EFFECTIVENESS OF PATROLLING PROTECTED AREAS AGAINST BUSHMEAT POACHING THROUGH SNARING IN THE SOYSAMBU CONSERVANCY, KENYA

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DECLARATION

This thesis is my original work and has not been submitted for award of a degree in any other University.

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Dedication

All things of value are defenseless.

Lucebert

To the fragile beauty of Kenya's disappearing Nature.

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Abbreviations

ANAW African Network for Animal Welfare

ASL Above Sea Level

ASTER Advanced Spaceborne Thermal Emission and Reflection Radiometer

AUC Area Under Curve

CBD Convention on Biological Diversity

CDF Cumulative Distribution Function

CE Collective Error

CITES Convention on International Trade in Endangered Species of Wild Fauna and Flora

CMA Cumulative Moving Average

CMS Convention on the Conservation of Migratory Species of Wild Animals

CPT Crime Pattern Theory

CPUE Catch Per Unit Effort

CWU Consumptive Wildlife Utilization

DEM Digital Elevation Model

DSWT David Sheldrick Wildlife Trust

ECDF Empirical Cumulative Distribution Function

EIA Environmental Impact Assessment

EIAL Environmental Impact Assessment License

EID Emerging Infectious Diseases

EMCA Environmental Management and Coordination Act, 1999

EPSG European Petroleum Survey Group

ESIA Environmental and Social Impact Assessment

GLECA Greater Lake Elementaita Conservation Area

GPS Global Positioning System

IBA Important Bird Area

IE Individual Error

IPBES Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

IUCN International Union for Conservation of Nature

JTC Journey to Crime

KFS Kenya Forest Service

KSh Kenyan Shilling (1 USD=108.30 KSh (Sep. 2020))

KWCA Kenya Wildlife Conservancies Association

KWS Kenya Wildlife Service

LEM Law Enforcement Monitoring

LEWSE Lake Elementaita Wildlife Sanctuary Ecosystem

MAB Multi-armed Bandit

MAUP Modifiable Area Unit Problem

MEA Multilateral Environmental Agreements

METI Ministry of Economy, Trade and Industry of Japan

NEMA National Environment Management Authority

NGO Non-Governmental Organization

OECD Organisation for Economic Co-operation and Development

OECM Other Effective area-based Conservation Measures

PA Protected Area

PD Prediction Diversity

RAT Routine Activity Theory

RCP Rational Choice Perspective

ROC Receiver Operating Characteristic

SDG Sustainability Development Goals

SDM Species Distribution Modeling

SMART Spatial Monitoring and Reporting Tool

TSS True Skill Statistic

UNEP United Nations Environment Programme

UNEP-WCMC UNEP World Conservation Monitoring Centre

UNESCO United Nations Educational, Scientific and Cultural Organization

UTM Universal Transverse Mercator

WCMA Wildlife Conservation and Management Act, 2013

WDPA World Database on Protected Areas

WHC Convention Concerning the Protection of the World Cultural and Natural Heritage

Definitions

Biodiversity "... the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems." (United Nations, 1992, Art. 2)

Bushmeat Meat from wildlife.

Bushmeat poaching Illegal hunting of wildlife for their meat.

Commercial poaching (ambiguous) Either bushmeat poaching on a commercial scale, or trophy poaching that aims to sell the acquired trophies illegally.

Corridor an area used by wild animals when Migrating from one part of the ecosystem to another periodically (Government of Kenya, 2013).

Desnaring Removal of wire snares by rangers, with occasional assistance from third parties.

Deterrence (broad sense) The likelihood that an offender is detected, arrested, prosecuted, and fined (Siegel, 2012).

Deterrence (narrow sense) The likelihood that a poacher is detected and arrested in the protected area.

Dispersal area Area adjacent to or surrounding protected areas into which wildlife moves during part of the year (Government of Kenya, 2013).

Displacement The shifting of crime in response to a prevention or enforcement campaign (Eck, 1993).

Environmental criminology A family of theories that study criminal events and the immediate circumstances in which they occur (Summers & Guerette, 2018).

Guardianship The cumulative product of the presence of guardians, their capacity to detect offenders, and their capability to intervene once offenders are detected (Reynald, 2009).

Human-wildlife conflict Interaction between wildlife and human beings that causes a negative impact, and often resulting to some form of loss (Ministry of Tourism and Wildlife, 2020).

Landscape connectivity The degree to which landscape facilitates or impedes movement among resource patches. (Taylor et al., 1993)

Leakage The spillover of poaching to areas adjacent to the protected area, following patrol efforts in the protected area (Ewers & Rodrigues, 2008).

National Park (Kenya) A protected area managed by KWS (Government of Kenya, 2013).

National Reserve (Kenya) A protected area managed by the County government (Government of Kenya, 2013).

Other effective area-based conservation measures A geographically defined area other than a Protected Area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in situ conservation of biodiversity, with associated ecosystem functions and services and, where applicable, cultural, spiritual, socioeconomic, and other locally relevant values (IUCN-WCPA Task Force on OECMs, 2019).

Patrol bias The bias created in patrol data due to selective patrolling of protected areas (Keane et al., 2011).

Patrol effectiveness The extent to which patrol efforts by rangers lead to deterring poaching inside the protected area.

Patrol efficiency The extent to which available resources are deployed to generate a patrol effort in a protected area.

Patrol relevance The extent to which deterrence of poachers by rangers influence wildlife populations in the protected area.

Poaching The illegal taking of wildlife (Lemieux, 2014).

Presence-only data The presence of an object or phenomenon is certain, but its absence is not, for example owing to low detectability (Guisan et al., 2017).

Protected area A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values (Dudley, 2008).

Ramsar Convention Convention of Wetlands of International Importance especially as Waterfowl Habitat (Ramsar Convention, 1971).

Recency bias The observer gives greater weight to recent events than to historic ones (Shanteau, 1989).

Resnaring Replacement of snares that were removed by rangers by poachers.

Sanctuary (Kenya) A protected area aimed at the conservation and protection of one or more species of wildlife (Government of Kenya, 2013).

Silent victim problem The underestimation of the true level of crimes against wildlife, as animals are utterly dependent on rangers as witnesses (Lemieux, 2014).

Species distribution modeling Extrapolation and prediction of species distribution data in space and time through statistical models (Guisan et al., 2017).

Survivorship bias A form of selection bias, in which the analysis focuses on people, objects, or phenomena that passed a selection process. The filtration process results in an incomplete and biased sample (Shermer, 2014).

Trophy poaching Illegal hunting of wildlife for parts of their body.

Unsupervised learning Machine learning from unlabeled datasets (Bramer, 2007).

Wetland (EMCA definition) Areas permanently or seasonally flooded by water where plants and animals have become adapted (Government of Kenya, 2015a).

Wildlife All forms of non-domesticated plants and animals in the wild (Lemieux, 2014).

Wildlife conservancy (Kenya) Land set aside by an individual landowner, body corporate, group of owners or a community for purposes of wildlife conservation in accordance with the provisions of ... [the Wildlife Conservation and Management Act, 2013] (Government of Kenya, 2013).

World Heritage Convention Convention Concerning the Protection of the World Cultural and Natural Heritage (UNESCO, 1972).

Abstract

Bushmeat poaching – illegal hunting of wildlife for their meat – is seen as a severe threat to wildlife by both the Kenyan Wildlife Service (KWS) and managers of Kenyan protected areas. Protected areas must therefore be patrolled against bushmeat poachers. However, patrolling effectiveness is not yet well-understood, measured, and monitored. In particular, the relation between deterrence of poachers and patrol effort is not clear. Ranger expertise, which may be used to improve patrol effectiveness, has not yet been assessed. No specific patrolling strategies for bushmeat poaching have been developed to date. Moreover, current research does not usually apply criminological theory to understand and predict bushmeat poaching.

The research was implemented to assess the effectiveness of patrolling a protected area against bushmeat poachers placing snares. Improved patrolling strategies were developed using environmental criminology. This is a group of theories that aims to explain crime from its spatial context. The Soysambu Conservancy at Lake Elementaita in the Kenyan Great Rift Valley was used as a case study; fieldwork took place from December 2018 to April 2019. First, the current patrolling and poaching patterns were assessed by comparing conservancy data with research data, consisting of 120 km desnaring transects, and interviews with 31 rangers and six community representatives. Snaring hotspots were identified using Stienen/Steiner sets and nearest neighbor spatial analysis. Distance from snare positions to park infrastructure and neighboring villages were calculated using Dirichlet tessellation. The frequency of poacher observations as stated in ranger interviews was compared with reported poacher observations as stated in the conservancy's observation book. Second, the ability of rangers to forecast snaring locations and densities was measured. Rangers estimated snare densities prior to walking desnaring transects, and the acquired estimations were analyzed through the diversity prediction theorem. Third, improved desnaring strategies were developed. The search area was reduced through the presence-only Maxent species distribution model, and the search method was improved by applying an epsilon-greedy policy for the exploitation-exploration trade-off.

Assessment of the conservancy data showed that (1) snare density in the conservancy is high (325 snares found; 50 snares per km^2), (2) patrol patterns are predictable, and (3) the number of reported poacher sightings (two per month) is lower than would be expected based on observed snaring densities. Snares are mainly placed in the transition of bushy to open areas (Area under curve (AUC)=0.782), near public roads (AUC=0.705), and near park infrastructure, such as gates, lodges, and staff settlements (AUC=0.655). Rangers thought that snares would be placed near park borders (two-sided exact binomial test, $x = 23, n = 27, H_0 \leq 0.5, \alpha = 0.05, p < 0.001$), near communities outside the conservancy (two-sided exact binomial test, $x = 24, n = 28, H_0 \leq 0.5, \alpha = 0.05, p < 0.001$), and not near park gates (two-sided exact binomial test, $x = 2, n = 26, H_0 \leq 0.5, \alpha = 0.05, p < 0.001$). Rangers estimated the snare density before walking a transect by assuming that all snares removed during previous desnaring operations were replaced by poachers (one-sided exact binomial test, $x = 19, n = 27, H_0 \leq 0.5, \alpha = 0.05, p = 0.026$). The estimation of snare density by rangers prior to walking transects was deemed satisfactory when ratio r of collective error CE and individual error IE is smaller or equal to 0.1. This estimation capacity was not significant (one-sided exact binomial test, $x = 3, n = 27, H_0 : r \leq 0.1, \alpha = 0.05, p = 0.718$) The majority of interviewees self-rated their capacity to deter poachers from entering the conservancy as low (two-sided exact binomial test, $x = 6, n = 26, H_0 \leq 0.5, \alpha = 0.05, p = 0.009$). Poachers are frequently observed (two-sided exact binomial test, $x = 24, n = 26, H_0 \leq 0.5, \alpha = 0.05, p < 0.001$) but not always reported (9/31 interviewees). No input from communities surrounding the conservancy has been sought, although rangers think that community members know the poachers' identities (two-sided exact binomial test, $x = 28, n = 28, H_0 \leq 0.5, \alpha = 0.05, p < 0.001$). The snare recovery rate can be improved by 4–9% by reducing the search area and optimizing the exploration-exploitation allocation (Wilcoxon ranked sum test, $p = 0.04, r = 0.344, U = 89.4$, AUC=0.853, TSS=0.587, Boyce index=0.91).

Several new approaches have been developed and applied in this research. First, the environmental criminology group of theories has been applied to improve the detection of snaring hotspots and reduction of the search area. Environmental criminology studies the spatial context in which crimes take place, but it has hitherto not been applied on bushmeat poaching. Second, a novel

method was developed to identify snaring hotspots using Stienen and Steiner sets which were derived from the snaring point patterns. Third, the capacity of rangers to contribute to the improvement of patrolling effectiveness was tested using the prediction diversity theorem. Fourth, deterrence of poachers by rangers was disentangled from confounding factors, such as spatial and temporal displacement of poaching activities. Displacement and low ranger morale were found to cause survivorship bias, a phenomenon which has hitherto not been identified in anti-poaching literature. Fifth, an improved snare search methodology was developed which decoupled the reduction of the search area using Maxent presence-only species distribution modeling and the balancing of visiting known snaring areas (exploitation) and discovering new ones (exploration) using an epsilon-greedy policy. The latter has not yet been applied in anti-poaching literature.

In conclusion, at least 3% of the conservancy contains snaring hotspots despite a large ranger density (day time: 5.1 km^2 per ranger; night time: 7.9 km^2 per ranger). Rangers under-report poacher sightings, patrol patterns are predictable, and the patrol strategy relies on desnaring. Patrol effectiveness based on reported poacher sightings is subject to survivorship bias, because poachers were either not available for the detection or detected but not reported. Five broad recommendations can be made. First, patrol effectiveness, based on ranger-collected data, should be interpreted with care since poaching signs may be filtered out. Second, rangers should be commensurately remunerated and equipped if reliable patrol data are to be obtained. Third, desnaring without efforts to detect and arrest poachers devolves into a symbolic exercise since poachers will replace the snares removed by rangers. Fourth, wildlife crime is a social phenomenon and thus calls for the application of criminological theory. This allows for the prediction of areas where poachers may place snares without extensive upfront data collection efforts. Fifth, patrol effectiveness must be analyzed in the context of environmental management and governance of the surrounding areas.

Chapter 1: Introduction

The illegal taking of wildlife for their meat – bushmeat poaching – is increasingly recognized as a threat to the savanna biome's biodiversity. This study aims to assess the effectiveness of patrolling protected areas for bushmeat poachers and explores methodologies that increase this effectiveness.

1.1 Background

An estimated one million plant and animal species are threatened with extinction because of human actions (Díaz et al., 2019; IPBES, 2019). The main human activities responsible for this mass extinction crisis are land-use change, followed by over-exploitation of animals, plants, and other species through hunting, logging, and fishing (IPBES, 2019). The principal strategy for conserving the world's remaining biodiversity is setting areas aside for long term *in situ* conservation of nature. These so-called protected areas were covering 15% of the Earth's land surface in 2018 (UNEP-WCMC et al., 2020), and therefore represent an essential investment in the conservation of the Earth's biodiversity.

Protection against poaching – illegal hunting of animals – is a primary concern for managers of protected areas (Schulze et al., 2018). Without effective protection, these areas can turn into empty landscapes, thus defying the purpose for which they were created (Ripple, Chapron, et al., 2016). The consequences of poaching are not limited to animals in the protected areas. First, hunting, collecting, and eating of wild animals can trigger outbreaks of zoonotic diseases, such as human immunodeficiency viruses (HIV), Ebola, Severe acute respiratory syndrome (SARS), and Coronavirus disease 2019 (COVID-19) (Andersen et al., 2020; Wilkie, 2006). Second, illegal wildlife trade converges with other forms of crime and is increasingly seen as a threat to peace and stability (INTERPOL-UN, 2016). Third, loss of biodiversity will compromise the feasibility of the Sustainable Development Goals (Díaz et al., 2019; Vasseur et al., 2017). Therefore, the protection of protected areas is not just a biodiversity issue but also one of health, peace, and development.

Providing wildlife security is one of the critical functions of protected areas (Pacifici, Di Marco, et al., 2020). However, the provided level of wildlife protection is often insufficient to counter poaching pressure. Only one in four protected areas has sufficient staff and budget (Coad et al., 2019), while patrolling can take up to two-thirds of the annual operational costs on African sites (Plumptre, 2019). Furthermore, poaching pressure is mounting as one in three protected areas faces intense human pressure (Jones, Allan, et al., 2018). Researchers are therefore considerably interested in investigating the effectiveness of patrolling protected areas (Rodrigues & Cazalis, 2020), understood here as the balance between poaching pressure and patrol effort.

Current research focuses on the poaching of large mammals in African protected areas for parts of their body, such as ivory, skins, and rhino horns (trophy poaching). Researchers are, however, also becoming increasingly concerned about poaching that targets animals for their meat (bushmeat poaching) (Duporge et al., 2020; van Velden et al., 2018). Bushmeat poachers often use wire snares (hereafter: snares), which are cheap to make and non-selective: their use often results in killing animals that were not targeted by the poacher (by-catch). Field practitioners and Non-Governmental Organizations (NGO's) call for more anti-snaring research given the widespread and serious effects of snaring on wildlife populations (Gray et al., 2018; Gray et al., 2017; Masolele, 2018).

1.2 Problem statement

The portfolio of African protected areas is extensive and expanding. These areas are poorly funded and subjected to mounting bushmeat poaching pressure. Current patrolling levels are insufficient to reduce or even monitor illegal activities in protected areas (Dancer, 2019). Therefore, the effectiveness of patrolling must be maximized: poaching pressure must be reduced as much as possible using the available resources. Maximization of patrolling effectiveness requires an understanding of the factors that are influencing it. Of particular importance are an understanding of (i) the extent to which an increased patrol effort results in decreased poaching pressure; (ii) the involvement of ranger expertises and rangers' ability to deter poachers from entering protected areas; and (iii) the scope for improvement of patrolling effectiveness, particularly concerning bushmeat poaching by snaring.

1.3 Justification of the study

Bushmeat poaching is increasingly recognized as a threat to biodiversity in the savanna biome (Lindsey et al., 2013; Lindsey et al., 2015). There is, however, limited understanding of the effectiveness of patrols that have to mitigate this problem in protected areas (Keane et al., 2011). Such insight is essential for at least three reasons. First, protected areas represent a significant investment in the conservation of biodiversity: terrestrial national parks alone cover 11% of the Kenyan land surface (UNEP-WCMC, 2020b), and conservancies cover an additional 11% (KWCA, 2016). Together, these areas contribute 21% of Kenya's foreign exchange earnings and create employment for approximately 1.5 million Kenyans (Ouko, 2018; The World Bank, 2019). Politicians, citizens, and managers of protected areas alike need assurance that this investment is well-protected. Second, patrolling protected areas is expensive and can consume more than half of the annual operational budget (Plumptre, 2019). Funders of protected areas need to know whether these resources are well-spent, also because most parks are under-resourced (Coad et al., 2019). Third, policy-makers cannot develop wildlife conservation measures in the absence of an understanding of the cause and effect of patrolling effectiveness. Development and implementation of such policies are urgently required given the rapidly decreasing biodiversity, both worldwide (IPBES, 2019) and in Kenya (Ogutu et al., 2016).

The concerns about the effect of bushmeat poaching are shared by the Kenya Wildlife Service (KWS), and managers of Kenyan protected areas (Kiringe et al., 2007; Secretariat of the Convention on Biological Diversity, 2011). Nevertheless, a task force on wildlife security found that KWS concentrates its surveillance mainly on trophy poaching, while bushmeat poaching was underestimated and therefore under-patrolled (Rotich et al., 2014). Moreover, just two peer-reviewed articles that examined bushmeat poaching in Kenyan protected areas were published in the last 20 years (Kimanzi et al., 2014; Wato et al., 2006). Therefore, the development of policies that aim to mitigate bushmeat poaching in Kenya is not underpinned by peer-reviewed research.

The justification for this research can thus be summarized as follows. Policy-makers have to make decisions on measures to mitigate bushmeat poaching in protected areas. Such decisions require an understanding of the effectiveness of patrolling, in other words, the extent to which rangers deter poachers. This effectiveness is currently not well-understood. The lack of information on these issues is especially serious in Kenya. Here, bushmeat poaching is considered a severe threat to biodiversity, while near-zero peer-reviewed research on bushmeat poaching is available.

1.4 Objectives

General objective

To evaluate the effectiveness of patrolling a terrestrial protected area against bushmeat poachers placing snares, based on which improved desnaring strategies are developed.

Specific objectives

1. To determine the poaching and patrolling patterns in the Soysambu Conservancy compared to reported poaching prevalence.
2. To evaluate the extent to which rangers are able to predict poaching patterns.
3. To develop improved desnaring strategies.

1.5 Scope and limitations

1.5.1 Scope

This study examines the effectiveness of ranger patrol efforts to deter bushmeat poachers from entering a terrestrial conservancy to place snares. The study area is the 190-km^2 Soysambu Conservancy, located between Lake Elementaita and Lake Nakuru National Park in Nakuru County, Kenya.

The scope of research is defined as follows. The research took place within the boundaries of the conservancy. Environmental policies to deter poaching in the wider Lake Elementaita region were studied as far as these could influence patrol effectiveness in the Soysambu Conservancy. "Rangers" are understood as "rangers employed by the Soysambu Conservancy" as opposed to government-appointed rangers from the Kenya Wildlife Service (KWS). "Deterrence" refers to the capacity of rangers to discourage poachers from entering the conservancy. This is deterrence in the narrow sense: deterrence in criminological theory includes not only arrest, but also prosecution and punishment. The latter two processes take place in the judiciary system outside the research area and are therefore out of this study's scope.

1.5.2 Limitations

A limitation of this study is that it took place in one relatively small conservancy. This was not a conscious choice: the research was planned to be carried out in the much larger Tsavo National Park. However, a requested research permit was informally rejected by KWS because the subject was "too sensitive"; a formal reaction to the request was never received. KWS was also unable to give clearance for the analysis of 10 years of patrol data collected by WildlifeWorks, a private company operating in Kenya's Kasigau Corridor. Likewise, the David Sheldrick Wildlife Trust (DWST) was either not able or willing to share patrol and desnaring data in the Tsavo area. In this context, it seems appropriate to cite the Kenyan Wildlife Strategy 2030:

> While the Kenya Wildlife Service is recognized as the key government agency tasked with implementing the strategy, it is important to recognize that this strategy's success depends on effective collaboration and engagement of all stakeholders across the sector for collective action (Ministry of Tourism and Wildlife, 2018, p. 106)

This research limitation is also a finding, albeit not within the scope for this study: collaboration between KWS, the custodian of Kenyan wildlife, and other wildlife conservation stakeholders is still in its infancy.

The lack of collaboration interest from KWS was discussed with the Kenyan Wildlife Conservancies Association (KWCA) (D. Kaelo, personal communication, July 26, 2018). KWCA suggested to approach the Soysambu Conservancy, as (1) this conservancy was known to experience severe bushmeat poaching, and (2) was interested to reduce this poaching, and (3) would be willing to host the research and make data available.

Chapter 2: Literature review

The cornerstone of biodiversity conservation is the creation of protected areas. Significant research gaps remain concerning the protection of these areas, despite their importance and extent. Here, the background of protected areas and their protection are explored, especially concerning bushmeat poaching by snaring and the extent thereof in Kenya. The overview finishes with a list of research gaps concerning the effectiveness of patrolling protected areas for bushmeat poachers placing snares and bottlenecks for improvement thereof.

2.1 The biodiversity crisis

Biodiversity – the diversity within species, between species and of ecosystems (United Nations, 1992) – is being lost at an accelerating rate. The extent and rate of biodiversity losses give rise to concerns about the effects on the economy, health, and cultural values. The loss of biodiversity will result in substantial damages of which the extent cannot yet be overseen.

2.1.1 The age of extinction

An estimated 1 million animal and plant species are driven to extinction by human actions, many of them within decades (IPBES, 2019). The mass extinction of species is mainly caused by over-exploitation and habitat conversion, driven by rapid growth of the human population and unsustainable per capita consumption (Ceballos et al., 2020; Díaz et al., 2019; Ripple et al., 2017). These drivers of extinction have been accelerating in intensity: in the last 50 years, the world population has doubled, and global trade has increased tenfold (IPBES, 2019). In particular, mammals are disappearing fast, as an estimated 52% of large mammal species is threatened with extinction (Ripple et al., 2019). One of the main extinction threats for mammals is that we are eating them (Milner-Gulland et al., 2003; Ripple et al., 2015; Ripple et al., 2019).

The catastrophic decline of wildlife populations is also registered in Kenya. Its rangelands had, on average, 68% fewer animals in 2016 than in 1977 (Ogutu et al., 2016). Locally, such as in West Pokot county, nearly the entire (99%) animal population was lost (Damania et al., 2019).

Kenya's human population has grown exponentially from 14.6 million in 1977 (The World Bank, 2020) to 47.6 million in 2019 (Kenya National Bureau of Statistics, 2019a, 2019b). This growth has been accompanied by encroachment in all over 100 migratory corridors and dispersal areas (Ojwang et al., 2017). Communal rangelands are privatized, fenced, and converted to agricultural use (Norton-Griffiths, 2007; Pearce, 2015). Animals have to rely increasingly on national parks that were designed to protect them only in the dry season (The World Bank, 2019; Western et al., 2009).

2.1.2 Consequences of biodiversity loss

The consequences of the ongoing mass extinction cannot be overseen yet, but are likely to be drastic.

The collapse of tourism in Kenya, for example, would result in an 8 to 14% decline of Gross National Product (The World Bank, 2017). The sector employs approximately 1.5 million people (The World Bank, 2019). It accounts for 21% of foreign exchange, and 75% of these earnings come from wildlife tourism (Ouko, 2018). Tourism is a crucial sector for achieving Kenya's goals formulated in its Vision 2030 strategy (Government of Kenya, 2007). However, Kenya's dwindling wildlife populations put the viability of this sector at risk. The point has been reached in which further road construction has become inadvisable given the effect on wildlife. The 20 km defaunation zones, which are typical for road construction, result in net long term losses for the economy through missed tourist revenue (Damania et al., 2019).

The financial and insurance sectors have recently realized that loss of biodiversity will incur staggering financial losses due to the loss of ecosystem services (Swiss Re Institute, 2020). The Dutch National Bank, for instance, estimates that financial institutions in the Netherlands would lose an estimated 510 billion euros from their portfolios as a result of biodiversity losses. This amount represents roughly one-third of their 1.4 trillion euro investment portfolio (De Nederlandse Bank; Planbureau voor de Leefomgeving, 2020).

Biodiversity loss is associated with Emerging Infectious Diseases (EID) (UNEP, 2020). Pathogens can be transmitted from animals to humans through the consumption of bushmeat (Bird & Mazet, 2018; Johnson et al., 2020). Approximately 43% of the 335 EID events between

1940 and 2004 originated from wildlife (Jones et al., 2008). Several of these zoonotic EID events resulted in global outbreaks, such as HIV, Ebola, SARS (Wilkie, 2006), and COVID-19 (Andersen et al., 2020; Lam et al., 2020). Recent research demonstrated that species remaining after land conversion (from relatively undisturbed land to cropland, pasture, urban use) are more likely to harbor pathogens (Gibb et al., 2020; Ostfeld & Keesing, 2020). Kenya, with its rich but rapidly declining biodiversity, has been identified as a potential future source of EID (Allen et al., 2017).

Perhaps the most ominous prediction is that we can expect ”surprises.” Global modification of the earth's ecosystem on this scale has never occurred before in human history (IPBES, 2019), and will have cascading effects which we cannot yet anticipate (Estes et al., 2011; Terborgh et al., 2001).

Finally, the effects of biodiversity decline on human health and economy refer to *use value* – value that can be expressed in monetary units. Another value category is *intrinsic value* – the value that the natural world has regardless of its economic value (Dasgupta et al., 2019; Pearce & Barbier, 2000). The preamble of the Constitution of Kenya defines the environment as "our heritage" (Government of Kenya, 2010c). The loss of Kenyan wildlife can therefore not be readily expressed in monetary terms; it is, per the Constitution, part of Kenyan identity and therefore irreplaceable.

2.2 Area-based conservation measures

Terrestrial protected areas cover approximately 15% and 22% of the land surface of the world and Kenya, respectively. However, declaring that land has protected status does not guarantee that biodiversity will indeed be protected.

2.2.1 Establishment of area-based conservation measures

The key strategy for *in situ* conservation of biodiversity is the establishment of protected areas (Convention on Biological Diversity, 2010b). Parties to the Convention on Biological Diversity (CBD) agreed to establish a network of protected areas (Art. 8a) and promote sustainable development in adjacent areas (Art. 8e) (United Nations, 1992). These objectives were elaborated in the Aichi Biodiversity Targets (Convention on Biological Diversity, 2010b). Target 11 stipulates that more than 17% of terrestrial and inland water areas must be protected by 2020. These areas must be ecologically representative, well-connected with each other, integrated into the wider landscape, and effectively and equitably managed.

The Aichi Biodiversity Targets distinguish "protected areas" (PA) and "other effective area-based conservation measures" (OECM), both of which are "area-based conservation measures". Protected areas are understood as "clearly defined geographical spaces that are set aside for long-term conservation of nature" (Dudley, 2008). OECM are areas other than protected areas that aim to achieve biodiversity conservation, without this being their first or sole objective (Donald et al., 2019; IUCN-WCPA Task Force on OECMs, 2019). OECM are thought to play an important role in the spatial coverage of biodiversity (Watson et al., 2016), especially for private conservation areas (Shumba et al., 2020). However, the definition of OECM is recent (IUCN-WCPA Task Force on OECMs, 2019), and the database developed to track their extent is not yet used by the Parties to the CBD (WDOECM, 2020). Therefore, both research and implementation of area-based conservation measures have focused on protected areas (Maxwell et al., 2020).

Approximately 15% of the earth's land surface was covered by over 200,000 protected areas in 2020, an area amounting to 20.2 million km^2 (UNEP-WCMC et al., 2020). The extension of protected areas coincides with an 80% home range contraction of almost half of the mammals for which sufficient data are available (Ceballos et al., 2017; Pacifici, Rondinini, et al., 2020). The fraction of the home range of these mammals covered by protected areas is thus increasing, which underlines the importance of protected areas for conservation (Pacifici, Di Marco, et al., 2020).

Registered terrestrial protected areas in Kenya accounted for approximately 13% of its land territory in 2020 (Table 2.1). The government manages approximately 11% of this area as forest reserves, national sanctuaries, national parks, or national reserves (UNEP-WCMC, 2020b). This total excludes proposed protected areas, internationally designated areas (wetlands of international importance covered under the Ramsar Convention, biosphere and World Heritage sites covered under UNESCO), and overlaps between protected areas (UNEP-WCMC, 2020a; Visconti et al., 2013). The remaining 2% consists of registered wildlife conservancies (hereafter: conservancies).

KWCA reported an area of 63,600 km^2 for its member private and community conservancies (KWCA, 2016). This suggests that the total coverage of privately or community-run conservancies is 11% of Kenya's terrestrial land surface, of which only 2% is registered and included in the UNEP-WCMC database. Private and community conservancies cannot register themselves as such because the regulations for doing so are in draft since 2015 (Government of Kenya, 2015b; Ministry of Tourism and Wildlife, 2017). Consequently, most OECM are not covered by the Kenya Wildlife Conservation and Management Act, 2013 (Government of Kenya, 2013, 2017, 2018).

Table 2.1: Overview of terrestrial protected areas (PA) and other effective area-based conservation measures (OECM) in Kenya.

Protected area type	n	km^2	%	Management	Type
Conservancy, registered	51	9,812	2	Non-state	OECM
Conservancy, not registered	105	53,549	9	Non-state	OECM
Forest Reserve	199	15,265	3	KFS	PA
National Park	22	28,762	5	KWS	PA
National Reserve	31	15,950	3	KWS	PA
National Sanctuary	6	36	< 0.01	KWS	PA
Not protected	N/A	445,765	78	KWS	N/A
Totals	414	569,140	100		

Sources: Registered protected areas from (UNEP-WCMC, 2020b), processed in accordance with (UNEP-WCMC, 2020a). Total of non-state conservancies is derived from (KWCA, 2016).

2.2.2 Current concerns about protected areas

The current 20.2 million km^2 of terrestrial protected areas represent a major investment in *in situ* biodiversity conservation. However, researchers point out that using a "square kilometer" indicator to measure success is inappropriate if insufficient attention is given to conservation outcomes, biological representativeness, and effective and equitable management (Barnes et al., 2018; Visconti et al., 2019). There are three groups of issues faced by protected areas.

First, patrolling costs can take up to 66% of the annual operational budget of protected areas (Johnson et al., 2016; Plumptre, 2019). However, only 22% of them receive sufficient funding (Coad et al., 2019), while 33% is under intense human pressure (Geldmann et al., 2014; Jones, Cusack, et al., 2018). This means, for example, that only 4–9% of animals are living in a protected area that has sufficient resources to protect them (Coad et al., 2019; Lindsey et al., 2018). Second, protected areas do not necessarily contain a representative sample of biodiversity within their perimeters. Many areas are located on land that seems to be selected for its marginality rather than for its biodiversity value (Baldi et al., 2017; Joppa & Pfaff, 2009; Venter et al., 2018). Third, and most relevant for this research, the establishment of protected areas does not necessarily guarantee that species within their boundaries are protected. There is evidence that protected areas are more effective in protecting landscapes (Joppa et al., 2009; Riggio et al., 2019) than species (Geldmann et al., 2013). On average, protected areas may conserve biodiversity within their confines, but there are large individual and regional differences between them (Barnes et al., 2016; Geldmann et al., 2018). Populations of large mammals in African protected areas, for example, have declined with 59% on average over the period 1970–2005 (Craigie et al., 2010), despite the protection that these areas were supposed to give them.

Kenyan protected areas face similar issues. The "huge under-funding of operational costs" caused KWS to divert funds intended to restore ecosystem losses caused by the construction of the standard gauge railway and the southern bypass through Nairobi National Park. The funds were used to cover recurrent costs instead (KWS, 2017). Severe wildlife losses have been observed both inside and outside Kenya's protected areas (Ogutu et al., 2016; Western et al., 2009). The parks are not representative of the year-round migration pattern of wildlife (Fynn & Bonyongo, 2011; The World Bank, 2019; Western et al., 2009), and are thought to cover only 30–41% of the home range of mammal populations (Ogutu et al., 2016; Tyrrell et al., 2020). Most animals can thus be found outside the protected areas at least part of the year, while encroachment occurs on all 100 wildlife dispersal areas and corridors (Ojwang et al., 2017; Said et al., 2016). The resulting fragmentation of the protected area network can lead to inbreeding ("genetic erosion"), as observed in the relatively small and wholly fenced Lake Nakuru National Park (Heller et al., 2010).

2.3 Bushmeat poaching

The importance of bushmeat poaching in the savanna biome is increasingly recognized as a threat to biodiversity. This problem is recognized in Kenya, but research has concentrated on bushmeat poaching in Tanzania. Managers of protected areas and researchers have become aware of the extent and seriousness of bushmeat poaching by snaring.

2.3.1 Definitions

The intense human pressure on protected areas does not yet factor in any poaching that may occur in them (Geldmann et al., 2014; Jones, Cusack, et al., 2018). "Poaching" is understood as "the illegal taking of wildlife", with "wildlife" defined as "all forms of non-domesticated plants and animals in the wild" (Lemieux, 2014). There are, broadly speaking, three forms of animal poaching. Animals can be killed for their meat (bushmeat poaching), for parts of their body (e.g., skins, horns, bones) (trophy poaching), or as retaliation for harm to crops, livestock, or humans ("human-wildlife conflict").[1]

Other terms than "bushmeat poaching" are used in literature to describe the illegal killing of wildlife, and each of them introduces ambiguity. For example, KWS uses the general term "poaching" in its annual reports, whereas it usually refers to "trophy poaching" (KWS, 2017). Others use "commercial poaching", which can stand for either "bushmeat poaching on a commercial scale" or "trophy poaching" (Kideghesho, 2016; Lunstrum & Givá, 2020). Finally, the term "bushmeat poaching" is at times replaced by various euphemisms such as "harvesting", "unauthorized resource use", and "hunting and collecting". Each of these terms obfuscates the fact that animals are killed illegally, often through hunting methods that cause severe animal suffering.

[1] The term "human-wildlife conflict" will be placed in quotation marks throughout this study. This is to indicate that the term is not taken literally, as it is conceptually impossible for animals to have a conflict with humans or vice versa (Peterson et al., 2010). Moreover, the term frames animals as adversaries ("problem animals") in what is, in many cases, essentially a spatial planning failure.

2.3.2 Bushmeat poaching in the savanna biome

Bushmeat is an important source of proteins for an estimated 150 million households in Asia, Africa, and Latin America (Nielsen et al., 2018). Consequently, the bushmeat trade is estimated to be worth several billion dollars per year (Brashares et al., 2011). On the African continent, bushmeat hunting is considered a significant threat to biodiversity in West and Central African forests (Milner-Gulland et al., 2003; Tranquilli et al., 2014). Here, the human population is increasing, with a corresponding increase in the demand for bushmeat. However, there are insufficient alternatives for bushmeat, while the forest biome has a lower meat production capacity than the savanna biome. In Central Africa, this has led to an annual off-take of 1–3.4 Mt of bushmeat, which equals approximately six times the sustainable harvest (Milner-Gulland et al., 2003; Nasi et al., 2011). The resulting defaunation (the "empty forest syndrome" (Redford, 1992; Wilkie et al., 2011)) led researchers to focus on bushmeat consumption and poaching in the humid tropics. Bushmeat poaching in African savannas was considered, until recently, to be a low-impact, subsistence activity (Barnett, 2002; Lindsey et al., 2013).

However, mounting evidence suggests that bushmeat poaching in savannas is neither low-impact nor restricted to subsistence hunting (Lindsey et al., 2013). The effects are possibly as harmful as habitat destruction (Barnett, 2002), and bushmeat poaching is resulting in ”empty savannas” (Bouché et al., 2012; Lindsey et al., 2012; Ripple, Abernethy, et al., 2016). Large mammals are particularly affected (Ripple et al., 2015; Ripple et al., 2019; Wilkie et al., 2016). Bushmeat poaching has thus become an important concern for managers of protected areas in savanna biomes (Lindsey, Petracca, et al., 2017; Lindsey et al., 2015; Schulze et al., 2018).

2.3.3 Prevalence of bushmeat poaching

The prevalence and extent of bushmeat poaching in Africa's savannas remains unclear to date, and must be inferred from (i) signs of illegal activities, such as detected snares; (ii) declining animal populations; and (iii) interviews with households and poachers (van Velden et al., 2018). A review of available bushmeat research in African savannas found that, on average, 52% (median) of respondents admitted to bushmeat consumption, and 15% (median) to bushmeat poaching (van Velden et al., 2018). Poachers often hunted for cash rather than for subsistence (Nuno et al., 2013; Travers et al., 2019; van Velden et al., 2020), and the bushmeat trade has become increasingly commercialized (Brashares et al., 2011; Lindsey et al., 2013; van Velden et al., 2018).

The research on bushmeat poaching is heavily biased towards Tanzania, particularly the Serengeti ecosystem (van Velden et al., 2018). Early 2000's 1 million people were living along its boundaries and increasing with 2.9% per year (doubling time: ~ 24 years) (Loibooki et al., 2002). Households surrounding the area were found to consume, on average, 2.2–2.8 bushmeat meals per week. Calculating backward from these data points, Rentsch and Packer (2014) concluded that the annual off-take of wildebeest by bushmeat poachers amount to 6–10% of their population (98–141,000 wildebeests per year). Researchers found that ~17% of respondents were involved in bushmeat hunting (Fischer et al., 2014; Nuno et al., 2013), although this was locally as high as 46% (Ceppi & Nielsen, 2014). Approximately 34% of traders depended entirely on bushmeat trade (Barnett, 2002). They bought their meat from an estimated 52,000–60,000 poachers who make at least one hunting trip per year (Loibooki et al., 2002). Similar percentages for households involved in bushmeat hunting (19-40%) were found in communities adjacent to national parks in Uganda and Malawi (Solomon et al., 2007; van Velden et al., 2020).

