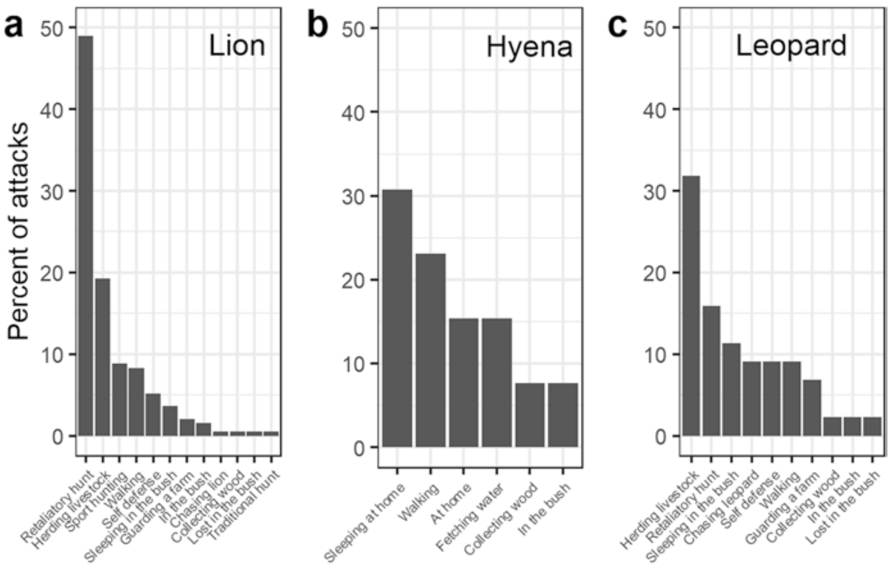

Fig. 14.3 Contexts of human attacks by (**a**) lions, (**b**) hyenas, (**c**) leopards in the Tarangire Ecosystem of northern Tanzania

villages demonstrating clusters of attacks while people in other villages were rarely subject to attacks by large carnivores (Table 14.1).

14.3.1 Context of Human Attacks by Large Carnivores

Figure 14.3 indicates that there are numerous contexts in which people fall victim to attacks by large carnivores, although these contexts vary from one carnivore species to another. For example, the majority of lion attacks (49%) occurred during retaliatory lion hunts, and 19% of attacks occurred while people were herding livestock (Fig 14.3a). In contrast, 31% of people subjected to hyena attacks were attacked while sleeping in their homes at night, or while walking from one location to another (23%). Interestingly, no person was attacked by hyenas following a retaliatory hunt or while herding their livestock (Fig. 14.3b). Leopard attacks occurred while people were herding livestock in the field (32%), or when people were engaged in leopard retaliatory hunts following a livestock depredation event (16%) (Fig. 14.3c). When combining all attacks by lions, hyenas, and leopards, 41% and 20% (total 61%) of people attacked were engaged in retaliatory lion or leopard hunting and livestock herding in the field, respectively. Therefore, these two activities are likely to pose the highest risk for attacks by large carnivores in the TE.

Hyena attacks on humans occurred mostly at night (Fig. 14.4), likely corresponding to the almost exclusively nocturnal behavior of hyenas in the communal areas of TE (**Kiffner et al**. Chap. 11). Because most carnivore attacks on humans were related to retaliatory hunts and herding livestock (Fig. 14.3a, c), it is not surprising that the majority of lion and leopard attacks on people occurred during the daytime (Fig. 14.4).

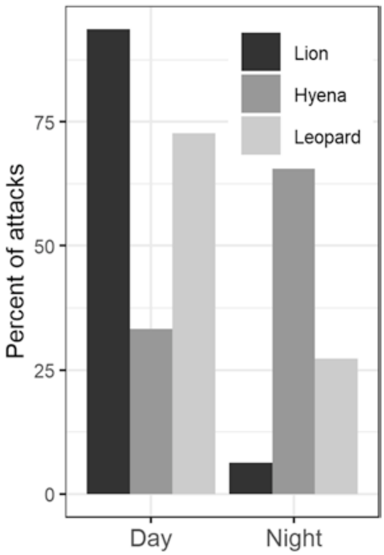

Fig. 14.4 Percentage of daytime and nighttime carnivore attacks on humans in the Tarangire Ecosystem of northern Tanzania

Fig. 14.5 Retaliatory lion hunting party of more than 100 Maasai warriors participating in the search for lions in Manyara Ranch in 2006, following a lion attack on cattle in a nearby village

Most human attacks by carnivores in the TE occured in the context of retaliatory lion or leopard hunts (Fig. 14.5) and while herding livestock in the field. This can be expected considering the pastoralist lifestyle of the Maasai communities, the dominant ethnic group in the ecosystem. Unprovoked lion hunts or hunting lions for their body parts is prohibited by wildlife laws in Tanzania. Nearly all lion killings by humans in the TE were provoked by livestock depredation, with no traditional lion hunts recorded (Kissui 2008). Traditional lion hunting practices in the TE have likely stopped in recent years due to law enforcement by the wildlife authorities, increased awareness by the community about the illegal nature of the practice, and ongoing changes in the lifestyle of the communities.

14.4 Patterns of Livestock Depredation

Livestock depredation by large carnivores is prevalent and one of the common causes of conflict, and an emotional subject when communities and conservation authorities interact in the TE. Numerous studies have been conducted to document the extent and consequences of this issue, and to identify socio-economic, environmental, and spatial factors influencing livestock depredation across the TE (Ukio 2010; Mponzi et al. 2014; Lichtenfeld et al. 2015; Mkonyi et al. 2017b, c; Kissui et al. 2019; Beattie et al. 2020). Livestock losses resulting from large carnivore depredation cause significant economic costs to livestock keepers. For example, Lichtenfeld et al. (2015) estimated that 64% of the financial loss experienced by pastoralists in the Simanjiro area of the TE was attributable to livestock depredation by lions. Lions were found to inflict greater financial loss because of their propensity to kill cattle which have greater financial value than smaller stock like goats or sheep (Kissui 2008; Lichtenfeld et al. 2015).

14.4.1 Spatial Variation of Livestock Depredation

Several large carnivore species contribute to livestock depredation in the TE and even large predatory birds such as eagles and hawks are known to attack poultry (Bencin et al. 2016). However, the most common carnivores involved in livestock depredation are lions, leopards, and hyenas. The other carnivore species (cheetahs, wild dogs, jackals) are comparatively rare and generally cause low levels of livestock depredation in the TE (Kissui 2008; Mponzi et al. 2014; Lichtenfeld et al. 2015; Mkonyi et al. 2017c).

Quantitative data on livestock attack events by lions, leopards, and hyenas collected between 2004 and 2017 by the Tarangire Lion Project include a total of 1447 livestock attack events across 29 villages (Table 14.2). Following Kissui (2008) a livestock attack event was defined as an incident in which a predator killed or injured one or more livestock.

Livestock depredation by large carnivores was highly variable between villages (Table 14.2), with some villages experiencing higher levels of livestock depredation (e.g., Selela, Emboreet, Loiborsoit) while other villages experienced much lower depredation levels (e.g., Mbuyuni, Noondoto, Terat). The reasons for the observed spatial variability in livestock depredation could be numerous (Hoffmann et al. 2019) but could be driven by the dynamic distribution of carnivores outside fully protected areas. For example, livestock depredation by lions in Manyara Ranch was concentrated in areas of high primary productivity and proximity to surface water during the dry season (Beattie et al. 2020). During the rainy season, livestock attacks occurred over a wider area and were more frequent (Mponzi et al. 2014; **Kiffner et al**. Chap. 11), suggesting that livestock depredation risk is largely driven by environmental factors. Interestingly, it has also been established that livestock attacks in bomas (enclosures commonly made of thorn bushes, set up to protect homesteads and livestock) seem to occur at random across the landscape without showing a particular spatial pattern (Hoffmann et al. 2019; Kissui et al. 2019).

Hyenas caused most livestock depredation events in the TE. Of the 1447 livestock recorded attack events, 50% were due to hyenas, 30% by lions, and 20% by leopards. In a separate study using monthly visits to livestock keepers to assess levels of livestock depredation in bomas, Kissui et al. (2019) found that hyenas contributed to more than 98% of livestock attacks in bomas while the other carnivore species contributed marginally to livestock losses in bomas. Similarly, TPW has recorded 2313 incidents of carnivore-livestock conflict since 2013. Of these, 80% were by hyena, 8% by leopard, 7% by lion, 2.1% by wild dog, and <2% by jackal and cheetah. Of TPW's data on 1433 carnivore incidents at bomas since

2013, 86% were by hyena, 8% by leopard, and 4% by lion. Of the 841 incidents reported at pasture, 69% were caused by hyena, 12% by lion, 6% by leopard, and 6% by wild dog.

Figure 14.6a shows that the recorded number of livestock attack events across the TE decreased from 2004 to 2017. This decline could be attributed to several factors. First, it may be due to the success in the ongoing human-carnivore conflict mitigation efforts such as the increased use of predator-proof bomas being advocated by conservationists. Secondly, it could be an indication of the changing livestock husbandry practices by the communities to adopt more effective measures leading to improved livestock security. Third, it could be an artefact of altered data collection efforts by the TLP team (between 2004 and 2012, TLP worked with dedicated enumerators in each village; from 2013 onwards, data were collected based on reports received from village leaders and community members). Fourth, the decline could indicate an overall shift in the lifestyle of the local communities.

14.5 Community Attitudes and Perceptions on Human-Carnivore Conflicts in the TE

The persistence of carnivores in human-dominated landscapes is highly dependent on successful human-carnivore conflict mitigation (Treves and Karanth 2003; Thornton and Quinn 2009; Thorn et al. 2014). As human population increase and the footprint of human activities expands across the landscape, interactions and conflicts between people and carnivores are likely to increase (Inskip and Zimmermann 2009; Nyhus 2016; König et al. 2020). Conflict mitigation efforts and ultimately carnivore conservation efforts are influenced by peoples' attitudes and perceptions of carnivores (Dickman 2010; Dickman et al. 2014; Kansky and Knight 2014; Kansky et al. 2014; Hazzah et al. 2017). Several studies have conducted assessments of attitudes and perceptions towards carnivores and carnivore conservation in the TE. Studies by Mkonyi et al. (2017a) and Fatael have both reported that several factors influenced attitudes towards carnivore conservation, including the level of education and experiences in economic losses caused by carnivores.

In the TE, peoples' perceptions of the frequency of human-carnivore conflicts were much greater (several orders of magnitude) than the observed levels of conflicts recorded during surveys in which the frequency of carnivore depredation events was monitored directly (Fig. 14.7). This mismatch is common among human-wildlife conflict studies and likely due to the complexity of the socio-economic and psychological factors underlying human perceptions and responses to human-wildlife conflicts (Gillingham and Lee 2003; Dickman 2010; Nyhus 2016). Multiple hypotheses have been put forward to explain such mismatches between the perception and reality of the frequency of negative interactions between humans and large carnivores, including extreme damage events which may indeed occur as evidenced by the loss of human life and the loss of cattle which represent economic and social value for pastoralists. Direct experiences of such losses or merely the knowledge

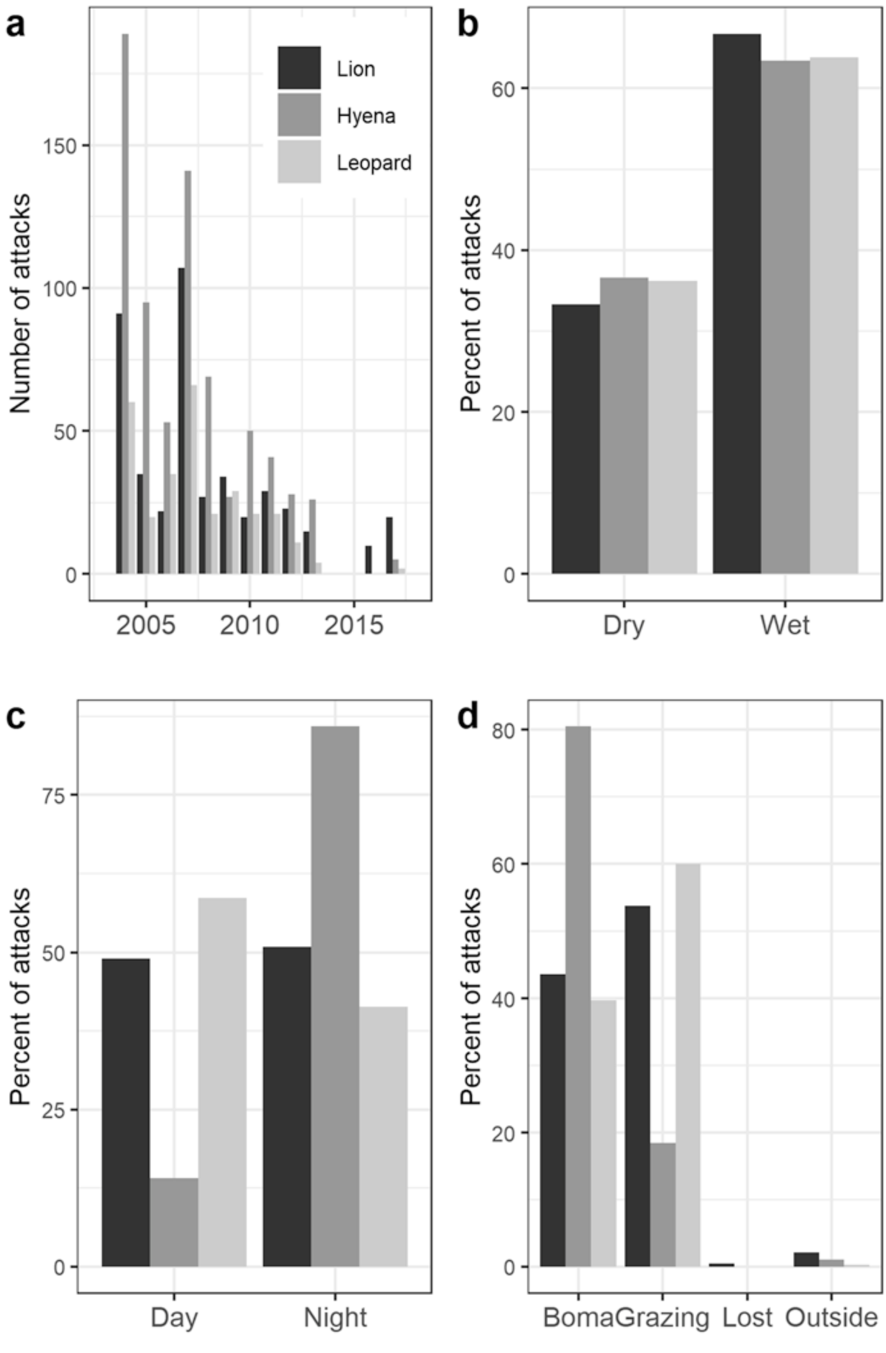

Fig. 14.6 (**a**) Number of livestock depredation events by carnivores (lions, leopards, and hyenas) in the TE from 2004 to 2017; (**b**) number of livestock depredation events during the dry and wet season; (**c**) number of livestock depredation events during day and nighttime; and (**d**) number of livestock depredation events by location/circumstance of the event

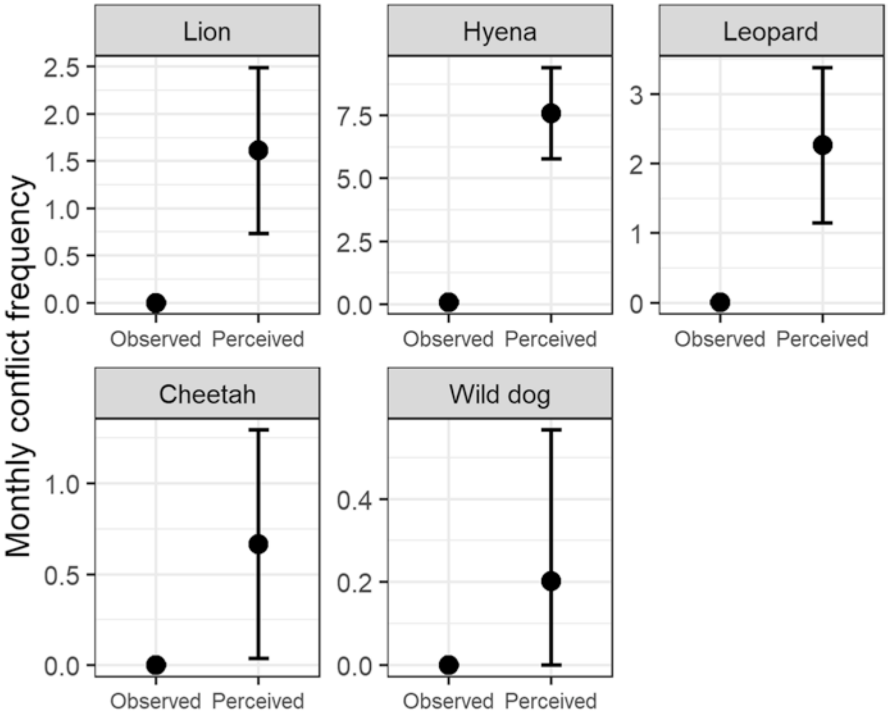

Fig. 14.7 Observed [based on monthly visits to 42 pastoralists' households from 2009 to 2013 (Kissui et al. 2019)] vs. perceived [based on 164 household interviews (Koziarski et al. 2016)] mean monthly frequency of livestock attacks by lions, hyenas (respondents did not differentiate between spotted and striped hyenas), leopards, cheetahs, and wild dogs in the Tarangire Ecosystem, northern Tanzania. Error bars denote 95% confidence intervals

about such potential repercussions of interactions with large carnivores may instill fear of these species (Dickman et al. 2014; Koziarski et al. 2016), and indeed, the majority of interviewees in the TE expressed fear of lions and leopards (Fig. 14.8). Furthermore, opportunity costs associated with livestock guarding practices [e.g. the construction and maintenance of predator-proof bomas is expensive and labor intensive (Kissui et al. 2019; Chaka et al. 2021)], as well as a feeling of powerlessness to deal with wildlife (which may emerge if livestock is depredated despite having livestock guarding measures in place), or deep-rooted conflicts and mistrust between pastoralists and management authorities of protected areas could underlie such mismatches. For example, retaliatory killing of lions in and adjacent to Manyara Ranch [a hotspot for human-lion conflict in the TE (Koziarski et al. 2016; Beck et al. 2021)] may have been caused by resentment towards grazing restrictions and overall disagreements with conservation politics (Goldman et al. 2013).

Due to its history as a culture- and wildlife-rich landscape, the TE has attracted diverse stakeholder groups engaged in conservation and management of natural resources, including local and national government wildlife authorities (Tanzania

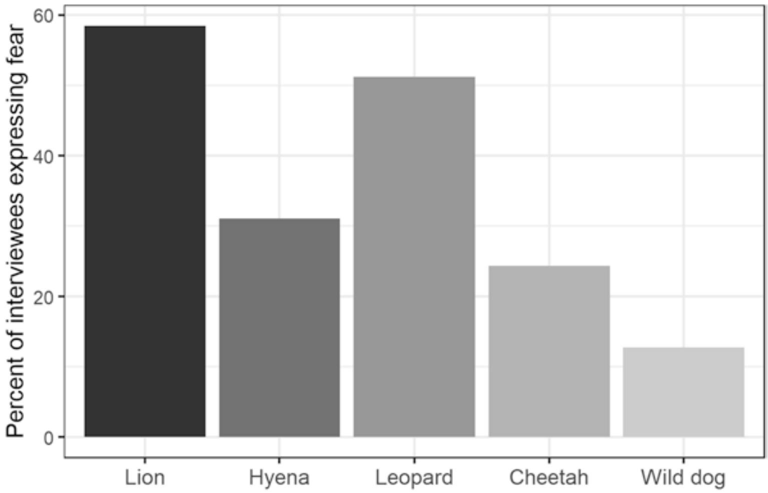

Fig. 14.8 Percent of interviewees who feared large carnivore species in the Tarangire Ecosystem of northern Tanzania. Percentages were based on 164 interviews (Koziarski et al. 2016)

National Parks Authority, Tanzania Wildlife Management Authority, regional/district game officers); conservation and rural development NGOs; community-based natural resource management groups; tourism and wildlife-based private investors; farmers; livestock keepers; and indigenous people. Having such a diverse group of stakeholders in the landscape creates a potential for differing interests and disagreements over the perceptions of human-carnivore conflicts and approaches for conflict mitigation.