2.3.4 Bushmeat poaching by snaring

The most frequently used bushmeat poaching method is the placement of snares (Gray et al., 2018; Lindsey et al., 2013). For example, the number of snares present at any time across the Serengeti ecosystem is estimated at around 137,000 (Rija, 2017). Snares can be made from various materials, such as utility cable, bicycle brake cable, winch cable, nylon, and sisal rope (Becker et al., 2013; Linkie et al., 2015; Mudumba et al., 2020; Woodroffe et al., 2014). The two main types of snare are neck snares and foot snares. Neck snares typically consist of a noose placed vertically above the ground and affixed to a tree that can withstand the struggling of the snared animal (Becker et al., 2013). Foot snares are designed to immobilize the animal and consist of a concealed noose on the ground which traps the leg of a passing animal (Noss, 1998).

Snares are cheap to make and difficult to detect (Ibbett et al., 2020; O'Kelly et al., 2018a; Rija, 2017). This form of hunting is non-selective, resulting in by-catch of non-target animals (Becker et al., 2013; Campbell et al., 2019; Loveridge et al., 2020). Snaring is a wasteful form of hunting: up to one-third of the snared animals escape with an injury. Approximately three out of four escaped animals die of snare-inflicted injuries (Loveridge et al., 2020). Moreover, a substantial proportion (25–63%) of snared animals is not recovered in time by the poachers and is lost through rot and scavengers (Mudumba et al., 2020; Noss, 1998).

A summary of snaring-focused research is included in Appendix A. A wider overview of poaching research can be found in Appendix B.

Research specifically looking into bushmeat poaching by snaring is scarce, despite the perceived severeness of the problem. Out of 112 papers that used primary data (transects, patrol data, camera traps) to analyze poaching, 26 papers mentioned snaring but placed this in the context of illegal activities in protected areas in general. Only 17 papers focused explicitly on snaring, and eight papers further zoomed in on bushmeat poaching.

Snares are often found in clusters (hotspots), near the boundaries of protected areas, and near roads (the so-called "edge effect") (Duporge et al., 2020). Clustering of crime locations is analyzed and predicted by criminological theory. However, none of the listed research papers placed their findings within this theoretical framework.

Researchers frequently detect snares near ranger posts (Jenks et al., 2012; O'Kelly et al., 2018b; Watson et al., 2013). Different and sometimes contradictory explanations are provided for the unexpected occurrence of poaching in the direct proximity of these crime deterrents. A first possible explanation is the involvement of rangers in poaching (Jenks et al., 2012). A second explanation, suggested by (O'Kelly et al., 2018b) was that ranger stations are set up in areas where poaching is more likely to occur; more poaching observations will thus be made near these stations. An opposite trend was found by Denninger Snyder et al. (2019), who found that there are fewer observations of poaching signs near ranger stations, presumably due to deterrence. Finally, Watson et al. (2013) suggested further research into this matter.

Three papers examined the detectability of snares under field conditions. Two relatively small field experiments (search plots of 0.25 km^2 and 22 km^2 respectively) found a snare detectability rate of ± 20% in forests (Ibbett et al., 2020; O'Kelly et al., 2018a). This rate refers to a known number of snares that were recovered by rangers and is further specified by Ibbett et al. (2020) as a search effort of a one hour two-kilometer transect covering 0.25 km^2. A larger field experiment (5,200 km^2) in the savanna biome found, however, a snare detectability of only 3% (Rija, 2017). Here, each ranger covered 140 km^2, whereas the covered area in the other experiments was much smaller (0.08 km^2/person in Ibbett et al. (2020) and 0.33 km^2/person in O'Kelly et al. (2018a) respectively).

2.3.5 Drivers and consequences of bushmeat poaching

A rapidly increasing population creates more demand for meat (van Velden et al., 2018). Uncontrolled growth of settlements near the borders of protected areas creates a larger pool of potential poachers near a source of meat supply (Lindsey et al., 2012; Lindsey et al., 2015; Rentsch & Packer, 2014; Ripple et al., 2015; Watson et al., 2013). People are more likely to poach in protected areas once the land outside them has been defaunated (Lindsey et al., 2013; Ripple et al., 2015). Protected areas with an insufficient budget for law enforcement can thus become open access resources (Nielsen et al., 2014; Ripple et al., 2015).

Interviews with poachers suggest that bushmeat poaching is a profitable and low-risk activity. In the Serengeti ecosystem, the risk of arrest was estimated at 0.07% per poaching trip. Just 4% of the poachers were arrested more than once (Loibooki et al., 2002). Furthermore, arrested poachers face a relatively small risk of conviction and punishment (Salum et al., 2018). The revenues of bushmeat poaching outstrip the costs of arrests with an order of magnitude (Hofer et al., 2000; Knapp, 2012). The net benefits of bushmeat poaching can be two to three times as much as agricultural revenues (Knapp, 2012). Consequently, bushmeat poachers do not belong to the poorest segment of the population (Knapp et al., 2017; Nielsen et al., 2014; Twinamatsiko et al., 2014; van Velden et al., 2020).

The costs from bushmeat poaching are shifted on to society and come in the form of lost revenue and taxes from tourism, lost biodiversity, economic and human costs from outbreaks of zoonotic diseases, and lost ecosystem services (Lindsey, Romañach, Tambling, et al., 2011; Rogan et al., 2017). Poaching and butchering of wildlife create ideal conditions for transmitting zoonotic diseases (Karesh & Noble, 2009; Wilkie, 2006). Indeed, bushmeat samples taken in the Serengeti from a wide range of animals contained traces of potentially dangerous zoonotic pathogens (Katani et al., 2019). Poachers do, however, not take precautions against possible contamination, even if they are aware of the possibility thereof (Alhaji et al., 2018; Dell et al., 2020).

2.3.6 Interventions to mitigate bushmeat poaching

There are three groups of interventions that aim to reduce bushmeat poaching, namely enforcement, community development, and sourcing alternative proteins. The frequency with which researchers assess or recommend these interventions is summarized in Table 2.2. Monitoring is not an intervention but is regularly proposed and has been included here for completeness.

Table 2.2: Proposed interventions aimed at reducing bushmeat poaching

	Researched		Recommended	
Intervention	n	%	n	%
Enforcement	22	46	84	39
Community development	16	33	82	38
Alternative proteins	10	21	24	11
Monitoring	0	0	24	11
Totals	48	100	214	100

Source: van Velden et al. (2018).

2.3.6.1 Enforcement approach

The enforcement approach (”fences and fines”, Songorwa (1999)) forms the bulk of researched and proposed interventions against bushmeat poaching. There are several issues with this approach. Sufficient enforcement budgets are not always a guarantee for wildlife protection, as even well-resourced wildlife authorities can be overwhelmed by poaching pressure (Barichievy et al., 2017; Rogan et al., 2017). When taken too far, enforcement can turn into militarization, with the subsequent loss of trust from communities (Duffy et al., 2019; Mabele, 2017). There is also no guarantee that law enforcement can keep up with the poaching pressure caused by the increasing populations surrounding the protected areas (Challender & MacMillan, 2014; MacKenzie et al., 2017).

Severe penalties for arrested poachers often accompany the "fences and fines" approach. Authorities may be tempted to introduce extreme fines when the poacher detection probability is small, for example, due to insufficient law enforcement funding (Lindsey et al., 2020; Plumptre, 2019). This approach is not supported by criminological theory. Poachers are not necessarily aware of penalty severeness, the costs associated with incarceration for society are high, and potential offenders are not necessarily deterred from committing offenses (Wilson & Boratto, 2020).

In recent years, researchers have shown increasing interest in technologies for the detection of poachers (Kamminga et al., 2018; Kretser et al., 2017; Martin, 2019; Spillane, 2018). For example, acoustic and camera traps are well-established technologies that are applied in the field. Both types of traps were able to collect more poaching signs than would be expected on the basis of patrolling data (Astaras et al., 2017; Hossain et al., 2016), and were recommended for supporting patrol data. Likewise, there has been much interest in the use of drones in wildlife conservation. However, the logistical challenges and costs are currently prohibitive for most PAs (Harvey, 2015; Oxpeckers, 2016; WWF, 2019) despite initial optimism (Jiménez López & Mulero-Pázmány, 2019; Martin, 2019). Besides, many African governments are reluctant to allow the operation of drones within their territories (Linchant et al., 2015; WWF, 2019).

A recent development (2020) which may assist rangers in detecting poachers is the combination of machine learning and cheap (< 10 USD) low-power (< 1 mW) micro-controllers (Warden & Situnayake, 2020). The miniaturization of machine learning programs allows the installation of deep learning (Patterson & Gibson, 2017) algorithms on embedded devices. This allows a sensor to classify ("recognize") objects, such as moving humans, and make decisions on the basis thereof. Such sensors may be connected, thus extending their reach and improving the resilience of the network of which they are part ("internet of things") (Kacprzyk, 2020). The combination of machine learning and the internet of things ("IoT") are being pioneered in several sectors, such as telecom, energy, and agriculture (Mathur, 2020), but hardly in the conservation of wildlife (Spruyt, 2017).

2.3.6.2 Community development approach

The community development approach seeks to reduce bushmeat poaching through community involvement in the management of natural resources (van Velden et al., 2018). This approach has been widely implemented over the years (Roe & Booker, 2019; van Velden et al., 2018).

Conservation through community development has, however, two important caveats. First, population growth at the edges of protected areas without sufficient enforcement budgets and spatial planning remains problematic (Lindsey et al., 2015; Watson et al., 2013). It leads to retaliatory killings ("human-wildlife conflict"), bushmeat poaching, illegal firewood collection, and illegal grazing (Lindsey et al., 2013). The benefit-generating capacity of protected areas is thus undermined by the very people with whom the benefits are supposed to be shared (Green et al., 2018).

Second, recent reviews looking into community-based development projects' effectiveness came to the startling conclusion that outcomes are seldom monitored or evaluated (Roe & Booker, 2019; Roe et al., 2015; van Velden et al., 2018). Out of 106 projects, only 21 were monitoring effects on biodiversity conservation, and just one project could demonstrate positive effects of its intervention (Roe et al., 2015). Researchers who examined the relation between snaring densities and community development found that community development projects do not reduce poaching, even when the presence of wildlife is vital for the local economy (Muchaal & Ngandjui, 1999; Twinamatsiko et al., 2014; van Velden et al., 2020; Watson et al., 2013). In sum, the benefits of community-based development projects for conservation seem to be self-evident to many policy-makers, but the supporting evidence is, at best, weak.

2.3.6.3 Alternative protein source approach

The last intervention type is the replacement of bushmeat with alternative protein sources. One alternative is to allow commercial use of wild animals in their own habitat (game cropping). Researchers who examined this option in Serengeti found that the benefits from bushmeat poaching dwarfed the legal benefits from game cropping (Holmern et al., 2002).

There are also conservation issues with this type of intervention. Legalizing game meat creates a market in which illegal and legal meat cannot be distinguished from each other. Furthermore, not all wildlife species will be commercially viable. This may lead to a conservation focus on selected species rather than ecosystems (Macnab, 1991).

2.3.7 Bushmeat poaching in Kenya

The threat of bushmeat poaching for biodiversity has been recognized by both KWS and managers of Kenyan protected areas. Nevertheless, there is almost no peer-reviewed research on bushmeat poaching in Kenya; evidence is anecdotal. Kenya's primary strategy for mitigating bushmeat poaching is an enforcement approach with increasingly severe penalties.

2.3.7.1 Prevalence of bushmeat poaching in Kenya

Bushmeat poaching in Kenya was seen as a misdemeanor until recently (Kahumbu et al., 2014; Secretariat of the Convention on Biological Diversity, 2011). However, a survey in 2002 revealed that 80% of the households in Kitui District (now: Kitui County) was consuming ~ 14 kg of bushmeat per month. Moreover, tribes who traditionally did not eat bushmeat were starting to do so. Consequently, the bushmeat trade had become commercialized (Barnett, 2002). Managers of protected areas in Kenya started to see bushmeat poaching as a significant problem (Kiringe et al., 2007; Okello & Kiringe, 2010). KWS stated that it considered bushmeat poaching as "one of the most serious threats to wildlife populations and wildlife-based community development" (Secretariat of the Convention on Biological Diversity, 2011).

The amount of peer-reviewed research on bushmeat poaching in Kenya is, however, minimal. Eleven scholarly articles on poaching in Kenya have been published in the period 2000–2020. Only two of these focus on bushmeat poaching (Kimanzi et al., 2014; Wato et al., 2006); the remaining nine articles discuss elephant poaching. Evidence for bushmeat poaching in Kenya remains anecdotal. Government-appointed task forces reported "unprecedented levels of bushmeat poaching" (Rotich et al., 2014) and "rampant illegal bushmeat trade", while simultaneously admitting that neither the magnitude nor the impact of the bushmeat trade are not well-understood (Okita-Ouma et al., 2019). The National Wildlife Strategy 2030 considers bushmeat poaching to be a "serious threat to species survival" but contains no references to support this claim (Ministry of Tourism and Wildlife, 2018).

Publicly available information on bushmeat poaching in Kenya is scarce. DSWT desnaring teams operating in the Tsavo area removed 5473 snares and arrested 50 bushmeat poachers in August 2019–August 2020 (DSWT, 2020b). A Task Force looking into wildlife security referred to a 2009 study in which 59 bushmeat poaching gangs in the Tsavo Conservation Area were identified (Rotich et al., 2014, p. 20). The report does, however, not cite the source of this information.

2.3.7.2 Interventions to mitigate bushmeat poaching in Kenya

Law enforcement is the main strategy for the mitigation of poaching in Kenya. Bushmeat poaching and trading is illegal, regardless of whether it occurs inside or outside a protected area, and regardless of whether it is for commercial, subsistence, or any other purposes (Government of Kenya, 2013). The revision of the Wildlife Conservation and Management Act, 2013 (KCMA, 2013) increased the penalty for bushmeat trading to at least three years imprisonment without the option of paying a fine. In contrast, buyers of bushmeat can face one-year imprisonment or one million KSh fine (1 USD=107.80 KSh), or both (Government of Kenya, 2018, s.98). Court cases for poaching are relatively rare. The numbers of trophy-related and bushmeat-related poaching cases were practically identical in recent years (bushmeat-related poaching cases in 2017: 209, ditto trophy-related: 212). The number of bushmeat-related cases decreased from 281 (2016) to 209 (2017) (Kahumbu et al., 2018).

A Task Force examining wildlife security in Kenya found that KWS has continued to focus its surveillance efforts on trophy hunting (mainly elephants and rhinos) (Rotich et al., 2014). Mitigation of bushmeat poaching did not seem to be seen as a priority. KWS was found to have insufficient resources for enforcement and intelligence; for example, suspects of poaching often have to be released because the collected evidence is insufficient. These findings were confirmed by an audit by Kenya's Auditor-General, who found that KWS has no monitoring and evaluation capacity and insufficient capacity for enforcement and prosecution (Ouko, 2018).

The objectives for the improvement of law enforcement are formulated in strategic plans. The KWS Strategic Plan 2012–2017 plans to "strengthen law enforcement & security" but provides no further details on how this would be implemented (KWS, 2013). The National Wildlife Strategy 2030 (NWS 2030) foresees to increase the coordination, capacity, and effectiveness of wildlife security units by expanding and modernizing them (Ministry of Tourism and Wildlife, 2018, Strategy 2.2). However, the formulation of these actions does not provide immediate insight into what will be done, by whom, and when. The NWS 2030 further emphasizes the importance of maintaining wildlife corridors. Nevertheless, the national spatial plan 2015–2045 does not contain any provisions for this (Government of Kenya, 2016).

Education and awareness of the population is emphasized in wildlife conservation policies which target reduction of "human-wildlife conflict" (Ministry of Tourism and Wildlife, 2018, 2020; Ongalo, 2019), However, these terms do not occur in the context of abatement of bushmeat poaching. The expectation that community members change their behavior when education and awareness are provided based on the "knowledge deficit model" (Simis et al., 2016). However, behavioral change occurs as a function of motivation and is not caused by the provision of environmental information alone (Schultz, 2011).

Game cropping was banned in 2002 but has come under renewed attention with the report of a task force on consumptive wildlife utilization (CWU) (Okita-Ouma et al., 2019). Bushmeat trade is included in CWU and therefore examined by the task force. The task force could only state that this trade is "rampant", without quantification or provision of data sources. Further research was recommended.

In summary, severe sanctioning of bushmeat poaching has been the primary approach to mitigate bushmeat poaching in Kenya. There is, however, no visibility on the effectiveness hereof as the monitoring and evaluation capacities of KWS are considered to be limited.

2.4 Estimating patrolling effectiveness

The estimation of patrolling effectiveness of rangers in a protected area involves (1) defining "effectiveness"; (2) defining "patrol effort"; and (3) assessing the relation between poaching signs and patrol efforts.

2.4.1 Patrolling effectiveness in context

"Patrolling effectiveness" is not defined in wildlife security literature. The term "effectiveness" is used in a wide range of biodiversity conservation contexts (Rodrigues & Cazalis, 2020). It may refer to "management effectiveness" (Geldmann et al., 2013; Lham et al., 2019), "relevance" (Andam et al., 2008; Bruner et al., 2001; Lindsey, Chapron, et al., 2017), "efficiency" (Jachmann, 2008; Plumptre et al., 2014), or "coverage effectiveness" (the extent to which the protected area represents biodiversity) (Chape et al., 2005; Nord et al., 2019) . The relation between efficiency, effectiveness, and relevance as defined by the Organization for Economic Co-operation and Development (OECD) (OECD/DAC Network on Development Evaluation, 2019) is shown in Fig. 2.1.

In this study, "patrol effectiveness" is interpreted as the extent to which poachers have been deterred from entering the protected area (outcome) as a function of patrol effort (output). Patrol effectiveness is a counter-factual term: it expresses poaching that would have occurred in the absence of patrolling (Ferraro, 2009; Rodrigues & Cazalis, 2020).

Another way to think about patrolling effectiveness is that the effective size of protected areas is reduced when it is insufficient, as summarized in the following quote:

> "Of course, if we want to know the effective size of a marine reserve, then we have to multiply its area by the probability that a rogue fisherman will be caught and punished if he poaches in the reserve."
>
> – Geoff Kirkwood, in Dobson and Lynes (2008).

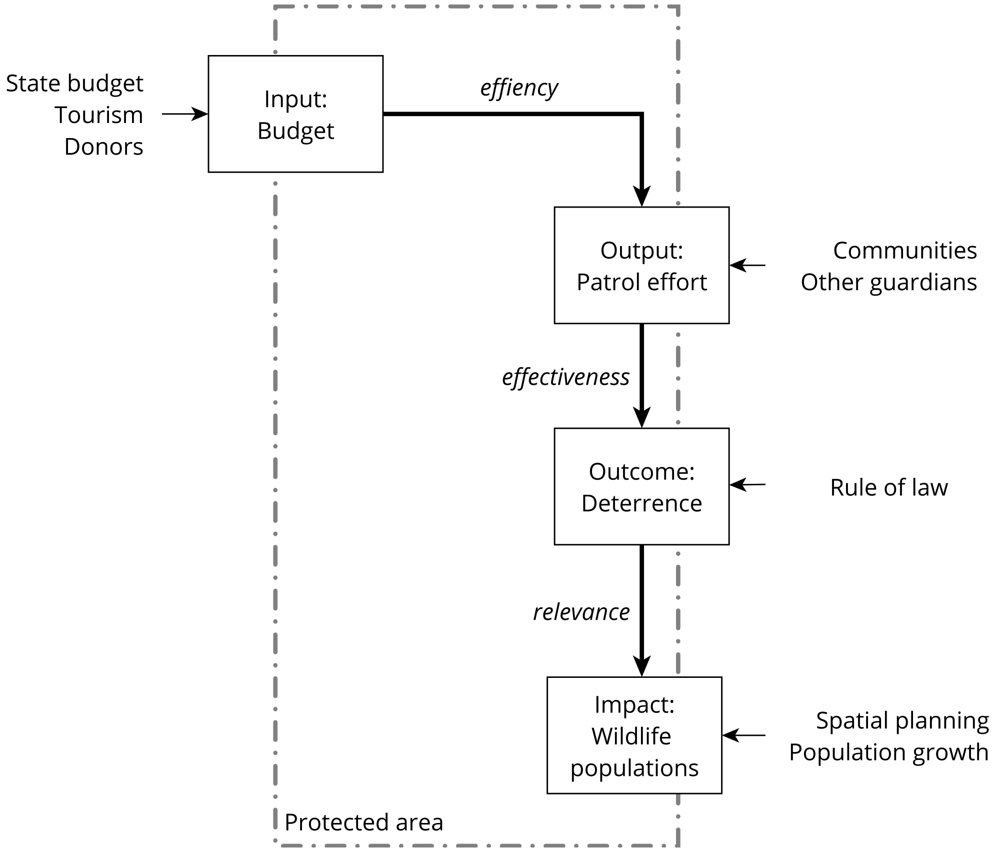


Figure 2.1: Relations between efficiency, effectiveness and relevance in the context of providing wildlife security in protected areas.

2.4.2 Patrol effort

"Patrol effort" can be characterized in three ways. *Ranger density* compares the size of the ranger force over the size of the protected area. The *catch per unit effort (CPUE)* is the quotient of the number of observed poaching activities and the (square) kilometers or hours patrolled. The *guardianship* concept considers patrol effort as a composite of ranger presence, ability to detect illegal activities, and capability to intervene.

2.4.2.1 Ranger density and unit effort

An estimation of patrol effort is obtained by dividing the number of rangers over the protected area's size, or vice versa. This results in a rough indicator of protection input, namely density in rangers/km^2 or km^2/ranger. The ranger densities in protected areas can be compared with recommended values. KWS recommends 6 km^2 per ranger (Ouko, 2018), while KFS recommends

14 km^2 per ranger (Kamau et al., 2018). Recommended ranger densities for rhino and elephant protection are 20 and 50 km^2/ranger respectively (Henson et al., 2016). Actual ranger densities in Kenyan protected areas are shown in Table 2.3.

Table 2.3: Overview of ranger densities in Kenya per management category.

Management	Area (km^2)^a	Rangers (n)	Density (km^2/ranger)^b	Source
KWS	44,748^c	3,569	13	Ouko, 2018
KFS	15,265	2,542	6^d	Kamau et al., 2018
Non-state	63,361	2,991	21	KWCA, 2016

[a] The areas are derived from Table 2.1.
[b] Densities were calculated as Area (km^2)/Rangers (n).
[c] Ouko (2018) restricted the covered area to direct KWS management.
[d] Kamau et al. (2018) calculated ranger density as 9.7 km^2 per ranger, using an unspecified area.

The ranger densities for both protected areas and OECM listed in Table 2.3 are within the recommended ranger densities for rhino or elephant protection. However, the ranger density indicator is crude because it does not indicate whether the rangers are actively patrolling in the field, assigned to administrative duties, or stuck at a ranger station due to logistical problems. An example of a KWS elite unit stationed in a remote location with insufficient fuel and unsuitable vehicles was given by Rotich et al. (2014, p. 69). The ranger density indicator is nevertheless still used by researchers (Bruner et al., 2001; Dancer, 2019; Ghoddousi et al., 2017) and enforcement agencies, such as KWS (Ouko, 2018).

The introduction of unit patrol effort as ranger hours in the field or kilometers patrolled is a partial improvement over the ranger density indicator. It leaves non-patrolling rangers out of consideration. Dividing the number of observed illegal activities (“catch”) over the patrol hours in the field or kilometer patrolled (“unit effort”) normalizes the observed poaching activity over the patrol effort and produces the ”Catch per unit effort” (CPUE) indicator. The CPUE indicator is increasingly used to interpret patrol effectiveness and prediction of poaching (Critchlow et al., 2016; Gholami et al., 2017; Kar et al., 2017; Moore et al., 2018). The use of the CPUE indicator has gained popularity in tandem with the introduction of software to register both patrol movements and observations, such as MIST (Ecological Software Solutions LLC, 2020) and

SMART (SMART Development Team, 2017). Interpretation of the CPUE indicator is difficult, because the number of observations is related to the quality of the patrol effort; this will be discussed in more detail in the next section.

A different perspective on patrol effort is provided through the guardianship concept (Reynald, 2009). Here, the patrol effort is decomposed in the components that define the roles of rangers, namely: (1) ranger presence in a protected area, (2) ranger capacity to detect illegal activities, and (3) rangers' capability to intervene upon detection. The guardianship concept can be used for a qualitative evaluation of deterrence (Hollis-Peel et al., 2012; Reynald, 2009). The disadvantage of guardianship is that it cannot be easily calculated as a single indicator.

Anyone present at the (potential) crime site reminds the (potential) offender that someone may be watching and detecting the crime and thus functions as guardian (Hollis-Peel et al., 2011). For example, tourists' presence has been proven to have a deterrent effect on poachers (Jachmann et al., 2011; Kablan et al., 2017; Kyando et al., 2017; Piel et al., 2015). Moreover, a similar effect has been found for the presence of researchers (Campbell et al., 2011; Kablan et al., 2017; Laurance, 2013; Piel et al., 2015).

2.4.3 Uncertainties and biases in observed poaching trends

The main approach for assessing patrol effectiveness in protected areas is the interpretation of the CPUE trend. The assumption is that increased patrolling will lead to either increased encounter rates (and arrests), deterrence of poachers, or both (Hilborn et al., 2006; Keane et al., 2011; Moore et al., 2018). Low or decreasing arrest rates would therefore indicate that deterrence occurred (Dobson et al., 2018). However, early research warned on the use of CPUE rates (Gray & Kalpers, 2005; Stokes, 2010; Walston et al., 2010). Several simultaneously occurring and interacting mechanisms can lead to a misinterpretation of the CPUE trend. These are shown schematically in Fig. 2.2.

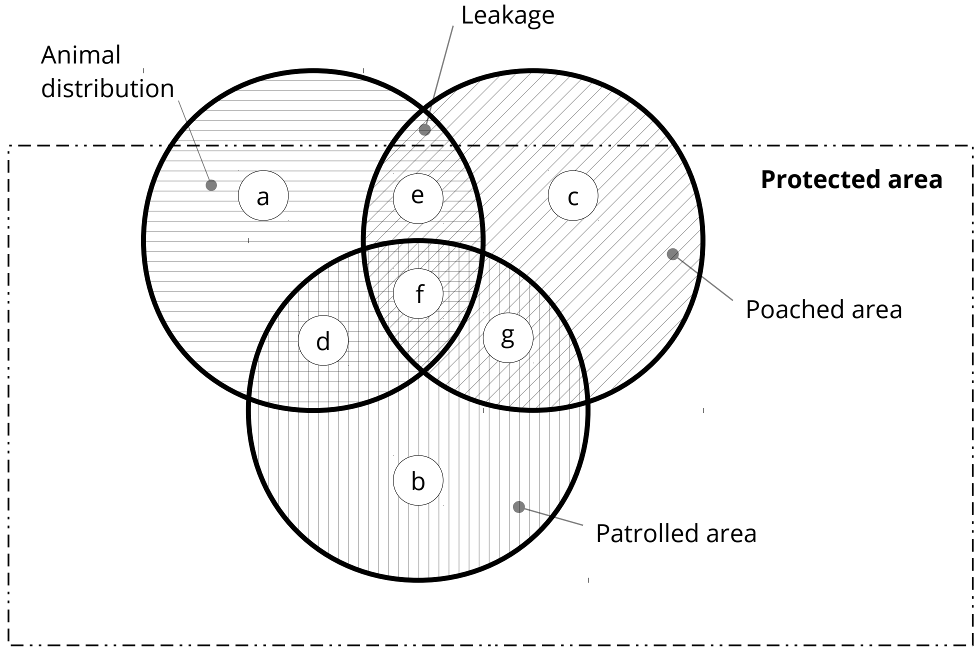


Figure 2.2: Spatial schema of overlap between animal distribution (a), patrolled areas (b), and poached areas areas (c) within a protected area. Poaching can occur in areas (e) and (f), Arrests of poachers can take place in areas (f) and (g) only. Observations of poaching activities originate from areas (f) and (g); poaching in area (e) is not visible for rangers. Neither poachers nor rangers are certain about the size of the spatial intersections.

All processes shown in the figure can coincide and interact. Isolating the deterrent effect of ranger patrols requires controlled experiments which are not feasible within individual protected areas (Rodrigues & Cazalis, 2020), whereas patrol efforts worldwide are currently considered insufficient for monitoring illegal activities (Dancer, 2019). Furthermore, the detection of changes in illegal activities within protected areas requires prohibitively large sample sizes, even when the effect is large (Jones et al., 2017). The situation is further complicated because animals are silent victims: they are utterly dependent on rangers and third parties for reporting crimes against them (Lemieux et al., 2014). This results in an under-reporting of true poaching prevalence, regardless of other processes that may be at work.

2.4.3.1 Changes in the population at risk

The absence of observed poaching signs in an area (f) can be caused by animals migrating away (Holmern et al., 2007; Rentsch & Packer, 2014) or extirpation of the animal population (Jachmann & Billiouw, 1997; Vanthomme et al., 2017). The habitat outside the protected area may have been destroyed, resulting in a population reduction within the protected area (Kritzer, 2004; Moore et al., 2018; Woodroffe & Ginsberg, 1998).

2.4.3.2 Fluctuations in poaching pressure

Poachers may not be available for detection because there is temporarily less poaching. For example, the demand for bushmeat rises during festive periods (Dong et al., 2018; Risdianto et al., 2016), and during lulls in the agricultural calendar (Maingi et al., 2012; Nyirenda et al., 2015; Wilfred & Maccoll, 2014).

2.4.3.3 Spatial displacement

Poachers can become aware of patrolling patterns and displace poaching activities to area (e). Rangers may not patrol parts of the protected area because of security issues (Gray & Kalpers, 2005; Nolte, 2016) or logistical problems (Ghoddousi et al., 2016; Rotich et al., 2014). Patrols may also prefer certain areas such as known poaching hotspots (Moreto & Matusiak, 2016). This creates *patrol bias*: not every place in the protected area is equally likely to be patrolled (Keane et al., 2011; Kuiper, Kavhu, et al., 2020; Kyando et al., 2017). Poachers can observe predictable patrolling patterns and displace their activities to a different area within the protected area (Herbig & Warchol, 2011; Hötte et al., 2016; Knapp et al., 2010; Mmahi & Usman, 2019; Moreto & Matusiak, 2016; Rija, 2017). Law enforcement within the protected area can also lead to the displacement of poaching activities to unpatrolled sites beyond the park's boundaries ("leakage") (Andam et al., 2008; Ewers & Rodrigues, 2008; Renwick et al., 2015).

2.4.3.4 Temporal displacement

Poacher awareness of patrolling patterns can also lead to the displacement of poaching during other parts of the day or year. Undetected poaching may increase when there is low ranger presence during the night, weekends, and holiday periods (Dong et al., 2018; Herbig & Warchol, 2011; Nolte, 2016; Ouko, 2018; Wilfred & Maccoll, 2014). For example, Kenya's Auditor General found that 90% of poaching in Kenya occurs during the night (Ouko, 2018).

2.4.3.5 Change of poaching method

Poachers can reduce detection rates by switching to a different poaching method. Researchers frequently observe a switch from noisy poaching methods, such as using guns, towards the use of snares (Henson et al., 2016; Holmern et al., 2007; Jachmann, 2008; Johnson et al., 2016; Nahonyo, 2009).

2.4.3.6 Detectibility

Poaching signs and poachers may be hard to detect. Some signs, such as tracks and carcasses, may disappear over time (Kahindi et al., 2010; Keane et al., 2011; Lemieux et al., 2014). Snares have a detectability in the 3–30% range and can therefore easily go undetected (Ibbett et al., 2020; O'Kelly et al., 2018a; Rija, 2017). Poachers can decrease detectability by operating during the night, especially when rangers do not have night vision equipment at their disposal (Ouko, 2018). Moreover, patrols can miss poaching signs because they patrol from vehicles rather than on foot (Rija, 2017). Some patrols may not be trained for snare detection or not focusing on bushmeat poaching (Wato et al., 2006). Poachers can also hide their tracks (Astaras et al., 2017; Rija et al., 2020; Wrangham & Mugume, 2000). Circling vultures above poaching locations can give away poachers' location, who therefore poison them (Mmahi & Usman, 2019).

2.4.3.7 Under-reporting

Rangers may see poaching signs without reporting them (Fig. 2.2, areas (f,g)). For example, rangers may remove snares during patrols without reporting the number and locations of these snares. Alternatively, signs of poachers and poaching signs may not be reported because rangers are demotivated or compromised. Ranger morale can be reduced by low conviction and punishment rates of poachers, who have been arrested with physical risks for rangers (Moreto, 2016).

Researchers who interview rangers invariably find that verbal and physical aggression is used against them. Nearly 73% of the surveyed African rangers have been threatened by poachers, and 66% has been attacked by them; 71% was threatened by community members (Singh et al., 2020). Similarly, many of the 2,061 African rangers who participated in a global survey reported that they had been subject to verbal abuse (38%), threats (42%), or physical violence (14%) from community members in the last 12 months alone. Also, nearly 2 out of 3 rangers thought they were not paid a fair wage (Belecky et al., 2019). Rangers may thus become involved in poaching ("inside poaching" (Moreto et al., 2015)), assist poachers by providing them with information or patrolling patterns (Lindsey, Romañach, Matema, et al., 2011), or become compromised by accepting bribes (Mmahi & Usman, 2019; Mubalama, 2010; Robinson et al., 2010). For example, Lindsey, Romañach, Matema, et al., 2011 found that 47% of arrested poachers in a protected area had received assistance from rangers.

2.4.3.8 Deterrence

The objective of patrolling is to discourage poachers from entering the protected area. Deterrence theory assumes that potential offenders weigh the probabilities and consequences of their actions (Becker, 1968). This assumes not only rationality but also that the potential offender knows the law and can make a reasonable estimate of the probabilities of being apprehended, convicted, and fined (Wilson & Boratto, 2020). Researchers who interviewed rangers found that these probabilities are low (Eustace, 2017; Hofer et al., 2000; Knapp, 2012). Increasing the punishments for non-compliance with wildlife laws will not be effective under these circumstances (Lab, 2010; Linkie et al., 2015; Wilson & Boratto, 2020). Low conviction and punishment rates demotivate the rangers who may have taken physical risks to arrest poachers (Moreto, 2016).

2.5 Use of ranger expertise for improving patrolling effectiveness

The role of rangers is in quantitative anti-poaching research is, in most cases, restricted to the supply of observation data. In recent years, studies have appeared with ranger involvement in evaluating poaching prediction models (Koen et al., 2017) and participation in the development thereof (Kuiper, Kavhu, et al., 2020).

The relative absence of ranger expertise results in a knowledge gap on patrolling strategies. In most cases, rangers patrol areas where they think poaching may occur; they do not take a representative sample of the protected area (Gray & Kalpers, 2005; Keane et al., 2011; Stokes, 2010). No research has been implemented to examine how rangers decide upon places and times to patrol sites within the protected area. Out of 105 articles that contain primary data on patrolling terrestrial protected areas, ten articles included interviews with rangers in the scope of their fieldwork (Appendix A). In just one of these studies, the choice of patrolling locations is touched upon: rangers stated a preference for repeat visits of poaching hotspots (Moreto & Matusiak, 2016).

Rangers are generally not involved in the use and management of data they generated (Kuiper, Massé, et al., 2020). Consequently, data quality problems will have to be found downstream in the reporting pipeline by environmental managers and researchers. Such problems are seldom discussed but can be substantial. For example, patrol data quality problems required discarding 20% of the ranger-collected observation data (Lemieux et al., 2014). In another case, all spatial patrol track data had to be omitted from analysis (Kablan et al., 2017).

The dominant approach to analyzing patrol effectiveness – statistical analysis – and the alternative approach – elicitation of ranger experience – both have their merits and shortcomings, as discussed in the following sections.

2.5.1 Strengths and limitations of statistical decision models

The implicit endorsement of statistical decision making for improving patrolling effectiveness is perhaps informed by the well-established fact that even simple decision models will outperform human experts under certain conditions (Dawes, 1979; Goldberg, 1970; Meehl, 1954). Statistical models are better than humans in finding weak regularities in predictor variables (Kahneman

& Klein, 2009). Besides, statistical models' classification accuracy is better than that of human expert judgments because the identified decision rules are applied consistently. Also, statistical models are not sensitive to a range of biases that can influence human decision-making (Tversky & Kahneman, 1974).

Statistical decision models have several limitations, which are usually not considered in anti-poaching literature. The development of statistical models may be feasible when (1) there is a complete and accurate set of clues (also known as predictors, independent variables, and explanatory variables) available, (2) a reliable and measurable criterion (also known as a response, dependent variable, and outcome) is available, and (3) the environment from which these variables are obtained is stable (Kahneman & Klein, 2009; Klein & Klinger, 1991; Shanteau & Stewart, 1992).

However, there is no scientific consensus on the set of predictors related to the occurrence of poaching. Research articles that applied statistical analysis (n=83) used 15 distance-related predictors, such as distance from poaching event to the nearest ranger post, and 55 other predictors, such as slope, wetness, season, and moon phase (see Appendix A for an overview). The predictor sets used in these models are not only often diverging but are also sometimes contradictory. There is, therefore, neither consensus on the selection of predictor variables nor the size and direction of the effect of these variables. Furthermore, the environment in which statistical models are developed may change, for example, due to population growth and spatial planning failures in and around protected areas. This can render the statistical model obsolete without the user necessarily realizing so.

2.5.2 Strengths and limitations of human experts

In field conditions, information is often incomplete, and there is insufficient time and resources to process all available options and probabilities (Klein, 2011; Klein & Klinger, 1991). Human decision-making involves the reduction of the number of cues used in order to reduce uncertainty (Gigerenzer & Gaissmaier, 2011; Hafenbrädl et al., 2016; Pachur & Marinello, 2013; Shanteau, 1992). For example, offenders may base the selection of their victims on a single clue (Garcia-Retamero & Dhami, 2009).

Judgments from experts are not necessarily reliable. Humans can build up reliable expertise only if (1) the environment is sufficiently regular and providing observable cues, (2) feedback is timely and accurate, and (3) there is sufficient time to learn the relations between cues and outcomes (Kahneman & Klein, 2009; Klein, 2015; Shanteau & Stewart, 1992). Rangers can therefore be expected to develop effective patrolling strategies if the poaching activity cues are visible and regular ("environment validity") and if they have had sufficient time to observe them (experience).

Researchers have not yet studied the cues which poachers use to decide where, when, and how to poach. This is surprising for two reasons. First, the inclusion of ex-poachers in patrols leads to an improvement of the detection of poaching activities (Moore et al., 2018). The expertise of these ex-poachers was apparently not captured in the business-as-usual patrolling strategies. Second, the development of statistical or game-theory models based on a broad set of predictor variables is counter-productive if poachers use a strongly reduced set of clues. These models do not mimic the actual poacher decision process and are therefore not likely to produce accurate results. Moreover, the enlarged set of predictor variables can lead to over-fitting. An over-fitted model performs well on known data (hindsight) but fails when new data are provided (foresight) (Gigerenzer, 2008; Goldstein & Gigerenzer, 2009). Overfit is unlikely to be detected in current wildlife research because field validation of predictive or descriptive models is both rare and under-powered (Jones et al., 2017).

Expert judgment can be made more robust by pooling forecasts from a diverse group of individuals, a phenomenon known as the *diversity prediction theorem* (Hong & Page, 2004; Page, 2007a). This phenomenon has been known for over a century (Galton, 1907) and has been applied to a wide range of decision problems, such as prediction of scientific research reproducibility (Dreber et al., 2015), market predictions (Arrow et al., 2008) and crowdsourcing (Guazzini et al., 2015; Noveck, 2017).