14.6 Conflict Mitigation

There is an important gap, referred to as the knowing-doing or research-implementation gap, between spheres of research and on-the-ground decision-making. Gray et al. (2020) found that much of the research conducted on human-carnivore conflict in East Africa was not being translated into action on the ground. Nevertheless, multiple stakeholder groups including government authorities, NGOs, and local communities have implemented conflict mitigation activities in the TE. However, the key stakeholders focusing on specific conflict mitigation measures are two NGOs, the Tarangire Lion Project and Tanzania People & Wildlife (Lichtenfeld et al. 2015; Mkonyi et al. 2017b; Kissui et al. 2019). Both lethal (e.g., targeted hunts and killing of problem carnivores) and non-lethal (e.g., the use of predator-proof bomas) conflict mitigation strategies have been utilized in the TE. The pastoralist communities in the TE are also still using traditional husbandry practices such as thorn bushes around bomas to mitigate conflicts with large carnivores.

14.6.1 Lethal Control as Conflict Mitigation Strategy

Historically, the use of lethal control has been a common method in the TE, especially in the form of retaliatory carnivore killings following livestock depredation events. Spearing, use of guns, and poisoning are the common tools used in lethal control of carnivores. Most of the lethal control attempts are practiced informally by aggrieved community members following the loss of their livestock or following a threat to human safety. Occasionally, lethal control of carnivores has been conducted by wildlife authorities, especially in cases where carnivores were involved in human injuries or death. However, lethal control is a controversial method to manage problem animals and only used as last resort in specific situations (Treves and Naughton-Treves 2005).

14.6.2 Non-lethal Mitigation

14.6.2.1 Predator-Proof Bomas

Several non-lethal conflict mitigation measures have been deployed to reduce livestock depredation by carnivores. Predator-proof enclosures (locally called bomas) are widely used across the ecosystem. The effectiveness of predator-proof bomas has been tested by several studies (Lichtenfeld et al. 2015; Mkonyi et al. 2017c; Kissui et al. 2019) and found to significantly reduce losses due to livestock depredation. Predator-proof bomas have also been found to be a cost-effective strategy towards reducing livestock losses which is likely to pay off the investment within a few years (Kissui et al. 2019). However, the effectiveness of predator-proof bomas is highly dependent on the condition of the boma, which in turn depends on the maintenance of the boma (Chaka et al. 2021). Some of the challenges restricting the wide use and maintenance of predator-proof bomas include the initial costs of purchase of materials for the construction of the bomas, scarcity of local materials such as poles used in the construction, and maintenance costs.

14.6.2.2 Husbandry Practices

Appropriate husbandry practices play a key role in human-carnivore conflict mitigation through improving livestock security (Ogada et al. 2003; Patterson et al. 2004). Residents in the TE generally agree that vigilant livestock herding provides added security and reduces the frequency of livestock depredation (Mkonyi et al. 2017c). In the Maasai community, livestock herding during the day is typically done by the warriors *(morans)*, but occasionally livestock herds are accompanied by small children who tend not to be mature enough to provide adequate protection against large carnivores. In Manyara Ranch, livestock of adjacent communities use the area for grazing and drinking especially during the dry season months. Beattie et al. (2020) developed a risk map which identified areas that were both productive and yielded relatively low depredation risk by lions. This analysis indicated that livestock depredation risk is particularly high near surface water. Thus scouting out the surroundings of the dams and hazing lions out of these areas (Petracca et al. 2019) *before* herding cattle to drinking water could be an effective way to reduce livestock depredation risk.

Some of the traditional livestock protection strategies such as the use of thorn bushes to construct bomas in combination with the use of guard dogs have been found to provide some security to livestock when thorn bush walls are properly constructed and maintained (Chaka et al. 2021). However, the thorn bush walls do not provide as much protection as bomas enforced with wire-fence (Lichtenfeld et al. 2015; Kissui et al. 2019).

14.6.2.3 Translocation of Problem Animals

Translocations have rarely been used in the TE to move lions that had attacked people. In 2016/2017 lions had repeatedly attacked people in Burunge Wildlife Management Area. The entire pride was captured, transported, and released in the Selous Game Reserve (in an area with minimal conflict potential). However, translocations are expensive and considered controversial as translocated animals may fail to establish at the new location (Linnell et al. 1997).

14.6.2.4 Education and Awareness

Despite substantial and sustained efforts to mitigate livestock depredation by large carnivores by reinforcing bomas and improving husbandry techniques, it is not possible to eliminate livestock depredation and attacks on people. For large carnivore conservation to be successful, it is thus inevitable that the residents of the TE tolerate some level of conflict with large carnivores (Inskip et al. 2016; Kansky et al. 2016). In the TE, overall tolerance levels towards wildlife species are relatively high (**Kiffner et al**. Chap. 1), yet the tolerance towards large carnivores is less apparent. Involving and engaging school pupils through environmental education provides a possible avenue to create more awareness and passion for wildlife and the environment and increase tolerance for wildlife including large carnivores (**Bond et al.** Chap. 16). Environmental education programs are implemented by government wildlife conservation agencies thorough their community-based conservation departments as well as multiple conservation NGOs.

14.7 Conclusions

Carter and Linnell (2016) define coexistence as a “dynamic but sustainable state in which humans and large carnivores co-adapt to living in shared landscapes where human interactions with carnivores are governed by effective institutions that ensure long-term carnivore population persistence, social legitimacy, and tolerable levels of risk". While our chapter illustrates key co-adaptations in both carnivores (e.g. increased nocturnal behavior in human dominated areas; **Kiffner et al.** Chap. 11) and humans (adoption of fortified bomas, improved livestock guarding methods; Kissui et al. **this chapter**) that facilitate coexistence, we also highlight key challenges. Finding effective solutions to prevent conflict with large carnivores requires collaboration and strong ties between science, management authorities, and local stakeholders. In the TE, large carnivore conservation will likely only succeed if conservation policies are supported by pastoralists. In turn, conservation will likely be most effective if landscape planning adequately addresses the needs of pastoralist communities, if pastoralists are effectively supported in reducing livestock depredation methods, and if conservation economies ensure a fairer distribution of costs and benefits associated with large carnivores (Blackburn et al. 2016).

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Chapter 15
Financing Conservation in the Twenty-First Century – Investing in Nature-Based Climate Solutions in Makame Wildlife Management Area

Marc Baker, St. John Anderson, and Christian Kiffner

Abstract Wildlife conservation in Africa has been dominated by protected areas (PAs) that largely excluded the interests of local communities. While this "fortress conservation" has succeeded in securing natural habitat and wildlife populations, it has come at a cost to local communities who forego access to natural resources on which their livelihoods depend and who obtain few direct benefits from the designated PAs. Concomitantly, climate change poses formidable challenges that require urgent attention to meet global climate goals. Combining finance mechanisms primarily intended for climate outcomes with community-based conservation models presents opportunities to integrate nature conservation and climate change mitigation and adaptation while providing direct income to local communities. In this chapter, we present an example of a results-based system of payments for ecosystem services – the purchase of verified emission reductions for use as carbon offsets. We outline the key steps for planning and implementing the REDD+ project of Makame Wildlife Management Area, and emphasize the monitoring of key parameters associated with climate, community and wildlife benefits. Our case study depicts an innovative, nature-based solution to climate change, wildlife conservation, and rural livelihoods for an African savannah rangeland where conventional approaches are insufficient to meet the costs of conservation.

Keywords Payment for ecosystem services · Wildlife monitoring · Climate change · Carbon offsets · Nature-based solutions

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15.1 Conservation in the Anthropocene

Biodiversity loss and climate change are arguably the greatest threats to our planet and fundamentally threaten human health, wealth, and wellbeing (Sala et al. 2000; Patz et al. 2008; Wheeler and von Braun 2013; Ceballos et al. 2017; IPBES 2019; IPCC 2019; Bradshaw et al. 2021). The common strategy to counteract biodiversity loss is the delineation and implementation of protected areas (Arcese and Sinclair 1997; Dinerstein et al. 2017). Historically, protected area establishment by governments was accompanied by evicting people and denying access to natural resource utilisation upon which the local population relied (Brockington 2002; Jones 2006). These practices also occurred while establishing the two national parks in the Tarangire Ecosystem (TE) during the second half of the twentieth century (**Bluwstein** Chap. 2; **Igoe** Chap. 3). At the same time, it is increasingly recognized that the current network of protected areas is insufficient to halt wildlife declines in East Africa (Craigie et al. 2010; Ogutu et al. 2016), mostly because the current extent of protected areas does not effectively protect the entire suite of resources that wide ranging wildlife populations require throughout the year (Fynn and Bonyongo 2011; Bond et al. 2017). Albeit Tanzania has expanded the extent of fully protected areas in recent years (Caro and Davenport 2016), a changing political landscape, population growth and the need for economic development across Africa renders the "fortress conservation" approach less suitable and harder to justify.

To address the dual goals of providing crucial habitat for wildlife and providing rural communities with direct benefits from natural resources, Tanzania initiated the establishment of wildlife management areas (WMAs) (Wilfred 2010; Kiwango et al. 2015). Tanzania's WMAs represent a form of community-based natural resource management (CBNRM) which aim to reduce poverty and conserve priority ecosystems (WWF 2014). Following the principles of CBNRM, the key underlying assumption of the WMA concept is that involving local communities in the management of wildlife resources in village lands (i.e. the communities implement a land-use plan that enables them earn income from wildlife though photographic and hunting tourism) promotes the long-term persistence of habitat and wildlife populations, and encourages rural economic development. While the ecological effectiveness of this approach has been demonstrated in two WMAs of the TE (Lee 2018; Lee and Bond 2018; Kiffner et al. 2020), socio-economic and governance aspects of WMAs have received substantial criticism (Bluwstein et al. 2016; Moyo et al. 2016; Bluwstein 2017; Brehony et al. 2018; Kicheleri et al. 2018; Kajembe and Treue 2021). A central issue is whether income through wildlife-based tourism is sufficient to cover the opportunity costs associated with foregone land uses in the wildlife areas and direct costs associated with damages caused by wildlife (Salerno et al. 2016). Organisations involved in conservation have struggled to accurately price the cost of conservation at both the local, national and continental level, and the costs of conservation have been met in many cases through donor funds and bilateral and multilateral overseas development aid arrangements (Packer et al. 2013; Lindsey et al. 2018).

The dual challenges of climate change and biodiversity loss are now recognised as being inextricably linked (Araújo and Rahbek 2006; Mantyka-Pringle et al. 2015). Recent IPCC (IPCC 2019) and IPBES (IPBES 2019) reports are unequivocal in demonstrating the critical interdependency of natural ecosystems and the climate system, and while climate concerns dominate political and economic discussions (UNFCCC 2015, 2017), the economic imperative to invest in, manage and protect biodiversity is gradually gaining traction across governments (Dasgupta 2021) and within the private sector. This has come with a recognition that a dramatic increase in funding for nature conservation is necessary to make this a reality and to avoid economic risks associated with the loss of the economic value contributed to the global economy by nature (World Economic Forum 2020). The 2015 Paris Agreement explicitly acknowledges that funding needs to come from a far wider range of sources than have been relied upon to date. Such sources include green taxes, regulatory frameworks for businesses and institutions, investment instruments like green bonds and government spending, subsidy and investment commitments.

Implicit in these approaches is a recognition that the costs of protecting and managing nature and natural resources should become embedded and integrated into our economic and socio-political systems at a structural level and on a long-term (preferably indefinite) timescale. Implementing strategies that address climate change (both mitigation and adaptation) through the use of nature and natural ecosystems are a subset of nature-based solutions (Chausson et al. 2020) known as natural climate solutions (NCS). One of these proposed NCS is formalized in the REDD+ (Reducing Emissions from Deforestation and forest Degradation) concept. REDD+ is a methodology framework that aims to curb climate change by stopping the destruction of forests. The "+" signifies the role of biodiversity conservation, community benefits, sustainable management of forests and enhancement of forest carbon stocks (Mollicone et al. 2007; Gardner et al. 2012; Jodoin 2017; UNFCCC 2017). The methodology for measuring and monitoring the reduction of deforestation (climate benefits) is predetermined by the standard (UNFCCC 2017). This methodology ensures the verified emission reductions are permanent, and accounts for leakage, i.e. the potential for the driver of deforestation to be shifted elsewhere. Verified emission reductions from a REDD project are issued post verification and thus known as ex-post emission reductions, meaning that the monitoring of climate benefits, community benefits and biodiversity benefits must be complete prior to issuance and therefore monetization of the emission reduction.

In this chapter, we outline how the creation of a carbon asset in Makame Wildlife Management Area through the implementation of a REDD+ monitoring framework underlying a sustainable financing structure has allowed long-term wildlife conservation to be integrated into rural socio-economic development plans.

15.2 Makame Wildlife Management Area

The Makame Wildlife Management Area (hereafter MWMA) is the largest WMA in Tanzania covering an area of 3643 km^2 of *Acacia-Commiphora* shrub and woodland. MWMA is composed of five villages, Irkiushiobor, Ndedo, Ngabolo, Katikati and Makame with a total population of c. 16,500 people. Predominantly Maasai pastoralists, these communities engage in livestock keeping and small-scale subsistence farming. While Maasai usually do not hunt, illegal hunting occurs in the WMA as neighbouring communities and migrants may attempt to hunt wildlife species for bushmeat and animal products (e.g. ivory) in the nearby areas. Fuelwood collection within the project area is for self-consumption only and no resources extracted from the forest are for commercial markets, but rather are used for a largely subsistence livelihood. The main agricultural crops in the WMA include maize and beans. MWMA constitutes the southern part of the greater Tarangire Ecosystem and forms a buffer between the Maasai pastoralist dominated rangelands to the north and west, and agricultural communities to the south. A camera trap survey indicated that the area provides habitat for numerous wildlife species including all large carnivore species found in northern Tanzania (Foley et al. 2018; **Kiffner et al.** Chap. 11).

As outlined in the introduction, WMAs in Tanzania have experienced mixed success, both in the context of protecting habitat and wildlife populations and providing economic benefits to participating communities (Wilfred 2010). Wildlife-based tourism, either photographic or trophy hunting, is often relied upon as the main revenue source to meet the costs of implementing the management of the WMA and provide benefits to the WMA communities (Igoe and Croucher 2007; Wilfred 2010; USAID 2016). Developed by donors and NGOs, often with little or no consultation with the local tourism industry, revenues have rarely been able to meet simple management needs such as paying the village game scouts (VGS), and consequently little or no revenue has been available to meet the needs of the communities within WMAs (USAID 2016); these revenues are key to ensuring that these communities see conservation of their resources as a valid choice.

Similar to other WMAs in Tanzania, the process of accrediting MWMA has been completed by conservation NGOs which have primarily focused on biodiversity conservation rather than treating the WMA as a business unit. In the case of MWMA, revenues have historically been earned from consumptive wildlife tourism. Revenue from trophy hunting, which only partly covers management expenses of the WMA, has occasionally been supplemented by donor-funded projects for specific activities and tasks. MWMA is relatively distant from the major tourism circuit of northern Tanzanian, making traditional photographic tourism approaches unlikely as a substantial alternative source of income (but photographic tourism is mentioned in the management plan as an option).

15.3 Investing in Landscape Conservation

Carbon Tanzania (CT) has developed a business approach to conservation that combines elements of conventional integrated conservation and development with a "payments for ecosystem services" model. This approach relies on the design, development and implementation of a REDD monitoring framework and uses the combined verified carbon standard (VCS) avoided deforestation methodology and climate community and biodiversity (CCB) standard to create verified emission reductions (VERs). The foundation for the creation of these emission reductions is the implementation of the resource management plan. This management plan designates activities within MWMA, the implementation of which leads to measurable emissions reductions. For example, in this case this means the protection and management of the grazing zones, or '*ronjo*' in the north of MWMA.

REDD is often understood narrowly as a system that promises conditional performance-based payments for ecosystem services (Sills et al. 2009), and while projects developed by CT are similarly premised on performance-based payments, we have gone further in developing an approach to project implementation that recognizes the part that can be played by tried and tested interventions. These include participatory land-use mapping through boundary determination, development of land-use plans and clarification of land tenure, as well as income-generating activities, local employment and community development.

The Makame Savannah REDD project is designed to *protect MWMA from land grabbing and encroachment, and the conversion of land from forest to agriculture – and thus to prevent land-use changes that have occurred in many parts of the TE* (Msoffe et al. 2011). *This REDD project protects the southern extension of the TE and in doing so contributes to the conservation of biodiversity, notable examples of which are rare and threatened species of resident and migratory megafauna* (Foley et al. 2018). *The project stops deforestation and prevents carbon dioxide emissions generated through tree harvesting and clearing land for agriculture. In doing so the Maasai communities living within the WMA enjoy strengthened land tenure, grazing rights and significant monetary income. In summary, the project aims to provide benefits for the climate, local communities and for biodiversity.*

In the next section, we outline how the Makame Savannah REDD project creates the revenue needed to ensure that Makame WMA functions as a viable business unit.

15.3.1 Developing the Carbon Asset

The value system created by a carbon project takes the form of a verified emission reduction (VER), measured in tonnes (t) of carbon dioxide equivalent (CO_2), where the reduction of 1 t of CO_2 from the atmosphere is equal to 1 VER. Developing a REDD project to the point of issuance of a VER requires a specific approach defined by the international standard that CT is applying to the project. The Makame

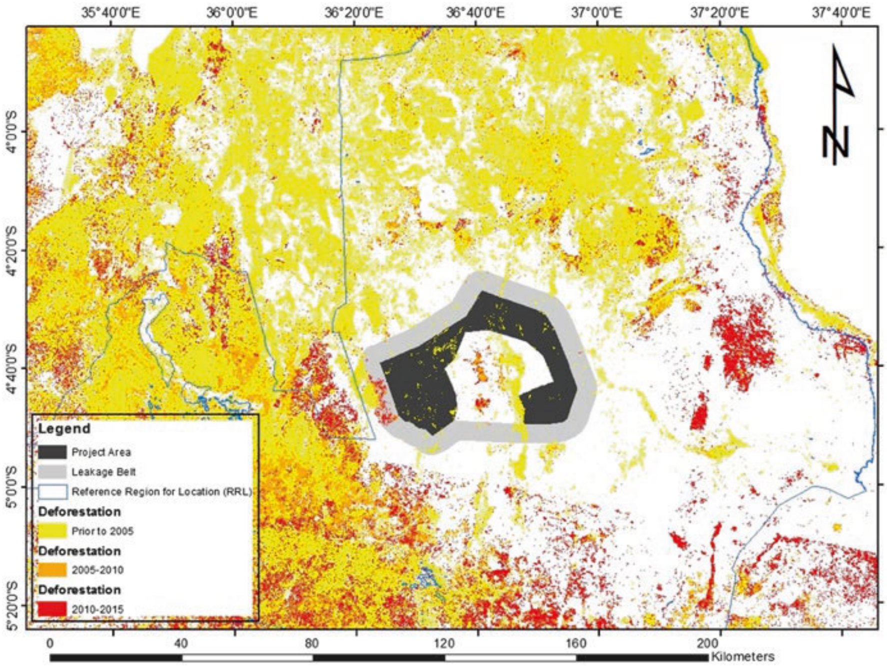

Fig. 15.1 Map of the Makame Savannah REDD project, its associated leakage area as well as deforestation in and around the project area. Landsat TM, ETM and OLI imagery, medium resolution remotely sensed spatial data from the years 2005, 2010 and 2015 were acquired for analysis. A Minimum Mapping Unit (MMU) of 0.36 ha (2 * 2 Landsat pixels or 60 m * 60 m) was used to most closely conform with the Tanzania DNA forest definition minimum area of 0.5 ha

Savannah REDD Project utilizes the avoided deforestation verified carbon standard REDD methodology, entitled, "VM0007: REDD Methodology Modules (REDD-MF)." The only eligible activity as part of this project is avoiding unplanned deforestation because the forest land is expected to be converted to non-forest land in the baseline scenario case. The project area that is accounted for in terms of carbon savings covers 104,065 hectares of *Acacia-Commiphora* shrub and woodland in northern MWMA as depicted in Fig. 15.1. Whilst this area was 100% forest at the start of the project in 2016 it is under increasing threat from deforestation due to shifting agriculture.

Methodological choices and approaches include activities relating to biodiversity monitoring and the delivery of community benefits which are closely linked to the preconditions within the project area. This includes land and resource ownership as well as the structural and practical process of conserving land. In this case, land and resource rights are stipulated by the Village Land Act 1999 and Wildlife Management Act 2012.[1] The Village Land Act designates the village councils and

[1] accessible via https://www.tanzania.go.tz/home/pages/61

village assemblies as the statutory management authorities over these village lands. This land tenure framework, in combination with Tanzania's local government structures, defines the rights and responsibilities of the village councils and village assemblies and provides a strong foundation for participatory management of communal land and resources such as forests. This is further supported by the Wildlife Management Act which describes the management framework. The primary project activity of CT is to assist in the development, submission and implementation of MWMA's Resource Management Zone Plan, henceforth referred to as the Resource Management Plan (RMP) (The Makame WMA Council 2017). The RMP describes the basic management and development philosophy of the WMA by providing a description of the current reality, and further defines the desired future of MWMA. Permitted and prohibited activities are clearly defined in the RMP, with specific reference to where such activities are and are not permitted, and it sets out the potential means for preventing prohibited activities and achieving the desired future as stipulated in the RMP. Implementation of the RMP will result in: the conservation of natural resources in the area, strengthening of governance in MWMA member villages, securing of land rights for local communities, and improving the livelihoods of local communities. Each of these actions will help to reduce the increasing pressures on the local forest resources – pressures that are apparent in Fig. 15.1.

The carbon methodology choices must follow VCS guidelines (Sills et al. 2009) that describe non-project and project scenarios, including an analysis of financial additionality, carbon biomass, as well as baselines of biodiversity and community indicators. Thus, to develop and operationalise a REDD project, CT proves that any activities leading to emission reductions are additional to the normal operational ability of the resource owners, in this case, MWMA. In this chapter we highlight the key processes for project development and implementation; the details and evidence of this process can be found in Makame REDD project documentation and monitoring reports (Carbon Tanzania 2016).

15.3.2 *Scenario Existing Prior to the Implementation of the Project*

The without-project scenario for MWMA depicts the situation in which the implementation of the WMA Resource Management Plan (WMA RMP) is ineffective. In this scenario, land tenure, WMA boundaries and land-use planning would continue to be poorly defined. As a result, the forested area within the project area would continue to be deforested mainly by conversion to shifting agriculture. Accordingly, deforestation would lead to increased carbon emissions from the standing biomass, pastoralists would lose seasonal grazing lands that were designated by the communities, and both resident and migratory wildlife species would lose habitat and likely decline in population size.

Table 15.1 Estimated annual greenhouse gas (GHG) emission reductions in Makame Wildlife Management Area from 2017–2026. GHG emissions are modelled using a spatially explicit neural network; the model takes into account non-linearity and assumes that areas near deforested areas are more likely to be deforested than areas further away from deforested areas. Thus, estimated emission reductions are not linearly scaled with project duration

Year	Estimated GHG emission reductions (tCO_2)
2017	3345
2018	26,461
2019	70,195
2020	116,044
2021	119,009
2022	162,904
2023	170,303
2024	207,288
2025	249,187
2026	240,301

15.3.3 Estimate of Greenhouse Gas Emission Reductions, and Potential Market Value

The project is estimated to generate an average of 136,504 t CO_2 in avoided emissions annually for the first 10 years of the project (Table 15.1). The cumulative net greenhouse gas (GHG) emission reduction over this same time period is therefore estimated to be 1,365,037 t CO_2. The estimated emissions reductions (Table 15.1) are validated at the project start date based on modeling both the rate and spatial distribution of deforestation within the reference region. Verified emission reductions are not issued until verification, which can occur at some point 1 year after the project start date. At this stage the actual emission reductions are measured and calculated and thus test the assumptions made in the model.

A functioning REDD project is premised on the sale of resulting VERs into what is broadly known as the Voluntary Carbon Market (VCM). This is an aggregation of individuals, companies, organisations and institutions that have committed voluntarily to compensate or reduce their carbon footprints through the purchase of equivalent emission reduction units generated by internationally certified projects, such as the Makame Savannah REDD Project.

From the high of 2008 when the market reached a total value of US$ 790 M the global VCM was worth approximately US$ 296 M in 2018, after a decade-long fall in the wake of the financial crisis and a drop in demand for VERs. Credits generated from the land-use, land-use change and forestry (LULUCF) sector represented 55.5% of the market, transacting US$ 171 M in 2018. This proportion is predicted to increase as net-zero carbon commitments by large corporates in 2020 have focused on the use of LULUCF credits, now described increasingly as “nature-based solutions (NBS)” or “natural climate solutions (NCS)” as the most desirable way to mitigate climate change. Of these credits, REDD units represented the majority at 60% by volume, and this is expected to increase with the formal inclusion of REDD as an approved climate mitigation strategy in the Paris Agreement.

Because VERs are not yet considered to be an exchange traded commodity it is not possible to accurately estimate the monetary value of VER from a specific project. The price that a project can obtain for their VERs depends on a number of factors including location, the variety and perceived importance of the biodiversity being protected and the tangible socio-economic benefits that the project delivers for the local community.

15.4 Project's Climate, Community and Biodiversity Co-benefits

The project's targets have been broadly defined based on what the MWMA community cares most about and what its members felt would be critical to achieving the goals of the project. These targets were identified during a series of community workshops and include aspects related to natural resources as well as to important social, cultural, economic or religious aspects of the community. Transparent and equitable management of these assets will result in the sustainable use of the resources that are most critical for long-term community and environmental sustainability in the MWMA.

15.4.1 Monitoring Community Co-benefits

Monitoring co-benefits of the project activities are designed to meet the standards of the Verified Carbon Standard (VCS) and the Climate Community Biodiversity Alliance (CCBA). CT developed a detailed community monitoring plan (Table 15.2). This plan provides an overarching and systematic framework for collecting, analysing, and reporting on social and community indicators to better understand and demonstrate the impact of the project activities and to fulfill the requirements of the CCB standards. CT measures project-related outcomes in the fields of education, governance and management, employment, empowerment of women and relations between CT and the MWMA community (Table 15.2).

15.4.2 Monitoring Biodiversity Co-benefits

The key biodiversity asset of MWMA is the large-mammal community. To assess whether the Makame Savannah REDD project provides co-benefits for biodiversity conservation, appropriate monitoring is a prerequisite (Newmark and Hough 2000;

Table 15.2 Overview of community monitoring plan implemented by Carbon Tanzania in the Makame Wildlife Management Area

Area	Objective	Activities	Term	Indicator/Output	Frequency of monitoring
Education	Increased community understanding of natural resource management and the project	Carbon Champions Program (Carbon Champions are employed by CT to educate community members in Maa on project activities	2016-	# people reached # people tested for knowledge on ??	Bi-annual
	Increased education opportunities in the WMA	Increase education budget	2019-	Funds available for education	Annually
Governance and Management	Improved management of WMA	Train WMA leaders in good governance	2016-	# training sessions # people trained	Annually
		Train local government leaders in good governance	2016-	# training sessions # people trained	Annually
		Implementation and awareness training for resource zone management as well as business plan	2018 2023 2028	Realisation of training session # people trained	Every 5 years
		Increase budget for management activities	2019-	Funds available for management	Annually
	Creation/renewal of management frameworks	Creation/Renewal of WMA resource zone management plan	2017 2022 2027	Existence of WMA resource zone management plan	Every 5 years
		Creation/Renewal of WMA business plan	2018 2023 2028	Existence of WMA business plan	Every 5 years
	Increased/maintained natural resource management infrastructure	Addition of solar electricity, radio communications and internet access to WMA infrastructure	2018	Implemented or not	Every 5 years

Area	Objective	Activities	Term	Indicator/Output	Frequency of monitoring
Employment/Income/ Livelihood	Increased control of resources	Allocate and maintain community grazing area	2017 2022 2027	Resource zone management plan	Every 5 years
		Non-timber (including carbon, medicine, honey) rights	2017 2022 2027	Resource zone management plan	Every 5 years
	Improved community livelihoods	Employ village game scouts	2018-	# people registered as village game scouts # people employed as village game scouts	Annually
		Increase village budgets for development	2019-	Funds available for villages	Annually
Women's Empowerment	Formal women's empowerment	Formation of regularly meeting women's forums in WMA member villages	2017-	# of forums # of forum members	Annually
	Female representation in WMA leadership	Women in leadership positions in WMA	2017-	# of women in leadership positions	Annually
Carbon Tanzania–Community Relations	Formal relations structure	Appeals and complaints committee	2017	# of appeals and complaints logged	Annually
	Informal relations structure	Bi-annual feedback meetings with WMA	2018	# of meetings held # of attendees	Bi-annually

Lindemayer and Likens 2010). If well designed and executed, such monitoring systems can inform us whether REDD+ projects ensure effective conservation of wildlife populations (Kiffner et al. 2019). To achieve this, wildlife monitoring schemes need to provide robust estimates of key indicators such as density or occupancy of species of conservation concern over time to assess the long-term trajectories of wildlife populations and their responses to variation in the environment and to anthropogenic interventions (Nichols and Williams 2006). For wildlife monitoring schemes to generate strong inferences about the ecological effectiveness of the project (i.e. defined by stable or increasing wildlife populations), they need to address three key issues. First, indicators (e.g. density or occupancy) in the REDD+ area need to be compared to the same metric assessed in a control area (i.e. an area not subject to the conservation treatment) to ensure that possible responses of wildlife populations were due to the intervention. Second, the selected indicators should be unbiased and thus their estimation needs to account for the fact that observers may not always detect target species when the species is actually present (i.e. address imperfect detection). Finally, indicators ought to be sampled at the appropriate spatial scale, and ideally rely on randomized or systematic sampling designs (Nichols and Williams 2006). Moreover, suitable wildlife monitoring needs to address challenges associated with generating sufficient detections of the target species. Dense vegetation (which limits visibility), trophy hunting (which causes animals to be wary and avoidant of humans), and generally low densities of target species cause few direct sightings of target species in MWMA (Table 15.3). Additionally, the dense vegetation and presence of potentially dangerous wildlife species prevent systematic placement of sampling units that can be accessed by vehicle or on foot.

As a trade-off between theoretical considerations, realities on the ground, and budget constraints, we designed a road transect-based monitoring protocol to systematically record direct sightings and signs of wildlife species inside and outside of the project area. We designed the monitoring scheme so that the resulting data can be analysed in an occupancy framework which allows direct incorporation of imperfect detectability, and spatio-temporal variation in a unified modelling framework (Mackenzie and Royle 2005).

We placed 52 one km long transects (separated by 200 m to allow for some level of independence between transects) in the project (n = 32) and the non-project (n = 20) areas. On four consecutive days in April 2019 and July 2020, we slowly (~5 km h^{-1}) drove each transect and recorded signs (tracks, dung/faeces) of elephants and all carnivore species (Table 15.3). To address possible detection bias associated with time of day, we reversed the order of transects on days 3 and 4. We recorded all mammal sightings and signs, where signs were detected and identified by experienced trackers of the Hadzabe ethnicity, sitting on the front of the vehicle (Fig. 15.2).

The surveyed area yielded a remarkable diversity of mammal species, yet direct sightings of species was generally low considering an effort of 252 km each year (Table 15.3), further reinforcing the concept of relying on signs for detecting trends of focal species. In particular the carnivore community appears to be complete, i.e. all species predicted to occur in the area (Foley et al. 2014) have actually been detected during the two annual surveys (Table 15.3). For initial analyses, we considered dynamic occupancy models implemented in the "unmarked" package (Kéry and Chandler 2016) in the R 3.6 environment (R Core Team 2016). While

Table 15.3 Wildlife species sighted and carnivore species detected via signs during 52 one km long road transects in Makame WMA. Each transect was driven four times in 2019 and 2020, thus survey effort was 208 km in each year

Common name	Scientific name	Sightings	
		2019	2020
Impala	*Aepyceros melampus*	4	11
Black-backed jackal	*Canis mesomelas*	0	3
Caracal	*Caracal caracal*	0	1
Vervet monkey	*Chlorocebus pygerythrus*	2	0
Zebra	*Equus quagga*	2	1
Slender mongoose	*Galerella sanguinea*	0	4
Giraffe	*Giraffa camelopardalis*	0	1
Dwarf mongoose	*Helogale parvula*	2	1
Bush hyrax	*Heterohyrax brucei*	2	0
Elephant	*Loxodonta africana*	2	0
Kirk's dik-dik	*Madoqua kirkii*	16	30
Warthog	*Phacochoerus africanus*	2	3
Bush duiker	*Sylvicapra grimmia*	0	1
Lesser kudu	*Tragelaphus imberbis*	2	3
		Signs	
Cheetah	*Acinonyx jubatus*	3	0
Bushy-tailed mongoose	*Bdeogale crassicauda*	1	2
Black-backed jackal	*Canis mesomelas*	80	85
Caracal	*Caracal caracal*	54	46
Civet	*Civettictis civetta*	13	1
Spotted hyena	*Crocuta crocuta*	41	87
African wild cat	*Felis lybica*	44	88
Slender mongoose	*Galerella sanguinea*	4	22
Common genet	*Genetta genetta*	35	25
Large-spotted genet	*Genetta tigrina*	5	0
Dwarf mongoose	*Helogale parvula*	12	5
Egyptian mongoose	*Herpestes ichneumon*	6	14
Striped hyena	*Hyaena hyaena*	3	2
White-tailed mongoose	*Ichneumia albicauda*	2	10
Serval	*Leptailurus serval*	42	9
Elephant	*Loxodonta africana*	260	95
African wild dog	*Lycaon pictus*	4	2
Honey badger	*Mellivora capensis*	18	5
Bat-eared fox	*Otocyon megalotis*	22	25
Lion	*Panthera leo*	21	21
Leopard	*Panthera pardus*	31	5
Aardwolf	*Proteles cristata*	1	22

Fig. 15.2 Hadzabe tracker at work, detecting and identifying animal signs along road transects in MWMA. (Photo by: Roshni Lodhia)

accounting for imperfect detection, these models allow inference on the occurrence of target species at the transect level and on how changes in occurrences at the transect level are driven by local colonization and extinction between the 2 years (Kéry and Chandler 2016). To fit these models, we created presence/absence matrices for each species based on unique detections of the target species. For simplicity, we considered only two competing models. We first considered a dynamic occupancy model with constant detection probability and constant occupancy across areas. In addition, we considered a model where occupancy was conditional on the area (project vs. non-project). We compared competing models based on AIC_c scores and selected the most parsimonious model and focused our analyses on three target species: elephants, lions and leopards. In the following, we do not focus on yearly trends because the 2020 survey was carried out during a different season (dry season) than the initial survey (wet season) and thus do not allow for meaningful temporal comparisons. In both felids, model selection suggests that occupancy did not differ between project and non-project areas whereas elephant occupancy differed spatially (Table 15.4).

Elephant signs were detected with a relatively high detection probability (0.66; 95% CI: 0.59–0.73) and elephant occupancy was considerably greater in the project vs. the non-project area (Fig. 15.3). Lions were detected at much lower detection probability (0.20; 0.11–0.33) and they were estimated to occupy c. a third (0.36; 0.17–0.60) of transects, irrespective of area status. Among the three species, detection probability of leopard was smallest (0.12; 0.08–0.17). Estimated occupancy was very high, yet associated with wide margins of error (Table 15.4; Fig. 15.3).

Table 15.4 Estimates for initial occupancy (2019), colonization and extinction rates (changes across transects between 2019 and 2020), and detection probability of elephant, lion, and leopard in Makame WMA. Model selection, suggested inclusion of the site covariate for the elephant model, whereas lion and leopard occupancy did not differ between the two sites. All estimates are on a logit-scale

Elephant	Estimate	SE	z-value	p-value
Occupancy: Non-project	−1.09	0.52	−2.10	0.04
Occupancy: Project	2.46	0.70	3.53	<0.01
Colonization	−1.04	0.51	−2.03	0.04
Extinction	−0.75	0.40	−1.88	0.06
Detection	0.68	0.15	4.50	<0.01
Lion				
Occupancy	−0.60	0.51	−1.17	0.24
Colonization	−0.21	0.70	−0.29	0.77
Extinction	0.17	0.97	0.17	0.86
Detection	−1.37	0.35	−3.98	<0.01
Leopard				
Occupancy	7.17	29.8	0.24	0.81
Colonization	−0.35	108.00	0.00	1.00
Extinction	1.15	0.61	1.89	0.06
Detection	−1.99	0.21	−9.47	<0.01

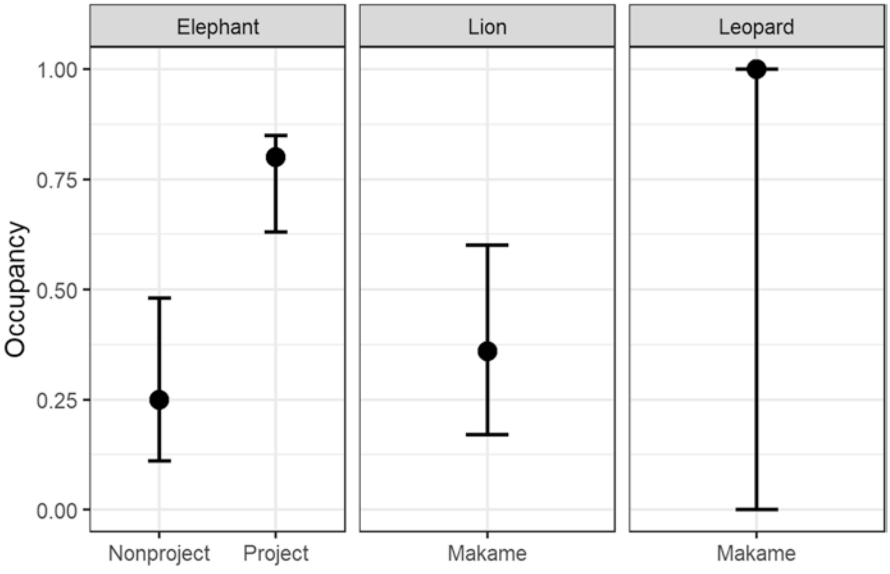

Fig. 15.3 Initial (2019) occupancy estimates (and associated 95% confidence intervals) of elephant, lion and leopard in the surveyed areas of Makame WMA, based on dynamic occupancy models. For elephant, model selection suggested that occupancy differed between survey strata; for both felids, there was no indication that occupancy differed across strata

15.5 Can REDD Projects Contribute to Human-Wildlife Coexistence?

The limitations of the REDD approach have been studied in the past (Mbatu 2016). However, many of these critiques have predominatly focused on UNFCCC REDD approaches and REDD implemetation by government agencies that develop and follow their own procedures (Angelsen et al. 2012). Perhaps one of the key limitations of REDD relates to how site selection is conducted. This includes understanding the cultural setting, the drivers and rate of deforestation, and the legal pre-conditions of a potential project area. These factors can all influence the additionality argument which considers what the 'business as usual senario' would be and asks what would happen in the absence of this project. Understanding these factors can have impacts on methodological choices and how a project is implementated. REDD initiatives such as Makame, follow international standards and approaches which ensure rigour within the emission accounting, and ex-post measurement of results. This includes conservative carbon accounting that accounts for leakage (movement of deforestation elsewhere) and risk, these two factors alone remove 30% from the issued emission reductions. Application of international standards ensures marketability, saleability and therefore the long-term financial sustainability of this approach. In addition, REDD is correctly perceived as an unsuitable approach for areas where the primary causes of deforestation are charcoal production and illegal timber cutting. In the case of the Makame REDD project, like much of Tanzania, 81% of deforestation is driven by shifting cultivation, considerably more than charcoal at 12% (Doggart et al. 2020).

In terms of providing co-benefits, REDD projects have also been subject to manifold criticisms. Scholars critiqued that biodiversity may not be adequately protected under REDD schemes (Collins et al. 2011; Caro and Borgerhoff Mulder 2016). To address this concern in the MWMA, the prioritisation of wildlife protection is supported by the Honeyguide Foundation which trains village game scouts in anti-poaching operations. Thus, the current project not only protects habitats but also explicitly addresses the control of illegal exploitation of wildlife to ensure that the diversity and abundance of wildlife species are conserved.

In other REDD projects, unequal benefit sharing has been highlighted as an obstacle to community benefits (Chomba et al. 2016). CT addresses this by strengthening the governance capacity of MWMA to ensure transparent and efficient management, and by ensuring that land-tenure rights are secure in the long term (Chhatre et al. 2012).

The Makame REDD project can serve as a new model for funding improved land management and biodiversity conservation, especially in the framework of WMAs in Tanzania. Investing in, designing, implementing and operationalising REDD projects requires a new and innovative way of thinking. To our knowledge, in all cases where REDD projects have been successful, they have been developed by organizations that are built on business principles and are dedicated to developing and managing REDD projects, and rarely by conservation organizations that are running REDD projects as subsidiary activities.

In this case and elsewhere in Tanzania, CT has shown that REDD can provide significant economic benefits to rural communities. To date, CT has transferred 800,000 USD to communities in Tanzania and simultaneously protected areas for biodiversity conservation (Kiffner et al. 2019). These successes have predominately stemmed from an approach that follows a business strategy to conservation, resulting in cost effective operations that focuses on long-term investment rather than donor-based funding.