2.6 Environmental criminology

Poaching is, by definition, an offense. Therefore, applying criminology in poaching research would be required to underpin empirical observations with a theoretical framework. One group of theories, environmental criminology, is of particular interest for the analysis of poaching. ”Environmental” in “environmental criminology” refers to the spatial context in which crimes occur and should not be confused with environmental crimes. These theories attempt to establish a relation between crimes and the (spatial) context in which these take place (Summers & Guerette, 2018). Environmental criminology’s central proposition is that crime events are not randomly distributed through space and time because crime opportunities are not equally distributed. If crime is not randomly distributed, it has a pattern; if it has a pattern, then patrolling resources can be allocated accordingly. However, environmental criminology is seldom applied to environmental crimes (Kurland et al., 2017). Three theories that have direct relevance to the analysis of poaching are highlighted in the following sections.

2.6.1 Rational choice perspective: crime calculus

The rational choice perspective (RCP) is based on the assumption that offenders weigh efforts, risks, and awards before committing a crime (Cornish & Clarke, 1987). This analysis may be rudimentary and made with limited or incomplete information (Johnson, 2010; Summers & Guerette, 2018).

Offenders become familiar with the environment, which reduces the risk of detection and leads to habituation (*crime scripting*), (Leclerc, 2017). This results in repeat victimization, where the same victims are targeted multiple times over a period of time. These repeats can be spatial, temporal, and tactical (Pease & Farrell, 2017). Applied to poaching, poachers can target the same area (spatial repeat) in a quick succession of poaching trips (temporal repeat), and apply the same hunting method, e.g., snaring (tactical repeat). The combination of these repeats results in *virtual repeats* or *contagion*. Here, the occurrence of a crime confers an increased risk for similar and nearby targets for an amount of time (Caplan & Kennedy, 2016). The repeated targeting of victims leads to spatial and temporal clustering of crime in hotspots.

The predicted spatial clustering properties of crime were confirmed in anti-poaching research. Each of the 43 articles containing a spatial poaching component found hotspots (Appendix A). Contagion has been studied in the context of poaching in a marine protected area (Weekers et al., 2020), but has not been researched in terrestrial protected areas.

2.6.2 Crime pattern theory: spatial patterns in crime

The routine activity approach (RAA) posits that crimes can occur when motivated offenders converge with suitable victims in the absence of capable guardians (Cohen & Felson, 1979).

Crime pattern theory (CPT) describes how this convergence takes place by abstracting (urban) landscapes as a set of nodes, paths, and edges (Brantingham & Brantingham, 1993b). The starting point of CPT is that potential offenders' spatial movements are not considered to be different from those of the general public (Brantingham et al., 2016). Movements of people cluster around *nodes*, places that people travel to and from (home, work, school, shopping centers), and the connections (*paths*) between these places. Places that are qualitatively different from each other, such as neighborhoods and parks, are separated by *edges*, such as walls, roads, and fences (Brantingham & Brantingham, 1995; Brantingham & Brantingham, 1993a; Song et al., 2017).

Crime opportunities cluster around frequently visited nodes and paths because this is where victims and offenders' presence may overlap (Johnson, 2010). Such frequently visited places are also likely to be guarded. Therefore, crime often clusters at edges a limited distance away from such places (Song et al., 2017; Summers & Guerette, 2018). A crime hotspot can build up when there is spatial clustering of crime opportunities ("terrain vulnerability") combined with near repeats (Caplan & Kennedy, 2016; Caplan et al., 2011). The individual decisions of offenders lead to the collective effect ("emergence") of hotspot stability and persistence. Hotspots can thus remain active even when individual offenders are arrested (Caplan & Kennedy, 2016).

Bushmeat poaching in protected areas will, according to CPT, cluster in three different forms. First, poaching will occur at and where the network of paths and nodes (villages, schools, shopping centers, and workplaces) intersects with the presence of wildlife. More poaching is expected near villages and in areas where illegal firewood collection and illegal grazing occur.

Second, hotspots can occur near park boundaries and near roads dissecting protected areas; both are forms of edges. Third, a poaching incident will confer an increased risk on the location and its surroundings for some time (*contagion*). The combined effect of these phenomena is that hotspots can be stable in time and place even when individual poachers are arrested (Herbig & Warchol, 2011). For example, (Critchlow et al., 2015) found that the best predictor for future poaching activities was the occurrence of identified hotspots.

Edge effects were found in 31 out of the 33 papers that used primary data (patrol data, transects, camera traps) to study poaching in protected areas (Appendix A). Research into snaring usually finds hotspots and edge effects, although these are often not singled out as a finding. For instance, edge effects may be visible on maps where snare positions are plotted (see e.g. Kimanzi et al. (2014)).

2.6.3 Journey to crime

Offenders are constrained by time and distance, which reduces their action radius (Summers & Guerette, 2018). Offenders will thus prefer crime locations that are not too hard to reach and not too far from their homes (Townsley, 2017). Therefore, criminologists hypothesize that the frequency of crime decreases monotonically with the distance between the offenders' home and the crime location, and that short trips are more frequent than long ones ("distance decay hypothesis") (Hammond & Youngs, 2011). Offenders may find that committing crimes at nearby sites is risky because this can be observed by rangers, villagers, and other passers-by. The "buffer zone hypothesis", therefore, assumes that crime locations are not situated near offenders' homes (Townsley, 2017) (Fig. 2.3). The shape of the function is hypothesized; the actual shape needs to be assessed empirically. There is limited empirical support for the buffer zone hypothesis and robust empirical support for the distance decay hypothesis (Bernasco & van Dijke, 2020; Wiles & Costello, 2000). The limited support for the former hypothesis is related to the ambiguous methodology for determining whether a buffer zone effect exists (Bernasco & van Dijke, 2020).

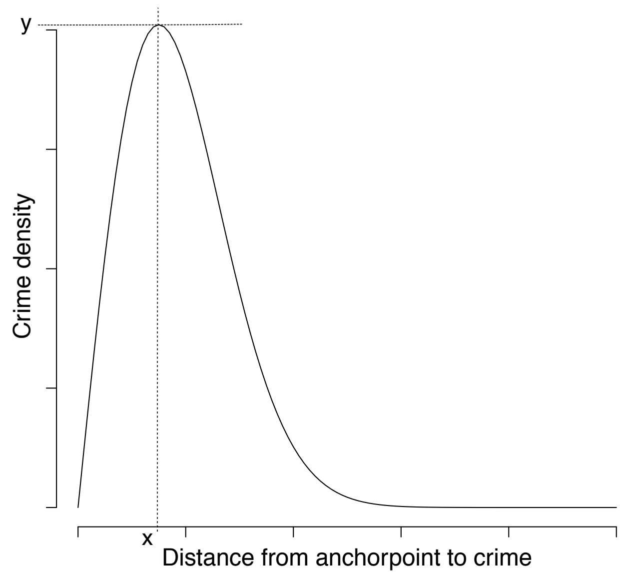


Figure 2.3: Density decay function as predicted by the buffer zone hypothesis, with maximum (mode) crime density (y) at distance x from the offenders' home.
Source: (O'Leary, 2011).

The distance decay function predicts that bushmeat poachers will place snares not too far from their homes. This was confirmed by Coad (2007), who found a mean distance of 2.2–4.5 km between traps and villages. The distribution of snare density versus distance to villages is expected to be left-skewed and tapering off with distance. The basis for these expectations is that each snare placed by a poacher constitutes a commitment to check it frequently. Snares have to be checked regularly because snared animals may die and start to rot, be eaten by predators, or break free by breaking the wire. Each visit reduces the possibility that rangers or outsiders do not see this, and long travel times increase exposure time. Thus, bushmeat poachers will prefer to minimize travel time and prefer hunting locations within easy reach from their homes (Faulkner et al., 2018; Mudumba et al., 2020).

2.7 Summary: Research gaps in anti-poaching research

There are, broadly speaking, three important and interrelated gaps in research that addresses bushmeat poaching, namely (1) incomplete understanding of the relation between patrol effort and poaching prevalence, (2) the potential role of using ranger expertise in the development of improved patrolling strategies, and (3) the application of environmental criminology on environmental crimes.

First, researchers propose measures to improve patrolling effectiveness while it is not clear what this effectiveness is and how it is measured. Moreover, current research assumes, explicitly or implicitly, that patrol efforts reduce poaching levels. Alternative processes that result in a decrease of observed poaching prevalence, such as displacement of poaching activity, changes in the population at risk, patrol bias, and under-reporting are not considered. As a result, the researcher or protected area manager may under-estimate the true poaching prevalence.

Second, researchers seldom elicit the expertise of rangers. Rangers select areas to be patrolled based on their knowledge and expertise; they are in the conservation front-line. Therefore, understanding their views on poaching patterns and poachers is crucial before any measures to improve patrolling effectiveness are proposed. At the same time, rangers are more often than not subject to violent threats from poachers and community members. This, in combination with generally low salaries and living conditions, can lead to lower ranger morale or collusion with poachers. Demoralized or compromised rangers are unlikely to report all poacher sightings.

Third, poaching in a protected area is by definition a crime, and the application of criminological theory would thus be in order. However, most quantitative research aims to find a fit between observations of illegal activities and environmental variables through statistical rather than criminological theory. Therefore, the reproducibility of anti-poaching research may be uncertain, as poaching events are understood post hoc through their covariates rather than through a causal chain predicted by criminological theory. Moreover, the body of knowledge on understanding and predicting crime that has been built up in criminology remains mostly ignored in the analysis of wildlife crime. This includes techniques for policing of crime hotspots and the identification of locations that are vulnerable to crime.

Chapter 3: Materials and methods

The methodology is organized into five parts. First, the study area is discussed. Particular emphasis is given to the environmental management and governance of the study area and its environs, given its multiple environmental designations. An overview of the research methodology is provided in the theoretical and conceptual framework. The remaining three parts describe the research methodology for each of the three specific objectives of this study. First, 82 km of desnaring transects were walked throughout the study area. Second, a survey was administered on a sample of 31 rangers concerning their perceived sufficiency in the workforce, capacity to detect poaching, and capability to detect poachers. This was supplemented by interviewing six representatives from communities adjacent to the study area. Finally, the results of the fieldwork were used to develop improved desnaring strategies. This involved modeling four alternative desnaring strategies and another 46 kilometers of transect walks to validate them.

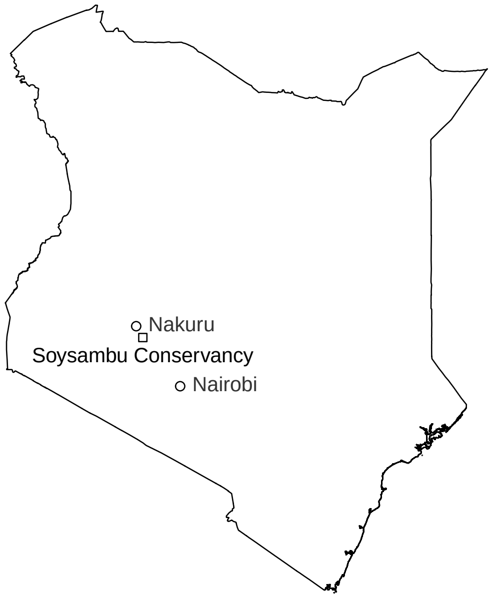
The Kenyan National Commission for Science, Technology, and Innovation Research authorized this research under License A21280.

3.1 Study area

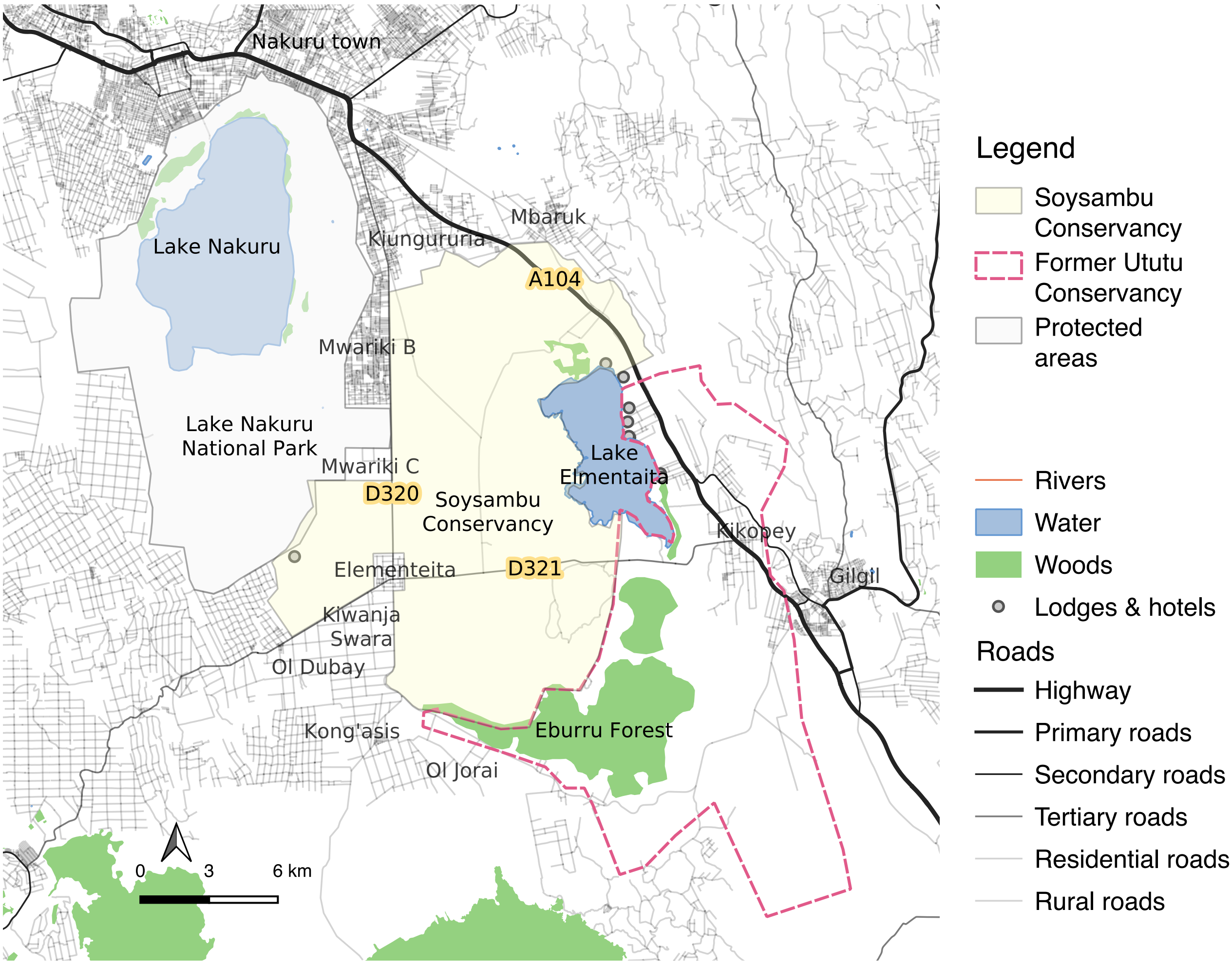
The study area is a wildlife conservancy in Kenya's Great Rift Valley. The description of the geography, climate, and mammal diversity is extended by discussing the conservancy's environmental management and governance and its environs.

3.1.1 Geography and climate

The fieldwork for this study was implemented on the 190-km^2 Soysambu estate (S 0°28.122′ E 36°11.408′, 1776 m+ASL) (Fig. 3.1). This farm was established in 1903 by Hugh Cholmondeley, whose descendants still live on the estate. The primary land use is livestock farming and haymaking. In 2006 the Board of Trustees agreed upon formalizing wildlife protection on the estate, followed by the establishment of Soysambu Conservancy Ltd. in 2007 (Zeverijn & Co, 2013).



(a) Location of Soysambu Conservancy within Kenya.

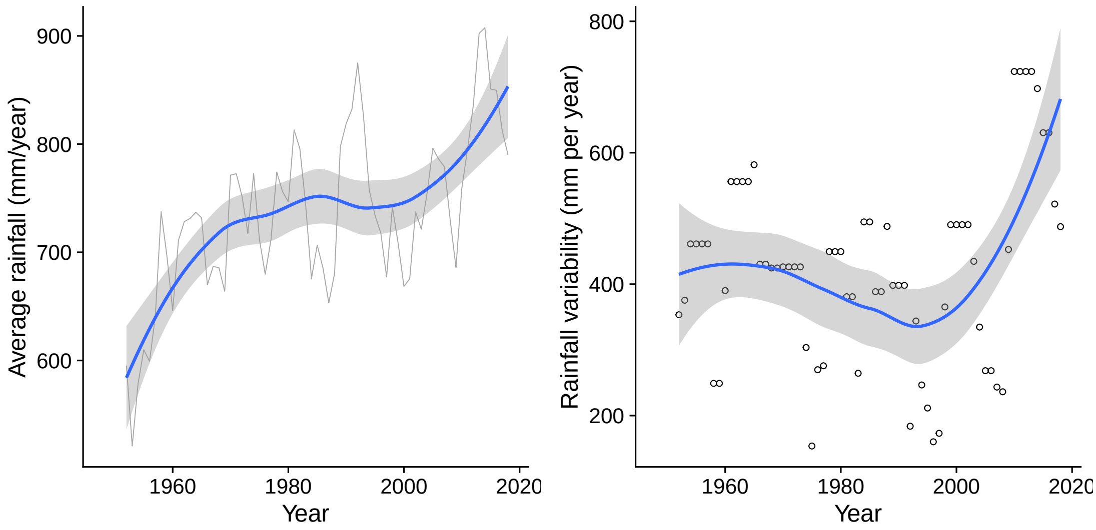


(b) Overview of Soysambu Conservancy and its environs.

Figure 3.1: Soysambu Conservancy: location within Kenya (a) and overview of environs (b).

Soysambu is bordered on the east side by Lake Elementaita and the north-western side by Lake Nakuru National Park. The latter is separated from the Soysambu Conservancy with an electric fence. The former Kekopey ranch and conservancy (known as Utut Conservancy, SPARVS Agency Ltd., 2008) borders the estate on the south-eastern side (Fig. 3.1). The northern and south-western borders of Soysambu consist of human settlements and marginal agriculture. The conservancy is dissected by three roads, namely the Nairobi–Nakuru highway (A104), the Elementeita–Nakuru road (D320), and the Elementeita–Kekopey road (D321). The A104 and the northern section of the D320 are enclosed with an electric fence.

Mean temperature is 18.2 °C, distributed in the 17.2 °C to 19.5 °C range, and the average reported rainfall is 600–700 mm, with long rains reported to be from April to June, and short rains in the period October-November (Government of Kenya, 2010b). The long-term (1948–2018) measured rainfall has a unimodal pattern, with most rain falling in April (Soysambu Ranch, 2019). The region is classified as a zone V agro-ecological area. This means that the region is semi-arid and considered to be best suited for extensive ranching, given the unreliable rains (Jaetzold & Schmidt, 1983). Nevertheless, small-scale subsistence culture (maize and beans) is increasingly practiced in the area. Crop failure is frequent, given the erratic rainfall (Ongalo, 2019). The Soysambu meteorological station has been measuring rainfall that is both increasing (Fig. 3.2a) and becoming less reliable (Fig. 3.2b).



(a) Mean rainfall **(b)** Rainfall variability

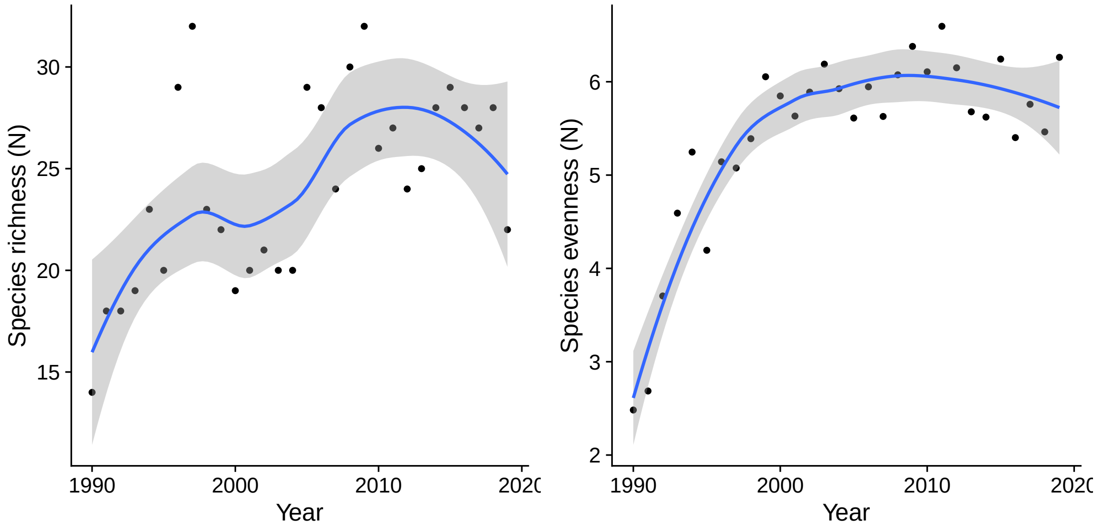
Figure 3.2: Rainfall trends in the Soysambu estate over the period 1948–2018 (five year moving windows). *Source*: Soysambu Ranch, 2019.

3.1.2 Flora and fauna

Most of the area consists of open bushland, consisting of acacia (mostly*Acacia seyal*) and leleshwa (*Tarchonanthus camphoratus*). The northern part of the lake and along the Mereronyi and Mbaruk rivers contain stands of yellow fever trees (*Acacia xanthophloea*). The northern part of the conservancy contains stands of *Acacia kirkii* and *Acacia tortilis*. The latter acacia type can also be found in the south-western side of the conservancy.

Stands of candelabra trees (*Euphorbia candelabrum*) can be found on the escarpments at the north-western side of Lake Elmentaita (Zeverijn & Co, 2013). The south-eastern and southern shores of the lake consist of swamps with sedges (*Cyperus laevigatus* and *Typha spp*) (Government of Kenya, 2010b).

The Soysambu conservancy carries out biannual animal censuses since 1990. Species richness – the count of different species – has increased since the start of the animal census but has been declining since 2010 (Fig. 3.3a). A similar trend is visible for species evenness (Fig. 3.3b), here expressed as the Hill number corresponding with Shannon entropy (Daly et al., 2018; Jost et al., 2010).



(a) Species richness.

(b) Species evenness.

Figure 3.3: Species richness and evenness for mammals in the Soysambu Conservancy. *Source:* Soysambu Wildlife Conservancy (2018a)

The population trends for ungulates (Fig. 3.4) show that there is a steady increase of (near-threatened) Rothschild 's giraffe (*Giraffa camelopardalis rothschildi*) and Burchell's zebra (*Equus quagga burchellii*). The populations of Duiker (*Sylvicapra grimmia*) and Thompson's gazelle (*Eudorcas thomsonii*) have been declining since the start of the censuses. The trends for the remaining species are either stable or downward. It is not clear whether the trend is permanent or part of a multi-year fluctuation in the latter case. Moreover, the trend may be the result of by out-migration, reduced in-migration, poaching, or other causes. Most downward trends have set in around 2005 (Kirk's dik-dik (*Madoqua kirkii*); eland *Taurotragus oryx*, Grant's gazelle (*Nanger granti*), Thomson's gazelle; warthog (*Phacochoerus africanus*), and waterbuck (*Kobus ellipsiprymnus*)). This downward trend since 2005 has also been observed for the wider Naivasha - Nakuru region (Ogutu et al., 2017).

Water is available throughout the estate in the form of water troughs. This is of particular importance for some species, like Burchell's zebras, which have to drink water every day. The zebra population is rapidly expanding (Fig. 3.4) and will exceed the estate's carrying capacity or has already done so.

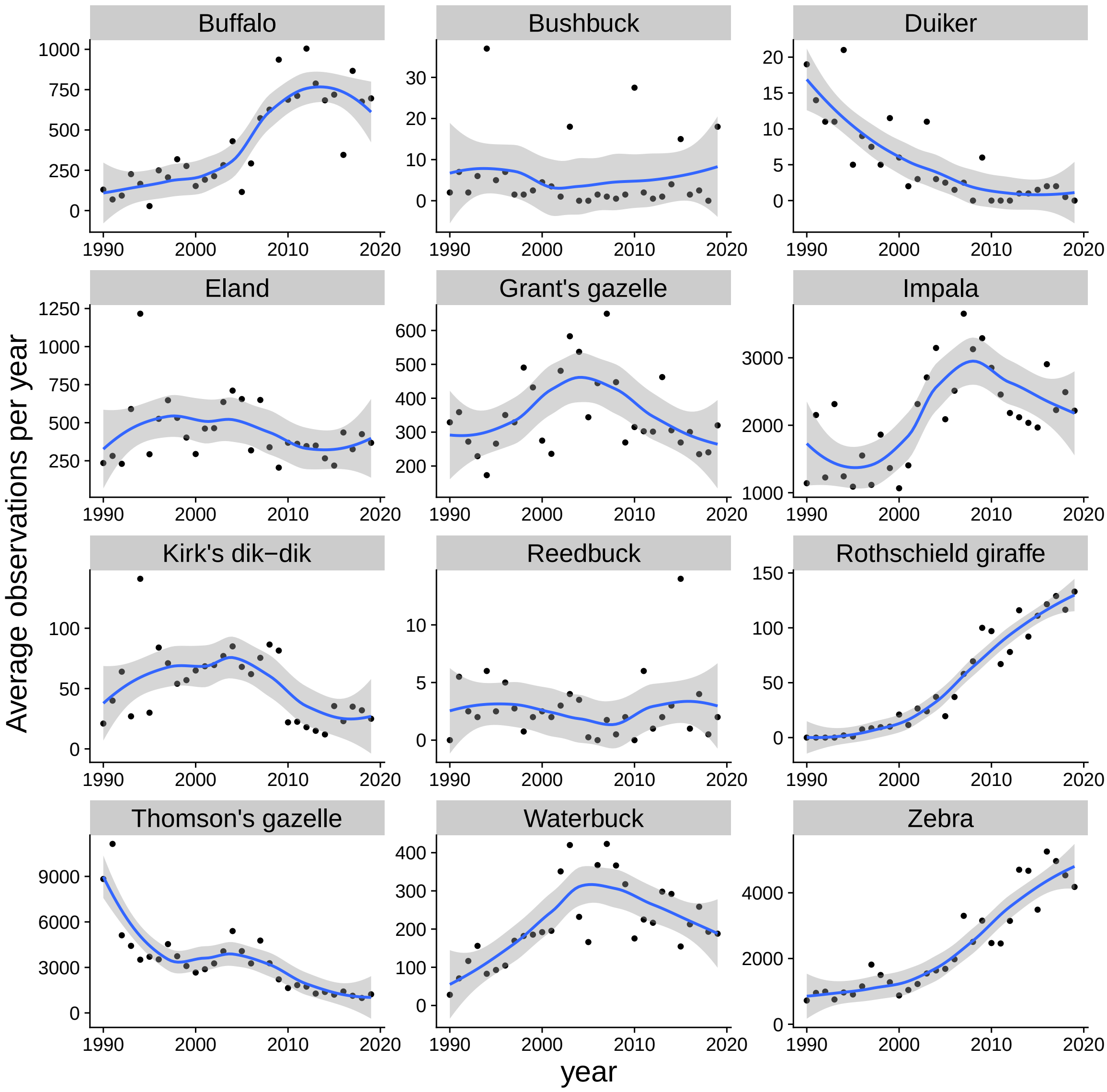


Figure 3.4: Count of ungulates per year in the Soysambu Conservancy. Count is average from two censuses per year. *Source*: Soysambu Wildlife Conservancy, 2018a.

3.1.3 Environmental management and governance

The Soysambu Conservancy and its surroundings are recognized for their importance in terms of biodiversity and landscape. The protection of both the Soysambu estate itself as the environmental management and governance of areas in its direct vicinity are discussed. The wider Lake Elementaita region is subject to several threats, such as uncontrolled development, habitat destruction, and landscape connectivity loss.

3.1.3.1 Patrolling in the Soysambu Conservancy

The estate employs 65 unarmed rangers who provide security to residents, visitors, and livestock in the estate. They also protect the conservancy against poaching, illegal firewood collection, and illegal grazing. No specialized internal or external training is required from or provided to the rangers. The rangers work in shifts (6 a.m. to 6 p.m., n=37; 6 p.m. to 6 a.m., n=24). Twenty rangers were recruited from local communities.

Rangers are allocated to different tasks and areas through a patrol plan. Foot patrols are conducted during the day by groups of two to four rangers (n=23). The remaining rangers (n=14) are allocated to park infrastructure, such as park gates, stores, and houses. Two vehicles patrol the estate at night, each staffed with a driver and a supervisor (n=4). The remaining rangers of the night shift (n=20) are allocated to park infrastructure objects. The plan foresees in bimonthly shifts of individual rangers to different duties. A Soysambu ranger can therefore be assigned to guarding a store in one week, while carrying out mobile patrolling against bushmeat poachers in another. A Soysambu ranger is therefore not engaged in wildlife protection throughout the year, and differs in that respect from a KWS ranger.

Rangers are supposed to report any observations made in the field to the control room, located at the estate's headquarters. Reports are handwritten in an observation book. No extract or tally is made from observations per category, e.g., illegal grazing, illegal firewood collection, or poacher sightings.

Rangers remove snares that were detected during routine patrols. The locations and numbers of these snares are not registered or reported. The conservancy organizes desnaring events on an ad hoc basis. Participants in such events are Soysambu rangers, KWS rangers, and volunteers from third parties, such as Projects Abroad and African Network for Animal Welfare (ANAW). The number and type of snares removed are registered and reported in desnaring reports. The GPS locations of snaring hotspots are reported; locations of individual snares are not.

The conservancy has limited contact with the surrounding communities, and few informers are available to help rangers preempt poaching activities or identify poachers.

3.1.3.2 Environmental designations

The Soysambu Conservancy is not yet registered as a wildlife conservancy under Art 40(3) of the Wildlife Conservation and Management Act, 2013 (Government of Kenya, 2013), because the legal procedure for doing so has been in draft since 2015 (Ministry of Tourism and Wildlife, 2017).

The current management plan for the conservancy (Soysambu Integrated Management Plan 2013–2016) does not foresee specific actions to mitigate bushmeat poaching. Instead, it defines an ambition level, namely to ”Ensure the development of a well resourced, well-trained and well-motivated security operation” (Zeverijn & Co, 2013, p. 36). A more detailed management plan, elaborating aspects of wildlife management and wildlife management user rights would be required for registration as a conservancy under the WCMA, 2013 (Government of Kenya, 2013, Art. 40(3)).

The wider Lake Elementaita area contains several areas with an environmental designation (Table 3.1, Fig. 3.5). However, only Lake Elementaita and neighboring Lake Nakuru National Park are gazetted and therefore legally protected. The Lake Elmentaita catchment, the Ramsar wetland of international importance, the Utut wildlife corridor, the UNESCO buffer zone, and the Soysambu Conservancy are not gazetted and therefore unprotected against development. IUCN flagged the weak environmental protection of the wider Lake Elmentaita ecosystem (IUCN, 2011).

Both Ramsar and UNESCO World Heritage designations have, in theory, environmental management consequences for the Soysambu Conservancy. First, UNESCO World Heritage sites must have an environmental management plan which specifies how the Outstanding Universal Value of the site is preserved for present and future generations (World Heritage Committee, 2017, Art. 108, 109). The proposed management plans identified bushmeat poaching as an important threat (Government of Kenya, 2010b; Ongalo, 2019). However, these plans have not been gazetted to date, and have therefore no legal status. The Soysambu management is not aware of any progress on proposed actions since 2010.

Table 3.1: Environmental designations of Soysambu Conservancy and the wider Lake Elmentaita ecosystem.

Designation	Status	Area (km^2)
Important Bird Area, Key Biodiversity Area [KE046][a]	Criteria for inclusion matched, 2001 (Birdlife International, 2019)	25.3
Wetland of international importance (Ramsar Convention) [1498]	Designated, 2005 (Ramsar Sites Information Service, 2019)	108.8
Soysambu Conservancy	Registered as Soysambu Conservancy, Ltd., 2007 (Zeverijn & Co, 2013)	194
UNESCO World Heritage Site [1060rev][b]	Nominated, 2010 (Government of Kenya, 2010b), Evaluated (IUCN, 2011), Inscribed (UNESCO, 2011, 35 COM 8B.6)	25.3
Lake Elementaita Wildlife Sanctuary[c]	Gazetted, 2010 (Government of Kenya, 2010a)	25.3

[a] No legal obligations forthcoming from this designation.
[b] This excludes the 35.8 km^2 buffer zone, which is not gazetted.
[c] The area and delineation of the sanctuary are identical to current UNESCO core zone.

Second, a National Environment Management Authority (NEMA) moratorium has been declared for the Ramsar area on 9 September 2015 following illegal developments around the lake (Mutwiri et al., 2017; UNESCO World Heritage Committee, 2019; Wahungu, 2015). This moratorium will remain in place until the UNESCO World Heritage environmental management plan has been gazetted. However, large-scale developments in the Ramsar area have continued. These developments include a hospital at the lakeshore (Lumbe, 2019) and a 220 kV power transmission line through the southern section of the conservancy in the immediate proximity of the UNESCO buffer zone (Mangat, 2019a, 2019b). A contingent of five KWS rangers placed in Kekopey (KWS, 2020) has not been able to stop illegal developments in the Elementaita region or the lake's riparian zone (Bett et al., 2016).

3.1.3.3 Environmental threats

Soysambu is experiencing threats to wildlife both from inside and outside the conservancy (Fig. 3.6). These are driven by bushmeat poaching and land conversion (Government of Kenya, 2010b). Soysambu is separated from Lake Nakuru National Park by an electric fence in the conservancy's western part. Electric fences put into place by Soysambu further limit the migration of wildlife in the entire northern section. Additional fencing is encouraged by KWS (Soysambu Conservancy, 2020). Furthermore, the Utut conservancy has been sold and parceled (Ongalo, 2019), and will be developed as holiday homes (Mutwiri et al., 2017). This will block wildlife migration, as this former conservancy is situated on the Naivasha–Elementeita wildlife corridor. Therefore, wildlife is becoming confined in the Soysambu Conservancy due to fencing and the loss of the migration corridor.

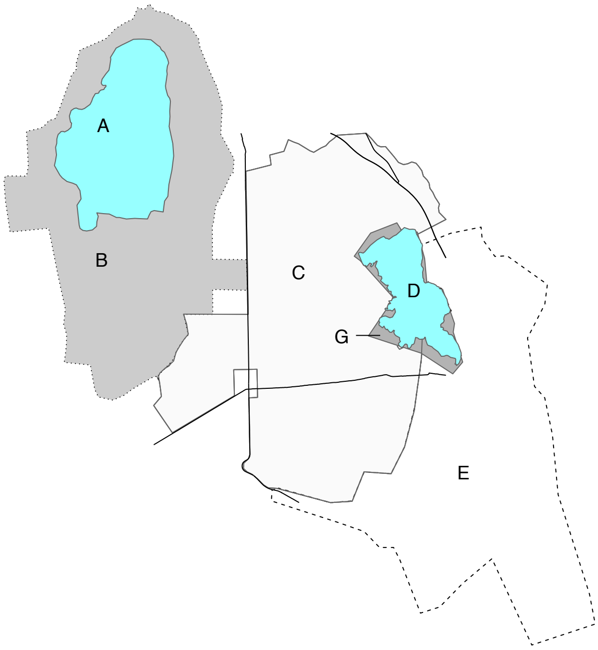
Human settlements and smallholder farms around the conservancy are rapidly expanding. The human population in the study area grows faster than the national average (Table 3.2), and will have increased by 80% between 2010 and 2027[1]. The first year, 2010, is the start year of the first proposed UNESCO World Heritage management plan (Government of Kenya, 2010b); 2027 marks the last year of the most recent management plan (Ongalo, 2019).

Table 3.2: Growth of human population national, regional, and around study area.

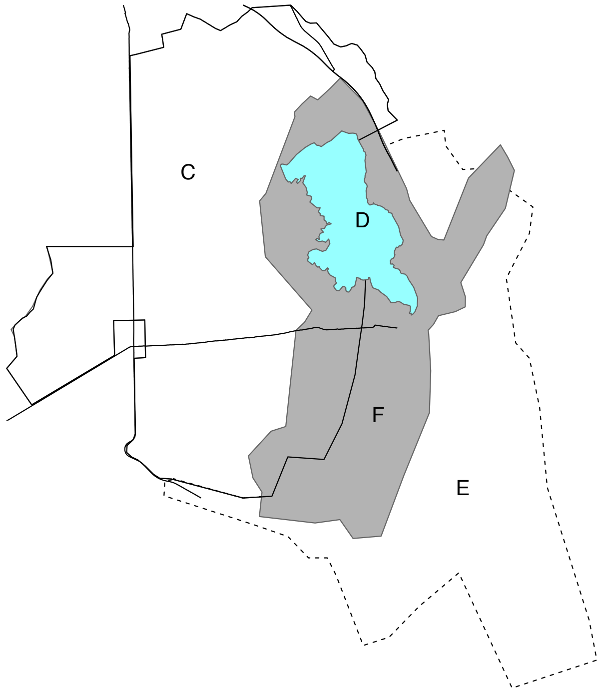
Location	population (2009)	population (2019)	growth (% per year)[a]	doubling time (years)[a]
Kenya	38,610,097	47,564,296	2.3	30
Nakuru County	1,603,325	2,162,202	3.3	21
Elementeita	1,500	2,338	4.9	14

[a] Calculations of annual growth rate and doubling time on basis of $N_t = N_0 \cdot e^{r \cdot t}$.
Source: Kenya National Bureau of Statistics, 2009, 2019a, 2019b; Zeverijn & Co, 2013

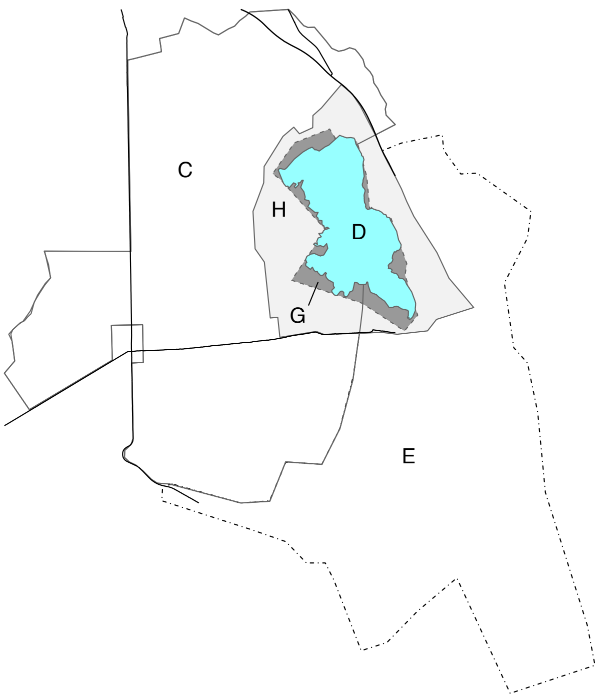
[1]Taking the Nakuru County population growth of 3.3% per year: $e^{r \cdot t} = e^{0.033 \cdot 18}) = 1.81 = 81\%$ population increase.



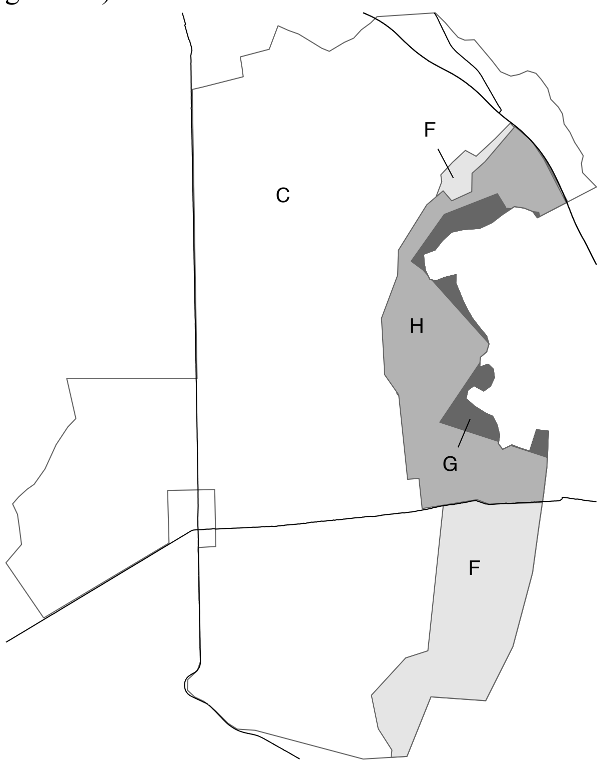
(a) Protected areas: Lake Nakuru (A), Lake Nakuru National Park (B); Lake Elementaita and Lake Elementaita wildlife sanctuary (G) .



(b) Location of Ramsar site (F, shaded) (not gazetted).



(c) Location of UNESCO World Heritage Site core zone (G, dark shade) and buffer zone (H, light shade).



(d) Environmental designations within Soysambu. Only the UNESCO buffer zone (G) is formally protected.

Figure 3.5: Environmental designations of the Soyambu Conservancy and the wider Lake Elementaita ecosystem.
A=Lake Nakuru; B=Lake Nakuru National Park; C=Soysambu Conservancy; D=Lake Elementaita; E=former Kekopey ranch/Utut Conservancy; F=Ramsar site; G=UNESCO core zone / National wildlife sanctuary; H=UNESCO buffer zone.

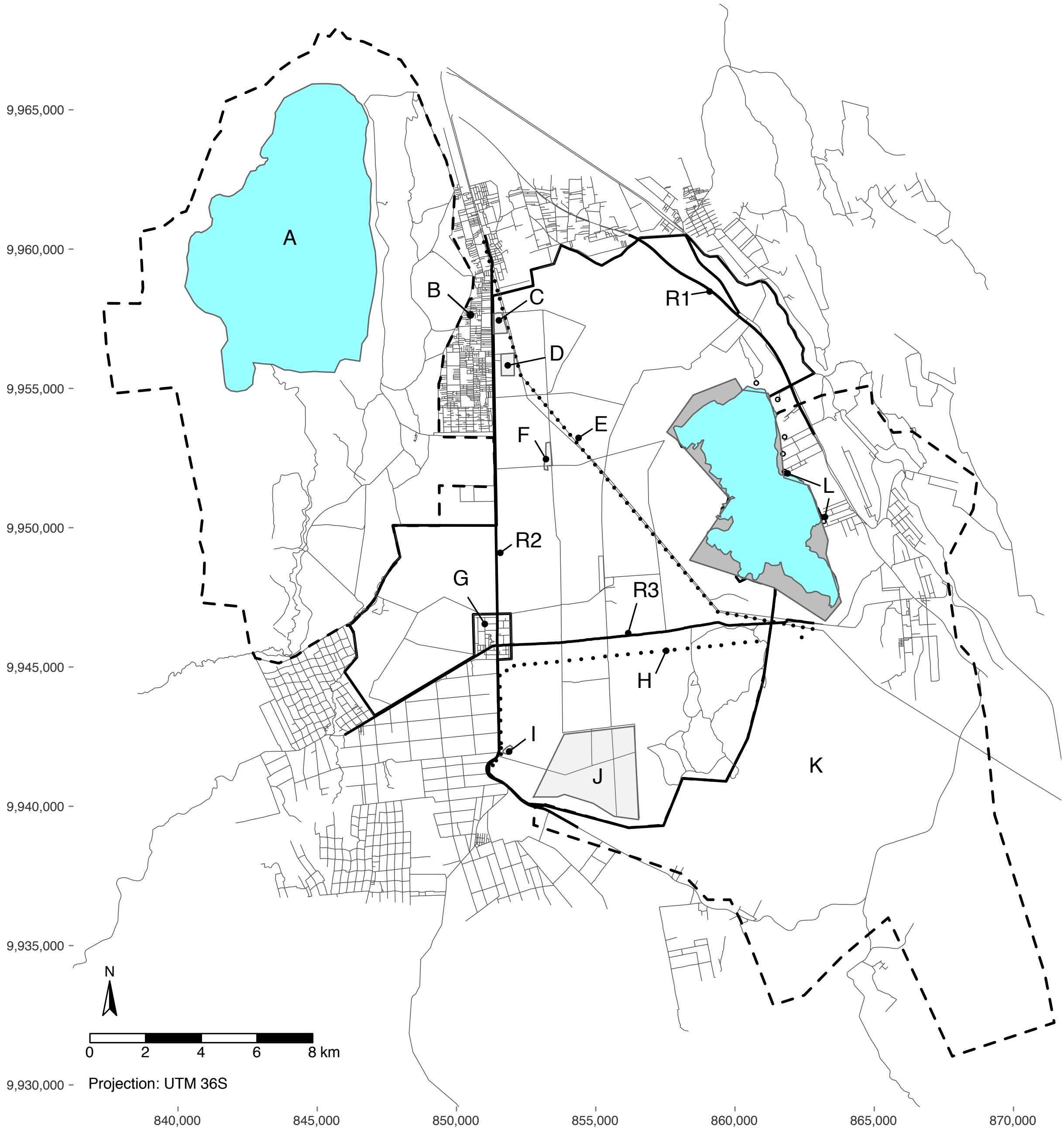


Figure 3.6: Soysambu Conservancy with selected environmental management features. A: Lake Nakuru National Park is separated from the Soysambu Ranch by an electric fence, and growing human settlements forming a wedge (B). C: Open-pit sand mining. D: Kenchic intensive chicken farm. E: Power line. F: Open-pit diatomite mine. G: Elmenteita village and agricultural areas. H: 220kV Power transmission line (constructed in October–November 2019). I: Quarry (not active). J: Crops. K: Former Kekopey ranch / Utut Conservancy, sold and to be developed as real estate holiday homes. L: Hotels on riparian land. The UNESCO core zone (shaded) does in some places not include any lake shores. Roads intersecting the conservancy: R1: Nairobi – Nakuru highway (A104), R2: Elementeita – Nakuru, R3: Elementeita – Kikopey.

3.2 Theoretical and conceptual framework

The central concept of the conceptual framework is guardianship, the collective effect of (1) ranger presence, (2) capacity to detect poaching, and (3) the capability to intervene once poaching is detected (Reynald, 2009). These guardianship components are used to structure data collection and the implementation of fieldwork (Fig. 3.7).

The data made available by the conservancy (patrol plan, observation book, and desnaring reports) are compared against observations from desnaring transects, ranger interviews, and interviews with representatives from adjacent communities.

Poaching is a crime and can therefore best be approached through criminological theory. Central to the theoretical framework (Section 3.2) is environmental criminology, particularly crime pattern theory (Brantingham & Brantingham, 1993a).

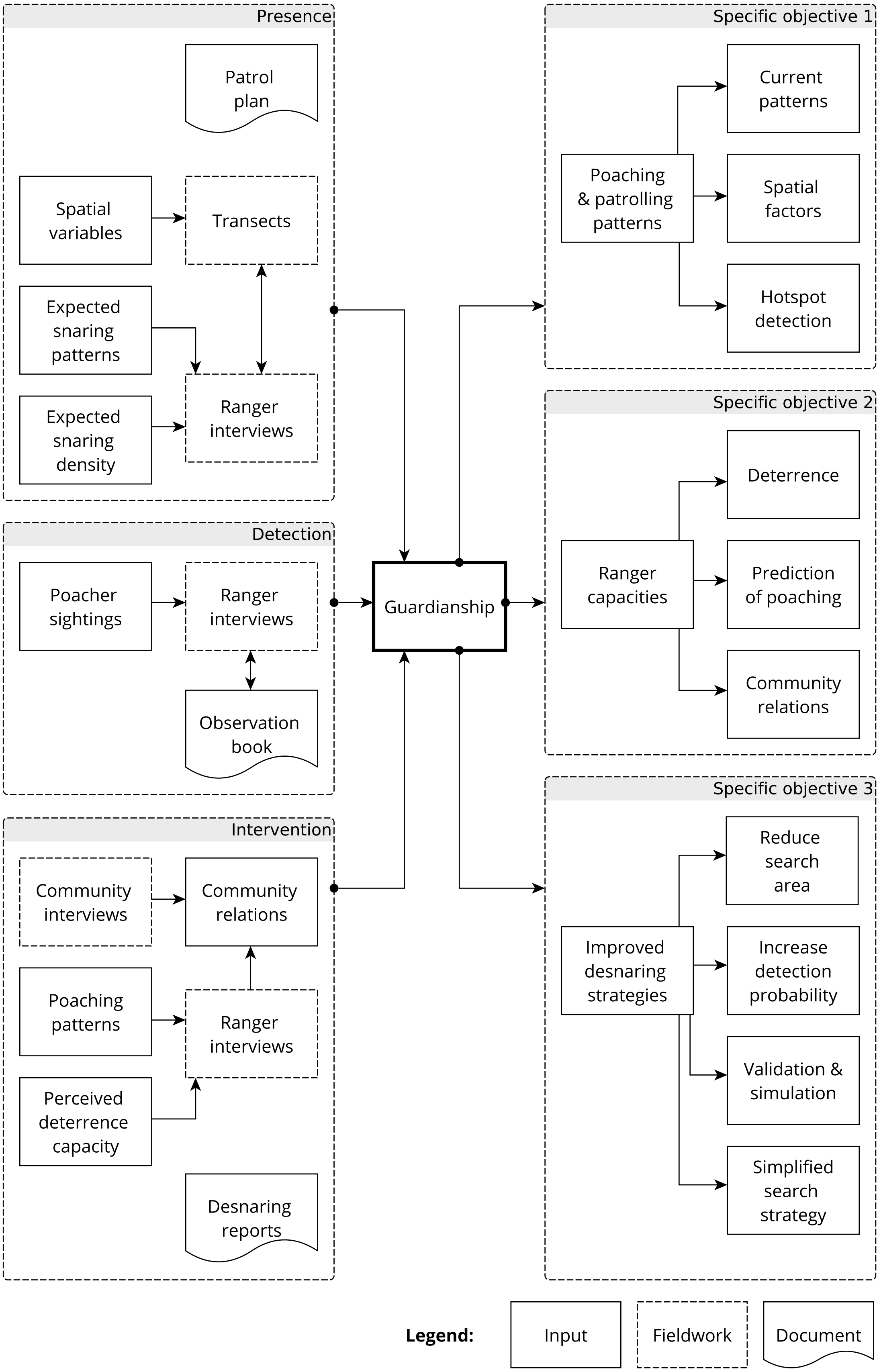


Figure 3.7: Conceptual framework for this research. The framework maps the guardianship concept Reynald (2009) over the three research objectives).

Table 3.3: Theoretical framework for this research.

Specific objective	Activity	Theory	Rationale or assumption	Reference
Assess poaching and patrolling patterns	Desnaring transects	Crime pattern theory	Crimes occur around nodes of activity and the paths between them. Poaching is thus likely to occur near edges (boundaries of the protected area, dissecting roads) and near nodes of activity (villages, workplaces).	Brantingham and Brantingham, 1993a
	Desnaring transects	Routine Activities Theory	Poaching can occur where there is a confluence of poachers and wildlife in the absence of rangers. Therefore, snares may be placed in the transition from bushes and open areas where animals seek shade and poachers can operate unseen.	Cohen and Felson, 1979; Thaker et al., 2010
	Desnaring transects	Journey to Crime	Poachers regularly visit snares and are bound by time and resource constraints. They will therefore place snares not further away from their homes than strictly necessary.	Townsley, 2017
	Desnaring transects	High-risk zoning	The dilation of observed snare positions with a nearest-distance cutoff results in a high-risk zone where the probability of finding additional snares is high.	Mahling, 2013
Assess ranger expertise and community opinions	Expected poaching patterns	Naturalistic Decision Making	Experienced rangers who receive sufficient feedback from their actions and observations (high validity environment) can estimate which cues are indicative for the occurrence of poaching.	Klein and Klinger, 1991
	Expected poaching density	Diversity Prediction Theorem	A diverse group of estimators can outperform individuals in estimation tasks. Thus, a group of rangers can estimate the number of snares that are likely to be found during a desnaring transect.	Page, 2007b
	Patrolling effectiveness	Snaring density is directly proportional to the frequency of poacher observations	A high prevalence of snaring indicates a high rate of incursions into the protected area by poachers since snares have to be inspected frequently. A high rate of incursions renders the probability that this goes undetected unlikely.	Becker et al., 2013

(continued)

Specific objective	Activity	Theory	Rationale or assumption	Reference
		Survivorship bias	Rangers are under the obligation to report all observed illegal events to headquarters, where the observation is entered into the site's observation book. The observation book contains filtered data where this is not the case.	Zabawski, 2019
		Patrol effort is directly proportional to poaching pressure	A higher patrol effort leads to a higher probability that poachers are detected and reported.	Hilborn et al., 2006
		Game theory	Patrolling a conservancy against poachers can be approached as a situation (game) in which multiple decision makers (players) can choose between different strategies concerning a conflict over natural resources.	Colyvan et al., 2011; Cumming, 2018
	Desnaring effectiveness	Deterrence	Desnaring is effective when the removed snares are not replaced and when the snaring activity is not displaced to other areas within the conservancy.	Moreto et al., 2015
	Regional integration	Equitable management	Equitable management leads to better (participation in the) protection of the conservancy. Where this is not the case, KWS assists in creating the necessary conditions as per WCMA, 2013.	Dawson et al., 2018; Government of Kenya, 2013
Develop improved desnaring strategies	Reduce search area	Presence-only species distribution modeling through maximizing entropy (Maxent)	The distribution of snares in a protected area can be extrapolated and predicted by approaching it as a species for which presence-only data are available and which maximizes entropy over the conditions that define its realized niche.	Phillips et al., 2004

(continued)

Specific objective	Activity	Theory	Rationale or assumption	Reference
	Maximize detection rate	Multi-armed bandit (exploration-exploitation), epsilon-greedy policy	The allocation of patrols between exploration and exploitation tasks can be approached via a strategy in which both the exploration/decision is subject comparing a random device with a noise parameter (epsilon). Where the outcome is “exploitation”, the hotspot to revisit is selected .	Press, 2009; Vermorel and Mohri, 2005

3.3 Poaching and patrolling patterns

Data on poacher sightings and removed snares reported by the conservancy were supplemented and checked against desnaring transects. The data from desnaring transects were used to analyze the association between snaring patterns and spatial features and the conservancy's vegetation density. Moreover, areas with high snare densities – hotspots – were detected and mapped.

3.3.1 Current patrolling and poaching patterns

The Soysambu Conservancy allocates rangers over different zones and objects in its area and carries out regular desnaring. The data provided by the conservancy on poaching observations by rangers and snares removed by desnaring teams were compared with the desnaring transects carried out during the study.

3.3.1.1 Patrol plan, desnaring reports and poacher sightings

The conservancy records the approximate location of snaring hotspots but does not record individual snares' GPS positions during desnaring. The number of snares found, specified by type (neck snares or foot snares), and snaring hotspot GPS locations are recorded in desnaring reports. These reports do not include snares that were removed by rangers during routine patrols; rangers are not required to report locations and numbers of detected snares. Animals with snare injuries are frequently observed by rangers, but not reported.

The ranger density for day and night shifts was calculated from the patrol allocation plan and plotted. The location of patrols is estimated since patrols do not track their routes or observations.

Reported poacher sightings were obtained from the conservancy's observation book. This book contains handwritten notes of rangers reporting signs of illegal activities to the control room. These notes are not entered into a database or further processed in any form.

The ranger densities and snare removal rates can be roughly compared to those reported by the DSWT desnaring teams. DSWT provides some publicly available information for desnaring reports over August 2019–August 2020 in the Tsavo region in Kenya (DSWT, 2020b), but declined to make detailed desnaring data available. Soysambu, meanwhile, has not yet established desnaring reports for 2019 and 2020.

3.3.1.2 Desnaring transects

Independent estimation of snaring prevalence in the conservancy was obtained by walking desnaring transects. The required length of these desnaring transect walks was calculated by comparing the number of known snaring hotspots with a Poisson distribution. The Soysambu conservancy has identified five snaring hotspots (Fig. 3.8). The probability distribution of the number of hotspots within the conservancy was approximated with a Poisson distribution in which $\lambda = 5$. The upper 95% of the confidence interval range for this distribution is ten snaring hotspots. The detection probability for single snares in mixed forests was estimated as p=0.14/2km=0.07/km transect walk by O'Kelly et al., 2018a. A total of 80 km transect walks was required to confirm the identified number of snaring hotspots at this snare detection probability. At this point, the upper 95% limit of the expected number of snaring hotspots calculated based on p=0.07/km transect walk would exceed the upper limit of the estimation based on $\lambda = 5$ that was found by the conservancy.

Desnaring transects were carried out by a minimum of three to five experienced rangers, who walked in a line with 20 meters distance between them. The swath width was therefore 60 to 100 meters with an average width of 80 meters. The total length of desnaring transects was 82 kilometers, spread over 27 transects. The average length of transect walks was 3 kilometers. The desnaring transects' locations are shown in Fig. 3.8 with the snare locations identified by the conservancy superimposed.

Armed KWS rangers assisted with desnaring in areas where the presence of buffaloes was deemed to be likely. The desnaring transect trajectory and any encountered snares or signs of illegal activities were mapped with a Garmin etrex 10 GPS device. The longitude-latitude EPSG 4326 coordinates were transformed to UTM projection EPSG 32736 (zone 36S, datum

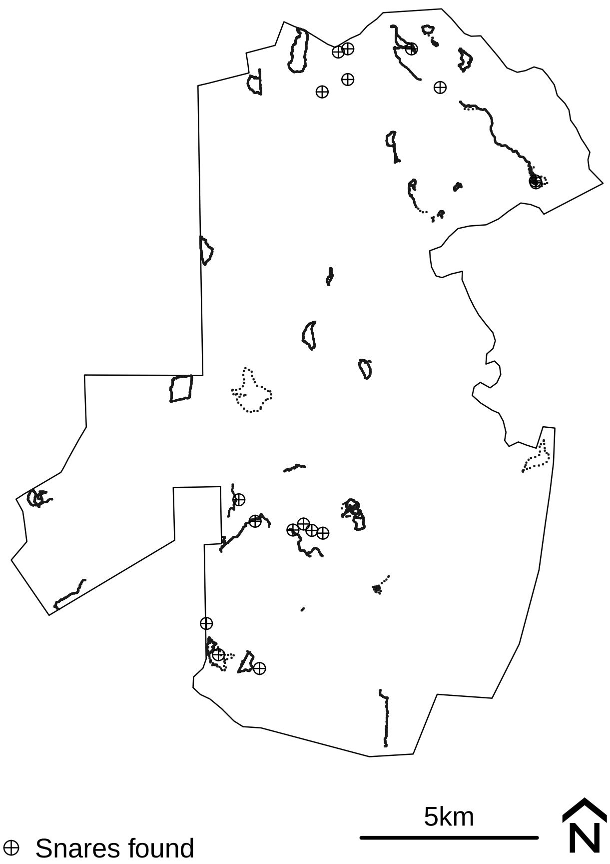


Figure 3.8: Location of desnaring transects and snaring hotspots identified by the Soysambu Conservancy.

WGS84), using either the `raster` or the `sf` package in the `R` statistical software (Hijmans, 2017; Pebesma, 2018; R Core Team, 2019). The EPSG 32736 projection has been applied to all spatial data.

Both desnaring track coordinates and encountered snare locations were plotted in `R`, using the `raster`, `sf` and `tmap` libraries (Hijmans, 2019; Pebesma, 2018; Tennekes, 2018). Vegetation maps were obtained from the Sentinel-2 platform, using L1C images (bands B2, B3, B4, and B8A) with 10 meter resolution (ESA, 2018). The soil-adjusted vegetation index (SAVI) (Huete, 1988) images were calculated from these Sentinel images using SNAP processing software (ESA, 2019).

Finally, unsupervised k-means clustering (Steinley, 2006) was undertaken to distinguish different vegetation types on the SAVI image. The results from this clustering were compared with Google Earth images (Google Earth, 2020). Vegetation classes consisted of forest (acacia forest north of Lake Elementaita), bush (acacia and leleshwa), water (Lake Elementaita and marshes at its lake shores), and open areas. The classification was made before the fieldwork and was used to plan the location of desnaring transects. The vegetation classes assigned by the k-means classification were verified during the desnaring transects.

3.3.2 Spatial factors of snare placement

The spatial variables to be assessed were selected based on earlier snaring research and underpinned by criminological theory (Table 3.4).

Variable	Motivation and data source
Distance to public roads	Crime pattern theory posits that crime hotspots are likely to occur at the edges of land use change (Brantingham & Brantingham, 1993a; Song et al., 2017). This has been confirmed by research into snaring in protected areas (Wato et al., 2006; Watson et al., 2013). *Data source*: OpenStreetMap (OSMF, 2019), read into `QGIS` (Open Source Geospatial Foundation Project, 2020) through the `Overpass` API (Trimaille, 2020).
Park boundaries	As predicted in crime pattern theory, researchers often find snares along the boundaries of protected areas (Loveridge et al., 2020; O'Kelly et al., 2018b; Wato et al., 2006). *Data source*: shapefile supplied by the conservancy.
Distance to water points	Scarcity of water causes animals to concentrate regularly around water points, where they are targeted by poachers (Watson et al., 2013). *Data source*: shape file supplied by the conservancy.
Distance to human settlements	Offenders will prefer to targets that are not further away from their homes than necessary (Summers & Guerette, 2018; Townsley, 2017). This is likely to apply to bushmeat poachers as well since snares have to be frequently inspected (Noss, 1998). Correlations between the presence of settlements and snaring locations were found in spatial snare research (Coad, 2007; Loveridge et al., 2020; Wato et al., 2006). *Data source*: See public roads variable.
Distance to infrastructure	Researchers found snares or traps near ranger posts in several cases (Jenks et al., 2012; O'Kelly et al., 2018b; Watson et al., 2013). *Data source*: See public roads variable.
Vegetation	Savanna ungulates, which were targeted by poachers, will seek shade in the bushes but will stay close to open areas so that they can escape predators (Thaker et al., 2010). The transition from open areas to denser bushes is also the area where poachers can operate unseen. Therefore, this is the area where routine activity approach would predict a higher likelihood of crime (Felson, 2016). A relation between snare abundance and vegetation density was found by O'Kelly et al., 2018b. *Data source*: Sentinel L1C image (ESA, 2018).

Variable	Motivation and data source
Elevation	The relation between poaching locations and elevation were assessed in snaring (Gurumurthy et al., 2018; Jenks et al., 2012; Linkie et al., 2015), and in trophy poaching (Park et al., 2016; Rashidi et al., 2018; Zafra-Calvo et al., 2016). *Data source*: Digital Elevation Model (DEM) from Advanced Space-borne Thermal Emission and Reflection Radiometer (ASTER) (METI, 2011)

Table 3.4: Variables used in the analysis of snare occurrence.

The strength of the spatial relationship between snare positions and spatial covariates was measured through a plot of the receiver operating characteristic (ROC) and the area under the curve (AUC) (Jiménez-Valverde, 2012). Both ROC and AUC were calculated in `R` with the library `spatstat`.

The relation between vegetation density and snare positions was also examined. This was implemented by isolating the bush-open area transition zone with a moving window operation on a SAVI image using the `raster` library of programming language `R` (Hijmans, 2019). This calculation step resulted in the ”degree of bushiness” in a $[0, 1]$ range. Here, 0 indicates open areas, 0.5 indicates mid-transition zone, and 1 indicates dense vegetation far from open areas. The values of interest are located around 0.5 since the expectation was that poachers would place snares in the transition from open areas and bushes. The bushiness value x was normalized with a min-max transformation:

$$t = 1 - \frac{|x - 0.5|}{0.5}$$

After this transformation, the maximum bush-open area transition zones had a value of 1, and either wholly bushy or open areas were assigned a value of 0.

3.3.3 Detection of snaring hotspots

The concentration of snares in clustered patterns results in ”hotspots”, areas with a larger than average poaching signs concentration. Two different methodologies for identifying hotspots were implemented and compared. Both methodologies were implemented in `R`, using the `spatstat` library (Baddeley & Turner, 2005; R Core Team, 2019).

The first method for detecting snaring hotspots, "high risk zones", is based on Mahling, 2013; Mahling et al., 2013. This methodology was carried out in three steps. First, snare locations' clustering properties were confirmed with a Hopkins-Skellam test (Hopkins & Skellam, 1954). Second, a nearest neighbor distance analysis was carried out. This distance was defined as the distance d_i in meters between all distinct pairs of points x_i and x_j (Baddeley et al., 2016):

$$d_i = min_{j \neq i} ||x_i - x_j||$$

From this, the empirical nearest neighbor distance distribution was established (Mahling et al., 2013):

$$G(r) = \frac{1}{n_y} \sum_i 1\{d_i \leq r\}$$

The nearest distance was plotted in a Stienen diagram (Stienen, 1982). The combination of the empirical nearest distance distribution and the Stienen set was used to calculate a quantile $P(Q)$, from which a dilation distance r was read. The union of discs resulting from dilation of each snare position u of snare set X with distance r produced the Steiner set (Baddeley et al., 2016):

$$X_{\oplus r} = \{u \in \mathbb{R}^2\} : d(u, X) \leq r$$

Third, the hotspot map was established. This map consists of a plot of the Steiner set (Baddeley et al., 2016), with the additional criterion of a minimum required number n of snares per union set.

The detection of hotspots is an ill-defined problem: there is no standard way of identifying them (Baddeley et al., 2016). Therefore, the obtained hotspot map was compared with an Allard-Fraley cluster set methodology hotspot map (Allard & Fraley, 1997). The comparison between the Stienen/Steiner and the Allard-Fraley approach took place by calculating the Jaccard index (Podani, 2000):

$$J(A, B) = \frac{|A \cap B|}{|A \cup B|}$$

3.4 Ranger capacities

The capacity to detect poaching and the capability to intervene once poaching is detected was assessed by administering a survey to 31 rangers. Additionally, six representatives from all seven communities adjacent to the conservancy were interviewed.

3.4.1 Deterrence

Deterrence was defined as the capability of rangers to deter poachers from entering the protected area. Factors related to deterrence were assessed in interviews with rangers.

A survey was administered, which included eight closed questions and three open questions. The questions are included in Annex D.

The closed questions were grouped into the categories (1) perceived sufficiency of ranger force, equipment, and transport, (2) self-rating of deterrence capability and predictability of patrolling patterns, (3) occurrence of meeting poachers during patrols. Additionally, background information on the occurrence of poaching in the conservancy and characterization of poaching and poachers was collected in four closed questions and seven open questions. These questions were grouped in (1) temporal and spatial occurrence of poaching, (2) perceived motivation of poachers, and (3) poaching techniques.

All closed questions were put in a 5 point Likert scale format, in which 1="disagree strongly" and 5="agree strongly". The open questions were grouped in the categories (1) actions that would enhance ranger capacities to deter poachers and (2) perceived bottlenecks in current patrol strategies and implementation thereof.

A sample size of n=30 rangers was required to detect a moderate effect size in the closed questions (h=0.5, 20–25% difference with neutrality (Likert score=3), power $1 - \beta = 0.8, \alpha = 0.95$). This sample was extracted from a total ranger population of N=65, in which the sampling units of individual rangers. The sample size was calculated with R package `pwr` (Champely, 2018; Cohen, 1988; R Core Team, 2019). The Likert test outcomes were statistically tested with an exact binomial test and presented in Likert diagrams (Boone & Boone, 2012). The results of the exact binomial tests were reported using APA-prescribed abbreviations (APA, 2019). Thus, statistical these tests are reported using the following symbols: success count is x out of n trials, assuming H_0 probability of success and 1-α confidence level.

The sample size of 30 interviewees was sufficient to discover topics represented by at least 10% of the interviewees at a 95% confidence level. This is calculated as:

$$n = \frac{ln(1-P)}{ln(1-R)}$$

with P=confidence level (95%) and R the probability that a theme was represented by an interviewee (Galvin, 2015). Solving for R with n=30 interviews and P=0.95 resulted in:

$$R = 1 - e^{\frac{ln(1-0.95)}{30}} = 0.10$$

A sample of n=31 interviewees was obtained by random selection from the list of rangers (N=65) employed by the conservancy. Each interview was individual, anonymous, face-to-face, and based on informed consent. All interviews were conducted in English if the interviewee felt (s)he was sufficiently proficient. The interview was carried out in Swahili, where this was not the case, using a staff member as a translator. The translator was not part of the ranger force and does not report to the rangers or vice versa. The protocol followed for the interviews complies with British Sociological Association guidelines (British Sociological Association, 2017) (Appendix C). The questionnaires are included in Appendix D.

The rangers were also asked to estimate the frequency with which they observe poachers during their work. This frequency was compared with reported poacher sightings in the conservancy's observation book. The time window over which this comparison took place equaled the period during which fieldwork was implemented (December 2018–March 2019).

3.4.2 Prediction of poaching

The ability of rangers to predict snare locations and densities was assessed. The consensus on expected snare locations was compared with actual snare positions, which became apparent during desnaring transects. The ability to give an ex-ante estimate of snare densities was tested by assessing whether group estimation performance outperformed individual rangers' estimation performance.

3.4.2.1 Poaching density

The extent to which rangers were able to estimate the snare density prior to walking a desnaring transect was calculated using the diversity prediction theorem (Page, 2007a, 2007b). This theorem is based on the observation that a diverse collective of independent estimators always makes more accurate predictions than individuals.

The collective error (CE) equals the difference between individual error (IE) and prediction diversity (PD) (Hong & Page, 2004):

$$CE = IE - PD$$

The individual error (IE) equals:

$$IE = \frac{1}{n}\sum_{i=1}^{n}(x_i - x_{true})^2$$

with x_i the individual estimation and x_{true} the actual value.

The prediction diversity equals:

$$PD = \frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})^2$$

with x_i the individual estimation and $\bar{x}$ the average estimation of the collective.

A diverse group always outperforms individual estimators. However, the effect can be assumed to be significant when the CE is at least ten times smaller than IE (Wagner & Vinaimont, 2010):

$$r = \frac{CE}{IE} \leq 0.1$$

Rangers participating in the desnaring transects were asked individually to estimate the number of snares found in the area to be desnaring before the transect started. The variables PE and IE were calculated by comparing the individual estimates with the number of snares found during the desnaring transect. Rangers were asked to motivate their estimation before and after the transect.

3.4.2.2 Poaching locations

Rangers were asked to estimate the likelihood of snare placement near specific spatial terrain features (park gates, lodges, roads, settlements, nearby villages, park boundaries, vegetation (open/closed)). The interview format, protocol, and interviewees were identical to that described in the previous section. The ranger-predicted likely snare locations were compared with the desnaring transects results and the results of desnaring reports established by the conservancy.

3.4.3 Community relations

Representatives of all communities surrounding the conservancy were interviewed. These representatives are either area chiefs or *nyumba kumi* representatives (Kioko & Okello, 2010). The locations of the communities are indicated in Fig. 3.9.

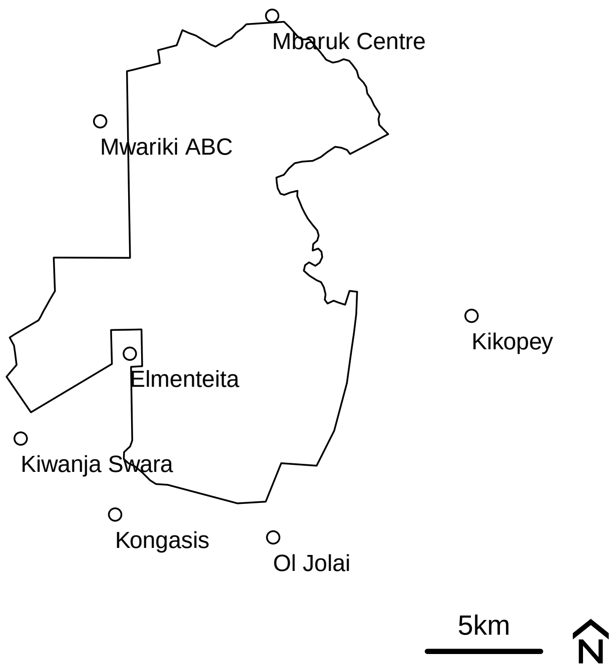


Figure 3.9: Location of communities around Soysambu

The interviews were administered with the help of the community development specialist of the Soysambu conservancy as a translator in case the interviewee deemed himself insufficiently proficient in English. This was the case in four out of six interviews.

Following an interview protocol that was identical to that followed during the interviews with the rangers, five open questions were asked. The interview questions were related to the relation between the community and (i) Soysambu conservancy, (ii) wildlife, and (iii) the input into the draft management plan for the Lake Elementaita Wider Ecosystem (LEWSE) (Ongalo, 2019). The interview protocol is included in Annex C. The list with open questions is included in Annex D.

Additionally, the group of 31 interviewed rangers was asked about the relations between (1) the conservancy and adjacent communities, (2) adjacent communities and poachers, and (3) the ranger force and poachers. These questions were formulated as four closed questions in Likert format with a 1–5 scale.

3.5 Improved desnaring strategies

”Desnaring strategies” were understood as a set of search actions to maximize the likelihood of snare detection. Improved desnaring strategies reduced the search area, increased the snare detection probability, or achieved a combination thereof.

An overview of the methodology for improved desnaring strategies is shown in Fig. 3.10. Three different improved desnaring strategies (OUT3, OUT4, OUT5) are developed based on the spatial analysis of the results of the desnaring transects (OUT1). The developed improved desnaring strategies were compared with a simplified desnaring strategy. This strategy deploys insights from environmental criminology, requiring a minimum of upfront information on snaring locations.

3.5.1 Reduce search area

The search area for snares was reduced through predictive mapping using a presence-only species distribution model (SDM) (Elith et al., 2006; Elith et al., 2011).

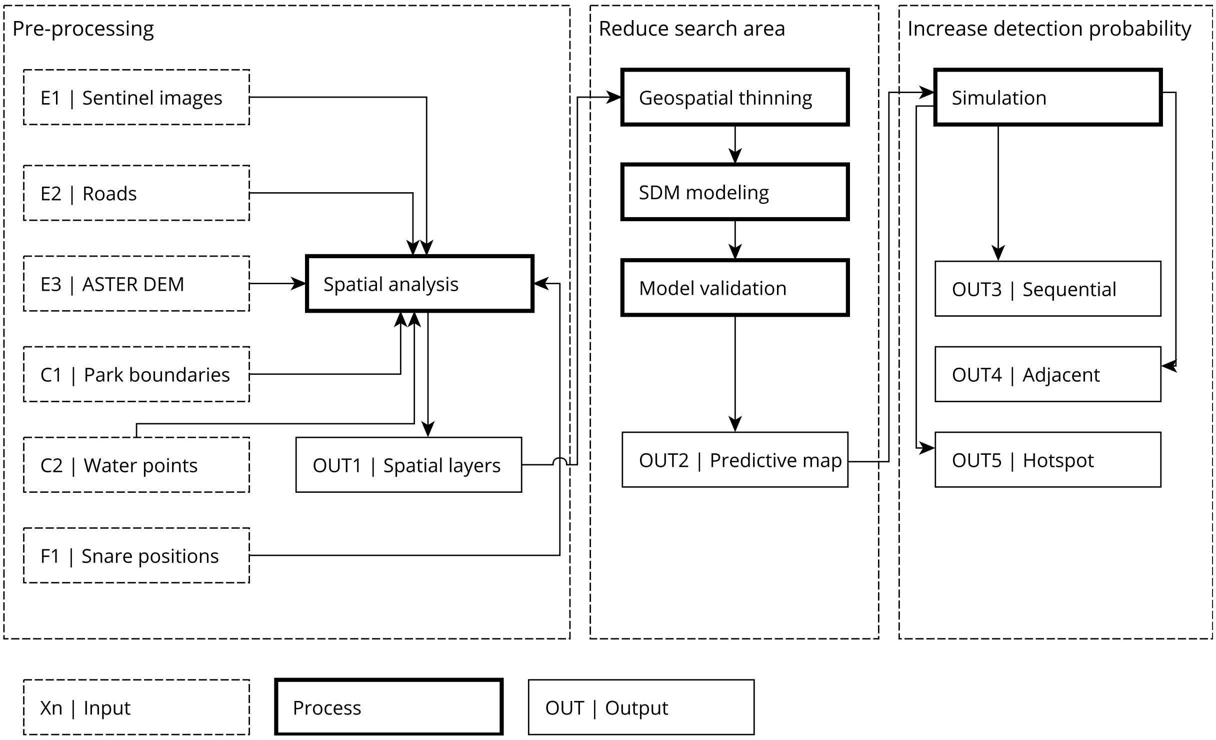


Figure 3.10: Data flow diagram of methodology for desnaring strategies.

First, geospatial thinning was applied to reduce sampling bias (Aiello-Lammens et al., 2015; Hijmans, 2012). The snare data set's thinning operation was carried out with the R `enmSdm` library (Smith, 2020).

The predicted snare distribution was then modeled through the Maxent SDM application in the R libraries `dismo` (Hijmans et al., 2017) and `SDMtune` (Vignali et al., 2020).

The predicted likelihoods of snare presence of the Maxent model were tested through 4-fold cross-validation. In each fold, 25% of the data was withheld for testing, and 75% of the data was used for training. Each model was tested on AUC and true skill statistic (TSS) (Youden, 1950). The TSS value was used as a threshold value for presence/absence on raw probability output values of the Maxent model (Allouche et al., 2006; Liu et al., 2013).

The model performance was visualized using the Boyce index (Boyce et al., 2002). The index has been modified by calculating the P/E ratio over a sliding window of habitat values (Hirzel et al., 2006). The calculation and visualization of the Boyce index's sliding window version were implemented through the `enmSdm` R library (Smith, 2020).

The validity of the Maxent model was tested in the field through 46 km of additional desnaring transects. Areas where the model predicted snaring presence, were visited and compared against areas in which snaring was predicted to be less likely. Both groups of transects (likely/unlikely snaring) had comparable vegetation cover. The snare density (snares/km transect) was calculated for both groups and statistically tested with the Wilcoxon Rank Sum test (McKnight & Najab, 2010) , and reported according to the APA reporting format. The required desnaring transect mileage was calculated as 45 km using the R `pwr` package (Champely, 2018). This sample was sufficient to detect a moderately improved snare recovery rate (Cohen's $d > 0.6$, power $1-\beta$=0.8, confidence level α=0.95).