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Chapter 16
Education as a Tool to Live in Harmony with Nature

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Abstract Environmental education (EE) can be an effective tool for developing meaningful conservation awareness and action. EE empowers people to explore environmental issues and engage in problem solving and actions to improve their environment. The Tanzanian government mandates that EE in primary and secondary schools be integrated into a range of subjects. In practice, lack of appropriate materials and teacher training limit EE implementation in Tanzania, and textbooks written in the U.S. or Europe may be less effective teaching aids in the Tanzanian context. We discuss the importance of effective communication and the importance of evaluating the impacts of environmental education interventions on knowledge and attitudes. We describe three innovative, culturally relevant, locally designed EE programs being implemented in schools in the Tarangire Ecosystem that fulfil the Tanzanian government's mandate while building community support for conservation efforts. We provide examples of media efforts for conservation on television

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and radio. Finally, we profile three case studies in the Tarangire Ecosystem that measured the impacts of: (1) a classroom education program; (2) a program that brings youth to Tarangire National Park; and (3) a conservation-themed gospel song played on the radio, and the lessons learned from the evaluations of each intervention.

Keywords Environmental education · Impact evaluation · Communication · Tarangire Ecosystem

16.1 Introduction

More than a half-century ago, the United Nations recognized the importance of and need for environmental education programs to address the challenges of environmental degradation across the globe (IUCN 1970). Specialists were encouraged to develop and implement environmental education (EE) as a science-centered multidisciplinary subject in which most, if not all, school subjects should be incorporated. Furthermore, EE was recommended as an essential component in teacher-training courses. The concept of EE is now widespread in educational policies, school curricula, and conservation strategies throughout the world (Rickinson 2001). The benefits of EE extend beyond just knowledge acquisition. Several literature reviews examining EE programs designed for school-aged students have documented positive outcomes in environmental knowledge, attitudes, dispositions, and skills, but EE also can positively affect outcomes less directly focused on environmental conservation such as improving overall academic achievement and civic engagement (Ardoin et al. 2018). Furthermore, positive environmental attitudes significantly increase individual motivation of students to learn science, demonstrating a synergy between EE and science education in general (Schönfelder and Bogner 2020).

Since 1995, the Tanzanian Ministry of Education and Vocational Training's Environmental Education Strategy has emphasized the importance of EE at all levels within school curricula. The official policies of the Environmental Education Strategy highlight teaching that includes active participation and cooperation with local residents through community activities (Kalungwizi et al. 2019). The idea is that schools can function as community centers for environmental education through participatory teaching. One significant challenge of putting EE into practice, however, is the orientation in Tanzanian schools towards lecturing and learning by memorization, and teachers are often not well-prepared for participatory teaching (Kalungwizi et al. 2020). Specific trainings for teachers that promote active, experimental, and participatory teaching, for example with tree planting and ongoing care for the trees, can help teachers and local communities to improve EE by engaging community members and sharing knowledge and skills with pupils, with the added bonus of helping to integrate groups within the broader community such as agriculturalists and pastoralists (Kalungwizi et al. 2019) and, of course, improving the environment.

In many parts of Tanzania, however, EE for young people is available only through extra-curricular programs such as the Malihai Clubs in secondary schools (see Box 16.1) because EE has not been formally incorporated into the school curriculum (McDuff 2000). Bruyere et al. (2011) noted that providing teachers with EE training and implementing EE in informal situations as well as formal classroom settings can improve students' knowledge about ecology. Further, the types of EE trainings and design of EE programs can influence consequent behaviors. Bruyere et al. (2011) outlined a model predicting responsible environmental behavior based on: (1) environmental sensitivity, knowledge, and attitudes (entry level); (2) personal investment, commitment, and understanding of the consequences of behavior (ownership); and (3) knowledge of and skill in using environmental action strategies (empowerment). Bruyere et al. then quantified the effects of a professional training program about marine and coastal environmental issues for primary school educators in Tanzania, and found that empowering teachers strongly predicted intention to act. Thus, teacher trainings should include strategies on what educators can do to take action, and then build their confidence and abilities to lead those actions (Bruyere et al. 2011).

Box 16.1: Malihai Clubs of Tanzania (MCT)

By Veila F. Makundi, Education Coordinator at Wild Nature Institute, Benjamin Kijika, National Coordinator at MCT, and Youthness Godfrey, Wildlife Officer at MCT

Malihai Clubs of Tanzania (MCT) was established in 1980 under the Ngorongoro Conservation Area Authority. It later shifted to Tanzania National Parks and then to the Tanzania Wildlife Protection Fund. In 2016 the Tanzania Wildlife Authority took over responsibility for most wildlife duties in the country and MCT was officially placed under its authority in October 2017.

MCT works in three zones: the Lake Zone, the Northern Zone, and the Southern Highlands Zone, with headquarters in the Njiro area of Arusha. MCT provides wildlife education through after-school environmental clubs, permitted under the relevant education authorities, and prioritizes Tanzanian communities adjacent to protected areas. Nationwide, there are total of 2434 registered clubs: MCT Lake Zone = 652 clubs, MCT Northern Zone = 1438 clubs, and MCT Southern Highland Zone = 344 clubs. MCT registers both private and government primary and secondary schools, as well as other higher learning colleges and universities.

Malihai is a Swahili word meaning 'Living Wealth' (including vegetation, mammals, birds, air, water and soil) and forms the hinge on which revolves environmental education that is disseminated to the younger generation. MCT clubs are designed to bring awareness to young children about the meaning, importance, and value of the environment that he/she is living in, and what he/she can do to make the environment better. MCT also works with other

(continued)

Box 16.1 (continued)
organizations (e.g., PAMS Foundation, Roots and Shoots, Wildlife Conservation Society). Through the clubs, children are provided access to textbooks, magazines, class lectures, and discussions and learn through audio-visuals which help to expand their knowledge about nature (Mariki et al. 2011). Youth are given opportunities to attend and speak their views and offer their contributions to solve current problems facing our communities.

Activities

Malihai Clubs of Tanzania trains, educates, and involves youth in communities living near protected areas to protect wildlife and conserve the environment. The guiding hypothesis is that through education people will create sustainable conservation actions.

Main Objectives

1. MCT teaches and encourages Tanzanian societies, especially youth, to understand the traditional, ecological, social, and economic values of natural resources.
2. MCT encourages youth in educational institutions to join and spread knowledge about how to protect nature and their environment to make the world a better place. MCT's slogan is "Let's Go Green."
3. MCT teaches communities about the traditional, economic, and scientific importance of natural resources.
4. MCT spreads and increases knowledge and awareness of natural resources among Tanzanian societies.

16.2 Behavior Change Needs Effective Communication

The ultimate aim of communication for conservation is to produce a change in behavior, both at the individual and community level. For a long time, conservationists have used the knowledge deficit model, assuming that to motivate people to change behavior, the conservation mission itself and the information about the behavior were sufficient. Unfortunately, this is not the case.

Behavior is determined by multiple factors, including a person's values, attitudes, relevant social and personal norms (Kidd et al. 2019), identity, and contextual factors such as socio-economic circumstances and infrastructure (Stern 2005). As such, behavior is very difficult to change, and to do so requires high motivation, generated by obtaining benefits that outweigh costs. To be effective, communication must go beyond simply providing training to the public; it must be strategically designed to make change desirable, and it must present an adequate benefit to the public (Heimlich and Ardoin 2008; Kidd et al. 2019). To transform communication from a mere transmitter of information to a tool that produces practical results, one must carefully shape it for each targeted audience.

The organization of a message can alter the perception of its content. Effective communication generally follows a sequence outlined in the Elaboration Likelihood Model, used by many environmental communicators, from getting the audience's attention to facilitating their comprehension, elaboration, and integration of new information with old, and finally shifting attitudes which ideally results in enduring behavior change (Jacobson et al. 2019).

16.2.1 How to Build Effective Messages

1. Initial assessment. The initial assessment is fundamental: how many subgroups are there in the group we are working with? What are the characteristics of each? What values do they have? What could prevent them from changing their behaviors? What could motivate them? The answers to these questions frame the context within which to shape the message.
2. Moving emotions. Emotions play a fundamental role in behavioral decision-making. If we succeed in making our audience 'live' an emotional experience through our communication, through words, images, metaphors, and stories, we will greatly facilitate the acquisition of the message, and will help generate the necessary substrate for change. Emotions can create connection to nature, which is the extent to which people feel integrated with nature. When someone feels connected to nature, he or she may be more inclined to protect it (Schultz 2002; Nisbet et al. 2009). In order to do this, it is essential to know and make the best use of the emotional and cultural code of the place where we are operating and to use mediators—be they people or communication tools—who are able to interpret the code in the best possible way.
3. Involvement. A good way to learn, or to develop the desire to learn more, is to do. For communication, action is one of the most effective tools to produce permanent changes. Unfortunately, producing this kind of communication requires economic and human resources that very often are not readily available. Involvement can be produced through activities such as experiences in the field, even on a small scale (for example, a school garden) or by urging the students to take on a positive identity and identify themselves with a mission for which they become the referents. Or again, a simple way to encourage involvement is to use hooks, in the communication process, to the specific daily life of our audience. This generates the sensation of being part of the context and of having the ability to understand the message and manage the situation.
4. Redundancy of the message. Several studies have shown that the use of different communication tools and the crafting of unique messages for each subgroup creates a 'redundancy' of the message that has positive effects on its memorization. The same goes for translating the message by making it available and assimilable through different senses: the same message in writing, pictures, and audio makes it easier to process and fix it in the memory.

For a message to be effective, it must be carefully modelled for the targeted audience. It should generate an emotion capable of giving the public a real ‘experience’ that is easy to remember. It should stimulate multiple senses, and be repeated through multiple channels, to create a redundancy that facilitates familiarization with the message. It should strive to change a specific behavior that will contribute to the desired goal for which the message has been created. And it should also be engaging, through direct practical experiences or even just through the everyday life of the audience, creating a bridge of ‘familiarity’ that opens the door and facilitates the passage of content. Feelings of involvement conveys confidence that enables one to do more than one thought was possible (Bruyere et al. 2011).

16.3 Impact Evaluations to Assess Effectiveness

In many ways, conservation is behavior (Schultz 2011). Because people are the reason why so many species are declining and ecosystems are being degraded, pro-environmental behavioral change among people interacting with species and ecosystems can also be the solution. To know that the work that we are doing is actually producing the desired behavioral change, we need to evaluate the behavioral changes that we aim to cultivate with our interventions. For effective behavioral change to occur within the field of environmental education, there is increasing recognition of the importance of evaluations to assess program effectiveness (Rickinson 2001; Ardoin et al. 2018).

Evaluation is the process whereby we systematically document changes in behavior or knowledge (usually) in relation to projects or interventions. The information typically includes the outcomes, impacts, and perceptions of a program, and with it we strive to determine how effective the program was, how it could be improved, or how to inform future decisions about ways to proceed with future implementations of that program (Patton 1987). As a consequence of the data that are collected, the success of an environmental education program can be assessed objectively (NOAA 2004).

Prior to the increasingly widespread use of evaluations in EE programs, the effectiveness of educational programs was often quantified using attendance, number of brochures or posters distributed, or other tools that merely adjudged the number of people that a program reached. In contrast to these reach measures, impact evaluation allows us to determine the changes created by a program, which comprises a more rigorous measure of effectiveness than simple reach measures (NOAA 2004). Impact allows us to differentiate how and why a program led to pro-environmental changes, not merely the number of people who participated. For example, thousands of people may walk past a presentation (a possible reach metric), but a better measurement would be to understand whether the presentation itself directly contributed to a knowledge or behavioral change among audience members based on their participation in the presentation.

Environmental education programs are more effective when evaluation is part of the process from the beginning, rather than as an afterthought to meet a requirement, often requested by a funding agency. By including evaluation from the outset into EE programs, we are engaging in evaluative thinking (Kellogg Foundation 2017). Evaluative thinking ensures that we are designing our EE programs so that we can best learn their value at present. This approach also makes it possible for us to improve upon what we have created, so that future program implementations will be even more effective at meeting our long-term goal. Evaluative thinking also ensures that our decisions are based on data, rather than impressions or hunches.

Most impact evaluations strive to understand how audiences may have changed with respect to knowledge, behavior, or attitudes, all of which should further the larger project goal. Of the three, knowledge is usually the easiest to change as it involves merely learning new ideas or facts. Behaviors are harder to change, but can be modified with adequate knowledge, social norming, and other outside experiences (Schultz 2011). Behavior usually requires both a change in attitude or perception about a situation, as well as the willingness and wherewithal to act upon that attitude change. As such, behavior is notoriously hard to change (Stern 2005; Kidd et al. 2019), and it is even harder to maintain. Most research has indicated that attitudinal change is the hardest to achieve. External societal forces including financial incentives, legal changes, regulations, and other larger matters are often required to facilitate attitudinal changes (Stern 2005; Schultz 2011; Kidd et al. 2019).

Behavioral change is facilitated when evaluative thinking frames the EE programs that we conduct and when the programs are aligned closely with the desired project goal. Clearly identifying these components, and then using a rigorous and ongoing evaluation approach helps to ensure that EE programs will be most successful. This is particularly important in ecologically rich but imperiled locations like the Tarangire Ecosystem.

16.4 Environmental Education in the Tarangire Ecosystem

The Tarangire Ecosystem (TE) is one of the richest areas on the planet for large animal diversity and abundance. The community lands surrounding the protected areas in the TE play a critical role in the health of the ecosystem and the wildlife it supports (Chaps. 8, 9, 10, 11, and 12). It is therefore important that people living in these areas understand the value of nature for wildlife conservation as well as for their own well-being and day-to-day survival. It is also important that they understand their roles as custodians of these areas, the importance of renewable natural resources and wildlife for Tanzania's economy, and how to minimize conflict and live with these species in harmony.

Because the TE is a human-dominated landscape yet supports economically and ecologically important wildlife and habitats, several non-governmental organizations have designed and implemented long-term environmental education programs in various schools and communities in the TE. The goals of these programs are to create a community of wildlife custodians, build community support for other activities such as anti-poaching ranger patrols and scientific research, and inspire the next generation of Tanzanian conservationists. In this chapter we profile three long-running environmental education programs in the TE:

- Wild Nature Institute's "Celebrating Africa's Giants"
- Tanzania People and Wildlife's "Youth Environmental Education"
- PAMS Foundation's "Living in Harmony with your Natural Surroundings"

We describe the focus and specific activities of each of these programs, and the reach in terms of numbers of participants. We highlight key successes and challenges faced. We also describe a creative media campaign to reach communities with wildlife conservation messages through music and video, and we present results from three case studies of social science assessments regarding the impacts of EE interventions.

16.4.1 Celebrating Africa's Giants

The Wild Nature Institute is a non-profit organization that conducts scientific research on endangered wildlife and inspires the public to protect wild nature. One program that seeks to achieve the latter is "Celebrating Africa's Giants" (CAG), a youth environmental education initiative for conservation of giraffes, elephants, rhinoceros, and wildebeests. The primary purpose of CAG is to inspire conservation action through EE for children in Tanzania.

Each of the CAG programs reaches children through a collection of educational and social lessons, activities, and materials focused on a particular species. Giraffes are used to teach adaptations to the environment and recognition of similarities and differences among individuals; elephants teach ecology and social behavior as well as empathy for others; rhinoceroses teach wildlife conservation and the importance of teamwork; and wildebeests teach migration and the need for landscape-level approaches to conservation.

<u>Initiating Contacts with Primary Schools</u>—The first CAG education project focused on wildebeests. As part of Wild Nature Institute's campaign to preserve the wildebeest migration corridor between Tarangire National Park north to the Gelai Plains (see **Lohay et al.** Chap. 13), in 2014 the Institute published a tri-lingual Maa (the language of the Maasai ethnicity), Swahili (the national language of Tanzania), and English children's book that educates children and adults about wildebeest migration, Tanzania's wildlife, and the ecological and economic benefits of conservation in the TE. The story explains why animals migrate and the challenges migratory populations face. By presenting a story with pictures and maps simultaneously in three languages, *The Amazing Migration of Lucky the Wildebeest* was designed to promote literacy, support conservation values in Maasai people, and provide greater understanding of the role wildebeests play in Tanzania's ecology, economy, and culture.

The distribution of these books was conducted by partners contracted from a local Maasai non-governmental organization. They held initial consultative sessions with the Monduli District Education Officer and head teachers at nine primary schools in the northern plains to introduce the book project and obtain approvals to use the book in classrooms. Understanding the shortage of children's books in Tanzania, particularly in rural areas, the head teachers commended and welcomed the initiative as they viewed it as an important step in adding value to extra-curricular reading materials.

After the initial consultations with the head teachers, the distributors held sensitization and dissemination sessions for 66 teachers in the nine primary schools. The aim was to orient the teachers to the book with regard to its context, structure, content, usage, and targeted pupils—in this case, Classes 3 and 4. The distributors also conducted a one-day seminar for 18 teachers, two from each of the nine schools, to develop a monitoring and evaluation tool for the use of the book. Participants brainstormed concrete suggestions on criteria and key elements of the evaluation tool and developed a simple form with 11 items. The teachers agreed to fill out the monitoring form each week for three consecutive months.

The book distribution process included a unified approach that collected profiles for each school: village name, school name, head teacher's name, phone/contact, signature, number of books distributed, date received, number of pupils and average age in Class 3 or 4, and the names of teachers present during the distribution. The distributors also documented all aspects of the distribution with photographs of important occasions, scenes, and landmarks in each school, including school signboards, the school environment, the distributors handing over the children's books to teachers, sensitization and dissemination seminars, pupils receiving and using books, and school infrastructure. By the end of 2015, 2600 books had been distributed to 13 primary schools in villages throughout the TE.

The distributors made follow-up phone calls to the initial nine schools each Friday to gather weekly information about the use of the book in accordance with the monitoring form. At the end of the 3 months, all the filled forms were collected from each school and data were analyzed to understand the usage, performance, and reach of the children's book in the initial target schools.

Expanding the Program—After the launch and success of *The Amazing Migration of Lucky the Wildebeest*, in 2016 the Wild Nature Institute developed additional educational materials for three of Africa's giant megaherbivores—giraffe, elephant, and rhinoceros—and created the name Celebrating Africa's Giants (CAG) for the program. The target audience was expanded from the Maasai community to all Tanzanian children in the TE regardless of ethnicity, and the subsequent materials were written in Swahili and English. The program was also expanded into secondary schools. The goal of CAG was to use the allure of giant animals as flagship species to inspire conservation actions for savanna habitats. The Institute assembled a team of scientists, educators, illustrators, and designers to develop the suite of innovative educational materials, and hired a permanent, full-time education coordinator to implement the program. The materials included not only storybooks, but also educational posters and targeted lesson plans and hands-on activities to accompany each of the books and posters. The conservation-themed EE materials and program were designed to be integrated into a range of subjects (e.g., math, language, geography, biology).

For CAG, the Wild Nature Institute produced three additional storybooks and three posters. The posters could be hung in any classroom or office and describe the anatomy, behavior, and other interesting information about each species in a visually compelling, easy-to-read format. Each segment includes a ‘call to action’ detailing what students and teachers can do to help each type of animal. Two Swahili-language activity books for younger children were also produced, featuring mazes, word searches, picture matching, vocabulary games, connect-the-dots drawings, and other learning activities. In-depth lesson plans for teachers were developed by an expert education consultant to accompany the books and posters. Activities in each of the lesson plans underwent a Primary and Secondary Curriculum Alignment by the consultant to clearly describe to teachers how each meets the Tanzanian curriculum requirements for various basic subjects, thus enabling the teachers to use the materials and implement the program within the classroom rather than simply as extra-curricular activities. All materials are available for download at: www.AfricasGiants.org

The Wild Nature Institute’s education coordinator and education consultant provide hands-on trainings for teachers on how to implement the CAG program and develop effective, conservation-oriented teaching strategies. Teachers are presented with the books and posters, learn the lesson plans, and practice the hands-on activities that they will use in the classroom. The Institute’s team had trained more than 200 teachers by the end of 2020. In addition, the Institute’s education coordinator visits the classrooms repeatedly to read the books and assist the teachers with implementing the lessons and activities. The program includes mobile education units that contain all the supplies needed to implement each of the activities (see Fig. 16.1).

Additional participatory activities are conducted to supplement the classroom EE curriculum. Wild Nature Institute organizes Giraffe Celebration Days in schools. Giraffe Day includes hands-on activities such as arts and crafts, a giraffe quiz, singing, and dramatic performances about giraffes, as well as sports and a school cleanup. Sports equipment and t-shirts that say “Tuwatunze Twiga, Fahari Ya Tanzania” (translation: We Protect Giraffes, the Pride of Tanzania) are distributed during the celebration days, and free lunches are provided. Thus far, hundreds of children have attended these events to celebrate the giraffes in their own backyards. Native trees are planted at secondary schools. Further, secondary school students are escorted on safaris to Tarangire and Lake Manyara national parks to see wildlife first-hand and continue the participatory learning outside the classroom.

The giraffe-themed education program was launched in October 2016 and as of December 2020 had reached approximately 26,128 students in 73 schools throughout Tanzania, ranging from ages 4 to 16, including more than 10,000 students in the TE (Table 16.1). At the end of 2020, the elephant education program was in its beginning stages of implementation, with one training conducted for four teachers from three primary schools to introduce the materials and lesson plans.

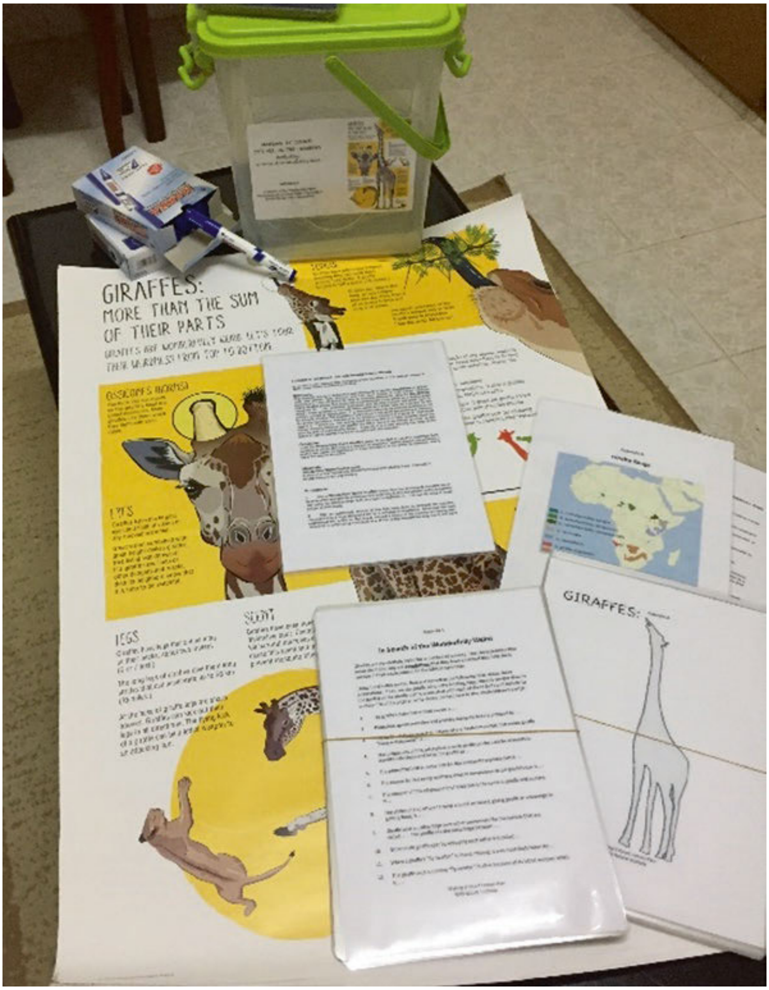

Fig. 16.1 Giraffe in a Box mobile education unit for classrooms in the TE

Successes—Teachers and ward education officers have received and supported the EE program quite positively. The *Lucky the Wildebeest* book was scaled up to older classes not intended in the initial plan. The program has expanded annually. In 2019, Wild Nature Institute's education coordinator visited 12 TE schools (6 primary, 6 secondary) a total of 68 times to implement the giraffe education program. Results from a preliminary evaluation of the reach and impact of the program in two classrooms indicated most students care about giraffes, but after the CAG program twice as many students understood that giraffe populations are in decline, suggesting the program increased knowledge about the need to protect giraffes (see Case Study 1, below).

Table 16.1 Year initiated, districts covered, number of primary and secondary schools and children reached, and successes and challenges of three education program sponsored by non-governmental organizations in the Tarangire Ecosystem of Tanzania

Education program	Year initiated	Districts	Number primary school children	Number primary schools	Number secondary school children	Number secondary schools	Successes	Challenges
Celebrating Africa's Giants (WNI)	2016	Monduli, Babati	7240	13	3063	12	Children demonstrated increased knowledge about giraffe population declines after experiencing program. Program has been requested by educators throughout Tanzania.	Difficult travel and communication in remote areas. Some unresponsive head teachers.
Living in Harmony with Your Natural Surroundings (PAMS)	2015	Babati, Simanjiro	na	na	16,279	8	Program is extremely popular and spread through word-of-mouth in one case.	Initial contacts with schools was difficult as the program was new.
Youth Education Program (TPW)	2009	Simanjiro	5808	11	793	2	Program activities reach thousands of other students and community members beyond the active Wildlife Club members through celebrations and events. Surveys and assessments show the program is creating sustainable positive outcomes in Simanjiro District.	Funding, scheduling, and logistical challenges slow the expansion to additional schools and districts. The selection of participants must use criteria that are fair, gender non-discriminatory, and non-political, while also allowing as many students as possible to participate in some activity.

Challenges—While the overall program has been largely successful, there have been several challenges to overcome. Communication was difficult with some of the more remote schools, with poor mobile phone network. Travel by teachers to workshops and meetings was also a challenge as some regions have no public transport and teachers had to board private cars or hire bikes and bicycles. In these cases, the best options are to meet the teachers at the schools and work with them on a school-by-school basis rather than organizing centralized workshops, and Wild Nature Institute changed its operations accordingly. Finally, some of the head teachers who were contacted were unwilling to respond, which deprived the teachers and children in those schools from being able to receive the free books and posters and participate in the program.

16.4.2 Youth Environmental Education Program

Tanzania People & Wildlife (TPW) is a non-profit organization working to protect wildlife, invest in people, and restore balance to Africa's vital ecosystems through effective conservation action, applied science, and collective impact. TPW focuses on mindset and behavior change programs specifically tailored to youth. With a long-term impact objective of creating a community-level conservation ethic, TPW supports wildlife clubs in primary and secondary schools, hosts annual environmental camps, and facilitates student visits to the nearby protected area, Tarangire National Park, to instill an appreciation for the natural world that reaches beyond the fear of predators.

Wildlife Clubs and Environmental Camps—TPW's youth environmental education program seeks to change the conventional model of conservation education by demonstrating both the economic and non-economic values of wildlife and reinvigorating the traditional value of coexistence. For instance, wildlife club activities and environmental camps aim to educate students about the ecological roles of apex predators and the top-down trophic controls they impart on the landscape. Particularly in pastoralist communities, where natural controls on wild herbivores are necessary to maintain healthy pastures for livestock, these lessons can shift the way youth view predators. Rather than wanting predators to remain in the national parks, young pastoralists can grow up with a greater tolerance for their presence in community rangelands and an appreciation of their ecological role.

TPW formed its first Wildlife Club in 2009 at a primary school in Simanjiro District—the Noloholo Simba Klabu (Noloholo Lion Club). Each club formed since then has taken on the name of a different iconic wildlife species. Now in 2020, TPW supports 13 Wildlife Clubs at 11 primary and two secondary schools throughout Simanjiro District (Table 16.1). Over 2640 students have participated actively in club activities, many of whom remain members of the clubs for their entire school career. Wildlife Clubs at each school partake in environmental lessons as well as weekly activities including tree planting, riparian area clean-ups, rainfall monitoring, track identification, and celebrations for Earth Day, Women's Day, and World

Lion Day. For several years, the World Lion Day celebrations have included talks by Maasai elders who killed lions in their youth as a cultural, coming-of-age practice or to gain status in the community. The speakers discuss their change in mindset that led them to protect lions rather than hunting them, they explain current threats to the lion population, and they tell stories about the cultural importance of lions. These celebrations are open to all community members so estimated total outreach of the program is over 8000 people.

Wildlife Club activities are led by a group of 16 Youth Environmental Mentors (YEM), former Wildlife Club members who have since graduated or left school. Since inception of the YEM program, 23 former Wildlife Club members received training in leadership skills, community organizing, and youth engagement.

Youth Environmental Mentors also assist with annual environmental camps for active Wildlife Club members. The first environmental camp took place in 2010 for 20 primary school students. Since then, 800 primary and secondary school students have participated. Students stay overnight at Noloholo Environmental Center for 7 days and partake in interactive lessons, wildlife-oriented games, small group projects and presentations, and field exercises such as bird watching, spoor tracking, and camera trapping. The week concludes with a hike through the communal rangelands surrounding Noloholo, often ending at the top of a nearby hill from which the students can look into Tarangire National Park and appreciate their proximity to a source of national pride. The camp lesson curriculum follows a trophic cascade model to highlight the ecological role of all living and non-living components of an ecosystem from water and grazing land through insects, reptiles, amphibians, birds, herbivores, and carnivores. The curriculum also includes sessions on anthropogenic impacts on the environment, climate change, conservation, and environmental careers. These lessons allow students to gain a solid understanding of basic environmental topics that they can bring back to their Wildlife Clubs after the camps.

Since 2017, environmental camps have included pre- and post-camp knowledge and attitude assessments to gauge the effectiveness of the camp curriculum on knowledge retention and changes in attitude about environmental issues. On average, students' scores increased by 36% on the post-assessment, providing evidence that the camp curriculum significantly improves students' environmental knowledge. Attitude assessments have also provided encouraging results: in an exercise measuring the importance of various aspects of the environment to students, perceptions of the value of carnivores, birds, and trees all increased post-camp. In particular, the perceived importance of carnivores in an ecosystem rose an average of 18% after camp. This change was attributed to the multiple camp sessions on the ecological importance of apex predators, specifically for pastoralists.

Young Explorers Tarangire National Park Trips—Trips to Tarangire National Park provide an opportunity for local youth to interact with wildlife in a positive and impactful way. Although these children live only miles outside of the park, most of them have never been inside the protected area to view wildlife in a relaxed and safe environment. Outside the park, conflict characterizes many interactions with species such as lions and elephants. Seeing these animals away from livestock and farms allows children to appreciate them as unique wildlife and not view them solely as threats. In addition, national park trips can provide an opportunity for students to learn about wildlife and wildlands as a national asset worth protecting. For secondary school students, national park trips also offer a platform for discussion about careers in wildlife conservation, protected area management, tourism and hospitality, and wildlife medicine. These trips help facilitate a positive relationship between protected area management, conservation organizations, and local communities.

Since 2016, TPW has facilitated nine Young Explorers Tarangire National Park trips for 292 primary and secondary school students. The selection process involves active Wildlife Club members of schools in Simanjiro District. The head teacher and Wildlife Club teacher at each school jointly select the students based on their overall performance in school and their participation in club activities. Due to space limitations at Noloholo Environmental Center where the students spend the first night of the trip and limited seating in the vehicles, each trip can support 32 students. Of the 292 students who have participated in Young Explorers trips, 44% have been female and 56% male.

Each student is equipped with an TPW water bottle, binoculars, and a Young Explorer's Guide booklet. The students complete wildlife-viewing activities in the Young Explorer's Guide booklet, assisted by Youth Environmental Mentors, teachers, and a professional guide. Later in this chapter we describe an impact evaluation for the Youth Explorer's program.

Scholarships—The cost of both private and government secondary school education in Tanzania can be restrictive to many rural students, especially girls, for whom education may not be considered a priority. TPW's scholarship program, initiated in 2009, provides selected students with a full 6-year tuition guarantee, enough for them to graduate from secondary school. The program provides scholars with tutoring, an annual scholars retreat at the Noloholo Environmental Center, and year-round mentoring with TPW's Youth Education Officer. In 2017, the program also began supporting university tuition for students who wish to continue on to higher education.

Since inception, TPW has supported 65 scholars, 45% female and 55% male. All scholars are from rural villages in Simanjiro District but they attend secondary schools and universities throughout Tanzania. As of November 2020, 73% of scholars who sat the Form 4 exam passed (n = 37) and 89% of the scholars who sat the Form 6 exam passed (n = 9). Seven scholars continued education through diploma or certificate programs, with six of them completing programs and the seventh currently working for TPW as a Youth Education Program Assistant. Eight scholars currently attend college or university, and five of them are TPW-supported university scholars.

This long-term investment in students' education: (1) instills in students a conservation ethic and understanding that humans are part of our ecosystem, breaking down the human-nature dualism from a young age; (2) supports continuity in education for girls, thereby counteracting deep-seated societal biases which threaten women's empowerment; (3) does not restrict beneficiary students to higher education in an environmental field, ensuring that future leaders in different disciplines have a strong foundation in environmental management; and (4) provides an environmentally non-consumptive alternative to lion spearing as a method of gaining social status for young men.

Successes—The TPW Youth Environmental Education program has been well received by communities, evidenced by the multiple request letters for additional programming by schools throughout Simanjiro and Monduli Districts. The program has expanded from one school in 2009 to 13 schools in 2020, with additional schools requesting Wildlife Clubs and scholarships. Further, the program activities often reach far beyond the active Wildlife Club members by including other students and community members in celebrations and events. TPW's Youth Environmental Education Officer and Assistant are regularly invited to attend school functions throughout Simanjiro District including graduation ceremonies, new building openings, and celebrations. Results from activity-specific surveys and assessments provide evidence that the Youth Environmental Education program is creating sustainable positive outcomes for children and communities.

Challenges—While TPW's Youth Environmental Education program has been largely successful, funding and logistical challenges have slowed its expansion to additional schools and districts. Despite requests from schools for additional programming, funding for the Youth Program is primarily through private donors and small foundations, which rarely have the capacity to support long-term expansion. This also holds true of the scholarship program where TPW commits to supporting scholars through 6 years of secondary school and potentially through university. Lastly, logistical and scheduling challenges restrict the number of environmental camps that can be held at Noloholo Environmental Center each year since camp facilities must also support trainings and events for other programs. Similarly, scheduling vehicle use for student transport can be a challenge since TPW vehicles are also used for other program activities. This limits the number of Young Explorer's Tarangire National Park Trips each year. These limitations necessitate a selection process for students who can attend environmental camps and park trips. The selection by teachers and TPW Youth Program staff is a challenge since they must develop selection criteria that are fair, gender non-discriminatory, and non-political, while also allowing as many students as possible to participate in some activity.

16.4.3 Living in Harmony with Your Natural Surroundings

PAMS Foundation is a nonprofit conservation organization whose mission is to empower the people who protect wildlife and wild places. PAMS has developed an environmental education syllabus called 'Living in Harmony with Your Natural Surroundings,' which is taught at secondary school Malihai Clubs near national parks, game reserves, and community conservation areas in various places across Tanzania, including Babati and Simanjiro Districts in the TE.

The 'Living in Harmony' syllabus includes a set of 16 lessons, each with its own objectives, teaching points, guidance on how to conduct the lesson, and activities. The following subject areas are incorporated into the syllabus:

Lessons 1 & 2—Woodlands and Forests
Lesson 3—Afforestation and Deforestation
Lesson 4—Tree Nursery Planting
Lesson 5—Wildlife Protected Areas
Lesson 6—Energy
Lesson 7—Water
Lesson 8—Soil
Lesson 9—Looking After Crops and Livestock (Human-Wildlife Coexistence)
Lesson 10—Wildlife
Lesson 11—Common Mammals of Tanzania
Lesson 12—Managing Resources for the Future
Lesson 13—Climate Change and Global Warming
Lesson 14—Waste Management
Lesson 15—Tourism
Lesson 16—Sports and Outreach Programs

PAMS employs two education officers, one for Babati and one for Simanjiro. The lessons are taught during after-school Malihai Clubs, whereby children can decide each week whether they want to attend the wildlife club, or another club. Each school has a lesson once per week during the after-school session. PAMS' education officers initially implement the lessons with a teacher from the school, with the goal that over time the teacher is capacitated to take over and teach the lessons on their own. During some sessions, the teaching syllabus is used. At other times, the students attend to nurseries where they grow seedlings to plant around their schools during a future session.

After they finish the course, all students receive a certificate of accomplishment. They also plant a tree at the beginning of the course. The tree serves as a permanent remembrance of their participation in the 'Living in Harmony with Your Natural Surroundings' program and their own role in helping to improve the school environment.

To complement the school-based environmental education program, PAMS brings approximately 10 participating children at a time on trips to Tarangire and Lake Manyara national parks to experience wildlife, where the wildlife and the people viewing the wildlife are relaxed and at peace. As noted previously, this is a very different experience compared to how they may experience these creatures outside the parks, when it is often associated with conflict. These park visits also enable the children to better understand tourism and the role tourists play in sustaining Tanzania's economy and environment, and the role of national parks in general. PAMS also organizes Fun Days where the children do activities such as sports, plays, art, and school clean-ups, with conservation messages combined with fun.

Initiating and Expanding the Program—PAMS' Living in Harmony EE program was first conceived and implemented in the Ruvuma region of southern Tanzania, where it is currently active in 16 secondary schools. The program was so successful there that PAMS then expanded to secondary schools near Tarangire and Lake Manyara national parks.

PAMS began the EE program in secondary schools in Babati District in 2015. Meetings were held with the Burunge Wildlife Management Area authorities and Babati District Education Officers for permission to initiate the program in three secondary schools. In 2017, an additional secondary school was included. A fifth school was added when a teacher at one of the participating schools was transferred to another school and helped establish the program in the new school. Overall, since its inception in the TE the program has grown to five secondary schools in Babati and has reached over 1000 children in the district (Table 16.1).

The EE program in Simanjiro District was first initiated at three secondary schools in 2017 and has reached several hundred children. PAMS is also partnering with Wild Nature Institute to implement the Living in Harmony program in Monduli District at three secondary schools, reaching hundreds more children.

Successes—Demand for ‘Living in Harmony with Your Natural Surroundings’ is high and very popular with the students.

Challenges—Initial contacts with the schools was difficult because the program was new.

16.5 Community Engagement Through Creative Media

Media can help promote a conservation ethic by sharing conservation-based messages on the radio and television in creative ways that can be easily heard and embraced. As an example, Wild Nature Institute has engaged in media efforts to promote giraffe conservation. The Institute produced Swahili-language videobooks of the four Celebrating Africa’s Giants children’s books that were aired on TBC1 Safari Television Channel two times per week for 4 weeks, in partnership with PAMS Foundation. They produced a giraffe conservation hip hop song and music video called “Okoa Twiga” (“Save Giraffes”) that played on television, radio, and long-distance buses across Tanzania, reaching millions of Tanzanians. The singer-songwriter Shubert Mwarabu of Music for Conservation conducted interviews on television and radio to promote the song and the CAG program. The Institute also commissioned and produced a gospel song focused on giraffe conservation, called “Tuwatunze Twiga” (“We Protect Giraffes”), and in partnership with PAMS Foundation arranged for the song to play on Clouds FM radio in northern Tanzania, reaching an estimated 9.5 million people in Arusha, Moshi, Manyara, Singida, Mara, and Tunduru districts. The section below describes an assessment that was conducted to determine the reach and impact of the gospel song in three communities in the Tarangire Ecosystem.

16.6 Case Studies of Social Science Assessments

In this section we profile three case studies in the Tarangire Ecosystem that scientifically measured the impacts of environmental interventions: a classroom giraffe education program, a youth explorers program taking schoolchildren into Tarangire National Park, and a giraffe conservation themed gospel song played on the radio. We describe the lessons learned from each of the evaluations and how these lessons can guide the way forward for more effective interventions.

16.6.1 Giraffe Education

The goals of Celebrating Africa's Giants' giraffe-themed education program are to guide students in learning about giraffe biology, to instill a sense of pride in giraffes as the national animal, and to understand what actions can be taken to protect the species. To determine how the intervention (i.e. giraffe-themed program) is reaching its primary goals and how it may be adapted to improve success, evaluation of its characteristics, components, and outcomes is essential. As part of the larger program evaluation, the Wild Nature Institute in partnership with The Living Desert carried out basic qualitative close-ended surveys to assess outcomes of the educational program in participating students. Outcomes are the short-term changes in program participants that result directly from the program. In this case, the survey investigates students' knowledge and opinions related to giraffes and begins to assess whether the CAG program influenced their cognition over both short- and long-term periods. Students completed questionnaires before and immediately following the program. This assessment sought to analyze the immediate and short-term effects of the CAG giraffe program on students' basic knowledge and opinions of giraffes and their conservation.