3.5.2 Increase detection probability

The baseline scenario ("sequential strategy") was compared with two improved desnaring strategies ("adjacent strategy" and "hotspot strategy"). Each scenario was modeled using the `R data.table` library (Dowle & Srinivasan, 2017). The general principles for the simulation, followed by a description of the desnaring strategies, are set out below.

3.5.2.1 set up of the simulation

An overview of the symbols used in this section is included as Table 3.5.

The conservancy was simplified to n meter raster cells, the size of which depends on the nearest distance analysis. A test landscape was set up using snare clusters found during field testing of the predictive model. This avoids information leakage, namely, the detection of snare cells used in the predictive model's training set.

A random number was generated when a desnaring team and a snare cluster coincide in the same raster cell. The snare was considered to be removed when the random number was lower than the preset detection probability. This random number, and therefore the program's snare removal decision, could vary between simulation runs. Therefore, these runs were repeated until the cumulative moving average (CMA) for detected snares per run was stable. The CMA was calculated as

$$CMA_n = \frac{x_1 + \cdots + x_n}{n}$$

Table 3.5: Symbols used in desnaring strategy algorithms.

Symbol	Explanation
n	Raster cells within conservancy
n'	Raster cells n for which snares were predicted to be present
s	Snares within conservancy raster cells n; $s \in n$
s'	Snares within predicted snaring likelihood n'; $s' \in n'$
s''	Replacement snare after removal of snare s'
R	Tally of detected replaced snares s''
S	Tally of detected and removed snares
V	Tally of visits to cells n'
V_l	Memory vector of visited locations
M_s	Memory vector of location of detected and removed snares
L_s	Length of memory vector M_s
N_m	Moore neighbourhood with radius $r = \{0, 1, 2\}$
X	Random sample of size 1
U	Uniform distribution
ε	Noise parameter for epsilon greedy search
$P(d_s)$	Probability of snare detection

where CMA=cumulative moving average, $x_1 \cdots x_n$=number of recovered snares in each run i, and n=number of simulation runs.

Poachers replace snares after rangers removed them ("resnaring"). These snares can be detected only in strategies where the same cell n'_i can be visited multiple times. The simulation tallied detection of first-time removed snares S, and replacement snares R separately. This allowed for the separation of snares found during the first visit (desnaring) and replaced snares found during subsequent visits (resnaring).

3.5.2.2 Development of alternative desnaring strategies

In the first strategy ("Sequential search"), each cell n' for which snaring presence by the Maxent model was predicted was visited. Snares s that were not located on raster cells with predicted snare presence n could therefore not be detected. Furthermore, the number of simulated visits V to each cell was precisely one. The number of detected replaced snares was, therefore, zero as no-repeat visits took place.

In the second search strategy ("Adjacent search"), detection of a snare triggered searching the Moore neighborhood ($N^M_{r=1}$) of the snare cell. The simulation was halted when the number of visits V reaches the length of predicted snare cells n'. This visit count was identical to the visit count in the sequential search strategy because each cell n' was visited exactly once. The standardization of the visit count per scenario made the search efforts comparable between strategies.

The third strategy ("Hotspot search") modeled the snare detection process as a spatial multi-armed bandit problem (MAB) (Vermorel & Mohri, 2005). Simulated desnaring in the hotspot search strategy took place as in the previously described strategies (sequential and adjacent search) if a preset noise variable, ε, was smaller than a generated random number (explore). The Moore neighborhood $N^M_{r=2}$ of a snare location sampled from snare memory M_s was searched (exploit) when ε was larger than the generated random number. The search strategy simulated here was annealing: ε increased when more snare locations were held in memory (White, 2013). The procedure for detecting and removing snares is shown in pseudo-code list Algorithm 1. The hotspot search strategy is summarized in pseudo-code list "Hotspot search" (Algorithm 2).

Algorithm 1: Detection of snares.

```
Input: raster object with s, s', n, n'
Output: R, S, V, M_s
begin
    if n'_i ∈ {s'} then
        generate X ~ U[0, 1]
        if X ≤ P(d_s) then
            set s' = NULL
            replace s' with s''
            set R = R + 1
            set S = S + 1
            set V = V + 1
            set V_l = {V_l, s'}}
            set M_s = {M_s, s'}
            if length M_s > L_s then
                remove M_1
            if strategy==hotspot search ∧ε ≠ ε_max then
                increase ε
            continue
        else
            set V = V + 1
            continue
    else
        continue
```

Algorithm 2: Hotspot search.

```
Input: raster object with s, s', n, n', ε
Output: R, S, V, M_s
begin
    for i = n ∈ n' do
        while i ≤ n' do
            generate X ~ U[0, 1] ;
            if X ≤ ε ;                                   // exploit
             then
                sample M_s ;
                if M_s ∉ V_l then
                    go to M_s ;
                    snare detection N^M_{r=2}
            else
                snare detection n'_i ;                   // explore
```

3.5.3 Simplified search strategy

A simple snare search strategy was developed separately from the three aforementioned desnaring strategies. This strategy was based on insights from environmental criminology. This strategy employs the degree of bushiness and distance between snares and settlements, park infrastructure, and roads. The rationale for these parameters' choice was set out in Section 2.6.

The cutoff distance S_x between settlements and snaring locations was determined by dividing the conservancy into zones through a Dirichlet tessellation (also known as Voronoï tesselation, (Nogueira de Melo et al., 2017)) using the settlements as anchor points. The cutoff distance from each settlement to the snares S_x found in each tessellation slice was calculated, and a cutoff distance based on empirical findings in literature was set. No distinction was made between external settlements (communities, villages) and internal human settlements (staff camps, gates, and lodges). The same procedure for calculating a cutoff distance was followed for park infrastructure.

A Dirichlet tesselation was also used to obtain the relation between snare density and distances to settlements outside the conservancy's perimeters (the so-called empirical density decay curves).

The distance from conservancy boundary to snaring positions B_x was estimated using a Lorenz curve and the Hoover index (Hoover, 1941; UN, 2015). The threshold for the distance between snare position and distance to roads was set at the Hoover index value.

Minimum and maximum cutoffs for the degree of bushiness $\{B_{min}, B_{max}\}$ were calculated during the spatial analysis of the result of desnaring transects. The choice of these values ensured that the selected area was neither completely open nor completely bushy.

The predictive snaring map was obtained by the union of the four raster layers S, I, R, B:

Distance to settlements layer: $S = \{x \in S | x < S_x\}$

Distance to park infrastructure layer: $I = \{x \in I | x < I_x\}$

Distance to roads layer: $R = \{x \in R | x < R_x\}$

Bushiness layer: $B = \{x \in B | x \geq B_{min} | x \leq B_{max}\}$

Union of layers: $L = (S \cup I) \cap B \cap R$

All calculations for the development of the simplified methodology were implemented in the R library `raster`.

Chapter 4: Results

The desnaring reports established by the conservancy reported 602 snares removed in 2018. Eight poachers were sighted in the period 30 November–31 March, during which the fieldwork took place. No poachers were arrested in 2018 and one in 2019, despite a high ranger density of 5.1–7.9 km^2 per ranger. Desnaring transects found a snaring density of 50 snares per km^2. Snares are placed in a clustered pattern near bush-open area transitions, near roads, and near park infrastructure. At least 3% of the conservancy is covered with snaring hotspots. Poachers replace snares removed by rangers within days to weeks. Rangers self-rate their capacity to stop poachers from replacing snares as limited, and 9 out of 31 interviewed rangers said they did not report sighted poachers. The effectiveness of desnaring operations can be improved by 4–9% by reducing the search area and improving the search strategy. Support from surrounding communities in environmental management and governance was not elicited by either the Soysambu Conservancy or KWS.

4.1 Patrolling and poaching patterns

The reported poaching signs were compared with the results of desnaring transects. Furthermore, reported poacher sightings and ranger allocations were analyzed.

4.1.1 Current patrolling and poaching patterns

The Soysambu Ranch deploys a patrol plan to allocate rangers over the conservancy area. The rangers report poacher sightings to their headquarters, and regular desnaring exercises result in desnaring reports.

The allocation of rangers to different areas or objects within the conservancy is set by the patrol plan (Soysambu Wildlife Conservancy, 2019). Allocation sizes per area or object do not change over the year. The ranger allocations, split out over day and night shifts is listed in Table 4.1 and shown in Fig. 4.1. The overall ranger density is 2.9 km^2 per ranger (190 km^2/65 rangers). This number falls to 3.1 km^2 per ranger when corrected for the average of four rangers on leave at any given moment. During the night, the estate's security concentrates on static objects such as gates, stores, and houses. The ranger density, corrected for rangers on leave and not assigned to direct estate duties, then falls from 5.6 km^2 per ranger to 10.6 km^2 per ranger. Foot patrols during nighttime are considered to be too risky because of possible encounters with wild animals. Therefore, mobile patrols are carried out by vehicle during the night.

Table 4.1: Allocation of rangers to day and night shifts and locations within the conservancy.

Allocation	Day shift	Night shift
Gates, lookout, control room	11	12
Mobile, assigned areas	18	1
Mobile, flexible	5	4
Other, not estate-related	3	7
Total	**37**	**24**
Density, overall (km^2/ranger)	5.1	7.9
Density, estate (km^2/ranger)	5.6	10.6
Density, mobile (km^2/ranger)	8.3	38

Rangers on day-shift duties reported eight sightings of suspected poachers during the period 30 November 2018–31 March 2019. There are no instances of poacher sightings or calls for reinforcement during night patrols in the observation books for this period. The Soysambu rangers did not arrest poachers in 2018. On 21 November 2019, one poacher was arrested by KWS rangers who participated in desnaring.

Desnaring in 2018 took place ten times, and is summarized in Table 4.2. The conservancy reported the removal of 602 snares its desnaring reports for 2018 (Soysambu Wildlife Conservancy, 2018b). This number excludes snares removed by rangers during patrolling.

4.1.2 Spatial factors of snare placement

The research involved mapping of snares throughout the conservancy. The relation between snaring patterns and spatial features were assessed.

Figure 4.1: Ranger allocation for day shift (left) and night shift (right). Size of dots indicate number of rangers. Black dots show patrols allocated to areas with known snaring hotspots. Dots on roads represent staffed gates. Mobile staff not assigned to a specific area is allocated to headquarters.

4.1.2.1 Snares removed during desnaring transects

The field research comprised 82 kilometers of desnaring transect walks in the period November 2018–March 2019. During these transects 325 snares and 2 guinea fowl traps were found (Table 4.3, Fig. 4.2). The distribution of dead versus live neck snares is approximately 50/50. Most dead neck snares showed no marks of pincers or pliers.

The lifted neck snares are made out of steel wire used in fencing and placed on animal trails on the edge of open areas and acacia bushes (Fig. 4.4). The snares are placed in a clustered pattern (Hopkins-Skellam test: A=0.042, p-value < 2.2e-16). The median distance to the nearest snares

Table 4.2: Summary of desnaring by the Soysambu Conservancy in 2018.

Month	Live snares[a]	Dead snares[b]	Foot snares	Sum
1	17	0	0	17
2	55	13	0	68
4	156	57	1	214
5	25	13	6	44
6	57	0	0	57
8	0	2	0	2
9	2	5	0	7
10	17	19	2	38
11	73	23	5	101
12	38	16	0	54
Total	440	148	14	602

[a] Live snares are snare that are functional: attached to a tree with an intact noose.
[b] Dead snares are snares that are either not attached to a tree or have no intact noose.

is 12 meters, and the 95% quantile is 250 meters. The transected area's snaring density was calculated as 4 snares per km walked or 50 snares per km^2 searched (average swath width: 80 m).

Table 4.3: Snares found in desnaring transects during the research.

Category	Dead snares	Live snares	Sum
Foot snare	2	15	17
Guinea fowl trap	2	0	2
Neck snare	150	158	308
Totals	154	173	327

Foot snares are placed on animal trails in open areas and are not placed in clusters. These snares consist of two holes and a car towing cable (Fig. 4.3). One hole is used to anchor the snare 0.6-0.7 meters into the ground with a wooden peg. The other hole is covered by a bucket cover with

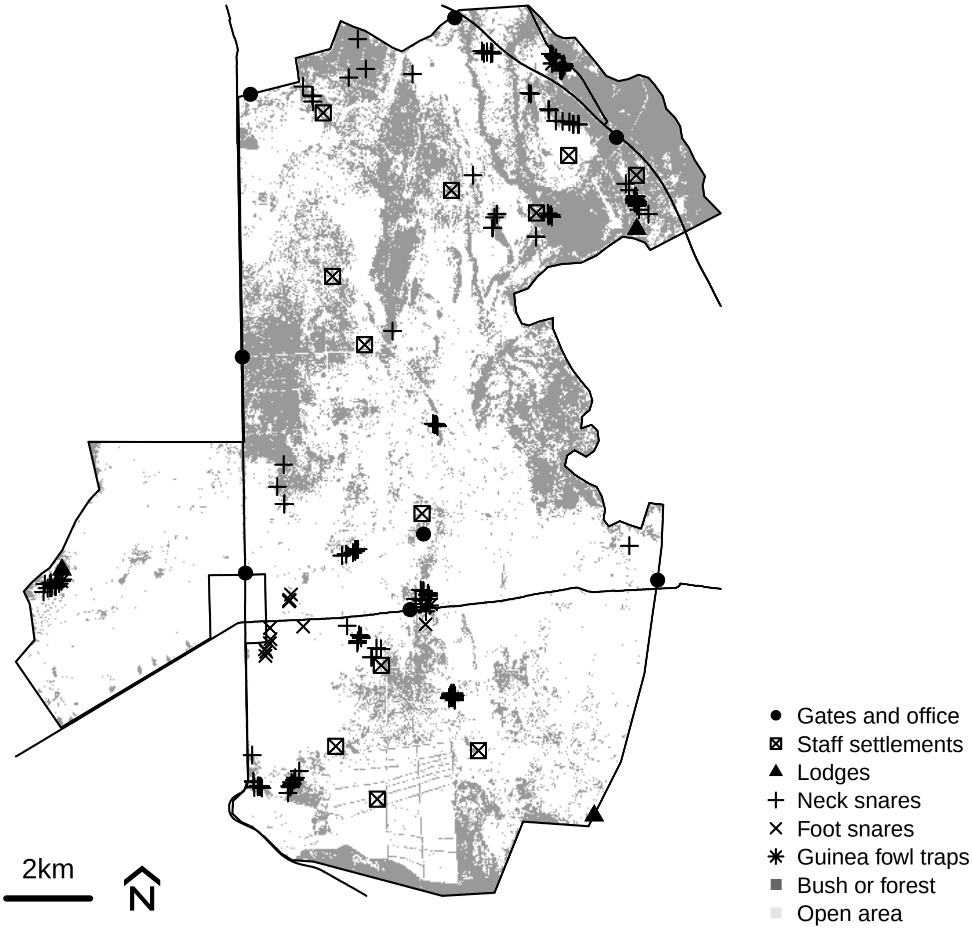


Figure 4.2: Locations of removed snares in the Soysambu Conservancy in relation to vegetation and park infrastructure.

a star-shaped incision. Once the animal steps on the bucket lid, the hoof sinks into the hole, thus trapping the animal. The lid is covered with soil, which makes the snare hard to detect. These foot snares are always placed directly on animal trails.

Several snared animals were found during the fieldwork. This included six zebras, one waterbuck, one Grant's gazelle, and one hyena. Furthermore, animals with snare injuries were frequently observed.

(a) Anchor hole and trap hole.

(b) Foot snare with cover removed.

(c) Foot snare cover.

(d) Top view of foot snare.

Figure 4.3: Overview of foot snare construction. Pictures by Henk Harmsen.

(a) Neck snare on animal trail.

(b) Removal of a snare from a hyena.

Figure 4.4: Neck snare (a) and by-catch, in this case a hyena (b). Pictures by Erik Klein Wolterink.

4.1.2.2 Location of snares

The strength of association between neck snare positions and covariates is shown in Table 4.4.

Table 4.4: Strength of association between snare positions and spatial features.

Variable	AUC[a]
Fraction bush/open area	0.782
Distance to roads	0.705
Distance to park infrastructure	0.655
Water troughs	0.581
Distance to communities	0.551
DEM / Elevation	0.549
Distance to boundaries	0.507

[a] Area under curve

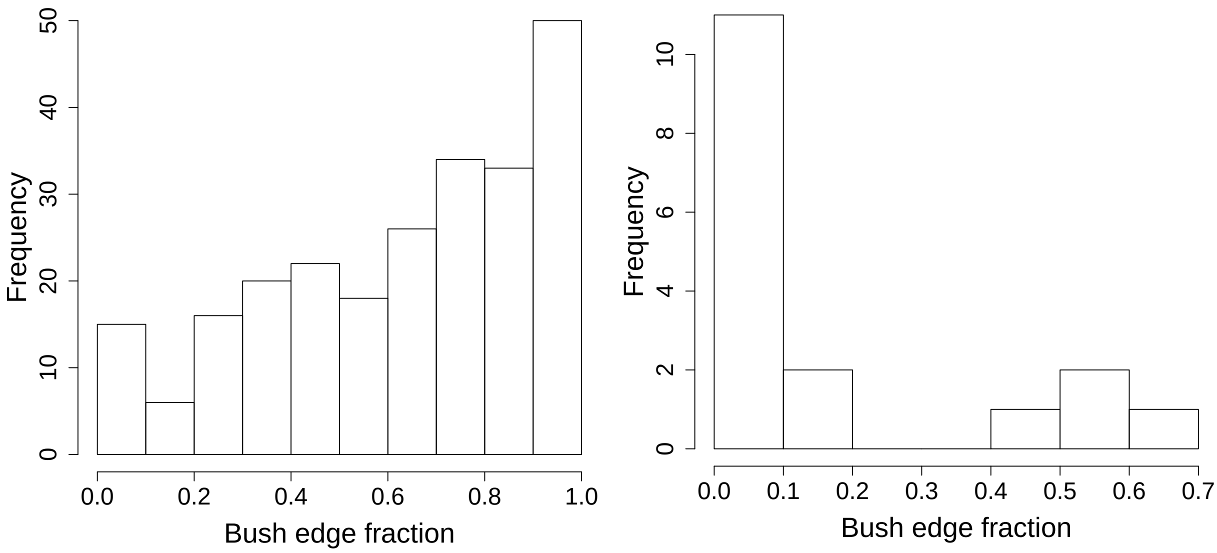
4.1.2.3 Fraction bush/open area

Fig. 4.5(a) shows the snare positions superimposed on the bush to edge transition, in which "zero" is either bush or open area, and "one" represents a 50% bush to 50% open area transition. Only the edges are plotted. Both the figure and the histogram for the bush edge fractions (Fig. 4.6(b)) shows that most snares are placed in the bush-open area transition.

Foot snares were generally placed in open areas around Elementeita village (Fig. 4.5(b)). The foot snares are placed on wildlife tracks and are hard to detect. The snares are of the "hole and lid" type (Fig. 4.3) around Elementeita village, but consisted of an above-ground loop deeper into the conservancy.

(a) Neck snares and vegetation density. **(b)** Foot snares and vegetation density.

Figure 4.5: Spatial positions of neck snares (a) and foot snares (b) over open-area bush transitions.

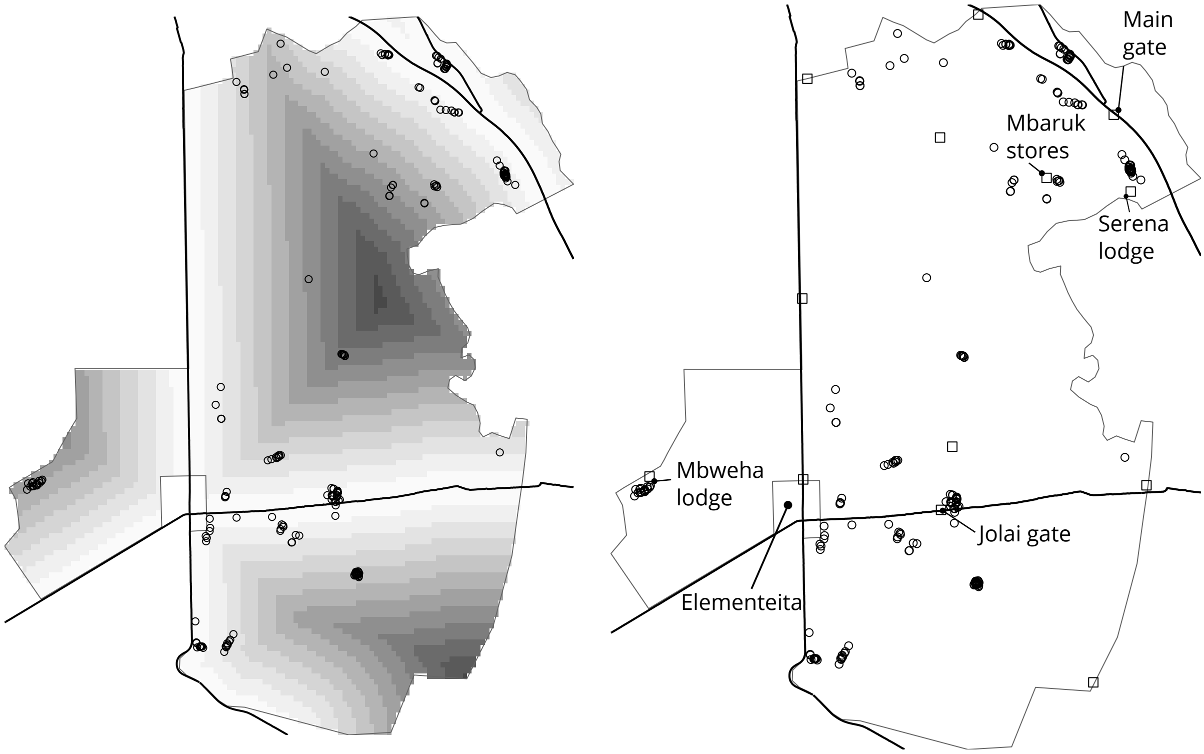


(a) Neck snare frequency distribution. **(b)** Foot snare frequency distribution.

Figure 4.6: Foot and neck snare density over vegetation density. (a) histogram for neck snare positions in relation to degree of bush-open area transition (zero: either bushy or open area; one: bush-open area transition); (b) ditto for foot snares.

4.1.2.4 Distance to roads and park infrastructure

Neck snares were often found near public roads (Fig. 4.7(a)), mostly within 1300 meters (Gini index=0.49, Hoover index=0.37 at 63% quantile).



(a) Distance to roads. **(b)** Distance to park infrastructure.

Figure 4.7: Distance of snares to roads (a) and park infrastructure (b).

The desnaring teams found neck snares near park infrastructure, such as gates, staff settlements, and lodges (Fig. E1(d)). The count of snares per tessellation tile (Fig. E2) showed that snares were generally found close to park infrastructure (50% within 1 km; 80% within 2 km) (Fig. E3). The relationships between snare positions and distance to watering troughs were moderate. No association could be established between snares and the boundaries of the conservancy. The snares occurred within a relatively narrow elevation band (1820–1835 m+MSL), although the strength of this association was found to be weak.

4.1.2.5 Distance to communities

The association between neck snare positions and their distance to settlements was weak. However, neck snares were placed near some settlements (Mbaruk Centre, Elmenteita, Kong'asis, Kiungururia). The relation between neck snaring and these villages were estimated through a Dirichlet tessellation, using the villages as anchor points (Fig. E4). The tile sizes do not take the effect of roads on reducing travel time into account.

The snare density as a function of distance from each village with the allocated tiles is shown in Fig. 4.8. The Elementeita density curve's shape shows a small peak for foot snares at 1 km and a peak for neck snares at 3–6 km. The Mbaruk stores density curve is bimodal with peaks at 2 km and 6 km from the village. Snares situated at the second peak (~ 6 km from Mbaruk village) are reachable from Kekopey town by the A104 Nairobi-Nakuru highway.

4.1.3 Detection of snaring hotspots

A methodology for the detection of snare clusters was applied and compared with an established cluster detection methodology.

4.1.3.1 Identification of hotspots

The nearest distances were visualized in a Stienen diagram; snare hotspots are shown as clustered small discs (Fig. 4.9). The snare positions were dilated by 250 meters. This corresponds with the nearest neighbor distance 95% cutoff.

The hotspots were determined by drawing a convex hull around the Steiner sets containing at least 5 snares within the amalgamation (Fig. 4.10). The identified hotspots (Fig. 4.9, shaded, n=16) cover 3% of the conservancy.

The Allard-Fraley hotspot methodology identifies fewer hotspots (n=10) and does not account for the increased risk of snaring around them. No snaring hotspots were identified that were not already found by the high-intensity hotspot methodology (Jaccard index=0.675).

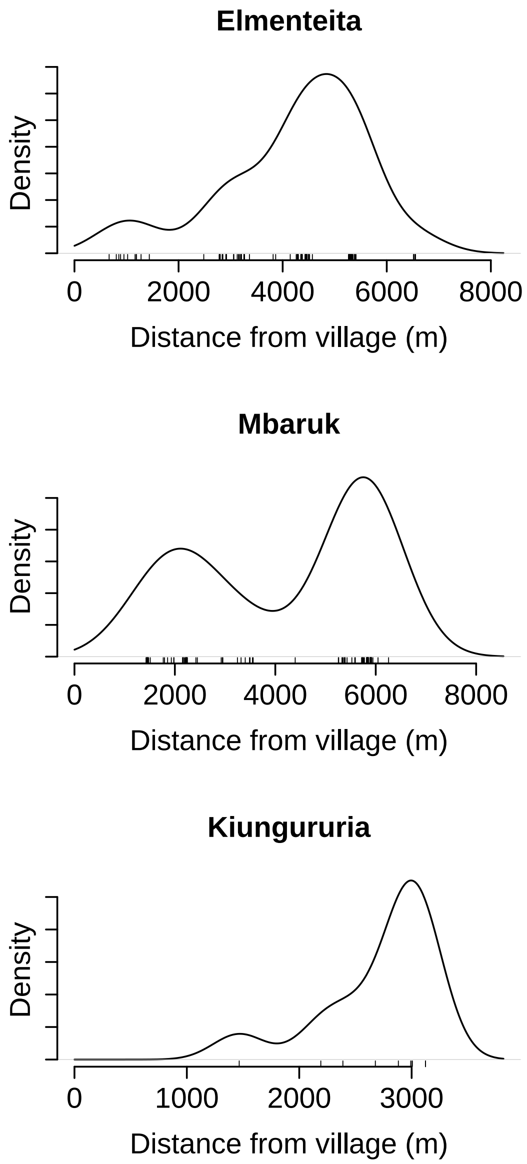


Figure 4.8: Snare density as a function of the distance from communities surrounding the conservancy. The maximum distance is constrained by the extent of the Dirichlet tile (Fig. E4).

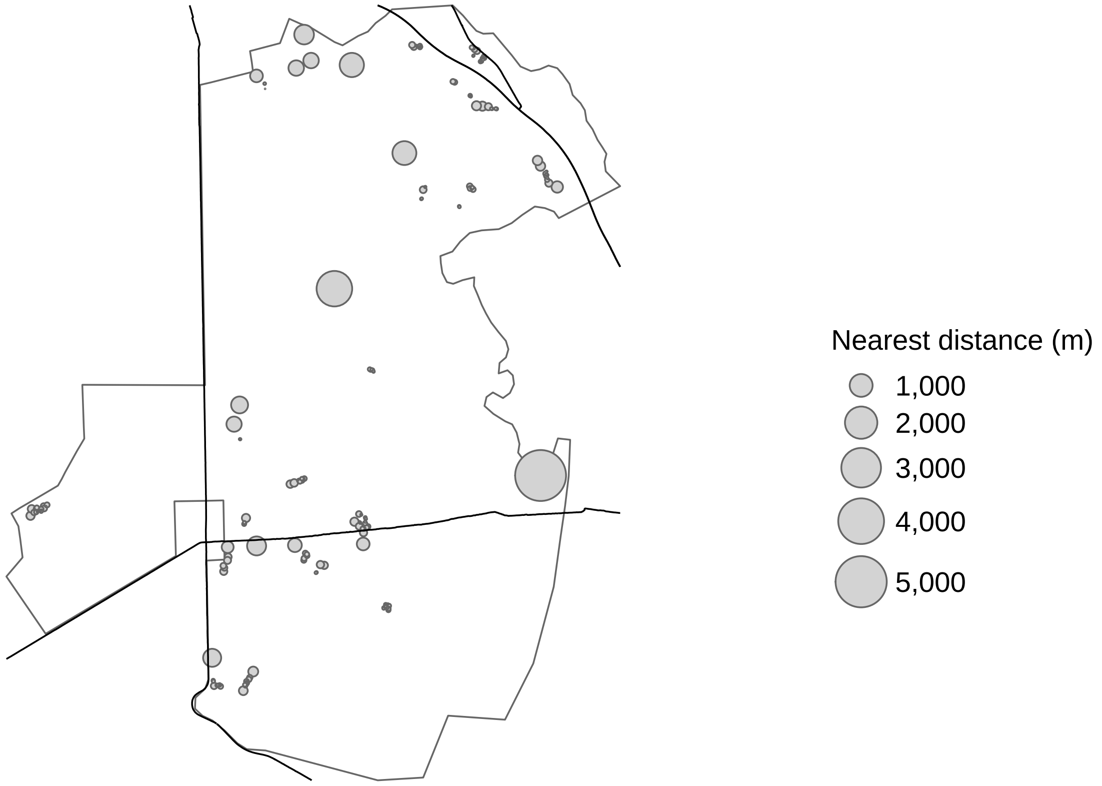
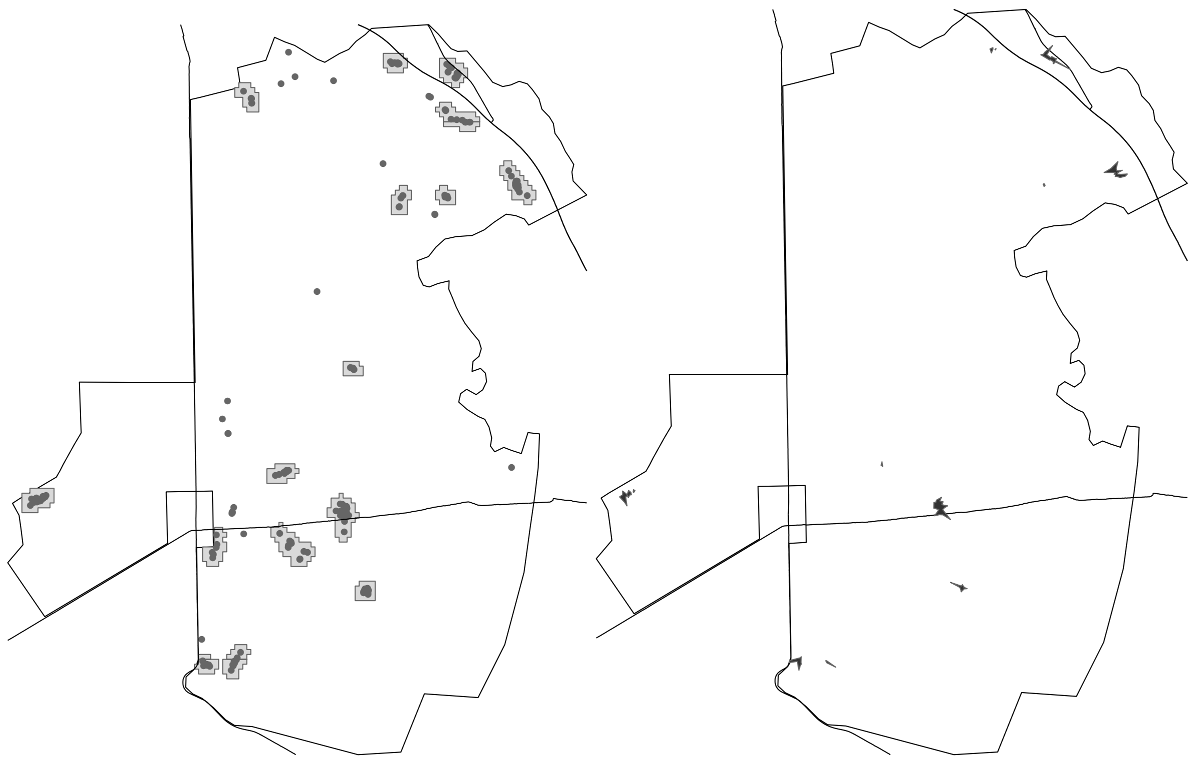


Figure 4.9: Stienen diagram for snare distances.



(a) Hotspots (high intensity zones).

(b) Hotspots (Allard-Fraley).

Figure 4.10: (a) Hotspots identification using the high-intensity zone methodology, based on nearest distance of 250 meters and a minimum of 5 snares per hotspot. (b) Hotspot identification with the Allard-Fraley cluster set hotspot methodology.

4.1.3.2 Re-snaring

A hotspot was visited three days after desnaring and was found to be repopulated with snares. Three known hotspots were subsequently visited at least three times (Fig. 4.11). The repeat visits showed that poachers replace snares which were removed by rangers (Table 4.5).

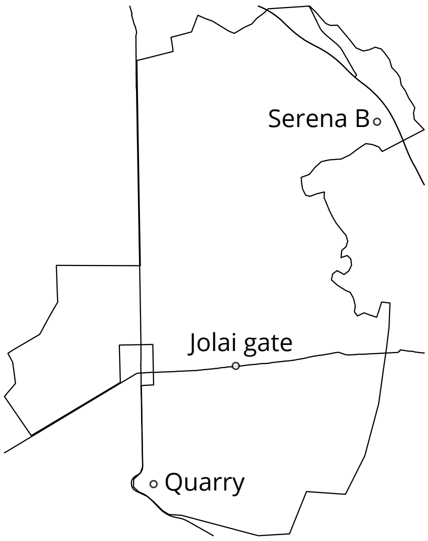


Figure 4.11: Locations of repeat desnaring transects locations.

Table 4.5: Number of snares found during repeat desnaring transects.

Date[a]	Hotspot name[b]	Snares found
2019-01-24	Serena B	20
2019-03-26	Serena B	13
2019-03-27	Serena B	3
2018-12-05	Jolai Gate	11
2019-01-17	Jolai Gate	19
2019-03-27	Jolai Gate	12
2018-11-22	Quarry	3
2018-12-06	Quarry	6
2019-01-25	Quarry	3
2019-03-26	Quarry	14
2019-03-27	Quarry	2

[a] Date of carrying out the repeat desnaring transect.
[b] Serena B is located near the Serena Lodge in the north-eastern part of the conservancy. Jolai Gate is found mid-way the D321 Elementeita – Kikopey road. Quarry is found in the south-western border of the conservancy.

4.2 Ranger capacities

Rangers reported that they frequently saw poachers but could not stop them from poaching in the conservancy. The reasons given were predictable patrolling strategies, lack of arms, and slow response times when reinforcement is called. The reported frequency of recorded poacher sightings in the conservancy's logbook was lower than the frequency of observing poachers as reported during the interviews.

4.2.1 Deterrence

Rangers are tasked with deterring poachers from entering the conservancy. Whether deterrence is achieved depends on poaching pressure, equipment, patrol pattern predictability, and the extent to which surrounding communities provide information on impending poaching operations or poachers' identity.

4.2.1.1 Perceived severity of poaching

Rangers thought that poaching in Soysambu at present is severe. However, they perceive that poaching was worse in the past (5 to 10 years ago) (Fig. 4.12). They are moderately optimistic about the reduction of poaching in the future. This is either because they think the maximum level of poaching has been reached, or because they think that deterrence will become more effective in the future, provided that management of the conservancy implements additional security measures.

Poaching is typically carried out by armed groups of at least three men in the age group 20–45 years, who concentrate their activities in hotspots (Fig. 4.13). These poachers were thought to originate from nearby communities. Poaching is considered to occur throughout the year, but its prevalence peaks in November–March. This period coincides with both festive periods (Christmas, New Year) and the required payment of school fees for parents with children. Rangers hypothesize that this increases the bushmeat demand by villagers and the cash demand for poachers who must pay school fees.

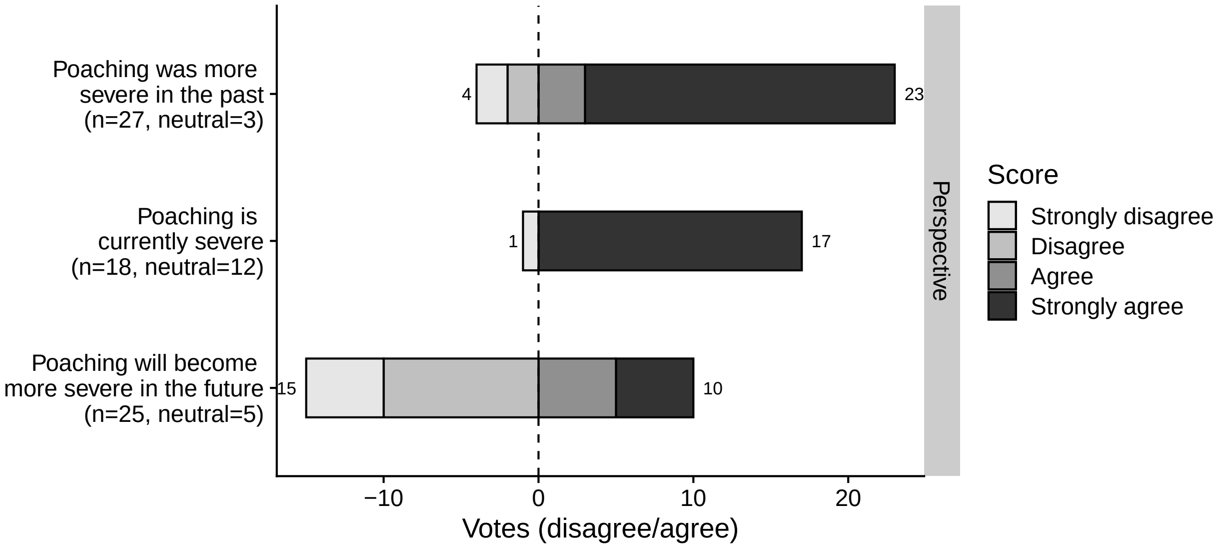


Figure 4.12: Past, current and future poaching levels as perceived by rangers. Results of statistical tests are included in Annex F.

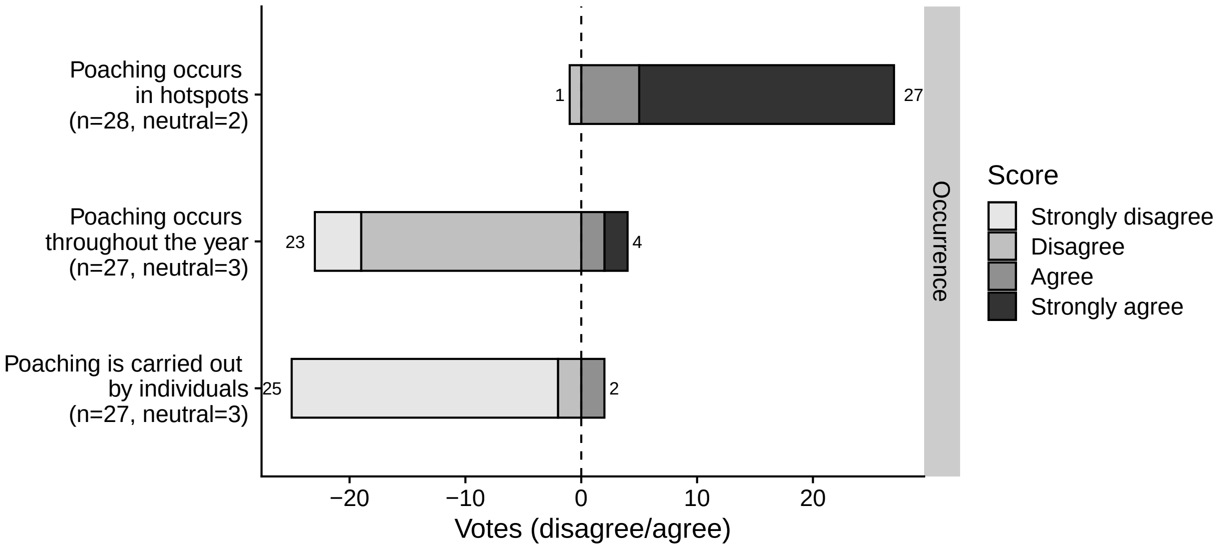


Figure 4.13: Occurrence of poaching within the conservancy. Results of statistical tests are included in Annex F.

4.2.1.2 Resource limitations and ranger morale

The rangers estimate that their capacity to deter poachers is limited (n=20) (Fig. 4.14). This is a significant majority (two-sided exact binomial test, $x = 6, n = 26, H_0 = 0.5, \alpha = 0.05, p = 0.009$). They are unarmed and face armed poachers who normally outnumber them. Furthermore, they feel that there are insufficient rangers for the area to be covered (n=18). This is not a significant majority (two-sided exact binomial test, $x = 10, n = 28, H_0 = 0.5, \alpha = 0.05, p = 0.185$). Transport and equipment are considered to be insufficient (n=24) (significant: two-sided exact binomial test, $x = 2, n = 26, H_0 = 0.5, \alpha = 0.05, p < 0.001$).) Rangers clarified that arms are the most urgently lacking equipment because they are unarmed and outnumbered (n=15) (not significant; two-sided exact binomial test, $x = 15, n = 24, H_0 = 0.5, \alpha = 0.05, p = 0.308$).). Therefore, according to some of the rangers, the issue with under-staffing is not so much an absolute lack of human resources but more a lack of firepower.

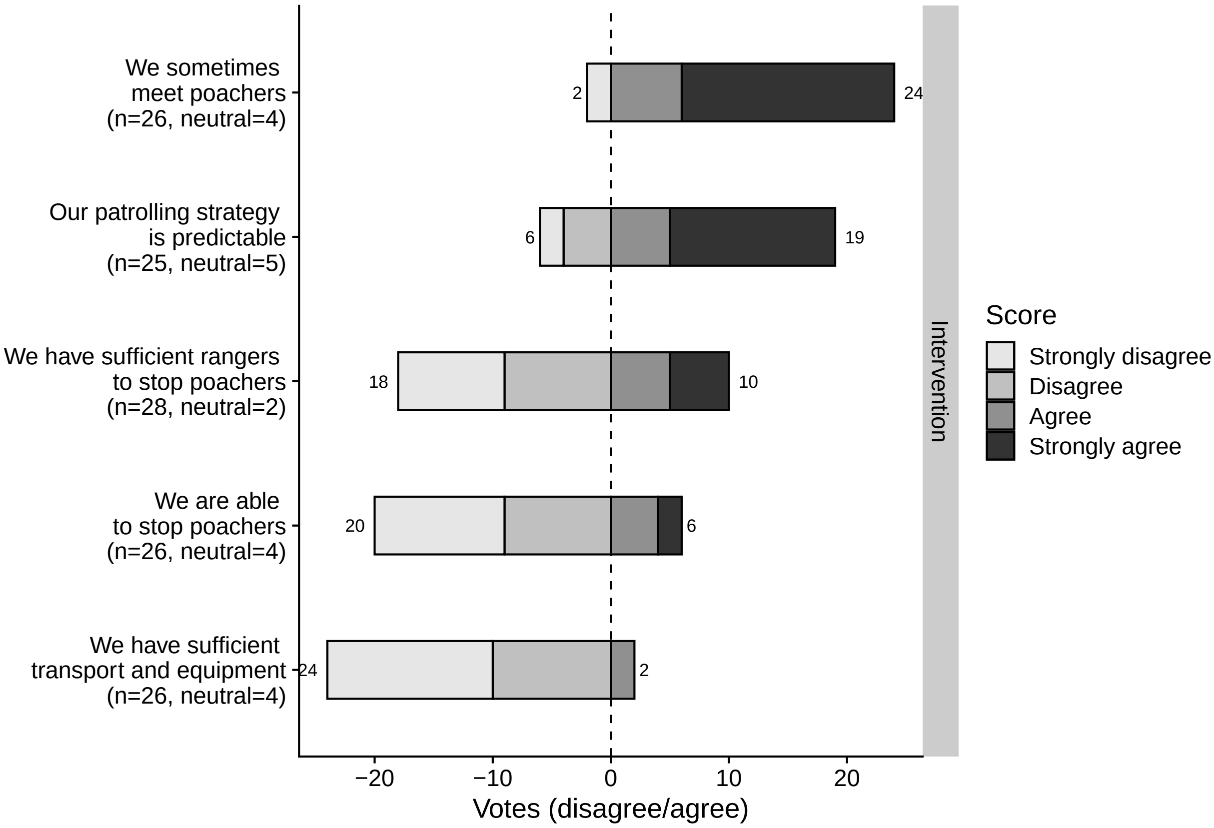


Figure 4.14: Rangers' capacity to stop poachers. Results of statistical tests are included in Annex F.

Transport was also seen as important equipment because of reinforcement response times (n=13). (not significant; two-sided exact binomial test, $x = 13, n = 24, H_0 = 0.5, \alpha = 0.05, p = 0.839$). Rangers can call for reinforcement if they see a group of poachers. Response times are slow because there are just two patrol vehicles available (n=11) ()significant; two-sided exact binomial test, $x = 11, n = 13, H_0 = 0.5, \alpha = 0.95, p = 0.022$). By the time reinforcement arrives, the poachers have disappeared.

There is almost no equipment available to detect poachers. Only rangers located at the observation point (located on Flamingo Ridge) have access to a pair of binoculars. No night vision equipment is available for night patrols.

The rangers point out that morale is not high. The salaries are deemed to be insufficient (n=16, raised during discussion of open questions). Some rangers have side-jobs and worry about their ability to maintain their family. This does not only result in fatigue during work but also in risk-averse behavior because ”it is not worth taking any risks”, for example, of physical injury during attempted poacher arrest. This is further aggravated by the suspicion that poachers know them and where they live (n=25; significant; two-sided exact binomial test, $x = 25, n = 28, H_0 = 0.5, \alpha = 0.05, p < 0.001$) and that they are considered to exact revenge after they are released from arrest. Punishments for poachers, once arrested, are thought to be light. Rangers, therefore, think that they will face released poachers probably sooner than later.

4.2.1.3 Patrolling predictability

Most rangers believe that their patrolling strategies are predictable (n=19 , significant; two-sided exact binomial test, $x = 19, n = 25, H_0 = 0.05, \alpha = 0.05, p = 0.015$) (Fig. 4.14). The allocation of rangers and vehicles can be inferred through observation or via information leakage (n=9, open questions). Such information leakage can occur when rangers share work and living spaces with other staff members. For example, the security control room is located within the office compound in which people can walk in and out; communication is over a shared radio channel so that everyone can listen in; people from outside the farm move in and out and sometimes stay within the area for extended periods. The working hours of rangers are organized in fixed shifts (day shift 6 a.m. to 6 p.m. and a reduced ranger density during the 6 p.m. to 6 a.m. night shift). This makes it easy for poachers to displace poaching activity. Three rangers alluded to collusion between rangers and poachers when information leakage was discussed .

Rangers evaluate the predictability of night-time patrolling as particularly high. During this time, patrolling is carried out using two vehicles. The patrol vehicles are manned[1] with a driver and a supervisor. The patrolling rangers have limited options when poachers are spotted. The driver has to stay with the car, and the supervisor cannot be expected to arrest a group of armed poachers on his own. Moreover, response times for reinforcements are slow. Rangers also observe that the patrol vehicles, once passed, are unlikely to return, which adds to the predictability of the

[1] Only men carry out patrol duties during the night.

nightly patrolling pattern. Finally, a car can be heard and seen from afar, which gives poachers ample time to hide in the bush. The rangers in the car are unlikely to see these poachers without night vision equipment.

4.2.1.4 Relations between poachers, rangers, and communities

The rangers were asked about familiarity between them, poachers and communities (Fig. 4.15, Fig. 4.16). The rangers are convinced that the communities know the identities of the poachers (significant; two-sided exact binomial test, $x = 28, n = 28, H_0 = 0.5, \alpha = 0.05, p < 0.001$). However, there are few informers, and communities do not appear to be willing to help the rangers to stop poachers. The rangers see four reasons for this. First, the communities may fear retributions from poachers if they would be exposed. Second, the poachers may be either family of other community members or family of Soysambu staff members. Third, poachers supply cheap meat to fast-growing communities and create “jobs”. Finally, there is no agreement between the rangers concerning assistance for communities by the conservancy, for example, in the form of water supply or addressing “human-wildlife conflict” (not significant; two-sided exact binomial test, $x = 10, n = 23, H_0 = 0.5, \alpha = 0.05, p = 0.678$). In turn, the communities do not feel obliged to inform the conservancy of impending poaching activity or the identity of poachers.

Most rangers believe that the poachers know them ()significant; two-sided exact binomial test, $x = 25, n = 26, H_0 = 0.5, \alpha = 0.05, p < 0.001$). By contrast, the rangers do not know many poachers, but rather “a few names and a few faces” (not significant; two-sided exact binomial test, $x = 7, n = 19, H_0 = 0.5, \alpha = 0.05, p = 0.359$). Rangers think that livestock herders may know at least some poachers. However, these herders do not often give information or give it very late; rangers think that they may be intimidated by the poachers. In summary, the relations between community members, rangers, and poachers as perceived by rangers are asymmetric, as shown in Fig. 4.16. Here, the relation between community members and poachers was measured through the question: ”Do communities know poachers?”; the relation between poachers and rangers by the questions ”Do poachers know who the rangers are?” and ”Do the rangers know

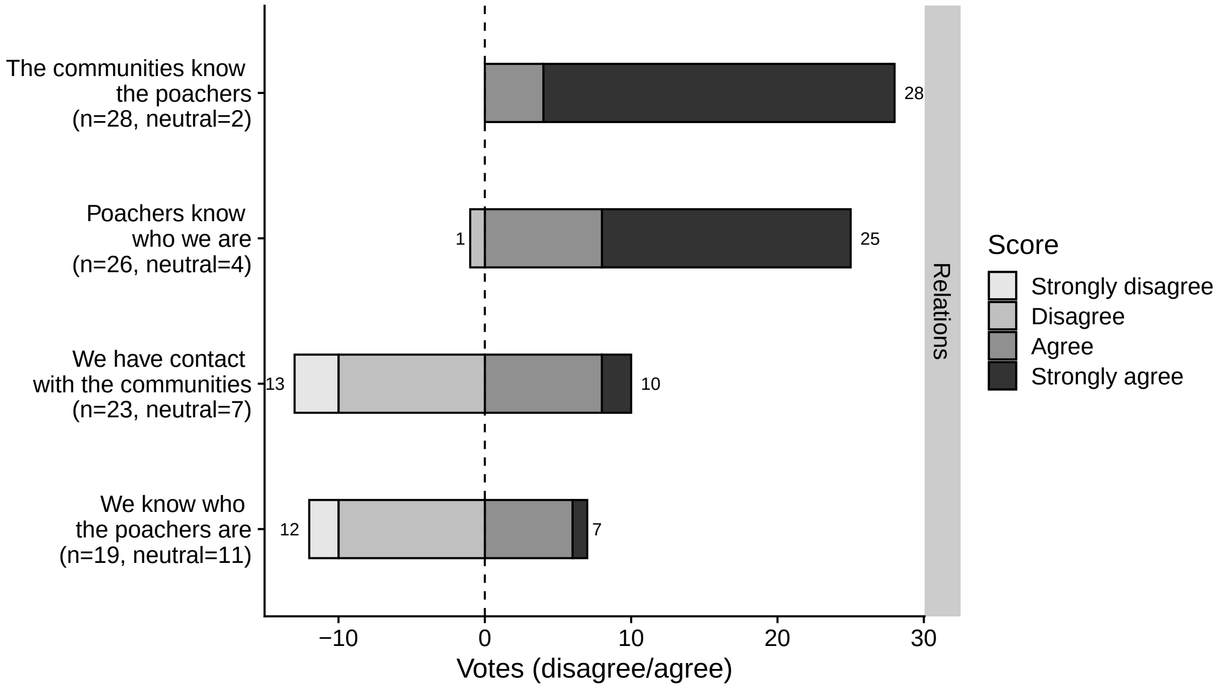


Figure 4.15: Relations between poachers, rangers and communities Results of statistical tests are included in Annex F.

who the poachers are?"; the relation between rangers and communities was measured through the question "Do you have contact with the communities?". Details of the statistical tests used (two-sided exact binomial, $H_0 = 0.5, \alpha = 0.05$ are included in Annex F.).

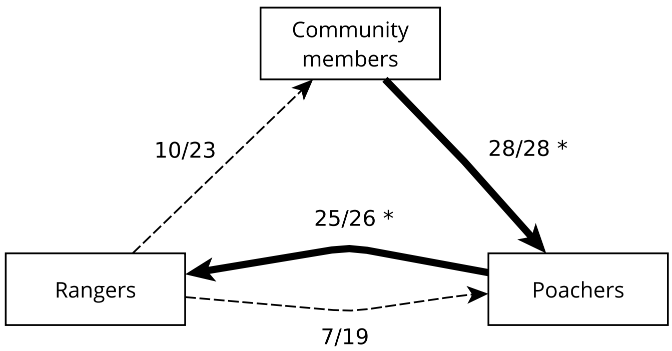


Figure 4.16: Schematic overview of relations between community members, rangers and poachers. Solid arrows and numbers marked with an asterisk represent significant relations. The numbers represent affirmative answers and total number of responses given. The counts correspond with the results shown Figure 4.15. The statistical test results are included in Annex F.

4.2.1.5 Poacher sightings

Rangers frequently sight poachers (Table 4.6) (significant; two-sided exact binomial test, $x = 24, n = 26, H_0 = 0.5, \alpha = 0.05, p < 0.001$). Confrontations between poachers and rangers in the field are rare. When seen, poachers generally move away from the rangers, often covering their faces. Rangers indicated during the interviews that they move away from the poachers as well. Furthermore, some rangers were threatened by poachers (n=5, open questions) and experienced violence at home or during attempted arrest (n=6, open questions).

The limited intervention capability, the perceived predictability of patrolling patterns, and general lack of ranger morale (low salaries, slow reinforcement response times, lack of equipment) are given as a motivation by some rangers for not reporting poacher sightings. These rangers admit that they do not intend to stop poachers or call a reinforcement, and instead look the other way ("We pretend that we don't see them" (n=5, open questions). Others indicated so implicitly ("You could see them every day if you want to") (n=4, open questions).

The frequency of poacher observations reported by rangers during interviews is summarized in Table 4.6. Based on these data, ten rangers would report more than double the two sightings per month (average of eight observations over four months) registered in the observation book.

Table 4.6: Frequency of poacher sightings per month by the interviewed rangers.

Frequency	n rangers	freq/month[a]
Twice per week	3	8.67
Weekly	7	4.33
Twice per month	6	2.00
Monthly	7	1.00
Quarterly	3	0.33
Yearly	1	0.08
Never	2	0
No opinion	2	n.d.
Total[a]	31	2.64

[a] The total frequency per month is calculated as a weighted mean.

Comparison of the observation book data and the ranger-reported sightings as appearing from the interviews show a mismatch (Table 4.7). The expected frequency of seeing poachers should match the maximum frequency of seeing poachers if each ranger would report his or her sighting.

Table 4.7: Comparison of reported poacher sightings (LEM data) versus poacher sightings as stated during interviews (Research data).

Variable	LEM data	Research data	Remarks
n rangers	65	31	Total ranger force (LEM) versus interviewed sample (Research data)
Average observations per month	2	2.64	Eight observations in four months (LEM) versus weighted average per month (Table 4.6)
Maximum	n/a	8.67	See Table 4.6

4.2.2 Prediction of poaching

Rangers expected snares to be placed near park borders and human settlements and far from park infrastructure (significant; $x = 25, n = 26, p < 0.001$ for placement near park borders; $x = 2, n = 26, p < 0.001$ for placement near offices and gates; both tests two-sided exact binomial, $H_0 = 0.05, \alpha = 0.05$). During desnaring transects, snares were found near park infrastructure and near roads rather than park boundaries. Rangers estimated snare densities before carrying out a desnaring transect by assuming that all snares removed during previous desnaring transects were removed.

4.2.2.1 Estimation of snare positions

The rangers were asked to rate the likely positions of snares vis-à-vis spatial features of the conservancy (Fig. 4.17, Table F5).

Water availability is not thought to have a strong local spatial influence as water troughs are available for livestock throughout the conservancy. The interviewed rangers think that snares are not placed near offices and gates, as this would be noticed by conservancy staff. There is no

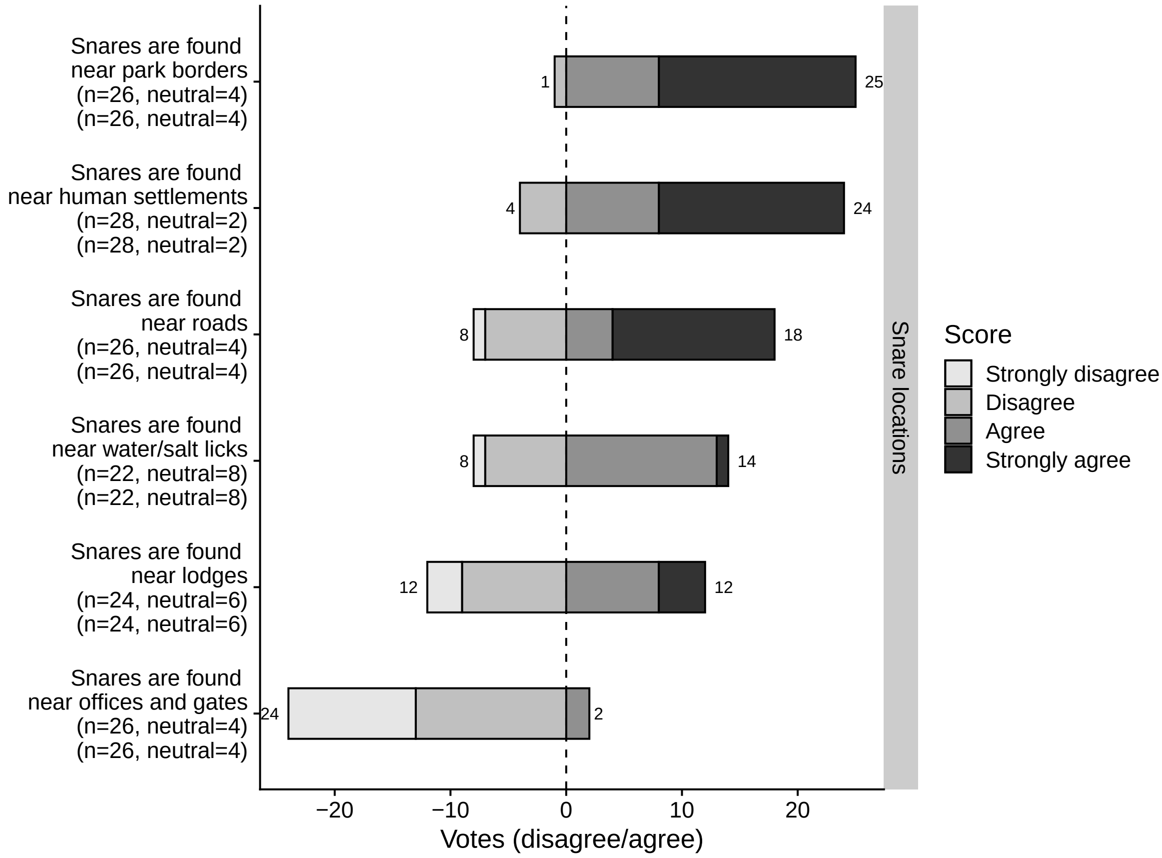


Figure 4.17: Rangers opinions on spatial and temporal prevalence of snaring. Results of statistical tests are included in Annex F.

consensus on the relations between snare positions and lodges (not significant; two-sided exact binomial test, $x = 12, n = 24, H_0 = 0.05, \alpha = 0.05, p = 1.000$). The most likely locations for snaring hotspots are deemed to be near park borders and human settlements.

The expected locations of snaring hotspots can be compared with discovered hotspots' actual locations during desnaring transects (Fig. 4.18). Hotspots were found near a gate and a lodge. Finding snares near a staff settlement and a park gate was unexpected for the rangers.

4.2.2.2 Estimation of snare densities

Rangers were asked to estimate the number of snares found in an area before the desnaring transect and motivate their estimation. The estimates have been compared with the actual number of snares found during the desnaring transects.

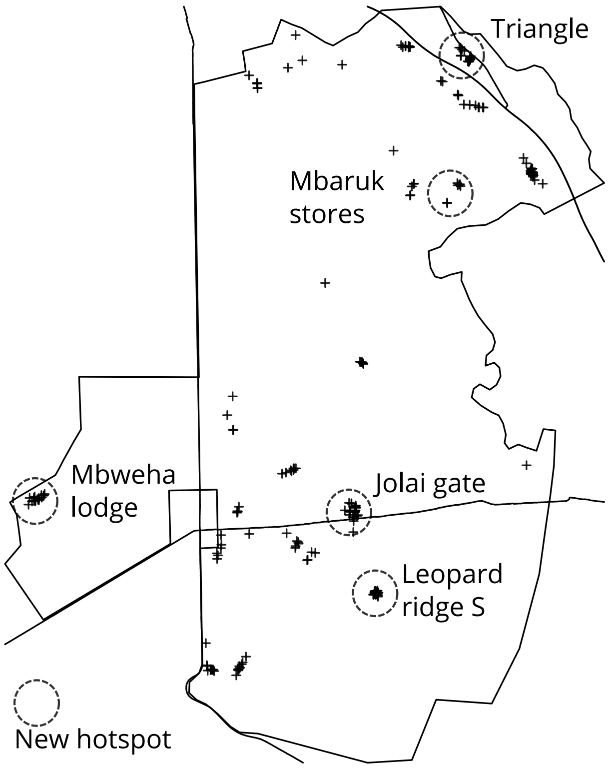


Figure 4.18: Snaring hotspots discovered during desnaring transects which were unknown to the conservancy.

The results were assessed with an exact binomial test. The ratio r of collective error (CE) and individual error (IE) is deemed satisfactory if $r \leq 0.1$. There are 3 such cases, and the probability that rangers estimate the true amount of snares in a transect with $r \leq 0.1$ is not significant (one-sided exact binomial test, $x = 3, n = 27, H_0 \leq 0.1, \alpha = 0.05, p = 0.0718$). The criterion for successful estimation can be relaxed by requiring that the range of estimates made by rangers $[min, max]$ includes the actual number of snares found and that the average of successes is larger than the chance value of 0.5. There are 19 such cases, and the probability of finding a group estimate that contains the true number of snares in its range is significant ((one-sided exact binomial test, $x = 19, n = 27, H_0 \leq 0.5, \alpha = 0.05, p = 0.026$).

The motivation for making a particular estimation was asked at the beginning of each transect. The estimates were based on the number of snares found during the last transect in the area if such desnaring had taken place. In some cases, the nearest community was evaluated to be more or less prone to poaching (n=9) as an additional criterion. The estimated snare count range did not contain the actual number of snares found in eight cases. Over-estimations occurred when no snares were found in a known hotspot (n=2). The conservancy staff under-estimated the snaring density in cases where snares were found in areas that were not previously desnared (n=6).

4.2.3 Community relations

Six interviewees representing seven settlements around the conservancy were interviewed.

Three out of the six interviewed community representatives reported that animals from Soysambu cause damage to livestock and crops. These incidents are reported to KWS, but no compensation for damages has materialized to date. Two communities are separated from the conservancy with an electric fence, and one community is situated next to an area where wildlife has been largely extirpated. The three remaining communities where damages occur saw this as a problem that is caused by the conservancy. They proposed that electric fences are mounted to shield them from wildlife.

Four out of six communities said that the conservancy has not set up projects or activities to help them. Elementeita village has piped water and a dispensary. Some more activities were organized through private initiatives of Soysambu staff, such as tree planting, improved stoves, the provision of school desks, and a greenhouse. One representative indicated that Soysambu had employed some youths for security and that hay is made available at a reduced price.

None of the community representatives had heard of or participated in the draft KWS management plan for the wider Lake Elementaita ecosystem submitted by the National Museums of Kenya to the UNESCO World Heritage Center. Each of the interviewees had concerns that would have been communicated to KWS if they would have been consulted. Three community leaders are worried about developments such as hotels in the riparian zone of the lake. Two representatives were concerned about waste from hotels flowing into the lake, and one representative stressed the importance of protecting Lake Elmentaita's catchment area.

When asked what could be done to improve the situation, four community leaders stated that Soysambu could help most by providing clean drinking water, as groundwater in the region is contaminated with fluoride. Two leaders stated that there are not enough jobs for their youths.

Community leaders said it was difficult to stop poaching if there were no jobs, no awareness of the importance of wildlife, and no compensation for crop and livestock damage. One leader made clear that wildlife would be killed if damages to crops and livestock remain uncompensated.

4.3 Improved desnaring strategies

The desnaring effectiveness was improved by reducing the search area, optimal allocation of rangers to search effort, and simplifying the search area reduction methodology. The improvement over the baseline scenario amounted to 4–9% for snare detection probabilities of 20% and 40%, respectively.

4.3.1 Reduce search area

The results of desnaring transects from the first phase of the fieldwork were used as input in a Maxent SDM model. The resolution of the raster stack containing variables was reduced to 250x250 meters for all input variables. This resolution corresponds with the 95% nearest neighbor distance found in the first phase of the fieldwork. The geospatial thinning operation reduced the number of snares per raster cell to one, resulting in 68 snare positions as presence data.

The four-fold cross-validation yielded AUC=0.853, TSS=0.587 and Boyce index=0.91 (Fig. 4.19). The values for each of these indicators suggest a performance which is better than a null model (AUC > 0.5, TSS > 0, Boyce index > 0.5). The variable importance found by the Maxent model is listed in Table 4.8.

Table 4.8: Variables used in Species Distribution Model (SDM) with percent contribution of each.

Variable	%
Bush-open area transition	61.9
Elevation	16.5
Distance to roads	12.7
Distance to park infrastructure	8.9

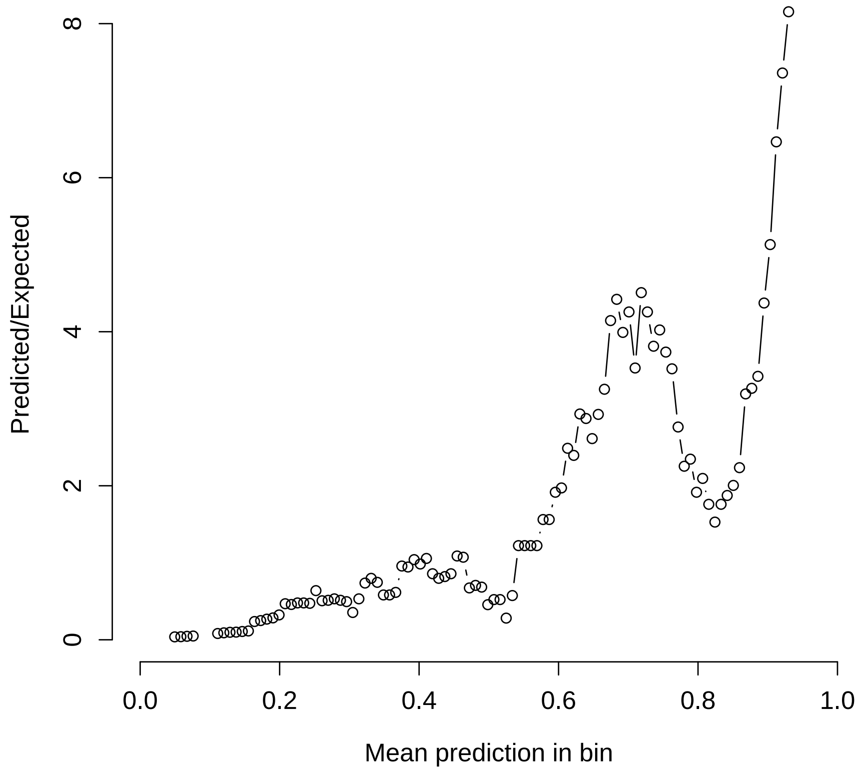


Figure 4.19: Boyce plot for Maxent presence-only Species Distribution Model (SDM) (Boyce index=0.91).

During 45 km of validation transects 92 snares were found (Fig. 4.20). The mean snaring density for low potential areas (as predicted by the Maxent model) was 0.94 snares/km transect, while this was 2.28 snares/km transect for the high potential areas (median: 1.87 and 0 snares/km transect, respectively). The difference in detected snare densities for low and high snaring potential areas was significant (Wilcoxon ranked sum test: p=0.04, r=0.344, α=0.95).

The addition of snare positions from validation transects to known snare positions from the initial desnaring transects did not alter the conclusion regarding the clustered pattern of snaring. The Hopkins-Skellam test also showed significant clustering when all detected snares are considered (A=0.052, $p < 2.2e-16$). Three new snaring hotspots were identified during the validation desnaring transects (Fig. 4.21).

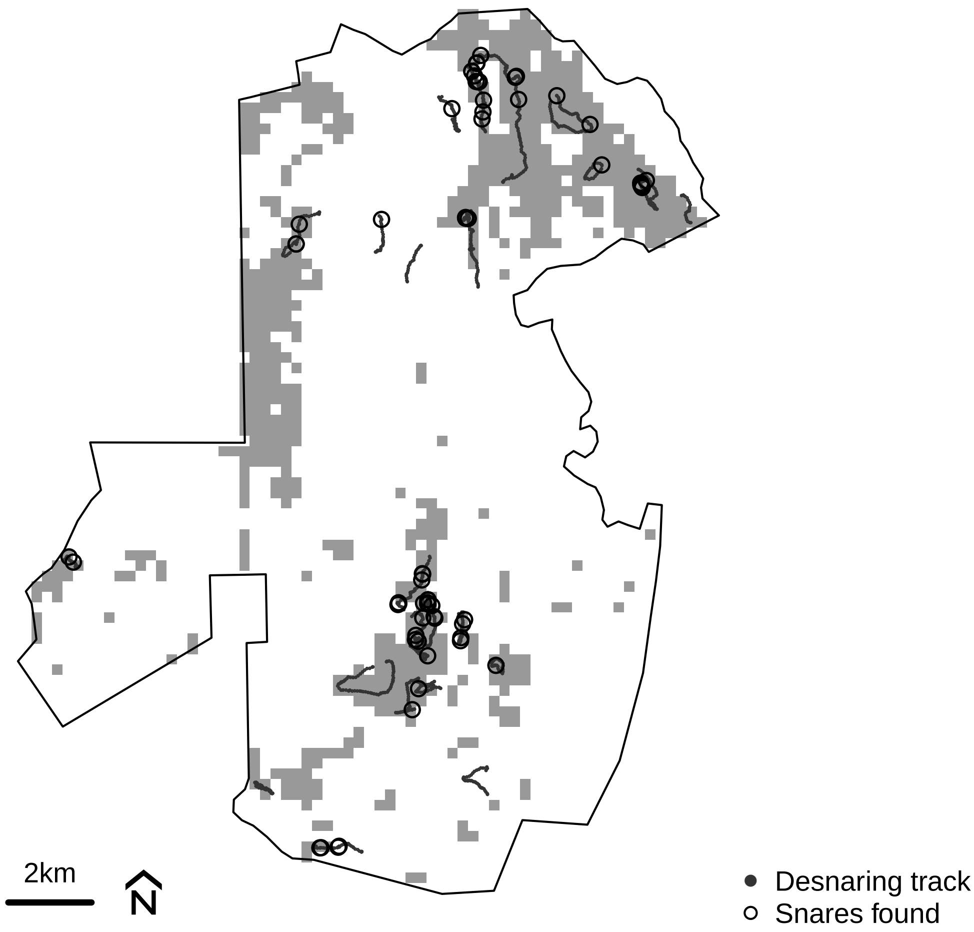


Figure 4.20: Likelihood of snaring as predicted by the Maxent model and validation desnaring transects. Shaded areas are predicted to have a higher likelihood of snaring.

4.3.2 Increase detection probability

The number of required computer simulation runs required to stabilize outcomes was found to be 1000. The diagnostic graph for the first 500 runs of the hotspot search simulation is shown in Fig. H1; here, the outcome (cumulative moving average of the fraction of detected snares) has already stabilized. Individual runs can produce higher and lower fractions of recovered snares, as can be glanced from the first 50 runs.

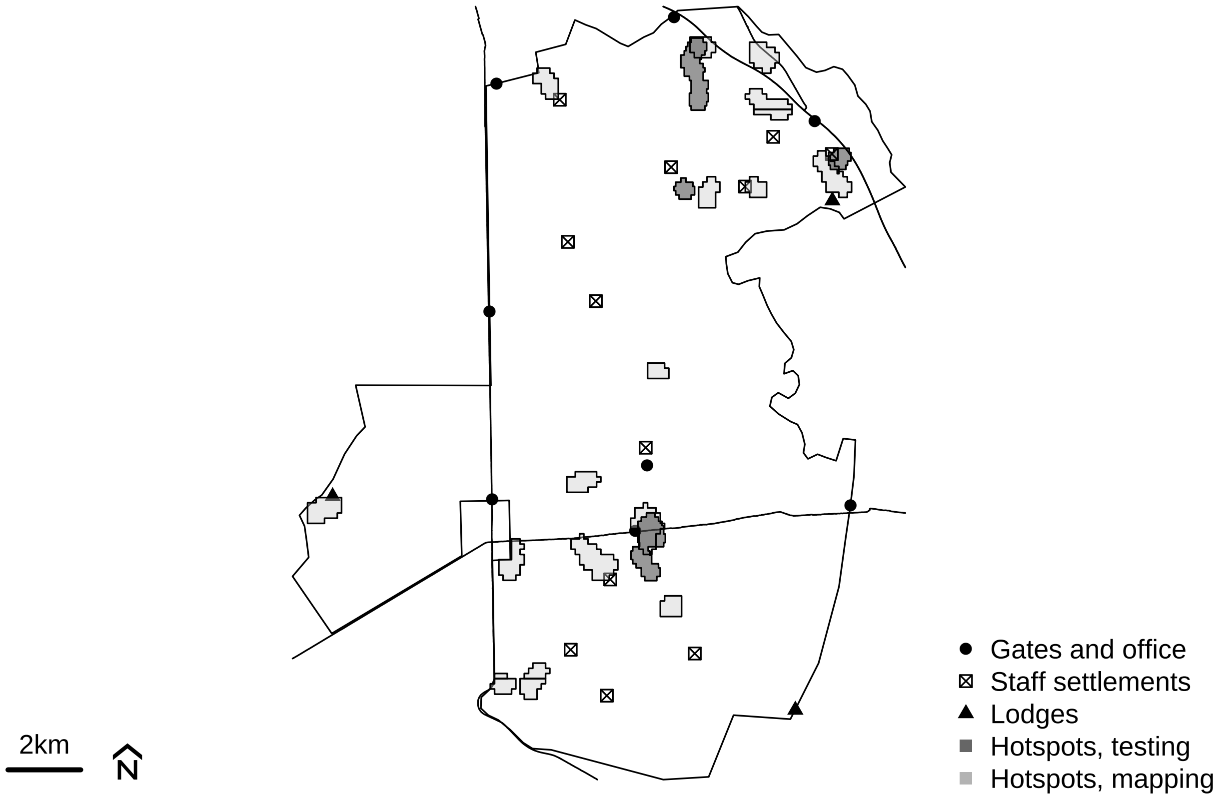


Figure 4.21: Snaring hotspots detected during initial desnaring transects (mapping) and testing of predictive model (field testing.)

The comparison of snare recovery performances (Fig. 4.22, Table 4.9) showed that the adjacent and the hotspot strategies performed better than the sequential search. The adjacent search strategy outperformed the hotspot search in desnaring performance but has a lower recovery rate for replaced snares. The three snare search strategies involved the increasing concentration of visits to raster cells (Fig. 4.23), and were thus increasingly likely to find replaced snares.

4.3.3 Simplified search strategy

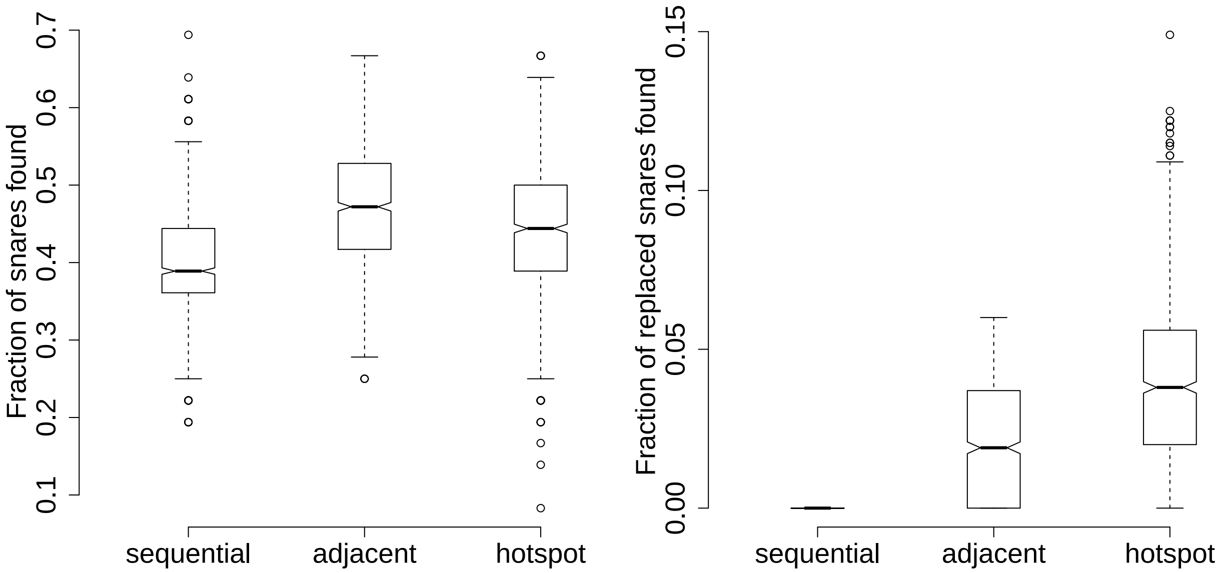
The maximum distance traveled from settlement to snares is approximately five kilometers. The distance from roads to snares was concentrated within 1300 meters from the roads, this is rounded off to 1500 m based on literature (Denninger Snyder et al., 2019).