A six-question survey, hereafter referred to as the questionnaire, was used to collect student responses from 24 pre- and 74 post-CAG program students. The following conclusions were gleaned from this preliminary evaluation:

- Knowledge of the giraffe as the national animal and personal importance of saving the species are high-scoring and remain generally unchanged from pre- to post-questionnaire responses.
- After the CAG program 50% of students correctly identified the state of wild giraffe populations. This is double the percentage of students who selected this response option in the pre-CAG survey, suggesting that the CAG program increases students' knowledge about the declining state of wild populations. The number of students leaving the program with this knowledge could be further maximized.
- Less than a third of post-CAG respondents correctly identified at least one of the two major reasons for declines in giraffe populations, indicating that more emphasis can be given to this part of the lesson.
- A majority of pre- and post-CAG students express that saving giraffes is important to them. An interest in the species even prior to completing the program is

an encouraging representation of the willingness of students to conserve the species.

- Students most frequently identify ‘telling others that giraffes are endangered and need help’ as an action they can take to help save the species. However, only about half of either pre- or post-program participants select this response.

16.6.2 National Park Visits

In 2020, TPW facilitated four Young Explorers trips with 120 students. During the week after each trip, the TPW Youth Environmental Education Officer administered an evaluation to the students to better understand their reflections from the trip and get recommendations to improve future trips. On average, the students gave the entire experience a quantitative rating of 4.79 on a 1–5 Likert scale (Likert 1932) with 5 representing a “very positive” experience (n = 95). TPW solicited qualitative feedback in the evaluation via semi-structured questions to reduce response bias and allow students to express their reflections in their own words. Common responses to a question on the students’ favorite aspect of the trip included:

- Learning facts about elephants like gestation period and seeing elephant calves
- Seeing the baobab trees and learning how they support different animals
- Seeing wild animals in person like cheetah, elephant, giraffe, and buffalo

When asked what they would tell their families and friends about the trip, common responses included:

- Wanting to go back to Tarangire National Park or other protected areas
- Seeing large herds of elephants along the Tarangire River
- Seeing a cheetah and wanting to go again to see lion and leopard

Most negative qualitative feedback concerned the rain, the poor road conditions, and the limited time in the park so that they were not able to see lions. Suggestions for improvement included visiting the park during the dry season when the roads are in better condition and offering more park trips so that more students can participate.

Six months after the trips, the TPW Youth Environmental Education Officer returned to the schools to conduct a post-hoc survey of students who visited Tarangire National Park. The purpose of this survey was to gauge the extent to which students retained lasting memories of the experience and how their views of protected area conservation and wildlife had developed. The students gave the experience an average quantitative rating of 4.80 on a 1–5 Likert scale with five representing a “very positive” experience (n = 117; this survey included more students than participated in the original survey). This suggests that memories of the experience remained just as positive 6 months after the trip as they were immediately upon return. Common responses to a question on what the students remember most from the trip included:

- Seeing the Tarangire River
- Seeing animals cool their bodies with mud at Silale swamp
- Seeing a cheetah

When asked if they would want to visit Tarangire National Park (or another park) again, and why or why not, common responses included:

- Wanting to go back to see lion and leopard
- Wanting to go to other protected areas like Serengeti and Ngorongoro to see more lions
- Wanting to visit other national parks to see different environments in Tanzania

Interestingly, there was little negative qualitative feedback reported in the post-hoc survey. When asked about their least favorite part of the trip, some students mentioned the poor road conditions and not seeing a lion, but 46% of the respondents said they had no negative feedback. This suggests that negative memories from the trip faded over time. Students generally ranked the trip as equally positive immediately afterwards and 6 months later, yet many students' negative feedback about rain and road conditions declined over time, providing evidence that the benefits of youth programming are sustainable even if there are logistical challenges during implementation. Youth participants in the program seemed to retain the positive memories of the Young Explorers trips more than they remembered the negative aspects.

16.6.3 "Tuwatunze Twiga" Gospel Song

Helping improve local peoples' positive perceptions of giraffes is key to ensuring improved conservation efforts for this endangered species. One recently implemented outreach innovation was the production of a gospel song in the Tarangire Ecosystem urging listeners to recall the giraffe's beauty and protect the species from poaching. The Wild Nature Institute produced the song and, together with The Living Desert, assessed how influential the song may have been in three communities where the song was aired on the radio (*sensu* Veríssimo et al. 2018). Details of the assessment can be found in Ocañas et al. (2020), and methods, results, and key take-home findings are summarized below.

In 2018 the Wild Nature Institute commissioned the Ngorongoro Hosea Kwaya Gospel Choir to write and record a 1-min song, "Tuwatunze Twiga," about the unique and beautiful characteristics of giraffes and urging people not to poach them. The song also emphasized that God created giraffes and we as humans are responsible for protecting them. "Tuwatunze Twiga" played on a local radio station once daily during the morning news, over the course of 2 months in December 2018 and January 2019.

The primary objective of the intervention (i.e. educational song) was to engage local peoples' interest in giraffe protection by exposing them to the song and its messages. The message was intended to influence listeners' perceptions about giraffes and the poaching that is contributing to local giraffe population declines. Short surveys of community members were conducted before and after broadcast of the song on the radio. The survey was designed to document people's baseline knowledge and perceptions of giraffes and quantify whether they were influenced by "Tuwatunze Twiga". The survey questions asked how residents liked giraffes, whether they perceived poaching as a threat, and whether they thought protection of giraffes was necessary. The questions evaluated whether any changes in perceptions from pre- to post-song sampling may have been influenced by the song. Survey results also captured an approximate idea of the percentage of people who heard the song.

In total, the pre-song and post-song intercept survey was administered to 237 and 240 people, respectively, with between 78 and 80 respondents from each of three focal towns in the TE—Makuyuni, Kigongoni, and Mto wa mbu. Surveys were conducted approximately 1 month before the song began playing (November 2018) and in the month after the song ceased playing (February 2019). A systematic probability sampling method was used to select respondents (Moring 2017). The surveyor stood in an active area of the focal community and approached every third passerby to request that they complete a 2-min survey. If multiple people were in a group, one person was randomly chosen to be asked to participate. If that person said no, the rest of the group was skipped.

The pre-song survey included five items: four were Likert-type items presented as statements with response options falling along a five-point scale ranging from 'strongly disagree' to 'no opinion' to 'strongly agree.' One open-ended question asked respondents to explain their thoughts about people working together to protect the natural resources of Tanzania, including giraffes. The post-song survey was administered after the song had been playing for 8 weeks, and included the same five questions as the pre-song survey but added a sixth asking if the respondent had heard "Tuwatunze Twiga." Likert-type responses were converted from text to numerical values where 'strongly disagree' = 1, 'no opinion' = 3, and 'strongly agree' = 5. These numerical values were used to calculate means or other values for statistical analyses. Kruskal-Wallis chi-square tests were conducted to determine differences between respondent groups. The two pairs of groups compared included all pre-song respondents versus all post-song respondents, and all post-song respondents who heard the song versus all post-song respondents who did not. It should be noted that the sample of respondents was not representative of the entire population (town) and therefore results are not generalizable. Below are the key take-aways:

- Thirty percent of post-survey respondents heard the "Tuwatunze Twiga" song. Impacts of the song might have been easier to detect if a larger portion of the sample had been exposed.
- Nearly all respondents agreed that giraffes are beautiful, with 89% agreement in the pre-song survey and 97% agreement in the post-song survey (a significant increase).
- Approximately 70% of respondents agreed that giraffes are threatened by poaching.

- People who have heard "Tuwatunze Twiga" were significantly more likely to agree that God tells some people that we should protect giraffes.
- There was significant variation among communities in their responses to the survey questions before and after the song.

Although detecting changes caused by exposure to "Tuwatunze Twiga" was challenging, results do reveal insights that can be used to design and inform future such interventions. First, clearly residents sampled believed giraffes are beautiful. This does not necessarily correlate to support for conservation of the species, but it would likely not facilitate conservation support if residents did not believe the species was beautiful (Knight 2008). Second, because only 59–78% of respondents agreed with the statement that giraffes and other wildlife are threatened by poachers, there may be an opportunity to increase public knowledge of these threats. Third, because 47–70% of respondents agreed that God tells people we should protect giraffes, such religious beliefs should be leveraged in other settings and messaging. It is likely the song appealed to people's pre-existing worldview (Stern et al. 1985; Stern et al. 1999) and reinforced the idea that giraffes are God's creatures and should be protected (Oreg and Katz-Gerro 2006).

Lastly, the qualitative responses indicate that most respondents seemed supportive of natural resource and giraffe protection efforts. Many respondents recognized the importance of protection efforts and some even suggested that these efforts should be better funded or improved. Therefore, community members would likely be supportive of future endeavors to increase protection of wildlife and other natural resources.

Members of the public can be responsive to musical messages and continued popularization of the song could become self-perpetuating if it becomes more prominently part of the culture. In the future, it may be additionally helpful to include a brief spoken one to two sentence campaign message at the end of the song to make the message even more explicit. The song also has potential to be used in other creative performance ways (e.g., live choirs or dances) in a more comprehensive education-entertainment intervention (Veríssimo et al. 2018) which may amplify the message it carries.

16.7 Conclusions

With carefully crafted messaging, active engagement with students and communities through participatory experiences, and rigorous assessments of impacts on knowledge, attitudes, and behaviors, EE programs can result in positive attitudes, a growth in knowledge and quality education, and investment in the future that improves conservation of nature. These EE programs take hard work and long-term commitment, which are often difficult due to funding and logistical constraints. However, the implementation and success of EE programs must be a priority because they complement and support other conservation efforts such as reducing human-wildlife conflicts using technical solutions, allocating land uses with a growing human and livestock population, and realigning community values and priorities.

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Part V
Synthesis

Chapter 17
Towards Human-Wildlife Coexistence in the Tarangire Ecosystem

Monica L. Bond, Derek E. Lee, and Christian Kiffner

Abstract In this final chapter we summarize the contributions to the book "Tarangire: Human-Wildlife Coexistence in a Fragmented Ecosystem." The 15 contributed chapters analyzed conservation and livelihoods issues from anthropocentric perspectives and from the wildlife lens, and explored aspects of human-wildlife interactions in the Tarangire Ecosystem (TE). With differing topics and perspectives, each chapter contributes in its own way to our understanding of key issues and challenges in the TE. We synthesize these multi-dimensional knowledge types according to complexity features that are characteristic of coupled social-ecological systems: non-linearity and thresholds; reciprocal interactions and feedback loops; time lags and legacy effects; resilience; heterogeneity; embedment and telecoupling; vulnerability; and surprises. Several examples highlighted in the book illustrate that planning for and managing human-wildlife coexistence remains a major and complex governance challenge. Learning from mistakes and successes of the past may offer guidance for more effective ways towards coexistence between people in wildlife in the TE and elsewhere. While our place-based analysis highlights that stakeholders and scholars differ widely in their opinions about *what* the specific solutions should be, there is overwhelming consensus about *how* such solutions

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should be planned and implemented: by employing interdisciplinary, collaborative, and equitable approaches that ensure that both people and wildlife can thrive together in the TE.

Keywords Complex socio-ecological systems · Stakeholder participation · Conservation conflict · Conservation solutions · EcoHealth

17.1 Synthesizing the Complexity of the Tarangire Ecosystem

No place on Earth is untouched by the imprint of humanity (Ellis et al. 2021), from the deepest depths of the ocean (Chiba et al. 2018) to the highest mountain tops (Napper et al. 2020). As we stated in the introduction to this book, humans interact with wild animals wherever we go, and the long history of these interactions has shaped human cultures, communities of organisms, ecosystem functioning, and evolution of both humans and wildlife. For as long as humans have existed, we have profoundly influenced and were profoundly influenced by wildlife.

Situated in the heart of East Africa, where anatomically modern humans likely evolved and supporting a remarkable diversity of large mammal species, the Tarangire Ecosystem (TE) is an excellent example of a dynamic social-ecological system, with waves of human occupation and exploitation of natural resources beginning tens of thousands of years ago and continuing to this day. The current landscape of the TE comprises small and large towns and scattered temporary homesteads, two famous national parks, game-controlled areas, a game reserve, a forest reserve, a ranch conservancy, and several community-based conservation initiatives including three Wildlife Management Areas as well as Simanjiro Conservation Easements and Certificates of Customary Rights of Occupancy, all embedded within an ecologically heterogeneous landscape. This system poses great challenges as well as great examples of and opportunities for coexistence between people and wildlife. As this book demonstrates, for several decades anthropologists have collected information about attitudes and perceptions of (mostly Maasai) people in the TE about conservation; wildlife scientists have monitored a diverse suite of wildlife species ranging from African savanna elephants (*Loxodonta africana*) and Masai giraffes (*Giraffa camelopardalis tippelskirchi*) to lions (*Panthera leo*) and antelopes; and NGOs have implemented community education programs and focused attention on resolving challenges and providing opportunities for both humans and wildlife. Insights gleaned from these studies of humans and wildlife and their interactions in the TE may have wide-reaching applications for addressing conflict in this and other coupled natural-human landscapes (Liu et al. 2007).

Conservation conflicts occur within the context of the social and cultural histories of the different people involved. A key driver of conflict is the different perceptions of reality stemming from those different histories. This is relevant to what has been termed the "Rashomon effect" (Levin et al. 2021), derived from a 1950 film in which a samurai is murdered, and four different witnesses provide four different equally believable yet contradictory stories of the murder. Levin et al. (2021) define the Rashomon effect in conservation as "the existence of multiple plausible but conflicting perceptions about the causes and underlying consequences of an urgent conservation challenge." To wit, this is not to say that there are different *truths* (which is not possible), but rather that there are different *perceptions* of the truth—or different and equally valuable ways of knowing about the external world—based on objectively collected mechanistic data (i.e., ecology), human experiential context (i.e., sociology), and human narrative (i.e., history). These various knowledge types each provide critically important components to help address complex conservation and human livelihood challenges.

In this book, we collated different knowledge types from a broad range of scientists and advocates who have worked with humans and wildlife in the TE over relatively long periods of time. Our intention is to provide interdisciplinary analyses that link three fundamental dimensions in the TE: the human perspective/system, the wildlife perspective/system, and the human-wildlife interface. The different knowledge types can and should be integrated to craft innovative and effective solutions to complex problems (Liu et al. 2007; Levin et al. 2021). Within the field of conservation science, researchers hail from various cultures, including social and natural sciences, each of us with our own unique background, training, and interests. The contributors to this book certainly fit this mold. Consequently, we offer our stories through not only objective observations and data but also through the subjective lenses of our own personal experiences and perceptions. There has, however, traditionally been a separation between the natural and social sciences (Liu et al. 2007). The primary challenge is in coming to terms with our traditional differences, embracing and respecting the plurality of opinions, integrating the natural and social sciences to better understand how coupled human-natural systems function, and moving forward in an interdisciplinary manner to achieve real success for both human livelihoods and conservation.

In this final chapter of the book we first summarize the previous chapters. With differing topics and different viewpoints and perspectives, each chapter contributes in its own way to our understanding of key issues and challenges in the TE (Fig. 17.1). Subsequently, we synthesize these multi-dimensional knowledge types according to complexity features pertinent to coupled social-ecological systems. By employing a systems-thinking approach, we hope to not only provide a better understanding of past and current conservation and livelihood challenges in the TE but also to identify possible pathways that enable both people and wildlife to thrive together.

Fig. 17.1 Word cloud in the shape of a tree based on the titles and abstracts of Chaps. 1–16 . The word cloud was generated with www.wordclouds.com

17.2 The Human Dimension

In recent centuries, the Tarangire Ecosystem has been inhabited largely by the Maasai people. Maasai are agro-pastoralists who rely heavily on livestock for economic, cultural, and spiritual reasons. The first two chapters of the Human Dimension section describe the origins and history of conflicts that arose from the Tanzanian government's 'statist' tactic of establishing protected area boundaries that excluded use by local people, especially the Maasai agro-pastoralists. As such, the focus is largely on human-human conflict. The third chapter in this section defines the conceptualization of wellbeing based on interviews with a collection of Maasai people from the Simanjiro area. The last two chapters describe two alternative approaches to community conservation that benefit both local pastoralists and wildlife in the area east of Tarangire National Park: local collective stewardship of rangelands through Certificates of Customary Rights of Occupancy, and a community-based Wildlife Management Area. Below we review the main points from each chapter.

In Chap. 2, Jevgeniy Bluwstein argues that ideas about people and nature have changed over time, yet colonial legacies regarding conservation and management have persisted in the TE. These legacies involve the separation of humans and wildlife. This antagonism between a state-centric paradigm of land control versus use by local Maasai pastoralists underpins much of the human-wildlife conflict in the region. Furthermore, Bluwstein contends that conservation in the TE has recently expanded beyond (contested) protected area boundaries, but the growing reliance on market-based approaches to protect biodiversity in these expanded areas while providing economic benefits may not be sustainable or fully address the needs of local people. This contested past and conflicted present points to the need for 'convivial' conservation efforts. Bluwstein outlines a set of governing principles for convivial conservation, including (i) democratic (local) engagement in conservation decisions; (ii) replacing 'protected areas' with 'promoted areas' without set boundaries; (iii) transitioning away from a market-based approach of payment for conservation and conventional short-term tourism; and (iv) forging a different type of relationship with the Tanzanian state.

Jim Igoe (Chap. 3) urges readers to consider that if conservation is viewed from the perspective of local Maasai pastoralists whose people have been present in and used the landscape for generations, then formally protected areas such as Tarangire National Park are, to them, *part* of the fragmentation of the ecosystem (along with farming, mining, and human settlements) rather than the only areas *free* of fragmentation. With state-imposed zoning and boundaries separating humans (other than tourists) from wildlife, the local people are denied access to resources they argue are their heritage. Igoe discusses the need to recognize compatibilities between conservation and pastoralism, and to support approaches such as collective, local-level resource management and stewardship.

In Chap. 4, J. Terrence McCabe and Emily Woodhouse delve into the conceptualization of wellbeing among the Maasai of Simanjiro. The results of interviews underscore the differences in attitudes between younger and older men and between men and women. Younger men wanted fewer cattle, wives, and children than older men, and younger men also considered education of children—including girls—more important than did the older men (but women were overall more adamant about the importance of girls' education than men). Women focused on the importance of livestock and cultivation not from a wealth perspective, but from the perspective of being able to provide milk and food for children, and a source of income for school fees and health care for children as well. Women valued children because they were important for their own social lives as women become dependent upon their children—especially sons. Women also believed that harmony within the household was critical for their wellbeing. Communal grazing lands were considered essential by all male respondents, and men and women all expressed distrust of the federal government, private investors, and some NGOs. Some of the respondents expressed frustration that they are encouraged to depend upon livestock and tourism and abandon cultivation, which poses a key conservation challenge. The romanticized image of the Maasai as nomadic pastoralists living in harmony with wildlife is not always accurate and this chapter clearly depicts the changing values and livelihoods among contemporary Maasai in the ecosystem.

In Chap. 5, Peadar Brehony, Alais Morindat, and Makko Sinandei describe an innovative program to secure land tenure rights, known as Certificates of Customary Rights of Occupancy (CCROs). This is a unique program developed in the Simanjiro region, in response to changes in traditional systems of land and livestock management among the Maasai people. CCROs cement traditional land use and governance practices with legal requirements for participatory land-use planning. The land-use plans are then overseen by village councils. This model creates a mixed-use coexistence landscape and constitutes a payment for ecosystem services model that seems to work for both people and wildlife alike.

In Chap. 6, Justin Raycraft reports high levels of community support for the Randilen Wildlife Management Area (WMA) in 2020. Furthermore, most interview respondents reported liking the WMA more than they did 5 years before. Strikingly, the vast majority of respondents stated that they trusted WMA authorities to act in their interests, and that they felt their community was included in WMA governance and management. Upwards of 90% of respondents viewed the WMA as a success and that it represented community-based conservation rather than a fortress conservation model. These results were based on structured interviews with a large, representative, randomly selected sample of men and women and quantitative analyses of the data. Perceptions of the benefits of a Wildlife Management Area adjacent to Tarangire National Park appeared to have changed over time, from negative to positive. This chapter provides strong arguments for quantitative analysis of conservation attitudes and suggests that attitudes towards conservation entities can change if people feel that they are involved in decision making and benefit from conservation efforts.

17.3 The Wildlife Dimension

The Tarangire Ecosystem supports one of the densest populations of African savanna elephants on the continent, one of Tanzania's most abundant populations of Masai giraffes, one of only a handful of long-distance migrations of wildebeests (*Connochaetes taurinus*) and zebras (*Equus quagga*) remaining in Africa, threatened yet ecologically still vital populations of carnivores including lions, cheetahs (*Acinonyx jubatus*), and wild dogs (*Lycaon pictus*), and a rich diversity of ungulates that shape the landscape's vegetation and provide food for predators and scavengers. The second section of the book addresses the wildlife dimension of the TE with insights gleaned from several long-term research projects whose breadth and depth rival research projects in Tarangire's more famous neighbor, the Serengeti-Ngorongoro Ecosystem. First, baselines for wildlife populations are discussed, followed by focused chapters on a suite of eight commonly detected ungulates, giraffes, elephants, and large carnivores.