The tentative predictive map (Fig. 4.24) is calculated as:

Distance to settlements layer: $S = \{x \in S | x < 5000m\}$

Distance to park infrastructure layer: $I = \{x \in I | x < 1500m\}$

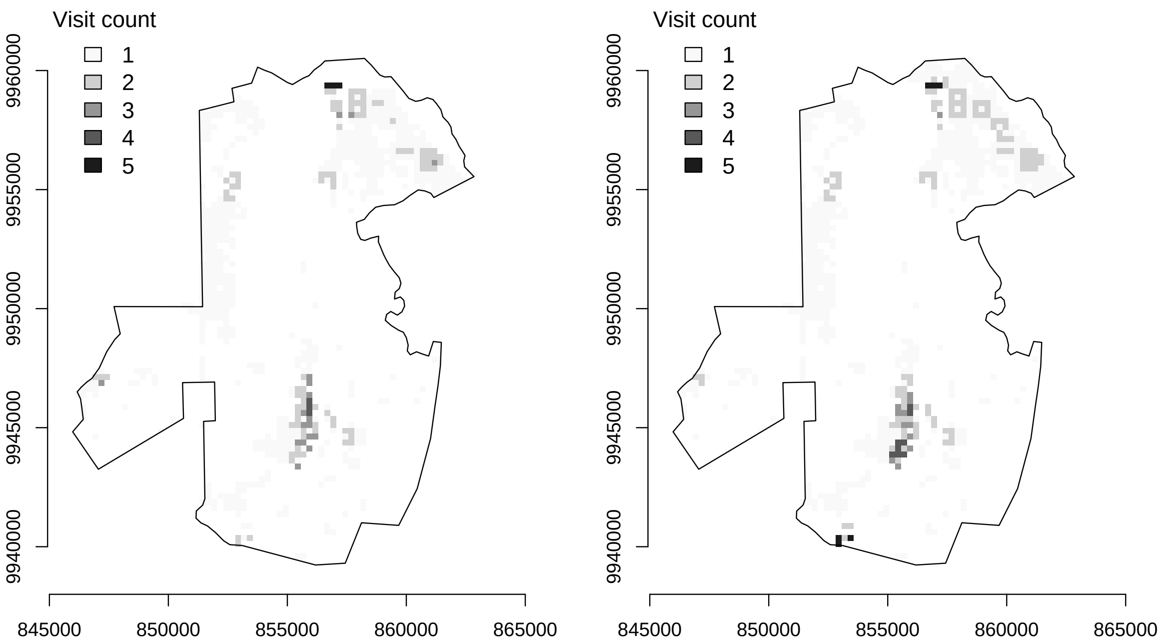
Bushiness layer: $B = \{x \in B | x \geq 0.1 | x \leq 0.9\}$



(a) Fraction of snares found.

(b) Fraction of replaced snares found.

Figure 4.22: Detection of replaced snares for three desnaring strategies (1000 repetitions, snare detection probability 40%).



(a) Adjacent search strategy.

(b) Hotspot search strategy.

Figure 4.23: Visit counts to high potential snaring cells by adjacent search strategy (a) and hotspot search strategy (b) respectively.

Distance to roads layer: $R = \{x \in R | x < 1500m\}$

Union of layers: $L = (S \cup I) \cap B \cap R$

Table 4.9: Desnaring simulation results.

Strategy	Detection probability[a]	Area visited[a]	Snare recovery[a]	Re-snare recovery[a]	Change rate[b]
Sequential	0.2	1.000	0.202	0.000	0.000
Sequential	0.4	1.000	0.402	0.000	0.000
Adjacent	0.2	0.894	0.238	0.008	0.044
Adjacent	0.4	0.825	0.467	0.022	0.087
Hotspots	0.2	0.818	0.230	0.026	0.054
Hotspots	0.4	0.787	0.448	0.040	0.086

[a] All numbers are expressed as fractions.
[b] Change rate refers to the difference with the baseline (Sequential) scenario (re-)snare recovery rates.

This map predicted 10 out of the 15 neck snare hotspots that were identified during the desnaring transects. Hotspots of neck snares located in the core of the conservancy or directly next to the Mbweha lodge (western boundary of conservancy) were missed.

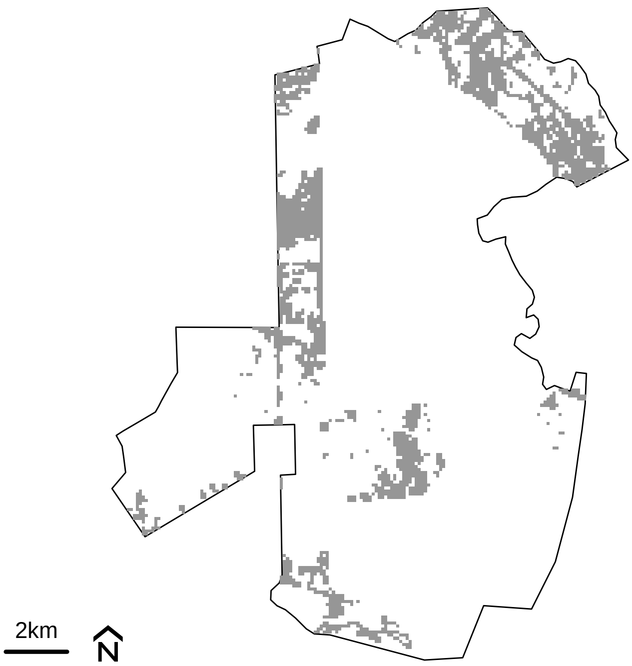


Figure 4.24: Reduced search area. The shaded area shows the locations which can be prioritized by desnaring teams.

Chapter 5: Discussion

This study aimed to assess the patrol effectiveness of rangers patrolling a protected area against bushmeat poachers. The findings suggest that ranger-collected monitoring data do not necessarily give an accurate representation of poaching prevalence. The LEWSE management plan proposed to use the indicators “number of arrests” and “observation book analysis”. These indicators are prone to survivorship bias. Poachers may have displaced their activities, managed to avoid ranger detection, or coerced rangers into under-reporting of observed poaching. When this is the case, reported poaching prevalence can be low, even when actual poaching levels are high. Moreover, desnaring as a standalone anti-poaching strategy is ineffective since snares are replaced by poachers soon after rangers removed them. The effectiveness of desnaring operations can be improved. However, improved snare recovery rates do not necessarily mean that poachers are deterred from replacing the removed snares. Improved desnaring operations will therefore require additional measures that discourage further incursions of poachers.

5.1 Patrolling and poaching patterns

Both desnaring reports of the conservancy and desnaring transects implemented during research demonstrated high snaring prevalence. Snares were placed in clustered patterns, which is consistent with expectations from environmental criminology.

5.1.1 Current patrolling and poaching patterns

Snaring densities in the study area are high, despite the presence of a relatively large ranger force. Moreover, fewer poachers are arrested than would be expected based on the poaching intensity.

5.1.1.1 Evaluation of patrolling densities in the conservancy

The overall ranger density in the Soysambu Conservancy is approximately twice the ranger density recommended by KWS and four times the actual KWS ranger density (Ouko, 2018). Furthermore, the Soysambu ranger density is 6–16 times the recommended ranger density for rhino and elephant sanctuaries (Henson et al., 2016) and 7 times the actual average ranger density in Kenyan conservancies (KWCA, 2016) (Table 5.1).

Table 5.1: Actual ranger densities in Soysambu compared with recommended ranger densities. KWS recommends 6 km^2 per ranger (Ouko, 2018), whereas 20–50 km^2 per ranger is recommended for rhino and elephant sanctuaries (Henson et al., 2016).

		As fraction of recommended (*)		
	Soysambu (km^2 per ranger)	KWS (-)	Sanctuary min (-)	Sanctuary max (-)
Overall	3.1	1.94	6.45	16.13
Day all	5.1	1.18	3.92	9.80
Night all	7.9	0.76	2.53	6.33
Day mobile	8.3	0.72	2.41	6.02
Night mobile	38.0	0.16	0.53	1.32

(*) Ratios, calculated as km^2 per ranger over recommended densities.

These comparisons underline the issues that arise when patrol efforts are expressed as ranger density per km^2. First, the overall ranger density (or recommendation thereof) masks the large differences that can occur between day-time and night-time ranger densities. Night-time patrol densities in Soysambu fall below the KWS-recommended densities, whereas the day-time densities do not. This is critical, as 90% of poaching in Kenya is estimated to occur during the night (Ouko, 2018).

Second, reported ranger densities do not necessarily correct for rangers that are not in the field, either because they are on leave, allocated to static duties, or not available for conservancy-related work. This becomes apparent in the sharp fall of allocated rangers in Soysambu when the overall ranger density is calculated using mobile patrols alone. However, the literature that employs ranger densities to evaluate patrolling or management effectiveness does not correct for these confounders (e.g. Bruner et al., 2001).

Third, ranger densities do not compensate for differences in equipment, such as availability of transport and arms. For example, KWS rangers are armed, whereas Soysambu rangers are not. Armed rangers were found to discover more poachers than unarmed rangers (Nahonyo, 2009). Moreover, lack of transport or infrastructure can immobilize rangers who were supposed to patrol in the field or demoralize them.

5.1.1.2 Desnaring and poacher arrests by conservancy

The desnaring rates in the Soysambu Conservancy can be expressed as snares detected and removed per day, as the conservancy does not track the kilometers walked or square kilometers desnared. This allows a tentative comparison with the desnaring efforts of DSWT, which monitors its efforts in the same units (days desnared and number of snares lifted). This comparison shows that the snare removal rate reported by the Soysambu Conservancy is 20 times higher than that reported by DSWT desnaring teams. In contrast, its arrest rate is 200 times lower (Table 5.2).

Table 5.2: Comparison between desnaring and arrests for DSWT and the Soysambu Conservancy.

	Soysambu[a]	DSWT[b]
Desnaring days	22	4,314
Patrol days	22,265	n.d.
Snares removed	602	5,473
Snares removed per day	27.36	1.27
Bushmeat poacher arrests	1	50
Arrests per year	0.02	4.23

[a] Reporting periods: snaring over calendar year 2018, and arrests over calendar years 2018 and 2019 (Soysambu Wildlife Conservancy, 2018b)
[b] Reporting period: August 2019–August 2020 (DSWT, 2020b)

The different snare removal rates can have several reasons, assuming (1) equal snare detection capacities for both DSWT and Soysambu staff, (2) equal detectability of snares in both Tsavo and Soysambu, and (3) equal poaching pressure between both reporting periods (the calendar year 2018 and August 2019–August 2020 respectively). First, the Soysambu rangers operate in a smaller area and tend to concentrate on known hotspots. These hotspots are normally found to be re-snared and thus provide a steady harvest of snares. The DSWT rangers may cover larger distances between snaring hotspots, which decreases the number of recovered snares per day. Second, the actual snare prevalence may be lower for Tsavo than for Soysambu.

In summary, Soysambu has a much higher snare recovery rate while simultaneously having a much lower arrest rate than the DSWT/KWS teams in Tsavo. The expectation would be that, all other things being equal, Soysambu would see high arrest rates given the high snaring intensity. The conservancy is relatively small, many active hotspots are known to the management, and the ranger density is high.

However, studies on bushmeat poaching by snaring have not found a relation between patrol effort and snaring levels (Becker et al., 2013; Campbell et al., 2019; Johnson et al., 2016; Kimanzi et al., 2014; Wato et al., 2006). This study is no exception; there are insufficient monitoring data from Soysambu and publicly available data to compare both patrol efforts and snaring intensities. Biases in patrol and snaring data can thus not be accounted for when using these data alone (Dobson et al., 2020). Therefore, no conclusion can be drawn regarding the effectiveness of desnaring based on the available data.

5.1.2 Research desnaring transect results

The results of the desnaring transects confirmed the high snaring intensities reported by the conservancy: at least 3% of the conservancy is covered with snaring hotspots. Moreover, new snaring hotspots were found near park infrastructure. The snaring patterns were consistent with those expected based on environmental criminology.

5.1.2.1 Snare densities

The number of snares found in an area can be calculated as snares per km^2 if the swath width of the desnaring team and walked distances are known. These variables are not often reported and subject to confounding factors.

First, the snare density varies considerably as a function of the inclusion of snaring hotspots. Random samples of desnaring locations resulted in snare densities of approximately 0.4 snares per km^2 (Mudumba et al., 2020; Wato et al., 2006). However, desnaring of known snaring hotspots results in higher values; a value of 4.58 snares per km^2 was found by Mudumba et al. (2020). Second, desnaring, which takes place over a longer period, does not include corrections for replaced snares. Such corrections are not made since re-snaring has hitherto not been reported or researched. For example, Kimanzi et al. (2014) found a snaring density of 165 snares per km^2 from desnaring operations stretching over three years. The high snaring density is possibly the result of counting snares removed by rangers and subsequently replaced by poachers. Third, the snare detection probabilities are not equal for different vegetation covers, desnaring teams, or snare types. For example, Rija (2017) found a snare detection probability of 3–4% in savannas, whereas O'Kelly et al. (2018a) found a detection probability of 15% for mixed forests and 26% for evergreen forests. Differences in desnaring team performance were found by Ibbett et al. (2020), and different detection probabilities per snare type (single snare, snare line) were found by O'Kelly et al. (2018a).

The snare density found in Soysambu during desnaring transects was 50 snares per km^2. This value is extremely high, given the 3% snare detection rate in the savanna biome (Rija, 2017). The snaring density of known hotspots found elsewhere was an order of magnitude lower (Mudumba et al., 2020).

5.1.2.2 Spatial factors of snare placement

Researchers have found associations between snaring patterns and roads, park boundaries, human settlements, water availability, ranger posts, and vegetation cover. In the study area, snares were found near the transition from bush to open areas, roads, and park infrastructure. The relation of snaring patterns with surrounding settlements, water availability, and park boundaries was less significant. This is not inconsistent with criminology theory.

In Soysambu, an association between snare positions and roads was found; the association with park boundaries was weak. Snares were often placed within 1300 meters from roads, a result which was also found by Denninger Snyder et al. (2019).

Researchers have found that snaring occurs near park boundaries ("edges", Duporge et al., 2020) and roads. Criminal pattern theory (Brantingham & Brantingham, 1993a) suggests that park boundaries and roads can both be considered as edges. In the Soysambu Conservancy case, roads are attractive edges for poachers since bushmeat can be transported quickly from the crime location to the consumer. Park boundaries may be less attractive than edges. The west boundary borders Lake Nakuru National Park, which is guarded by armed KWS rangers. The north boundary borders a conservancy area where animals have been extirpated. The area around Elementeita village and the southwestern section of the conservancy consists of open areas, with few trees available to affix snares. Therefore, crime pattern theory predicts that edges for this conservancy are more likely to be roads than boundaries. This theory can also distinguish cause and effect: edges can be *associated* with wildlife crime, but do not *cause* it. In other words, snares are not necessarily found near boundaries when more suitable edges are available in the form of public roads dissecting the conservancy.

Both management plans for the greater Lake Elementaita region express concerns about the poaching of animals that move out of patrolled areas (Government of Kenya, 2010b; KWS, 2019). This leakage effect can be considered as a particular form of an edge effect (Fig. 5.1).

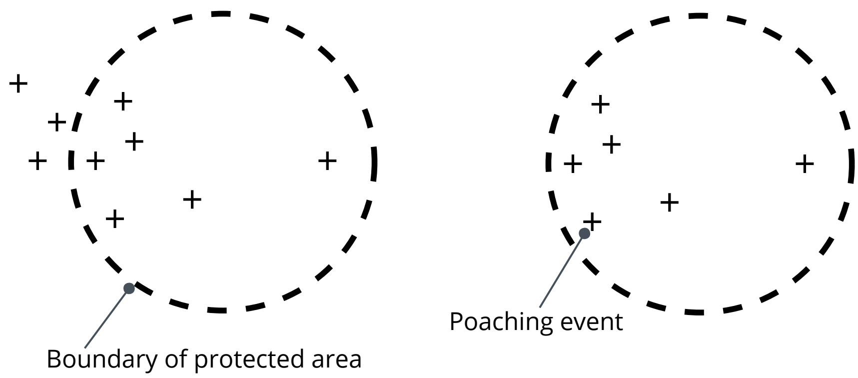


Figure 5.1: Leakage outside protected area edges. Left: poaching events in a protected area. Right: poaching events considered when the research scope is limited to park boundaries.

This leakage effect is not unique to the study area or bushmeat poaching. For example, elephant poaching often occurs just outside the boundaries of protected areas (Ouko, 2013). However, research into poaching in protected areas is often limited to the park boundaries (e.g. Denninger Snyder et al. (2019), Gholami et al. (2017), Thiault et al. (2020) and this study), when areas outside the protected area are not patrolled; observation data are not available in such cases.

Limiting the scope of research to the boundaries of the protected area itself further contributes to survivorship bias: an important source of poaching is not included in the sample and may be overlooked.

5.1.2.3 Human settlements

Researchers frequently detect snares are near villages (Becker et al., 2013; Loveridge et al., 2020; Mudumba et al., 2020; O'Kelly et al., 2018b; Watson et al., 2013). Similarly, snares were detected near some villages around the Soysambu Conservancy. The Journey to crime theory predicts that a limited action radius for poachers (Summers & Guerette, 2018), with short trips more likely than long ones. Indeed, distance decay curves consistent with such travel patterns were found for the villages Elementeita, Mbaruk Center, and Kong'asis surrounding the conservancy. The estimated average (mode) travel distance would be 3–5 km from the village to the snaring locations. However, not every village had snare locations in its immediate vicinity. Different explanations are possible: snares were placed but not detected, the area was less attractive for poaching, or the villagers are not involved in bushmeat poaching. Either way, this research does not support a general association between snaring positions and human settlements. In other words, the presence of villages near the Soysambu Conservancy does not *cause* poaching to occur. However, in cases where snares were found near settlements, travel patterns as expected by environmental criminology were found.

5.1.2.4 Park infrastructure

Snares were found near Soysambu park gates (Jolai Gate, Main Gate), internal settlements (Maendeleo, Mbaruk Stores), and lodges (Serena, Mbweha Camp). The proximity of snaring to ranger posts has been observed elsewhere (Jenks et al., 2012; O'Kelly et al., 2018a; Watson et al., 2013). The explanations given by these researchers for the presence of poaching signs near poaching deterrents are inconclusive. Moreover, other researchers come to the opposite conclusion, namely that poaching does not occur near ranger stations, allegedly due to the deterrent effect thereof (Denninger Snyder et al., 2019; Ghoddousi et al., 2016). Others find varying effects between ranger stations (Beale et al., 2018; Hossain et al., 2018).

Therefore, a first tentative conclusion is that poachers do not estimate that the rangers will arrest them after detecting poaching signs. Such reasoning is supported by the Rational choice perspective, which contends that offenders weigh the costs, benefits, and risks of crimes before committing them (Cornish & Clarke, 1987). A second tentative conclusion is that the proximity of park infrastructure and snaring hotspots is remarkable. Snares have to be checked frequently, and the likelihood that this is not observed by rangers or estate staff decreases cumulatively with each poacher incursion into the conservancy. The proximity of snares to park infrastructure is even more striking when park infrastructure locations are taken into account. All 16 identified hotspots are located within 2 km from park infrastructure, and six out of these were found within 500 m. Hence the remark of an interviewed ranger about poacher sightings: *"You can see them every day if you want to"*.

5.1.2.5 Vegetation cover

Neck snare positions in Soysambu were found to be strongly associated with vegetation cover. Poaching research frequently supports this finding (Jenks et al., 2012; Mudumba et al., 2020; O'Kelly et al., 2018b; Rija, 2017). The bushiness indicator applied in this research considers the degree of transition from an open area to dense bushes. This indicator is novel and can be justified from the Routine activities approach (Cohen & Felson, 1979). Ungulates seek shade in bushes but want to remain close to open areas in order to be able to escape from predators. This open area-bush transition zone is also the area where poachers can operate unseen by rangers; offenders and victims thus coincide in the absence of guardians. As predicted by the routine activities approach, neck snares were seldom found in dense vegetation, while open areas do not contain trees to attach neck snares.

Foot snares are placed in open areas on animal paths near Elementeita village. Rangers report that villagers can observe snared animals from outside the conservancy. This allows them to recover the snared animal during the night with minimal risk of detection. The placement of foot snares in open areas is therefore not inconsistent with the routine activities approach.

5.1.2.6 Presence of water

The association between water availability and snares was found to be weak. Elsewhere, researchers found a strong association between snares and available water (Duporge et al., 2020; Kimanzi et al., 2014; Lewis & Phiri, 1998; Mudumba et al., 2020; O'Kelly et al., 2018b; Rija, 2017). In the case of Soysambu, water points are distributed throughout the conservancy, and water availability is thus less spatially concentrated than would be the case in most protected areas. The water troughs are frequented by wildlife and livestock and their herders and are mostly located in open areas. The visibility of these locations and herders' presence presumably makes water troughs a less interesting place to put snares, as predicted by the routine activity approach.

5.1.3 Detection of snaring hotspots

Poachers were found to place snares in clusters (hotspots). Poaching hotspots were found in nearly every poaching study with a spatial component, whether it concentrates on trophy poaching, bushmeat poaching, or human-wildlife interactions (Angelici, 2016; Becker et al., 2013; Kimanzi et al., 2014; Kyale et al., 2014; Kyando et al., 2017; Mbau, 2013). This finding has several practical and theoretical implications, both for the Soysambu Conservancy and protected areas elsewhere.

First, the clustered snaring patterns implies that the detection probabilities of finding individual snares are not independent. As predicted by environmental criminology and observed empirically, both spatial and temporal auto-correlation have to be considered when analyzing poaching events. Soysambu rangers are aware of this when they pause the desnaring transect upon discovery of a snare in order to subject the surroundings to a more intensive search. In other words, the probability of detecting a snare is not independent of detecting the next one. The detection probability of a snaring hotspot is thus expected to be larger than that of individual snares. This characteristic can be used to develop improved desnaring strategies.

The second set of consequences appear when specific forms of hotspots are considered. Edge effects, spatial repeats, temporal repeats and near repeats are specific forms of hotspots that have all been found in the Soysambu Conservancy and protected areas elsewhere. In the case of

Soysambu, snares were found near roads (edges), detected hotspots were often found near other hotspots (near repeats), and desnared areas were rapidly re-snared by poachers (spatial repeats and temporal repeats). The question for the management of the Soysambu Conservancy is how the area should be patrolled, given that hotspot patterns are apparent. The question for the KWS is how leakage effects are controlled in the wider Lake Elementaita region. Such leakage is flagged by both the GLECA and LEWSE management plans (Government of Kenya, 2010b; Ongalo, 2019), but an assessment of its occurrence and extent were not included in the scope of this research.

Different patrolling strategies have been developed in the urban environment and proposed by wildlife researchers. These include hotspot patrolling (Sherman et al. (2014), proposed by Ferreguetti et al. (2018), Kyale et al. (2011), Rashidi et al. (2015); problem-oriented policing (Surakitbanharn et al. (2018), proposed by Anagnostou et al. (2020), Moreto and Charlton (2019); and zero-tolerance policing (Wilson and Kelling (1982), proposed by Ngene et al. (2013)). However, research into the effectiveness of specific patrolling strategies concerning deterrence of poachers and displacement of poaching has not yet taken place.

Third, the observed replacement of snares by poachers after rangers have removed them (spatial repeats) raises questions about the deterrent effect of desnaring operations. No research into re-snaring has been implemented to date. The current assumption is that desnaring removes the direct threat to animals (Rija, 2017), or more informally, that lives of animals were saved (DSWT, 2020a; Kar et al., 2017). Therefore, the implicit claim is that snare removal leads to a permanent or semi-permanent reduction of the total number of snares in an area if more snares are moved by rangers than replaced by poachers (Rija, 2017). Here, ”replacement” seems to refer to the total population of snares in a protected area in general, rather than replacement of removed snares in specific hotspots. The Soysambu Conservancy observations refute this claim: removed snares are rapidly replaced in the same snaring hotspot. This concern comes on top of earlier reservations on the effectiveness of desnaring, namely the low detectability and low costs of snares (Gray et al., 2018; Ibbett et al., 2020). In summary, desnaring risks to become a symbolic exercise if no complementary measures are taken to deter poachers. The existence of spatial repeats demonstrates that snaring hotspots are well-suited as ambush locations.

Fourth, considering wildlife crime as a social phenomenon requires a specific approach to detecting and mapping hotspots. For example, smoothing, splining, or kriging methods can lead to predicted snaring in Lake Elementaita itself, as poaching would then be approached as a continuous physical phenomenon (Chainey & Ratcliffe, 2005; Eck et al., 2005). The use of thematic maps for hotspot analysis (e.g. the census counting blocks used in Soysambu) would create a Modifiable Area Unit Problem (MAUP) (Openshaw, 1984). Here, the predicted poaching likelihood changes when the boundaries of the thematic map units (e.g. census blocks or administrative boundaries) are changed (e.g. Nyirenda and Chomba (2012), Rashidi et al. (2015)). Therefore, the high-intensity hotspot mapping methodology applied in this study used only the snaring spatial point pattern, combined with a dilation around the detected snare positions.

5.2 Ranger capacities

Rangers expected snares to be found near borders and not near park gates. However, desnaring transects found snares in park infrastructure's immediate vicinity (gates, lodges, and staff settlements). Rangers estimate snare density by assuming that snares removed during previous desnaring exercises were replaced, and self-rated their capacity to deter poachers as low. The communities are thought to be aware of poachers' identity, but community representatives showed little interest in wildlife conservation.

5.2.1 Deterrence

As appearing from interviews with rangers, the poaching problem in the Soysambu Conservancy can be described as follows. Bushmeat poaching by snaring is severe and is carried out by groups of poachers operating mainly at dusk and during the night when the ranger density reduces sharply. Rangers are unarmed and are outnumbered by these poaching gangs. Reinforcement takes a long time because there are not enough patrol vehicles. Furthermore, rangers feel underpaid and are not willing to risk potentially violent encounters with poachers. The rangers feel unsupported by the communities, who they feel know the poachers but are unwilling to denounce them. Moreover, poachers are thought to know the rangers, and they occasionally issue violent threats against them. Predictable patrolling patterns further undermine poacher deterrence. Poachers displace their activities and avoid detection by observing the reduction of ranger density and the absence of foot patrols between 6 p.m. and 6 a.m.. As a result of the factors listed above, rangers self-rate their capacity to deter poachers from entering the conservancy as low. They are neither willing to nor feeling capable of stopping poachers entering the conservancy, although poaching gangs are sighted regularly. One-third of the interviewed rangers reported that they tend not to report sighted poachers to their headquarters because they consider this to be a wasted effort. This under-reporting results in a mismatch between registered sightings of poachers (observation book) and actual sightings (interviews).

The individual components of the described are mentioned in individual studies elsewhere, but seldom combined or made the main subject of the study. However, the situation is alarming for two reasons. First, based on the available literature, there are no reasons to assume that this situation is limited to the Soysambu Conservancy. For example, 90% of poaching in Kenya is thought to occur during the night (Ouko, 2018). Second, predictability of patrolling patterns and under-reporting of poaching signs result in an under-estimation of the true poaching prevalence by environmental managers and researchers. In other words, low reported poaching prevalence may indicate the exact opposite.

The deterrence bottlenecks identified by the Soysambu Conservancy rangers can be split into three groups: issues related to (1) predictability of patrolling, (2) low ranger morale, and (3) community relations. The former two issues are discussed below; the latter will be discussed in Section 5.2.3.

5.2.1.1 Patrolling predictability

Patrol patterns can become predictable when observed directly by poachers or when information is obtained from compromised rangers (Herbig & Warchol, 2011; Mmahi & Usman, 2019). Additionally, vehicle patrols are predictable because they are limited to roads or when the lack of transport limits the action radius of rangers (Kyando et al., 2017; Mubalama, 2010; Rotich et al., 2014). Several Soysambu rangers alluded to "information leakage" – the passing on patrol strategies from rangers to poachers. No evidence of this was sought or found during this study.

The Soysambu rangers interpret patrolling patterns' predictability as a combination of night-time under-staffing, audibility and visibility of cars during the night, and road-bound patrolling patterns. This combination of factors has hitherto not been considered by researchers.

5.2.1.2 Ranger morale

A majority of protected areas (78%) has an insufficient budget available (Coad et al., 2019). Consequently, two out of three African rangers complain about insufficient salary, and 38% about a lack of basic equipment (Belecky et al., 2019). Affecting morale is also the unfortunate fact that most African rangers are threatened by poachers (73%) and community members (71%) (Singh et al., 2020). In these respects, the observed situation in the Soysambu Conservancy is not different from most African protected areas. Two aspects of patrolling in Soysambu are, however, different from the situation described by researchers elsewhere. First, most surveyed African rangers (~ 80%) deemed themselves sufficiently armed (Belecky et al., 2019), whereas Soysambu rangers listed lack of firearms as the main reason for their inability to stop poachers entering the conservancy. Second, lack of transport is seen in connection with reinforcement response times. The low reinforcement response times leave rangers exposed to poaching gangs. This situation, in turn, reduces ranger morale. Since lack of available transport has been observed by other researchers (Kyando et al., 2017; Rotich et al., 2014), there is no reason to assume that this situation is limited to the Soysambu Conservancy.

A payoff matrix for the interaction between rangers and poachers predicts that rangers will not attempt to arrest poachers once these are sighted (Table 5.3).

Table 5.3: Payoff table for ranger-poacher confrontations. Nash equilibrium is marked with an asterisk.

(a) Soysambu rangers (unarmed)

	Poachers	
	Cooperate	Defect
Rangers		
Cooperate	(0,0)	(0,2)*
Defect	(0,0)	(-10,-10)

(b) KWS rangers (armed)

	Poachers	
	Cooperate	Defect
Rangers		
Cooperate	(0,0)	(0,2)
Defect	(0,0)	(1,-10)

In game theory terms, poachers can “cooperate” by refraining from entering the conservancy, or “defect” by entering the conservancy to place snares. Rangers can “cooperate” by ignoring sighted poachers, or “defect” by attempting to arrest them. The payoffs for both rangers and poachers are zero when poachers choose to cooperate. Violent confrontations can occur when unarmed rangers attempt to arrest groups of armed poachers. Poachers risk at least three years imprisonment (Government of Kenya, 2013, 2017, 2018), whereas rangers risk physical injury. Actual (and potentially violent) encounters constitute the lowest payoff outcome for the unarmed rangers-poachers game. The maximum payoff in this non-zero-sum game is obtained when poachers defect, which gives them meat sales revenue, and when rangers cooperate, which does not produce any costs or benefits for them. This payoff is the Nash equilibrium: it predicts the optimal choice for both parties (Schelling, 1980). Note that the payoffs are hypothetical. However, of relevance for the analysis of the rangers-poachers interactions is only that the ordinal ranking of the payoffs is CD > CC | DC > DD.

The situation changes when rangers are armed or when armed rangers are included in the patrol, as is the case with DSWT desnaring teams. Poachers would still be able to rationally decide that poaching is the best option, depending on the risk of arrest and their risk appetite. The expected value of poaching benefits in relation to cooperating rangers (CD) does not change the payoff value much, given the low probability of arrest (0.07%, Loibooki et al., 2002). However, risk-sensitive poachers could refrain from poaching (CC, DC) since this choice guarantees the minimum risk under a minimax decision criterion (Gigerenzer & Garcia-Retamero, 2017). Armed rangers are now more likely to attempt the arrest of poachers when they see them. This is consistent with the research finding of Nahonyo (2009), where armed rangers seemed to observe poachers more frequently than unarmed ones.

5.2.1.3 Survivorship bias in relation to deterrence

The predictable patrolling patterns and deteriorated ranger morale observed in the Soysambu Conservancy are not unusual and have been documented elsewhere. Predictable patrolling and low ranger morale are both likely to result in an underestimation of true poaching prevalence. In the former case, poachers are either not available for detection due to predictable patrolling patterns. In the latter case, poachers are detected but not reported by demotivated or compromised rangers. Both are consistent with the high snaring densities found during the desnaring transects combined with the low number of reported poacher sightings. Both are examples of survivorship bias. This form of sampling bias is insidious because it creates a counter-intuitive situation. Low reported poaching rates might, in fact, mask heavy poaching activity from poachers who have outsmarted the management of the protected area. Research into such counter-patrol measures by poachers is both rare and recent (Mmahi & Usman, 2019; Rija & Kideghesho, 2020).

5.2.2 Prediction of poaching

The effectiveness of patrolling may be improved by including ranger knowledge. This study found limited support for the reliability of rangers' expertise. Both the estimations of expected snaring locations and density were not always accurate. The assessment of ranger expertise, however, brings two benefits. First, the comparison of actual snare locations and predicted locations gives some insight into patrol bias. Second, the estimation of snaring density assessment revealed that rangers implicitly assume a desnaring effectiveness of zero.

5.2.2.1 Prediction of poaching locations

The rangers' ability to estimate the likely locations of snaring hotspots was assessed. This was done by comparing the snare locations predicted by the rangers with the actual snare locations which were found during the first phase of the research.. Rangers expected that snares are most likely to be found near park borders and settlements; poachers are not likely to place snares near park infrastructure.

These ranger expectations are consistent with what is found by researchers, and consistent with the following assumptions. First, poachers are not likely to invest more travel time than necessary for their frequent snare inspections. Poaching signs would thus be found mainly near settlements (Watson et al., 2013). Second, they would also minimize the probability of detection by not venturing too far into the conservancy, thus remaining relatively near the park boundaries (Kyando et al., 2017). Finally, park infrastructure is continuously manned by rangers, which constitutes a deterrent for poachers (Denninger Snyder et al., 2019).

These assumptions are not always correct under all circumstances. First, poaching may be associated with the village vicinity, but this does not automatically imply that all villages are associated with poaching. In the Soysambu Conservancy, snares were found near some villages, but not near each of them. Second, as discussed in the previous section, environmental criminology predicts that public roads dissecting the Soysambu Conservancy are more attractive as edge than the park boundaries. Third, the assumption that rangers' presence in park infrastructure deters poachers requires an underlying assumption: poachers have to perceive the threat of arrest and prosecution as credible. Soysambu rangers are not armed and appear to be demoralized; poacher may have concluded that their detection by rangers is not likely to result in arrest.

Erroneous expectations of poaching locations can lead to patrol bias. Here, ranger focus on patrolling areas where they expect poaching (Moreto & Matusiak, 2016). Areas where poachers are not expected may be under-patrolled or not patrolled at all (Keane et al., 2011; Kuiper, Kavhu, et al., 2020; Kyando et al., 2017). Such patrol bias is likely to be present in Soysambu: the immediate vicinity of a staff settlement, a lodge, and a park gate was never patrolled because no poaching was expected there. Poachers can exploit patrol bias since it results in predictable patrolling patterns.

5.2.2.2 Prediction of poaching density

The ability to estimate snaring densities was estimated by comparing rangers' ex-ante estimates with results of desnaring transects.

The results suggest that groups of rangers outperform individuals in estimation tasks, but also that this effect is not significant. Most rangers used a rule of thumb to make their estimation: they use the number of snares found during the last desnaring exercise. However, this decision rule fails when snares are found in unexpected locations or in known snaring hotspots that have become inactive.

Three observations can be made regarding the results. First, using the most recent desnaring results to estimate the results of desnaring transects in the same sub-area constitutes recency bias. Here, rangers weigh the most recent desnaring results more heavily than previous ones (Shanteau, 1989). Second, the implicit assumption under the decision rule is that desnaring has no deterrent effect. The number of snares present in an area is assumed to be constant, regardless of the ranger effort to reduce it. This suggests that rangers implicitly self-rate their deterrence capacity as zero, which is consistent with their responses on this subject in interviews. Third, ranger experience is insufficient to develop snare density estimation expertise. The environment must also provide reliable feedback that allows rangers to link poaching observations with predictor variables (Kahneman & Klein, 2009). In this case, the environment is too noisy, possibly because of displacement and patrol bias.

5.2.3 Community relations

Interviewed representatives from villages surrounding the Soysambu Conservancy made three claims. First, they have not been compensated for damages to crops and livestock by wildlife, although such damages were reported to KWS. Second, communities want to be separated from the animals in the Soysambu Conservancy by an electric fence. Third, they have not been approached by KWS for input into the LEWSE management plan. The issues of compensation of wildlife damages, fencing of wildlife conservation areas, and community participation in environmental management and governance are discussed below.

5.2.3.1 Compensation of wildlife damages

Compensation from damages caused by wildlife is a requirement under the WCMA, 2013 (Government of Kenya, 2013, 2017, 2018). Compensation claims are not followed up in Kenya, and lack of compensation for wildlife damages is therefore not specific to the Lake Elmenteita region (Mwangi et al., 2016; Ouko, 2018). Without compensation, farmers may turn to retaliatory killings (Ontiri et al., 2019), and as one interviewee stated: "...it won't be possible to stop poaching without jobs and compensation".

The (lack of) compensation for wildlife damages, population growth, and spatial planning are related. Both bushmeat poaching and retaliatory wildlife killings are driven by population growth combined with an absence of spatial planning; both drivers are present in the study area. The population around Soysambu Conservancy doubles every twenty years. During the time period covered by a 10-year management plan the population has therefore increased by 39%. Furthermore, no spatial planning or enforcement of existing environmental laws has been implemented in the Lake Elementaita region since the inception of the first UNESCO management plan (GLECA, Government of Kenya (2010b)).

In summary, increased human settlement in wildlife corridors and dispersal areas around the conservancy is likely to increase bushmeat poaching and reduce wildlife populations. This puts pressure on patrol resources and eventually reduces patrol relevance, as there will be fewer animals to protect.

5.2.3.2 Fencing of wildlife conservation areas

Related to the wildlife damages compensation issue is the desire to put fences between humans and wildlife. Part of the study area is already fenced, and representatives of villages surrounding Soysambu, the conservancy itself, and KWS promote further fencing (Soysambu Conservancy, 2020).

Fencing of the eastern and southern parts of the greater Lake Elementaita region is proposed in the LEWSE management plan to limit "human-wildlife conflict" (Ongalo, 2019, Action 2.2). Elsewhere in the plan and in the Wildlife Sanctuary boundary variation report proposed by KWS (Bett et al., 2016), fencing is flagged as a problem for landscape connectivity. It is not immediately evident from the LEWSE management plan how the authorities foresee implementing fencing since it is brought forward as both a solution and a problem.

Fencing off the Soysambu Conservancy will have particularly severe effects since three public roads dissect the area. Construction of fences will thus fragment the area into four, relatively small, sub-areas (core area: 114 km^2, south (Jolai): 55 km^2; west (Congreve): 23 km^2; north (Triangle): 8 km^2). Moreover, the conservancy is separated from Lake Nakuru National Park by an electric fence, and the wildlife corridor which connects Soysambu with Lake Naivasha is lost to real-estate developers. The LEWSE management plan claims to protect wildlife connectivity (Ongalo, 2019, Action 2.8). However, the claim is vaguely worded ("...efforts will be made to design and protect a wildlife corridor where only conservation-compatible activities will be permitted...") and not monitored or budgeted.

In summary, there is no clear vision on the role and consequences of further fencing in and around the conservancy. Fencing will reduce animal movements, thus reducing the viability of the remaining animal populations, as observed in Nairobi National Park (Said et al., 2016). The reduced landscape connectivity is in direct contradiction with Aichi Target 11, which calls for "... well-connected systems of protected areas and other effective area-based conservation measures, ...(which are)... integrated into the wider landscapes ..." (Convention on Biological Diversity, 2010a). Patrolling effectiveness becomes less relevant because there will be fewer animals to protect.