Herbert H. T. Prins and Joost de Jong's Chap. 7 focuses on the dynamic ecohistory of the TE, the shifting baseline syndrome, and whether the system has passed beyond a threshold where the natural state cannot be restored. The ecohistory is placed in context of a discussion on shifting baselines, whereby a previous reference point to measure change in a system is itself already a change from an even earlier point in time. The baseline of an ecosystem should represent its natural state, but this is difficult to establish in East Africa where humans have lived for hundreds of thousands of years. Prins and de Jong propose a reference baseline of 1935 for wildlife in the TE, when wild animal populations were likely at a zenith after recovering from rinderpest, human numbers were low, and before widespread hunting, poaching, and habitat loss began.

Ungulates are ecologically and economically significant in the TE. In Chap. 8, Monica L. Bond, Christian Kiffner, and Derek E. Lee review and discuss historical and current data on population trends of eight species of ungulate: zebra, eastern white-bearded wildebeest (*C. t. albojubatus*), common eland (*Taurotragus oryx*), common waterbuck (*Kobus ellipsiprymnus*), impala (*Aepyceros melampus*), Grant's gazelle (*Nanger granti*), Thomson's gazelle (*Eudorcas thomsonii*), and Kirk's dik-dik (*Madoqua kirkii*). The chapter flags some problems with comparing data derived using different methodologies. Despite this, recent monitoring studies in Tarangire National Park, Manyara Ranch, Wildlife Management Areas bordering Tarangire and Lake Manyara national parks, and the Simanjiro Conservation Easements suggest relatively stable ungulate populations in these protected areas over the last decade. Overall the available evidence indicates that ungulate populations in the TE are probably well below the 1935 baseline suggested by Prins and de Jong, and remaining ungulate migration routes are threatened, but populations have apparently stabilized recently. On a positive note, the chapter provides evidence that additional collaborative conservation efforts (particularly the establishment of Wildlife Management Areas) contributed to localized wildlife population increases.

The Masai giraffe is the national animal of Tanzania and a globally iconic megaherbivore. Chapter 9 describes the population structure, social structure, and demography of Masai giraffes in the TE based on almost a decade of research. The long-term study was designed to understand the influence of humans on giraffe demography and social relationships in a coupled human-natural landscape. Using photographic identification to monitor individual giraffes over time, Derek E. Lee and Monica L. Bond quantified demography (survival, reproduction, and movements) of subpopulations defined by either administrative boundaries (national parks, Manyara Ranch, Wildlife Management Areas) or by social relationships among the giraffes. Results revealed that natural factors such as predation and season as well as humans influence survival, reproduction, and sociality of giraffes in complex ways. For example, female giraffes have lower survival if they live near to towns that are densely populated by people (some who poach giraffes for meat) and surrounded by farms. On the other hand, although female giraffes have weaker and more exclusive relationships with each other near Maasai bomas, they aggregate near bomas to reduce natural predation risk on their calves. Thus, the presence of pastoralists appears to be compatible with giraffe population persistence. Volcanic soils in northern Tarangire National Park and Manyara Ranch are good quality habitats for giraffes, with high calf and adult survival despite proximity to people, suggesting that giraffe conservation in the TE could be facilitated by protecting habitats on volcanic soils and maintaining connectivity. The Lake Manyara National Park giraffe subpopulation is isolated and has low calf and adult survival, but the size of the subpopulation has remained stable over many decades. Finally, similar to other ungulates, Wildlife Management Areas show greater giraffe densities as well as improved survival rates, indicating community conservation success. This bodes well for human-giraffe coexistence in the TE.

Charles A. H. and Lara S. Foley's Chap. 10 covers the history of the iconic African savanna elephant population in the TE from the early 1900s until today. The population suffered greatly during the 1970s when ivory poaching soared, which dramatically altered the movements and ranges of the elephants for two decades as they crowded within Tarangire National Park for safety. Government anti-poaching efforts and an international trade ban on ivory in 1989 alleviated much of the poaching and subsequently the elephant population expanded rapidly from 1990 to 2020. During this period of high population growth, the age structure of the population changed substantially, with more older males and females, and ranges expanded into Manyara Ranch, Burunge and Randilen Wildlife Management Areas, and beyond. Long-term individual-based elephant research by the Foleys revealed three subpopulations in the TE based on wet season ranging and association patterns. In the 1990s and early 2000s, the southern subpopulation had a significantly lower infant-to-mother ratio and congregated in significantly larger aggregations than the northern subpopulation, which was attributed to higher levels of human-induced stress from continuous, albeit low, levels of poaching in the south. However, since 2002 evidence suggests that the southern subpopulation is reverting to traditional grouping patterns, possibly due to a reduction in poaching. The increase in elephant movements outside Tarangire National Park into adjacent community lands has resulted in increased conflicts with people—especially crop raiding—thus necessitating the implementation of mitigation/conflict-reduction measures. With the amelioration of poaching and population expansion, Tarangire's elephants represent a true conservation success story, but the consequent increase in conflicts points to the critical importance of cooperation between wildlife authorities and local communities to ensure the safety and wellbeing of both humans and elephants.

Large carnivores evoke strong emotions among humans, being admired and feared alike. These top predators exert influential effects on their prey and as such they shape ecosystem processes. In Chap. 11, the last of the wildlife dimension chapters, Christian Kiffner, Charles A. H. and Lara S. Foley, Robert A. Montgomery, and Bernard M. Kissui synthesize available data on distribution and abundance of six species of large carnivores across the conservation gradient in the TE: lion, spotted hyena (*Crocuta crocuta*), striped hyena (*Hyena hyena*), leopard (*Panthera pardus*), cheetah, and wild dog. All but the cheetah and wild dog were widely detected throughout the ecosystem, and densities of these rarest of carnivores were relatively low. Spotted hyenas reached the highest densities of all the species, followed by leopards and striped hyenas. Tarangire National Park supported the highest densities of lions, followed by Makame Wildlife Management Area, with other protected areas supporting low densities of the largest carnivore species. Interestingly, species-specific densities were not strongly or clearly correlated with conservation status of an area. Most carnivore species moved outside Tarangire National Park during the rainy season, bringing them into greater contact with humans. Essentially, these species follow their prey, occupying areas with higher prey density and catchability but being less frequently detected in human-dominated areas. The lion population is particularly dependent on conservation efforts. The spotted hyena stands out from the other species in that occupancy of areas is positively associated with human population densities—and is responsible for most livestock predation events in the TE. The chapter concludes with two key elements that must be addressed to achieve human-carnivore coexistence: negative interactions with large carnivores such as livestock depredation and subsequent retaliatory killing of carnivores must be reduced using sustainable, cost-effective, and socially accepted non-lethal methods, and conservation measures must protect essential habitats for prey species.

17.4 Human-Wildlife Interactions

Exploring patterns, causes, and consequences of interactions between humans and wildlife can help guide appropriate policy and management decisions that consider the needs of both. The last section of the book focuses on a variety of issues dealing with the human-wildlife interface, from wildlife movements through human-dominated landscapes and coexistence between people and elephants as well as people and large carnivores, to strategies for using a results-based system of payments for ecosystem services and community education to inspire conservation ethics and promote effective solutions to coexistence.

A growing human population and associated land-use changes in the TE contribute to a lack of habitat connectivity which can hamper dispersal, gene flow, and the ability of wildlife populations to respond to climate change. In Chap. 12, George G. Lohay, Jason Riggio, Alex L. Lobora, Bernard M. Kissui, and Thomas A. Morrison describe movement patterns of wildlife among key habitat areas, from the core Tarangire and Lake Manyara national parks and Manyara Ranch to the Wildlife Management Areas, Game Controlled Areas, and Game Reserves. Data from telemetry, photo mark-recapture identification, aerial and ground count surveys, and DNA analyses can elucidate past connectivity, recently used movement paths, and movement probabilities. Early descriptions of wildlife movements across the TE indicate a vast wet season dispersal of ungulates in all directions from the dry season ranges in Tarangire and Lake Manyara national parks. However, by the 1980s movement routes west of Tarangire were largely blocked by agriculture, and linkages to the north and northwest were diminishing rapidly. By the 2000s only seven wildlife corridors remained in the TE, with the majority in critical danger of being lost. Recent data show large mammals including elephants, giraffes, wildebeests, and lions still make long-distance movements throughout the ecosystem and functional and genetic connectivity remains, but Lake Manyara National Park exhibits troubling signs of isolation. Wildlife populations are threatened by continued habitat loss, poaching, vehicle collisions, and conflicts with humans. Thoughtful, science-based land-use planning to protect safe movement corridors for wildlife—planning that is driven by community conservation efforts—could maintain connectivity and sustain these wildlife populations well into the future.

Elephants are one of the primary 'conflict' animals in the TE, given their propensity to move outside of protected areas and the danger posed by their massive size. Typically, studies of the human-elephant interface focus on farmers, as elephants often raid crops, but these pachyderms also inhabit rangelands utilized by pastoralists. In Chap. 13, John Kioko, Sophie Moore, Kathleen Moshofsky, Anne Nonnamaker, Blaise Ebanietti, Katharine Thompson, and Christian Kiffner characterize the pastoralist-elephant interface in Manyara Ranch. The authors interviewed cattle herders in Manyara Ranch about their perceptions of elephants, and observed elephant reactions to sound playbacks of humans, cattle, and other wildlife species. The vast majority of herders (nearly 90%) supported the presence of elephants in the ranch and generally perceived elephants as a minor threat to their cattle and themselves, compared to other wildlife species such as lions, buffalos, and hyenas. Elephants—especially groups with calves—reacted most to sounds of herders and domestic dogs, typically fleeing into nearby closed habitats. The relatively positive herder perceptions of elephants suggest that interactions with cattle are not based in conflict, and indeed herders often allowed their cattle to intermix with elephants (although they personally kept a distance), demonstrating potential for coexistence in rangelands. Overall, focus should remain on mitigating crop raiding by elephants.

Large carnivores are another major 'conflict' taxa in the TE. In Chap. 14, Bernard M. Kissui, Elvis L. Kisimir, Laly L. Lichtenfeld, Elizabeth M. Naro, Robert A. Montgomery, and Christian Kiffner summarize information on incidences of human-carnivore interactions in the TE based on surveys and reports dated back to 1943. Data included type of interaction (attack on human or livestock), carnivore species involved, where the interaction occurred (which village, in a boma, in the bush), and the human activities at the time of the interaction. The number of reported interactions increased from the early 1980s to the 2000s, possibly due to increase of the human population and decrease in natural prey populations. Nearly all large carnivore attacks on humans were on males, especially younger males ($\leq$30 years of age), and most were by lions. Older people were particularly susceptible to attacks by hyenas which most often occurred at night in the home. Carnivore attacks on humans were concentrated in just a few villages over the 66-year timespan, and most people were attacked during retaliatory lion or leopard hunts during the day, and to a lesser extent when livestock herding in the field—these two activities are likely to pose the highest risks to humans. In contrast, the majority of livestock predation events were caused by hyenas, and to a lesser extent by lions and leopards. Records indicate decreasing livestock attack events from 2004 to 2017, for various possible reasons. People's perceptions of the frequency of human-carnivore conflicts were much greater than observed levels of conflicts, possibly indicating the fear of extreme damage events (e.g. loss of livestock and associated economic and social losses; loss of human life) and deep-rooted conflicts and mistrust between pastoralists and management authorities. Key behavioral co-adaptations in carnivores, such as increased nocturnal behavior in human-dominated areas, as well as in humans, for example adoption of fortified bomas and improved livestock guarding methods, can facilitate coexistence.

In Chap. 15, Marc Baker, St. John Anderson, and Christian Kiffner profile a results-based model of payments for ecosystem services—the purchase of verified emission reductions for use as carbon offsets in the REDD+ project of Makame Wildlife Management Area in the southern TE. REDD+ (Reducing Emissions from Deforestation and forest Degradation) is a framework to curb carbon emissions by encouraging communities to preserve forests. Emission reductions from a REDD+ project are issued only after verification, thus climate, human, and biodiversity benefits are completed before the emission reductions are monetized. The Makame Savannah REDD project meets the standards of the Verified Carbon Standard and the Climate Community Biodiversity Alliance and was developed by Carbon Tanzania, a Tanzanian NGO, to protect the WMA's forests from conversion to agriculture. The project's targets for community and biodiversity co-benefits were developed during community workshops and thus were inclusive of the needs of the local people to the extent possible. The targets were also designed to meet global standards, and a monitoring framework was developed to systematically collect, analyze, and report on social and wildlife indicators. This approach follows a business strategy for conservation rather than reliance on donor funding and thus represents a long-term investment that can provide significant economic benefits to rural communities.

In Chap. 16, the last of the human-wildlife interaction chapters, Monica L. Bond, Karakai Barisha, Krissie Clark, Ferdnand D. Chugu, James M. Madeli, Revocatus Magayane, Alejandrina Ocañas, Anna Sustersic, and James Danoff-Burg outline ways to promote positive behavioral changes and foster support among human communities for conservation in the TE through environmental education programs. Behavioral changes that benefit conservation require effective communication that assesses the targeted audience, elicits emotions, activates involvement, and incorporates redundancy of the message. Education programs must include impact evaluations to assess effectiveness at changing knowledge, attitudes, and behaviors. The chapter describes three innovative long-running primary and secondary school education programs operating in the TE: Wild Nature Institute's 'Celebrating Africa's Giants', Tanzania People and Wildlife's 'Youth Environmental Education', and PAMS Foundation's 'Living in Harmony with your Natural Surroundings'. These programs use specially designed curricula and materials relevant to the local area and people, fun hands-on activities such as tree plantings and community events, and visits to Tarangire and Lake Manyara national parks. They have reached thousands of schoolchildren in the TE and have had positive impacts on attitudes towards local wildlife according to impact evaluations. Media such as animated videos on television and songs on the radio are sharing wildlife conservation messages in creative ways. With carefully crafted messaging, active engagement with students and communities through participatory experiences, and rigorous assessments of impacts on knowledge, attitudes, and behaviors, environmental education can result in positive attitudes, a growth in knowledge and quality education, and investment in the future that improves conservation of nature.

17.5 Assessing Sustainability in the Tarangire Ecosystem

Ostrom (2009) noted that resources used by humans are embedded in complex social-ecological systems, composed of subsystems such as resource systems (e.g. rangeland or national park), resource units (grasses, trees, wildlife, water), users (pastoralists, farmers, tourism operators), and governance systems (institutions and laws that govern resource use). These subsystems interact to produce emergent outcomes at the social-ecological systems level. These outcomes can be measured with social and ecological performance measures, but as Ostrom noted, "ecological and social sciences have developed independently and do not combine easily." Indeed, often vastly different frameworks, theories, and models are used by the different disciplines to explain the parts of the complex whole. The various contributions to this book underscore the diversity of narratives and opinions from both social and natural scientists about the origins and history of human-wildlife—and human-human—conflicts in the TE; about the measures proposed and implemented to reduce conflicts and conserve the biodiversity of this ecosystem; and about the (perceived) successes and failures of these measures. Using a common framework enables variables to be identified and quantified to study a particular social-ecological system in an interdisciplinary manner. Such a common framework also enables comparison with similar systems in other places.

17.6 Key Challenges and Opportunities for Human-Wildlife Coexistence in the Tarangire Ecosystem

The chapters of this book highlight numerous interdependencies within social and ecological systems as well as couplings between the two systems. Here we summarize current circumstances and challenges in the TE according to some of the key features of complex systems in the framework outlined by Liu et al. (2007) and Carter et al. (2016). These features include nonlinear dynamics and thresholds, reciprocal feedback loops, time lags, resilience, heterogeneity, embedment and telecoupling, and surprises. Feedback loops, legacy effects, and embedment among social and ecological components are, by definition, fundamental aspects of "coupled" social-ecological systems, whereas understanding thresholds, resilience, heterogeneity, and surprises can provide additional insights into addressing some of the challenges associated with human-wildlife and human-human interactions.

17.6.1 Non-linearity and Thresholds

It is critical to identify when relationships are nonlinear or when there is a threshold of collapse, to understand when conditions may be permanently altered beyond historical conditions. One of the most obvious examples may be the substantially altered mammal species composition in areas that have been subject to human development. While elephants, hippopotamus (*Hippopotamus amphibius*) and other large mammal species reportedly thrived just north of Lake Manyara National Park 50 years ago, these areas are now occupied by irrigated agriculture and settlement (Kiffner et al. 2015b). While some wildlife species persist in these areas, other species only occasionally visit those areas and then typically come into conflict with people. Certainly, some areas in the TE have been lost as wildlife habitat for at least the next few generations (**Prins and de Jong** Chap. 7). In contrast, pastoral areas such as Manyara Ranch and Wildlife Management Areas support mammal species communities that are similar to those observed in adjacent national parks. Thus, mammal community structure seems resilient to some degree of human impact but beyond a threshold of human impact, the mammal community becomes impoverished.

One of the most devastating conservation thresholds is extinction of a species from an ecosystem. Fortunately, the TE supports most native species of larger mammals, yet excessive poaching during the 1970s and 1990s has caused the eradication of black rhinoceros (*Diceros bicornis*) across the entire TE; during earlier times, other species had been lost from the TE already, and it is currently unlikely that any of these species will be restored anytime soon (**Prins and de Jong** Chap. 7).

In the TE, barriers to wildlife and livestock movements may have exceeded thresholds and historical movement patterns may now be fundamentally altered. For example, Bond et al. (Chap. 8) documented changing patterns of use of areas by wildebeests, with more animals in Burunge WMA and Manyara Ranch than in Tarangire National Park during the dry season, and Prins and de Jong (Chap. 7) noted that African buffaloes (*Syncerus caffer*) no longer move between Lake Manyara and Tarangire national parks as they had historically.

Furthermore, connectivity of elephants between the Tarangire Game Reserve and Lake Manyara National Park that was observed in the 1960s (**Foley and Foley** Chap. 10) has been lost today (**Lohay et al.** Chap. 12). Similar isolation of Lake Manyara National Park has likely affected giraffes as well (**Lee and Bond** Chap. 9, **Lohay et al.** Chap. 12). It remains to be seen whether removing recently created anthropogenic barriers and re-establishing connectivity will facilitate historical movement patterns again, but these barriers are probably permanent, at least in the foreseeable future. Likewise, the establishment of protected areas fundamentally altered movement and grazing regimes of agro-pastoralists and their livestock (**Bluwstein** Chap. 2, **Igoe** Chap. 3) and it is currently unlikely that national park policies will be adjusted to reverse this.

17.6.2 Reciprocal Interactions and Feedback Loops

In East Africa, feedbacks between tsetse flies (*Glossina* spp.), bush vegetation, fire, livestock, people, and wildlife fundamentally shape savanna ecosystems (Sinclair et al. 2015). Tsetse flies can transmit *Trypanosoma brucei* parasites that cause Trypanosomiasis (sleeping sickness) in humans and livestock, a fatal disease if left untreated. The flies thrive in areas with thick bush cover (Nnko et al. 2021), and these areas are typically avoided by pastoralists. Flies therefore act as protectors against overgrazing by livestock. More than a century ago, human settlements and cultivation and associated activities such as lighting fires and grazing livestock had reduced bushlands and kept the tsetse fly at bay. Epidemics of rinderpest, smallpox, and cholera in the late 1800s and early 1900s devastated human and livestock populations and enabled bushlands to expand, thus increasing tsetse flies, and colonial campaigns continued to separate people from the tsetse-dominated areas (**Bluwstein** Chap. 2, **Prins and de Jong** Chap. 7).

Dynamics among ungulates, large carnivores, humans, and livestock are also reciprocal and subject to feedback mechanisms. The observed decline in ungulate populations is likely associated with multiple underlying reasons: illegal hunting to satisfy the demand for bushmeat (Kiffner et al. 2015a), habitat loss due to conversion to agriculture (Msoffe et al. 2011b), and restricted access to key resources such as surface water and grass are likely operating in concert (**Bond et al.** Chap. 8). These losses were compounded by the previous large scale culling of zebra and wildebeest populations at the end of the 1990s (Foley and Foley 2014). In turn, the reduction of wild ungulate populations may also be partially responsible for a greater frequency of livestock depredation events by large carnivores (**Kissui et al.** Chap. 14) possibly because some large carnivores now rarely encounter wild prey (Khorozyan et al. 2015).

In contrast, increases in wildlife populations could also mediate the spatial distribution and frequency of human-wildlife interactions. After the TE elephant population was released from severe poaching-related mortality, the population growth and associated spatial expansion into previously unoccupied habitats (**Foley and Foley** Chap. 10) caused increases of human-elephant conflicts (in particular crop raiding by elephants) in many areas of the TE, especially in areas of Burunge and Randilen WMAs and villages bordering Manyara Ranch (**Kioko et al.** Chap. 13). Thus, what can be labelled a success from a conservation perspective may be a serious livelihood issue from an anthropocentric angle.

Another example of reciprocal human-human interactions in the TE is when people moved closer to Tarangire National Park and developed farms out of concerns the park boundaries would be expanded (**McCabe and Woodhouse** Chap. 4). The local communities reacted in response to the federal government's delineation of the national park boundaries by further exacerbating loss of wildlife habitat, which has resulted in ongoing conflict with both the government and wildlife (**Igoe** Chap. 3).

17.6.3 Time Lags and Legacy Effects

Impacts of prior couplings on later conditions are a result of time lags or legacy effects. For example, massive poaching during the 1980s in Lake Manyara National Park (**Prins and de Jong** Chap. 7) may have driven some expansion of bushlands, which still affects the ecology of the park to this day (**Bond et al.** Chap. 8).

The legacy impacts of colonialism included the introduction of rinderpest and smallpox epidemics that killed many local people who had previously kept tsetse flies at bay through their land management activities. The loss of local people resulted in the expansion of bushlands and tsetse flies which then kept pastoralists out of the infested areas. This led to continued separation of people and wildlife in the habitat reserves, as a result of both tsetse flies and government separation policies. Thus, the current tsetse fly distribution and even national park boundaries can be considered legacy effects of past colonialism.

17.6.4 Resilience

Some species are highly resilient to human disturbances, one example being spotted hyenas which are more abundant closer to human settlements (**Kiffner et al.** Chap. 11). Giraffes (**Lee and Bond** Chap. 9) are resilient to low-impact human settlements such as Maasai bomas, but not high-impact areas such as the towns of Mto wa Mbu, Makuyuni, Kibaoni, and others in the TE. Elephants are resilient when poaching is curbed, as evidenced by rapidly rebounding elephant numbers (**Foley and Foley** Chap. 10). Wildlife populations can begin to recover once protected from poaching and released from competition with livestock, as demonstrated by wildlife monitoring efforts in the TE's WMAs (**Lee and Bond** Chap. 9, **Kiffner et al.** Chap. 11, **Baker et al.** Chap. 14).

People in the TE can be highly resilient and survive despite the sometimes harsh environmental conditions such as drought and even though they have been pushed out of historical ranges (**Igoe** Chap. 3). Economic resilience of conservation efforts is important to sustainability for both people and wildlife. CCROs allow resilience because income is not tied to tourism or foreign investment (**Brehony et al.** Chap. 4). The Makame Savannah REDD+ project is an example of a long-term business strategy that is also resilient to the need for tourism dollars. WMAs were considered controversial previously, but operations have also proven to be less dependent upon income from tourism than national parks (Damien Bell, pers. comm). Furthermore, Raycraft (Chap. 6) showed that people's attitudes shifted from earlier distrust towards support of Randilen WMA.

17.6.5 Heterogeneity

Not surprisingly, heterogeneity is a primary feature in complex systems, and can be expressed in many ways, such as the dynamic seasonal distribution of resources and wildlife, differences in people's incomes or use of the land, even the diversity of opinions on a subject.

The dominant driver of vegetation heterogeneity in savanna ecosystems such as Tarangire is rainfall (Lehmann et al. 2011), with wildlife and humans also playing important roles (Msoffe et al. 2011a). Precipitation ultimately determines whether an area is mostly covered by trees, bushes, or grasslands (Lehmann et al. 2011). Savanna ecosystems are inherently dynamic with annual, decadal, and millennial changes in rainfall, along with fire, wildlife, and human actions pushing the system towards or away from a more woody or grass-dominated state (Higgins et al. 2000; Grady and Hoffmann 2012). These factors must be acknowledged and dynamism embraced in this era of rapid climate change.

Spatial heterogeneity in soil nutrient concentrations, along with protected areas that primarily cover only dry season ranges rather than the year-round requirements of migratory wildlife, is possibly the key underlying reason for most human-wildlife interactions in the TE. Further, heterogeneity in large carnivore behaviors influences human-carnivore interactions: most livestock depredations are caused by hyenas, but most large carnivore attacks on humans are by lions and leopards (**Kissui et al.** Chap. 14).

There can also be strong differences of opinion among people, such as between Maasai men and women residing in Simanjiro about concepts of wellbeing, in that women tend to focus more importance on the needs of children (**McCabe and Woodhouse** Chap. 4). How interactions with wildlife are perceived differs widely depending on the wildlife species considered (**Kiffner et al.** Chap. 1). From a human perspective, coexisting with giraffes is unproblematic, but coexisting with large carnivores and elephants is challenging and outcomes of interactions are strongly mediated by human behavior (**Kioko et al.** Chap. 13, **Kissui et al.** Chap. 14).

17.6.6 Embedment and Telecoupling

Another facet is the degree to which coupled systems are embedded within other systems or connected with distant systems. For instance, Wildlife Management Areas are coupled systems that are embedded in village structures and local governance. Protected areas are embedded in national protected area policies, and Lake Manyara is a UNESCO biosphere reserve—a global designation.

One of the major telecoupling aspects is the disproportionate distribution of wildlife-related costs and benefits. The costs of living with wildlife mainly accrue in poor, rural segments of the society whereas most benefits are realized in government treasuries, the bank accounts of people investing and working in the tourism sector (who often live in urban centers), and the pleasure of foreign tourists who enjoy the wildlife from the safety and comfort of luxury safaris (**Igoe** Chap. 3).

Another major telecoupling is climate change which is primarily driven by unsustainable economies of a few industrial countries and whose impacts will likely cause many impacts on human livelihoods and wildlife in the TE. On that note, REDD+ projects (projects designed to mitigate the effects of climate change) are telecoupled to foreigners who wish to offset carbon emissions (**Baker et al.** Chap. 15). Sedimentation of Lake Manyara is influenced by land-use decisions made in the Karatu highlands (de Bisthoven et al. 2020). Further, the catchment of the Tarangire River is located in the Kondoa highlands; therefore the dry season concentration of wildlife in the TE is dependent upon the protection of forests in that area. Thus, the TE is not an insular area unaffected by decisions made beyond its borders.

17.6.7 Vulnerability

Vulnerability is the likelihood the coupled system experiences harm from changes due to internal or external forces. For example, as outlined in Prins and de Jong (Chap. 7) and Foley and Foley (Chap. 10), market forces driving demand for ivory strongly influenced the Tarangire elephant population. Pastoralists can no longer access several wetlands in the TE because they are located in protected areas or have been converted to agriculture. During times of severe droughts, livestock populations typically decline with cascading effects on peoples' nutrition, wealth, and wellbeing (**Bluwstein** Chap. 2, **Igoe** Chap. 3, **McCabe and Woodhouse** Chap. 4).

Another particularly relevant example of vulnerability in the TE is how the COVID-19 pandemic substantially reduced income from tourism, which in turn reduced income to national parks, anti-poaching programs, and local people who are directly and indirectly benefitting from tourism. Concomitantly, the pandemic also abruptly stopped some wildlife monitoring efforts so that potential impacts can possibly only be detected well after this book has been published.

17.6.8 Surprises

When complexity is not well understood, people may be surprised at the outcomes. Such surprises include unintended consequences or perverse results. An example of an unintended consequence in the delineation of protected areas in the TE was that the fear of exclusion drove Maasai to shift towards agriculture, as plowing a piece of land is a way to secure land in the Tanzanian context. Surprisingly, human-wildlife conflicts—although obvious in the case of large carnivores and elephants—do not represent the core issues of concern to many of the people who were questioned in interviews throughout the TE. Primary issues for people seem to be land tenure and security (**McCabe and Woodhouse** Chap. 4). Indeed, most herders in Manyara Ranch expressed support for elephant presence despite frequently coming into contact with them (**Kioko et al.** Chap. 13).

In another positive surprise outcome, giraffe calves and their mothers were more likely to be detected near Maasai bomas, likely due to the lower risk of natural predation afforded by being near to pastoralists. This unintended consequence of pastoralists disrupting lion behaviors outside protected areas has helped giraffes and people to coexist (**Lee and Bond** Chap. 9).

17.7 Solutions for Human-Wildlife Coexistence

Understanding the complex features of coupled systems allows stakeholders to look to the past to develop better solutions for the future. As evidenced by the contributions to this book, a common theme is that the roots of conservation conflict in the TE stem largely from historical delineations of protected areas which did not consider seasonal movements of either wildlife or pastoralists, so-called 'colonialist' mentalities of separating people from wildlife rather than promoting coexistence, expansion of land uses such as large-scale agriculture that are incompatible with the needs of wide-ranging wildlife species and livestock, failures to involve local communities in land-use decisions, and dangerous active interactions with wildlife such as retaliatory killing of large carnivores. The examples in our book illustrate the manifold impacts of human-wildlife conflicts on wildlife populations, on food security, and on the physical and emotional wellbeing of residents of the TE, and how these conflicts reflect issues of inequity and are a source of social conflict between stakeholders.

Finding the middle ground for coexistence between humans and wildlife is a global challenge and "as much a humanitarian concern and an issue for social and economic development as it is a conservation issue" (Gross et al. 2021). Simple solutions to complex systems are unlikely to work for such deeply enmeshed problems. We believe that a first necessary step towards solutions is to move the discussion away from whether wildlife or human needs should come first, towards identifying solutions that work for both people and wildlife by quantifying the tradeoffs among wildlife-related ecosystem services (Kareiva et al. 2007).

As the different contributions to this book demonstrate, scholars disagree on the challenges that different elements of the system face and suggest different ways to address these challenges. It is not unusual to disagree about potential solutions to conservation problems (Lute et al. 2018) but we need to come to terms with such differences (Levin et al. 2021). The antidote to the Rashomon effect is to develop a shared logical framework so stakeholders can better understand the various points of view, all of which are valuable but which can be merged to offer the most effective ideas.

We can begin by agreeing about that which we disagree: for instance, the extent that the presence of livestock aligns with wildlife conservation goals is a matter of dispute. Whereas some scholars claim that livestock is largely beneficial to wildlife, the scientific literature rather suggests that wildlife-livestock interactions can be both facilitative and competitive and these relationships are strongly dependent on season (Odadi et al. 2011) and densities of livestock (Kowal et al. 2019). There is also disagreement that allowing people unrestricted use of land and resources is compatible with wildlife conservation, and fundamental disagreement on hard boundaries separating people and wildlife. The debate over benefits and costs of separating people from wildlife is not confined to the TE alone, but is a global disagreement. For example, some scholars have called for fencing around protected areas, both in the TE (**Prins and de Jong** Chap. 7) and throughout Africa (Packer et al. 2013; Di Minin et al. 2021). However, what fencing would do in the TE can be anticipated by the fate of declining wildlife populations in Lake Manyara National Park, and some scientists have predicted that fencing Tarangire National Park might cause the collapse of one of the world's last remaining migrations of wildebeests (Voeten et al. 2010; Morrison et al. 2016).

We can also define areas of agreement. Both social and natural scientists appeared to agree that rangelands can support both people and wildlife, provided that people have a low ecological footprint. Wildlife and people mostly avoid each other at appropriate spatio-temporal scales as evidenced by: elephants and herders avoiding each other in rangelands (**Kioko et al.** Chap. 13), pastoralists keeping their livestock in safe pens at night (**Kissui et al.** Chap. 14), and Maasai herding their cattle distant from calving grounds of wildebeest to avoid transmission of malignant catarrhal fever virus (Lankester et al. 2015). There is a broad agreement that the Simanjiro Conservation Easements, and now Certificates of Customary Rights of Occupancy, work for both wildlife and people. Some authors noted disapproval of Wildlife Management Areas in earlier studies, but a recent study suggests that attitudes among people residing in WMAs became supportive over time—and all research in the TE indicates that wildlife populations in WMAs can rebound once conservation measures are in place. Social and natural scientists agree that previous top-down decisions were inadequate to cater to the needs of people and wildlife in the TE. Some parts of the TE have been lost as habitat for large mammal species and livestock grazing, and now serve other human uses such as for settlement or agriculture. While the human population in the TE is still growing (National Bureau of Statistics 2013), it would be too simplistic to blame the historical decline of rangelands and wildlife populations on this alone (Bluwstein et al. 2021): various examples in this book point to drivers outside of the TE (e.g. poaching driven by international demand for ivory, large-scale agriculture driven by international food markets, and international development policies). One thing is certain: the health of humans, animals (domestic and wild), and ecosystems are inextricably linked. This was clearly demonstrated to the world by the COVID-19 pandemic, which likely stemmed from human interference with wild animals that host coronaviruses. EcoHealth (see Box 17.1) is a concept that uses interdisciplinary research and practices to understand and promote health and wellbeing for all levels of the system—from humans and their livestock to wildlife and plants to the entire ecosystem. If we fail to adopt such interdisciplinary approaches, we may all suffer the consequences.

Box 17.1: EcoHealth: An Interdisciplinary Approach

Douglas R. Cavener, Pennsylvania State University

Prior to the European colonial period and big game hunting of the nineteenth and twentieth centuries, native peoples of Tanzania lived in dynamic harmony with nature. Big game hunting of large mammals with firearms marked the beginning of the decline of this harmony. Although big game hunting in Tanzania is now highly restricted, the rapid human population growth and accompanied agricultural and pastoral expansion occurring in the past 50 years coupled with climate change now pose a much more serious and persistent challenge to the health of the ecosystem. Where once hundreds of millions of wild large mammals lived in harmony with a few million people, now nearly 60 million people and 50 million livestock dwarf the remaining few million wild large mammals. Ironically, these remaining wild animals—which include the charismatic giraffe, elephant, zebra, chimpanzee, lion, leopard, and cheetah—are responsible for the lion's share of Tanzania's economy through tourism. For Tanzania to survive and thrive as a nation of people and as one of the most important ecosystems on the planet, it will need to embrace and promote the health of the entire ecosystem including humans, wildlife, livestock, land, and water. Two key interdisciplinary concepts, One Health and EcoHealth, describe the underlying principles and the key role that people must play to achieve the goal of humans and nature living in harmony.

One Health is a biomedical approach focusing on animal and human health and includes both veterinary and human medicine (Lerner and Berg 2017). Lerner and Berg (2017) noted that the core values of the One Health concept relate somewhat narrowly to human health and the health of animals that directly influence human health. An expanded concept is EcoHealth, which encompasses the health of humans, animals, and ecosystems synergistically. EcoHealth has been defined as "a field of research, education, and practice that adopts systems approaches to promote the health of people, animals, and ecosystems in the context of social and ecological interactions" (Parkes et al. 2014). Importantly, EcoHealth embraces wellbeing and not merely the absence of disease. As such, the EcoHealth approach includes more social science and humanities—including local and indigenous knowledge—than the One Health approach. The core values of EcoHealth are population health (of humans, animals, and ecosystems) as well as biodiversity and sustainability (Lerner and Berg 2017).

Such core values underscore the importance of interdisciplinary approaches to health and wellbeing of not only humans but animals, plants, and the ecosystems in which we are all embedded. How might the concept of EcoHealth be applied in the Tarangire Ecosystem? Traditional health studies might report the incidence of malignant catharral fever in domestic cattle, which is spread by calving wildebeests (Lankester et al. 2015). But what are the human social/economic impacts, and potential solutions? Pastoralists either avoid

(continued)

Box 17.1 (continued)

wildebeest calving areas and shift their livestock elsewhere or chase off wildebeests from desirable rangelands. These actions can cause problems for people, wildlife, and the ecosystem. Integrating research disciplines can reveal potentially successful means of maintaining the wellbeing of pastoralists and their cattle, thriving populations of wildebeests, and the critical ecosystem services provided by thousands of migratory large mammals.

As much as coexisting with wildlife creates many challenges, it also provides opportunities. In the TE, many steps, most notably the establishment of conservation easements, CCROs, and WMAs, have been taken to conserve and restore wildlife populations during the last decades. To make full use of these opportunities we need to learn from our past mistakes.

These interventions have likely contributed to stopping wildlife declines, and wildlife populations slowly show signs of recovery in these areas (**Bond et al.** Chap. 8), but wildlife populations in the TE are likely well below their historical baselines (**Prins and de Jong** Chap. 7). While it may not be possible to restore the full community and abundance of wildlife across the entire TE, we believe that there is still substantial potential for ecosystem restoration provided that such attempts take into account the coupled social-ecological complexities (Fischer et al. 2021). Ecological restoration efforts in the TE also resulted in income-generating mechanisms and provided opportunities for employment and for communities to invest in infrastructure such as schools or dispensaries that contribute to the wellbeing of people and sustainable development of the region. Several scholars object that such monetary contributions are insufficient and we agree that there are multiple ways to make sure that benefits associated with wildlife accrue to people who actually live with wildlife. Since grazing rights are so important for Maasai (**McCabe and Woodhouse** Chap. 4) and limited grazing may be compatible with long-term persistence of wildlife populations (exemplified by Manyara Ranch; **Bond et al.** Chap. 8), restoration efforts in the TE are likely most effective if they take into account the needs of pastoralists as well as the needs of wildlife.

For coexistence to work, we anticipate that participatory and consensus-based approaches for planning and managing human-wildlife coexistence are a suitable way to find integrated and holistic solutions for people and wildlife to coexist in the TE (König et al. 2020, 2021). The establishment of conservation easements, CCROs, and WMAs during the last decades is a step in this direction, yet there are many areas where managing the different aspects of the human-wildlife interface could be done in a more holistic way. Tanzania is one of the first countries to enact national legislation on protecting wildlife corridors (**Lohay et al.** Chap. 12)—a laudable effort that hopefully contributes to maintaining seasonal wildlife and livestock movements and to facilitate anticipated range shifts of wildlife (Payne and Bro-Jørgensen 2020). This national legislation implicitly recognizes that humans and non-humans alike depend upon a healthy environment for both to thrive and fosters a land-sharing approach in Tanzanian landscapes. Specifically, strong and immediate efforts must be made to secure two key remaining wildlife migration corridors in the TE: from the dry season range in the national parks and Manyara Ranch north to Lossimingore and the Gelai plains, and east to the Simanjiro Plains. The large majority of these two corridors is already covered by CCROs, and several NGOs are working to fill in the remaining conservation gaps, most of which are now quite small, through additional CCROs or other means. Protecting these critical wildlife movement corridors will go a long way towards safeguarding the integrity and function of the TE as a whole, which benefits both wildlife and humans. Without this step, there may be sequential faunal collapse over the next decades.

For human-wildlife coexistence to work in the sense of Carter and Linnell (2016), effective institutions are required to ensure "population persistence, social legitimacy, and tolerable levels of risk". Several contributions of this book point to the idea that approaches to govern human-wildlife interactions have not always been effective in the past due to inadequate spatial scales of administrative responsibilities for wildlife, lack of resources for implementing effective technical solutions to prevent or reduce negative human-wildlife interactions at scale, and often also due to lack of trust between stakeholders. Thus, we anticipate that national endeavors to conserve connectivity (one of the key prerequisites for abundant wildlife populations and associated ecosystem services in the TE) will be most successful if they are accompanied by a national and collaborative human-wildlife coexistence program that could possibly be funded through income generated from ecotourism or payments for ecosystem services. Such a program could ensure that methods to prevent wildlife damages are developed, refined, and made available at scale. Empowering people to reduce human-wildlife conflict, by using cost-effective, socially acceptable, sustainable, and scalable methods such as predator-proof bomas to protect livestock and chili pepper fences to protect crops from elephants, would be a key component of such a program (Kissui et al. 2019; Kiffner et al. 2021). Foremost, however, such a program would need to make sure that land tenure issues are effectively addressed and that stakeholders are adequately involved in decision-making and adaptive management of wildlife corridors and human-wildlife interactions (Carter et al. 2021).

In this synthesis, we have outlined the issues affecting this human-dominated landscape, and the disparate opinions on the challenges affecting both people and wildlife. Too many other ecosystems have lost long-distance migrations or the vast majority of their large wild mammals; one only needs to think of the fenced reserves of South Africa and Kenya where wildlife migrations are now only distant memories of past ecosystem processes. Thus, we cannot stress enough the uniqueness of the TE. Despite all of the human development in the landscape, it is remarkable that this ecosystem is still ecologically functional. It hosts hundreds of thousands of people, millions of livestock, large mines, booming towns, two major tarmac roads, and a patchwork of agricultural fields—and yet still supports one of the most significant long-distance migrations of wildlife remaining in the world, much of it taking place on community land. Wildlife numbers have declined historically, but the mere fact that many populations are stable, and some are increasing, despite all the odds, is testament to the singularity of the place, and demonstrates that humans and wildlife can indeed coexist.

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