5.2.3.3 Public participation in conservation area management plans

The establishment of a management plan with community involvement is a requirement of both the WCMA, 2013 (Government of Kenya, 2013, 2017, 2018) and the UNESCO Convention Concerning the Protection of the World Cultural and Natural Heritage (UNESCO, 1972, 2019). The observed absence of community input is surprising because both the cover letter and the LEWSE Draft Management Plan submitted to the UNESCO World Heritage Center explicitly stated that stakeholder participation was foundational to the development of this plan (Ongalo, 2019). The management plan cannot be gazetted without taking the input of communities into account because doing so would violate Art. 44 of the WCMA, 2013 and its associated fifth schedule. Gazetting of the LEWSE management plan is, in turn, a requirement for lifting the moratorium of developments in the Ramsar area, which was announced by NEMA on 9 September 2015 (Wahungu, 2015).

Likewise, the Soysambu Integrated Management Plan contains no visible signs of participation from surrounding communities (Zeverijn & Co, 2013). With the exception of Elementeita village, representatives from surrounding settlements stated that the conservancy does not contribute much to solving their problems. The lack of involvement of and interaction with the surrounding settlements is a possible explanation for the lack of informers. When combined with traditional law enforcement, the use of informers is more effective in reducing poaching activities than through law enforcement alone (Jachmann & Billiouw, 1997; Linkie et al., 2015; Risdianto et al., 2016). Recent research demonstrated that this notion is shared by rangers (Anagnostou et al., 2020; Moreto & Charlton, 2019).

In summary, neither the Soysambu Conservancy nor KWS appears to involve communities much into the environmental management and governance of the Lake Elementaita area. This contradicts both Aichi Target 11, which requires the management of protected areas to be equitable (Convention on Biological Diversity, 2010a; Dawson et al., 2018; Schreckenberg et al., 2016), and the WCMA, 2013, which requires management plans to be developed in consultation with local stakeholders. The lack of community involvement reduces patrolling effectiveness by limiting the development of an informer network.

5.3 Improved desnaring strategies

The desnaring effectiveness can be improved by 4–9%. This is achieved by reduction of the search area and searching the surrounding areas when snares are found. Snares replaced by poachers can be found back with a hotspot search strategy, but the improvement over the baseline strategy is limited (4%).

5.3.1 Reduce search area

The effectiveness of patrolling can be approved by reducing the search area. Here, the areas where desnaring can take place are prioritized using a presence-only species distribution model. SDM has been applied to predict poaching events in both marine protected areas (Bisi et al., 2019; Thiault et al., 2020) and terrestrial protected areas (Denninger Snyder et al., 2019; Jenks et al., 2012). The approach followed in this study differs in three aspects from these earlier studies. First, the outcome was tested with the Boyce index, which has been specifically designed for this purpose. Second, the validity of the model was statistically tested in the field. Third, the obtained reduction of the search area was not a final outcome but used as an intermediate step in developing a patrolling model. These differences with previous approaches are discussed below.

5.3.1.1 Application of the Boyce index

The results of SDM are usually tested by k-fold cross-validation and calculation of AUC and TSS. The Boyce index has not yet been applied in the prediction of poaching through presence-only SDM, although it has been specially developed for this purpose (Boyce et al., 2002). The application of the Boyce index and its associated P/E curve has specific advantages over the

AUC/TSS indicators. First, the robustness of the P/E curve is expressed as the Spearman correlation over the habitat suitability range. In other words, the required monotonic increase of the P/E curve is explicitly expressed as a performance indicator (Pearce & Boyce, 2006). Second, the P/E indicator gives an indication of the suitability of habitat classes (unsuitable: P/E < 1; marginal: P/E = 1; suitable: P/E > 1) (Hirzel et al., 2006). Third, the maximum value of the P/E curve reached shows how much the model outperforms a chance model, in other words, how far the model deviates from randomness (Hirzel et al., 2006).

The Boyce index obtained for the reduction of the snaring search area was excellent (0.91) and demonstrated a monotonic increase of the P/E curve. A predicted higher habitat suitability class would, therefore, indeed correspond with a higher value of the P/E curve. The P/E curve itself is concave and reaches a maximum value of 8. Therefore, the model's resolution is not constant; higher habitat suitability classes are predicted with more certainty than lower ones. The model outperforms a random habitat suitability prediction with a factor of 8. The P/E = 1 value is obtained at a habitat suitability class of 0.5. This value is expected and often used as the default cutoff value for presence/absence in SDM (Hijmans & Elith, 2017).

5.3.1.2 Validation through field testing

The SDM models developed by other researchers to predict poaching events were not validated in the field (Bisi et al., 2019; Denninger Snyder et al., 2019; Jenks et al., 2012; Thiault et al., 2020). Field testing of statistical models in poaching research is rare and not always accompanied by statistical testing.

Field testing of a predictive model in the field was implemented by Critchlow et al. (2016). Other researchers carried out field test and compared CPUE values of areas with high and low predicted likelihood of illegal activities, respectively (Gholami, 2018; Gholami et al., 2017; Kar et al., 2017). These comparisons were not accompanied by statistical testing and are therefore indicative.

The comparison between areas with a high or low predicted likelihood of snaring in Soysambu yielded a weakly significant result (p=0.04). The variable importance identified by the Maxent model confirmed the strength of association, as calculated through the AUC from the earlier fieldwork. The bush-open area transition, distance to roads, and distance to park infrastructure were identified as variables with a higher predictive value. Three additional snaring hotspots were discovered during field testing of the Maxent model. All three were situated in the immediate proximity of park infrastructure (staff settlement and gates).

5.3.1.3 Limitations

Two methodological issues exist with the application of SDM modeling and field testing. First, SDM models are based on the assumption that the entropy of the species distribution maximizes in suitable habitat areas (Guisan et al., 2017), whereas crimes are predicted to cluster in small locations (Weisburd, 2015). The assumed entropy in SDM models and crime event patterns thus move in opposite directions. Whether or not this compromises the suitability of applying SDM models on crime point patterns has not yet been assessed to date and is, therefore, a matter for further research. Second, observing even moderately large trends in illegal activities within protected areas requires prohibitively large samples (Jones et al., 2017). Observation and analysis are further hampered by confounding variables (displacement, deterrence, under-reporting, leakage, patrol bias). These factors increase the requirement for an increased sample power since a low p-value may indicate either a significant difference between two populations, an insufficient sample size, or both (Ellis, 2010; Ioannidis, 2005).

These methodological bottlenecks have been partially circumvented in this study by using the reduced search area as input for an improved search strategy. The snaring prediction map is therefore not a final outcome but an intermediate one. The search strategy has been designed to update and prioritize the search area through reinforcement learning and is discussed below.

5.3.2 Search strategy

The SDM model resulted in a prioritized search area within the conservancy, within which the likelihood of snaring is estimated to be high. The search within this area is implemented through different methodologies: a baseline method (sequential search), a method in which areas in the vicinity of detected snares were more intensively searched (adjacent search), and an annealing epsilon-greedy policy for the multi-armed bandit problem (hotspot search).

5.3.2.1 Effectiveness of search strategy

Both the adjacent search strategy and the hotspot search strategy outperformed the sequential search, which served as baseline scenario with 4–5% (20% snare detection rate) and 9% (40% snare detection rate), respectively. The snare recovery rates are a gross improvement; the net improvement depends on an unknown snare replacement rate by poachers. This result suggests that desnaring patrols can operate more effectively but should not be the only strategy put into place to deter poachers.

The sum of de-snare and re-snare recovery rates is comparable for the adjacent and hotspot scenarios. The area covered in the adjacent scenario is larger than that in the hotspot scenario, which results in more recovered snares. The hotspot scenario has the advantage of finding more replaced snares while being unpredictable for poachers. This scenario may therefore outperform the adjacent search strategy under field conditions.

Stopping poachers from placing snares remains the main objective in anti-poaching strategies. Desnaring is ineffective when rangers are neither willing nor capable to arrest poachers once these are detected since the removed snares are replaced. Also, desnaring is resource-intensive: a relatively small area is searched for snares with a low detectability rate. In conclusion, the hotspot search strategy should be the preferred patrolling strategy for desnaring, but it must be used in conjunction with other anti-poaching strategies.

5.3.2.2 Application of environmental criminology

Both the adjacent search and the hotspot search employ insights from environmental criminology. The clustering of snares in hotspots means that individual snares' detection probabilities are not independent of each other. The snare detection triggers an increased and intensified search in the cell where the snare was detected. Furthermore, nearby snare cells are searched in anticipation of near repeats. Finally, cells where snares were found can be revisited in view of possible spatial and temporal repeats.

The existence of near repeats in poaching was recently confirmed for the Great Barrier Reef marine protected area (Weekers et al., 2020), but not combined with a specific search strategy. Likewise, spatial repeats were found by Critchlow et al. (2015), who found that the location of previous wildlife crimes was the most critical predictor variable. They did, however, not translate this observation into a specific strategy for the detection of poaching. No research to date exists which leverages the occurrence of both spatial and near repeats.

5.3.2.3 Flexibility and unpredictability of search strategy

The reduction of the search area and the search strategy itself have been decoupled in this research. This has the advantage of flexibility. First, methods other than SDM (e.g. occupation modeling, statistical pattern analysis) can be used as input for the search strategy. Second, the search strategy uses reinforcement learning rather than machine learning. Implementation of the search strategy, therefore, involves a series of incremental steps (reinforcement learning) rather than a preset and integrated "prioritize-and-search" strategy (machine learning or statistical learning; see e.g. (Critchlow et al., 2015; Gholami et al., 2017; Kar et al., 2017)).

Reinforcement learning, as implemented in the hotspot epsilon-greedy policy, results in a simple and flexible desnaring strategy. This is an important advantage over machine learning or statistical models. Rangers work in a noisy environment, where feedback signals are neither complete nor timely. Their ability to estimate snare locations and density was therefore limited. The literature on bushmeat poaching is consistent with this finding. There is no agreement on a standard set of predictor variables, nor in the general effect size and direction thereof. Reinforcement learning is arguably a better choice in a noisy environment than a machine learning model. Here, the field observations are evaluated in each step, rather than batch-processed for a static model.

5.3.3 Simplified search strategy

The previously described methodology to reduce the search area requires a considerable upfront research effort. This may be prohibitive for other conservancies who have neither the data nor the resources to collect the required information. Therefore, a simplified search area reduction methodology was developed to reduce the data required to a minimum. This methodology can be combined with the epsilon-greedy detection policy described above, and represents the first attempt to develop such based on environmental criminology.

The simplified search area reduction method covers ten out of sixteen snaring hotspots. Whether this is acceptable or not for managers of protected areas will depend on the strategies that accompany desnaring. First, the search area contains the locations which are most accessible for poachers. The presence of rangers will require them to go further into the protected area, which increases both their probability of detection and time spent on poaching. The conditions for this to be effective are that (1) the ranger presence is unpredictable, rangers have sufficient capacity to detect poachers, and (2) have sufficient capability to arrest poachers once this detection has taken place. Second, poachers' arrest will not go unnoticed by other poachers and any colluding park staff and will thus have a deterrent effect. In other words, coverage of all snaring hotspots is desirable but not directly necessary for this approach to have a deterrent effect.

Chapter 6: Conclusions and recommendations

6.1 Conclusions

This study's objective was to evaluate the effectiveness of patrolling a terrestrial protected area against bushmeat poachers placing snares, based on which improved desnaring strategies could be developed. ”Patrolling effectiveness” is defined here as the extent to which poachers are deterred from entering the protected area as a function of patrol effort. It is commonly expressed as the count of observed poaching activities, normalized over a quantitative expression of patrol efforts, such as ranger density, kilometer patrolled, or frequency of visits to an area. This quotient is the catch per unit effort (CPUE).

The following conclusions can be drawn based on the research findings:

i The deterrence of poachers depends on the quality of the patrol effort. Rangers must be (1) present in the area, (2) have the capacity to detect poaching activities, and (3) have the capability to arrest poachers once these are detected. The CPUE is not a reliable indicator of patrolling effectiveness because the quality of patrol effort affects the count of observed poaching activities. The nominator and denominator in the CPUE are thus not independent from each other.

ii Poachers have developed several strategies to counter patrol efforts. Predictable patrolling patterns are leveraged to avoid ranger presence. Detection by patrols is avoided by operating at night. Arrest attempts are thwarted by issuing violent threats to demoralized rangers. Together, these actions result in survivorship bias in reported poaching prevalence: actual poaching events are either not available for detection, or they are detected but not reported.

iii Each of the phenomena predicted by environmental criminology was observed in the study area. Snares are placed in clusters (hotspots), near roads (edge effects), and near other hotspots (near repeats). Poachers replace the snares removed by rangers (spatial and temporal repeats).

iv Rangers have limited capability to predict snaring locations and density. Snares near lodges and park infrastructure were not expected by rangers, but were found during desnaring transects. Rangers predicted snare density by assuming that all snares removed during the previous desnaring operation were replaced. They thus implicitly assume that desnaring has an effect size of zero.

v Community involvement in environmental management and governance is minimal and has also not been sought by either the conservancy or the Kenyan Wildlife Service.

vi Snares have to be inspected regularly, and the location of these hotspots near staff settlements, tourist lodges, and park gates in combination with a low count of reported poacher sightings is, therefore, an indication of low patrolling effectiveness.

vii The effectiveness of desnaring can be improved by (1) reducing the search area and (2) optimal allocation of ranger efforts between re-visiting known snaring hotspots and discovery of new ones. Desnaring as a standalone strategy is not an effective patrolling strategy because (1) the improvement of snare recovery rate is limited, and (2) snares which were removed by rangers are replaced by poachers if no other strategies for deterring poachers are implemented.

viii Patrolling effectiveness must be considered in the context of environmental management and governance of the areas surrounding the protected area. In the absence of such management and governance, patrolling effectiveness becomes less relevant because there will be fewer animals to protect.

6.2 New applications of methodology

This study contains several novel elements. Each of these items represents the first time that a methodology was applied in the context of bushmeat poaching research.

i Deterrence was disentangled from alternative options that poachers have at their disposal when confronted with patrols: spatial and temporal displacement and intimidation of rangers. The existence of survivorship bias in patrol data was demonstrated.

ii A novel method was developed to identify snaring hotspots, involving a combination of Stienen plots, Steiner sets, and nearest-neighbor distances. This method used the clustering properties of snaring: the combination of near repeats and spatial repeats (contagion).

iii The transition of bushy areas to open areas (bushiness) indicator has been developed during this study. It proved to be the strongest predictor for neck snare occurrence.

iv Environmental criminology has been applied for the first time on bushmeat poaching by snaring. Its application resulted in practical insights used for hotspot detection, reduction of the snare search area, and improved detection of snares. This group of theories was also used to explain different types of edge effects (placement of snares near borders and public roads). The current understanding of edge effects are based on statistical theory and can therefore not explain causality.

v The capacity of rangers to predict poaching density was tested using the diversity prediction theorem.

vi Rangers' lack of willingness to stop poachers has been explained through game theory.

vii A practical and effective desnaring methodology was developed. Novel elements are the decoupling of search area reduction and actual search; the development of a simplified search area reduction methodology solely based on environmental criminology; and applying an epsilon-greedy policy for solving the exploitation-spatial problem (multi-armed bandit problem) in a spatial context.

viii The replacement of snares by poachers – re-snaring – has not been described to date. Re-snaring has hitherto been interpreted as replacing snares distributed over the entire protected area rather than in specific snaring hotspots.

ix Patrolling effectiveness was put in the context of environmental management and governance. Lack of spatial planning and environmental governance and management can result in reduced wildlife populations, thus reducing patrol relevance. The point was made that patrol relevance supersedes patrol effectiveness.

6.3 Recommendations

i Ranger-collected law enforcement monitoring data should not be used to interpret or predict poaching prevalence without insight into the context in which these data were generated.

ii Environmental management plans must foresee adequate compensation, support, and equipment for rangers.

iii Unarmed rangers do not deter groups of armed poachers. Therefore, the conservancy must choose between arming its ranger force or seeking increased collaboration with the Kenya Wildlife Service.

iv Desnaring, when deployed as a standalone strategy, addresses symptoms (presence of snares) rather than causes ((re)placement of snares). The identification of snaring hotspots is useful where it informs complementary deterrence strategies.

v Poaching is a crime, and therefore calls for the application of environmental criminology to explain and predict poaching.

vi Poaching events cluster, both in time and in space. Analysis of poaching point patterns, such as regression and mapping of hotspots, must therefore only apply statistical techniques that do not rely on linearity and spatial or temporal independence. Further research is required to confirm whether the clustering of crime events is compatible with species distribution modeling techniques.

vii Poachers are likely to base their decisions on where, when, and how to poach on a limited number of cues. Simplification of the currently used statistical models is required to avoid over-fitting of data. Such simplification requires insight into poacher's decision strategies. Research into these strategies is therefore required.

viii Deterrence of poachers can lead to increased poaching outside the boundaries of the protected area (leakage). Research into patrolling effectiveness, wildlife protection, and wildlife protection policies should therefore extend its scope outside the protected area boundaries where possible.

ix The involvement of stakeholders in environmental management and governance is a legal requirement for equitable management of protected areas. Stakeholder involvement must be sought by both the conservancy and the Kenya Wildlife Service.

x Environmental management and governance must be evidence-based to evaluate policy interventions' efficiency, effectiveness, and relevance. Such evidence, particularly the extent and seriousness of bushmeat poaching, must be collected to assess whether the proposed measures to limit bushmeat poaching in the Lake Elmentaita ecosystem are effective and whether the proposed monitoring indicators are relevant.

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Appendices

Appendix A: Overview bushmeat poaching literature

Table 1: Summary of literature focused on snaring. Blanks mean that variable was not included in methodology. "Patrol effort" refers to the relation between patrol effort and snare abundance. "Ranger posts" refer to the spatial influence of ranger posts to snaring abundance. "Terrain" refers to terrain characteristics such as slope and roughness. Symbols for strength of association: "++": strong positive, "+": moderate positive, "o": neutral, "-": moderate negative, "- -": strong negative.

Author	Country	Focus	Edge effects	Habitat	Hotspots	Patrol effort	Ranger posts	Roads	Vegetation	Water	Settlements	Seasonality	Terrain
Becker et al. (2013)	Zambia	iconic species			++	- -				++	++	++	
Campbell et al. (2019)	Indonesia	iconic species			++	- -							
Ibbett et al. (2020)	Cambodia	detection								++		- -	- -
Jenks et al. (2012)	Thailand	bushmeat	++				++	++	++				
Johnson et al. (2016)	Lao	iconic species				- -							
Kimanzi et al. (2014)	Kenya	bushmeat	++	++	++	-		++		++			+
Lewis and Phiri (1998)	Zambia	bushmeat	++		++					++			
Loveridge et al. (2020)	Zimbabwe	iconic species	++	++	++						++		
Muchaal and Ngandjui (1999)	Cameroon	bushmeat										++	
Mudumba et al. (2020)	Uganda	bushmeat		++	++			++	++	++	++		++
O'Kelly et al. (2018a)	Cambodia	bushmeat			++					++			++
O'Kelly et al. (2018b)	Cambodia	detection	++		++		++	++	++		++		
Rija (2017)	Tanzania	detection							++	++			
Risdianto et al. (2016)	Indonesia	iconic species			++							++	
Wato et al. (2006)	Kenya	bushmeat	++			- -		++		- -	o		
Watson et al. (2013)	Zambia	bushmeat					++	++		+	++		
Wrangham and Mugume (2000)	Uganda	iconic species		++								++	

Appendix B: Overview of anti-poaching literature

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Astaras et al., 2017	Africa	Poaching with guns	NA	Gunshots	Acoustic traps	Detection
Aziz et al., 2017	Asia	Tigers and their prey	Nearest ranger post, nearest river, nearest human habitation	Protection status	Transects	Descriptive
Barichievy et al., 2017	Africa	Rhinos	NA	Duration and frequency of patrolling vis-a-vis poaching events	Patrol data	Descriptive
Beale et al., 2018	Africa	Elephants	Travel costs from neighbouring villages, contrasting patterns of distance to ranger posts	Primary productivity; standing water (satellite), tree cover, close to wet season; elephants: carcass density, live density	Aerial surveys	Descriptive
Becker et al., 2013	Africa	Snaring by-catch of elephants, lions, wild dogs.	NA	Game trails (77%), edge water (15%), around flowering trees (4%). Mainly woodland and snares typically attached to trees. Season: hot dry season more snares, either because there is more poaching or because the snare detection rate is higher.	Patrol data	Descriptive
Campbell et al., 2019	Asia	Snaring of tigers and by-catch of tapirs	Nearest distance between evidence of poaching	Relation between tiger poaching and by-catch of tapirs	Patrol data	Descriptive
Critchlow et al., 2015	Africa	Illegal activities	Roads, rivers, villages, towns	NPP, wetness, slope, wildlife density, habitat (forest, savanna, other), travel costs	Patrol data	Predictive
Critchlow et al., 2016	Africa	Illegal activities	Near boundaries, near water channels	Animal tracks; sites where illegal activities occurred previously	Patrol data	Predictive
Dajun et al., 2006	Asia	Detection of animals	For some species, distance to conservation station or route, but not conclusive	NA	Camera traps	Detection

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Dong et al., 2018	Asia	Illegal activities	NA	Season: rangers do not want to patrol during the Lunar New Year and during the rainy season. However during the festive season the demand for bushmeat increases	Patrol data	Descriptive
Eloff and Lemieux, 2014	Africa	Rhinos	Closeness to roads and borders	Moon phase	Incident reports; patrol data	Descriptive
Eustace, 2017	Africa	Poaching	NA	Poaching occurs late evening/early morning, rarely at night; by motorcycle with 2 persons using a spear	Interview with poacher	Social, poaching
Fang et al., 2015	Asia	Illegal activities	NA	NA	Patrol data and Camera traps	Predictive
Faulkner et al., 2018	Africa	Poaching	Distance from poachers' addresses to crime	NA	Patrol data and addresses of poachers	Descriptive
Ferreguetti et al., 2018	South America	Poaching	Edge of PA, water, road, trail, settlement	Moonphase, frequency of seeing wildlife	Camera traps	Detection
Gandiwa et al., 2013	Africa	Bushmeat	NA	NA	Patrol data and interviews with communities	Descriptive
Gandiwa et al., 2014	Africa	Poaching	NA	NA	Interviews with rangers	Social
Ghoddousi et al., 2016	Middle East	Urial sheep and Persian leopard	Distance to park border, ranger stations, villages, water sources	Urial sheep and Persian leopard populations, NDVI, slopes	Camera traps	Detection

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Gholami et al., 2017	Africa	Snaring	Patrol post, town, rivers	Animal density, slope, forest cover, Net Primary Productivity	Patrol data	Predictive
Gholami et al., 2018	Africa	Snaring	Border, roads, towns, water, salt licks	Slope, animal density, NPP, forest cover	Patrol data	Predictive
Gholami2019	Asia	Snaring	Geographical positions of illegal activities	NA	Patrol data	Predictive
Gray and Kalpers, 2005	Africa	Illegal activities	NA	Seasonal: more illegal activities in June/July	Patrol data	Descriptive
Gurumurthy et al., 2018	Asia	Snaring	Streams, villages, patrol posts, rivers, marshes, village roads, provincial roads, national roads, highways, PA boundaries	Land type, elevation, slope, patrol length (last season)	Patrol data	Descriptive
Haines et al., 2012	North America	White-tailed deer	Towns, roads, rivers	land cover (riparian, forest), weather conditions and time; temp, visibility, cloud cover, rain, wind. No weather data for 8 poaching days	Poaching event reports; patrol data	Descriptive
Herbig and Warchol, 2011	Africa	Poaching	NA	NA	Interviews	Social
Hilborn et al., 2006	Africa	Poaching	NA	NA	Patrol data, arrests, buffalo population	Descriptive
Hill et al., 2015	Africa	Snaring	For animals, hunters, rangers: number, speed; for animals: decay level, poachers: nr of snares, rangers: patrol times	NA	Agent Based Modeling; indirectly: patrol data	Predictive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Hofer et al., 2000	Africa	Herbivores	Major rivers, drainage lines	Meat price, hunting days, prob of catching animal with snare, wildlife density, travel time, costs for weapons, travel, prob of arrest, fine, relief and habitat	Patrol data, interviews, terrain data	Descriptive
Hossain et al., 2016	Asia	Illegal activities	Access, spring/neap tide	NA	Camera traps	Detection
Hossain et al., 2018	Asia	Tigers	Settlements, shipping routes, international borders, ranger stations, illegal activities	Tiger track encounter estimates	Tiger tracks, terrain data	Descriptive
Hötte et al., 2016	Eurasia	Amur Tigers	Poaching along road passing through the reserve was common	Seasonal: e.g. increased patrolling along rivers during salmon runs	Patrol data	Descriptive
Hough, 1994	Africa	Illegal activities	Patrols avoid remote areas and stay close to their stations. Motorized and use major trails; therefore predictable	Major information leakage, collusion and hostility from local communities. Outnumbered by poachers	Interviews with rangers	Social
Ihwagi et al., 2018	Africa	Elephants	NA	Night-day speed ratio of elephants	Elephant tracking and mortality data	Detection
Ihwagi et al., 2015	Africa	Elephants	NA	Land use type, elephant density, year of death, killed illegally/legally (response), no correlation with elephant densities	Aerial counts (live elephants), nomads, researchers, KWS, private ranch managers, community scouts (elephant carcasses)	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Jachmann and Billiouw, 1997	Africa	Elephants	NA	Law enforcement budget/km2, salary/scout/month, km2/carrier, effective patrol days/km2, investigation days, nr bonuses paid, avg bonus rate. Response var = count of illegally killed elephants	Patrol data	Descriptive
Jachmann, 2007	Africa	Illegal activities	NA	Seasonal: limited accessibility and visibility during the rainy season	Patrol data	Descriptive
Jachmann, 2008	Africa	Illegal activities	NA	Human population densities, senior staff visits/camp, operational budget	Patrol data, expenditures, senior staff visits, human population densities	Descriptive
Jachmann et al., 2011	Africa	Illegal activities	NA	Presence of tourists	Patrol data	Descriptive
Jacob et al., 2018	Africa	Illegal activities	NA	NA	Park expenditure, arrests, prosecutions, staff strength	Descriptive
Jenks et al., 2012	Asia	Illegal activities	Distance to head quarters, ranger stations, boundaries	Elevation	Camera traps	Detection
Johnson et al., 2016	Asia	Tigers	NA	NA	Signs of tiger presence, patrol data, camera traps	Descriptive
Jones et al., 2017	Africa	Illegal activities	Nearest village and international border with Liberia	Altitude	Grid sample	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Kablan et al., 2017	Africa	Mammals	Edge of park, tourist/research camps	Transects, signs of presence of animals. Illegal activities. Patrolling effort. Vegetation, rainfall	Patrol data	Descriptive
Kahler, 2018	Asia	Poaching	NA	NA	Interviews with communities	Descriptive
Kar et al., 2017	Africa	Snaring	Roads, water bodies, patrol posts, villages	Animal density, terrain: habitat, roughness, slope, poaching signs	Patrol data	Predictive
Kimanzi et al., 2014	Africa	Roan antilope	Water resources, salt licks, park boundary, far from roads, burnt vegetation, security gates	Slope, animal density	Patrol data	Descriptive
Knapp et al., 2010	Africa	Poaching	NA	Seasonal: agricultural calendar	Household interviews	Social, poaching
Knapp, 2012	Africa	Poaching	NA	NA	Interviews with poachers	Social, poaching
Kyale et al., 2011	Africa	Elephants	Patrol bases, park gates, boundary, park roads, rivers, waterholes	Elevation, slope, vegetation cover type, NO live elephant data (not available)	Land cover type, slope, surface water, elevation, elephant distribution, distance to park roads, gates, boundaries, patrol bases, elephant mortality database.	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Kyale et al., 2014	Africa	Elephants	Land cover, roads, rivers, waterholes, elevation, slope, patrol bases, park gates, park boundary	Seasonality	Elephant mortality data, land cover types, roads, rivers, waterholes, elevation, slope, patrol bases, park gates, park boundaries.	Descriptive
Kyando et al., 2017	Africa	Elephants	More poaching along the edges of the protected area: less patrols	Seasonality: more poaching during the wet season. Less visibility and accessibility; no tourist operations	Patrol data	Descriptive
Leader-Williams et al., 1990	Africa	Rhinos, elephants	NA	Abundance of elephants and rhinos	Patrol data, arrests, sightings of elephants, rhinos and carcasses.	Descriptive
Lemieux et al., 2014	Africa	Illegal activities	Border, water, road, big river, seasonal river, village	Ranger observations in grid cell, poaching observations in grid cell (response variables)	Patrol data	Descriptive
Lewis and Phiri, 1998	Africa	Snaring	Assumed snaring took place near villages (<2 km from garden boundaries); along river; sometimes lagoons and floodplains	Dry season is preferred for snaring	Transects & interviews	Descriptive
Lindsey, Romañach, Matema, et al., 2011	Africa	Bushmeat	NA	NA	Interviews with poachers, bushmeat buyers and ranchers	Social, poaching
Linkie et al., 2015	Asia	Snaring of tigers and their prey	Distance to nearest road, distance to forest edge, distance to nearest village	Elevation, slope, protected area status	Patrol data	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Lockwood, 2010	Africa	Rhinos	Distance to roads (inside and outside property)	Rhino density, property area, land management type, terrain type, housing density	Rhino population data, management reports, patrol data, maps, questionnaire	Descriptive
MacKenzie et al., 2012	Africa	Illegal activities	NA	Signs of illegal activity: harvested trees, illegal trails, grazing livestock, charcoal making, fires, encroachment	Transects, household surveys	Descriptive
Maingi et al., 2012	Africa	Elephants	Roads, river, waterholes, mgt point (park offices, gates, lodges, patrol bases, outposts), settlements	Elevation, land cover, EVI, elephant mortality (response), wet season/dry season	Patrol data	Descriptive
Marescot et al., 2019	Asia	Large mammals	Distance to ranger station, distance to road	Observed animals and snare positions; patrol effort, stream length crossing each grid cell	Patrol data	Descriptive
Massé et al., 2017	Africa	Rhinos	NA	NA	Interviews with community scouts	Social
Moore et al., 2018	Africa	Illegal activities	Near ranger post, tourist trail, boundary	Elevation (in categories/bins), ranger visits	Patrol data	Descriptive
Moreto and Matusiak, 2016	Africa	Ranger-community relations	NA	NA	Interviews	Social
Moreto and Lemieux, 2015	Africa	Ranger views on law enforcement	NA	NA	Interviews	Social
Moreto, 2016	Africa	Occupational stress among rangers	NA	NA	Interviews	Social

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Moreto and Matusiak, 2016	Africa	Law enforcement culture and operations	NA	NA	Interviews	Social
Moreto et al., 2015	Africa	Actions that undermine the efficacy of patrolling	NA	NA	Interviews	Social
Moreto et al., 2014	Africa	Illegal activities	NA	NA	Patrol data	Descriptive
Mubalama, 2010	Africa	Illegal activities	Roads, rivers, villages, park infrastructure, habitats, animal distribution	Land cover, socio-economic data	Patrol data	Descriptive
N'goran et al., 2012	Africa	Monkeys	Distance to villages and number of villages, distance to roads, distance to tourist site, distance to research station, percent primary forest in a 2x2 km neighbourhood	Human population size, duration of patrolling and location of patrol routes	Transects and patrol data	Descriptive
Nahonyo, 2009	Africa	Illegal activities	NA	Probability of discovering poaching gangs ~ days on patrol, arms carried, season (wet/dry)	Patrol data	Descriptive
Ngene et al., 2013	Africa	Elephants	Distance to settlements, wildlife, livestock, charcoal kiln, river, water points, elephant carcasses	NA	Aerial surveys, remote sensing data and management regime data (from visits to areas)	Descriptive
Nguyen et al., 2016b	Africa	Illegal activities	NA	Patrol effort (coverage) and poaching signs; other, e.g. slope and habitat	Patrol data, geographical data	Predictive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Nguyen et al., 2016a	Africa	Illegal activities	Distance to rivers, roads and villages	Habitat, slope, animal density	Patrol data, geographical data	Predictive
Nolte, 2016	South America	Illegal activities	NA	Equipment, staff, infrastructure, patrolling intensity and patterns (incl predictability), land tenure situation; list of priority threats	Interviews, workshops	Social
Nyirenda and Chomba, 2012	Africa	Illegal activities	NA	Length of patrols in days	Patrol data	Descriptive
Nyirenda et al., 2015	Africa	Elephants	Distance to rivers	Elephant carcasses, season (dry/wet, agricultural)	Patrol data	Descriptive
O'Kelly et al., 2018b	Asia	Snaring	Distance to villages and markets, distance to ranger stations, terrain ruggedness, distance to boundary, distance to international border	Forest density, patrol effort (number of patrols per site per year in the previous year)	Site sampling of snares	Descriptive
O'Kelly et al., 2018a	Asia	Snaring	NA	NA	Sample test sites	Descriptive
Ouko, 2013	Africa	Elephants	Roads, rivers, boundary	Rain, livestock, poverty, populations, slopes, soils, DEM, NDVI	Patrol data	Descriptive
Park et al., 2015	Africa	Rhinos	Distance to water, roads, houses/buildings, sources of vegetation	Elevation, steepness, number of rhino visits to cell	Animal movement data, location of snares, location of carcasses; patrol data	Predictive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Park et al., 2016	Africa	Rhinos, elephants	NA	Animal movements, illegal activities, elevation, snares, date, vegetation	Animal movement data, location of snares, location of carcasses; patrol data	Predictive
Piel et al., 2015	Africa	Chimpanzees	Distance to camp, refugee camp	Vegetation type, year, season	Transects	Descriptive
Plumptre et al., 2014	Africa	Illegal activities	Road, edge, settlement, river, patrol post	Land cover, rainfall, elevation, slope, soil, agr land, fire freq. Maxent bias grid based on sampling effort	Patrol data	Descriptive
Rashidi et al., 2018	Africa	Elephants	roads, int'l border, rivers/streams, settlements	Elephant density, livestock density, NDVI, SD NDVI, slope, elevation, waterhole density, wet/dry season	Elephant populations and poaching incidents from KWS; patrol data	Descriptive
Rashidi et al., 2016	Africa	Elephants	roads, int'l border, rivers/streams, settlements	Elephant density, livestock density, NDVI, SD NDVI, slope, elevation, waterhole density, wet/dry season	Elephant populations and poaching incidents from KWS; patrol data	Descriptive
Rashidi et al., 2015	Africa	Elephants	Nearest distance of poaching events	Time between poaching events	Elephant populations and poaching incidents from KWS; patrol data	Descriptive
Rauset et al., 2016	Europe	Bears, lynx, wolves	Mortality locations and times of large carnivores	Steep terrain, forests,permanent human activity, protection status of areas	Mortality data	Descriptive
Rifaie et al., 2015	Asia	Sumatran tiger	Nearest distance of poaching events	NA	Tiger poaching data, spatial data; patrol data	Descriptive
Risdianto et al., 2016	Asia	Sumatran tiger and their prey; snaring	Distance between traps	Time of observation of snares	Patrol data	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Rotich et al., 2014	Africa	Poaching	NA	NA	Interviews	Descriptive
Shaffer and Bishop, 2016	Africa	Elephants	Water, roads	Land cover type, poaching events (response)	Patrol data	Descriptive
Sharma et al., 2014	Asia	Tigers	Distance to railroads, roads, tiger habitats, tiger trading hub districts	NA	Database with tiger poaching occurrences; patrol data and third parties	Descriptive
Sibanda et al., 2016	Africa	Elephants	Distances from roads, rivers, boundaries	Seasons, NDVI	Elephant carcasses; patrol data	Descriptive
Steinmetz et al., 2014	Asia	Poaching	Water, trails, streams, edges	NA	Patrol data, interviews	Social, poaching
Stokes, 2010	Asia	Tigers	NA	NA	Patrol data	Descriptive
Tranquilli et al., 2012	Africa	Great Apes	NA	Size of protected area, years of protection, GDP/cap, armed conflicts, human population density, degraded area, conservation efforts (tourism sites, research sites, presence of guards), NGO presence	Literature, questionnaires, personal communications	Descriptive
Vanthomme et al., 2017	Africa	Bushmeat poaching	Distance to roads	Net Primary Productivity; percentage of wetlands, hunter travel costs, access restriction, traffic index, forested land cover; distance to plantations; rainfall	Patrol data on sampling plots; magnetic vehicle counters	Descriptive

(continued)

Author	Region	Focus	Cues.distance	Cues.other	Input.data	Analytics
Wato et al., 2006	Africa	Snaring	Highway, transnational border, farmland, pastoral, ranches, towns	Habitat type	Transects (snare and animal counts), patrol data	Descriptive
Watson et al., 2013	Africa	Snaring	Nearest road, nearest crops, water, boundary, nearest cultivated and/or settlement	NA	Grid sample	Descriptive
Wiafe, 2016	Africa	Illegal activities	NA	Agricultural calendar	Patrol data	Descriptive
Wilfred and Maccoll, 2014	Africa	Illegal activities	NA	NA	Transects (by car)	Descriptive
Wrangham and Mugume, 2000	Africa	Poaching	NA	NA	Patrol data	Descriptive
Yang et al., 2014	Africa	Snaring	Distance from (international) border	Animal density, terrain	Model, Patrol data	Predictive
Zafra-Calvo et al., 2016	Africa	Elephants	NA	Elephants (dead, alive), T, P, vegetation (detectability), elevation, roads, villages, agricultural land use, logging, population density, management type	Elephant count data and carcass data; spatial and climate data	Descriptive

Appendix C: Interview protocol

Fieldwork involves interviews with rangers and representatives of villages. These interviews must comply with ethical standards, such as informed consent. The procedure set out below complies with the BSA Statement on Ethical Practice (British Sociological Association, 2017).

1. The interview's goal is explained to the ranger or village representative to be interviewed (hereafter: interviewee).
2. If the interviewee is proficient in English, permission is asked to conduct the interview in English. If not, then permission is asked to interview Swahili with the help of an interpreter. The interviewer is a junior staff member of the conservancy who does not work in the same organizational unit as the interviewee. The further procedure outlined below outlines the steps if consent from the interviewee is obtained.
3. The goal of the interview is explained. The interviewer ensures that the interviewee understands the goal. The interviewer explains he is interested in *poaching* (the activity), not in *poachers* (the individuals).
4. The interviewer explains to the interviewee that participation is voluntary.
5. The interviewer explains that the interview is conducted on condition and guarantee of anonymity. Reporting to Soysambu management about the results of the research are done on an aggregated basis, viz. the collected and aggregated opinion of the interviewees is presented.
6. The interviewer asks whether the location of the interview is permissible. Generally speaking, the interview is held in the field, set apart from any other staff members. Alternative locations are offered if the interviewee expresses dissatisfaction with the present interview location.
7. Each answer from the interviewee is summarized by the interviewer to ensure that the answer is well-understood.

8. The final question of the interview is: “Is there anything I should have asked but didn’t?”

Appendix D: Questionnaires

Table 3: Closed questions for rangers.

Nr	Question
1	Poaching occurs here
2	Poaching occurs throughout the year
3	We have sufficient rangers to stop poachers
4	We have sufficient rangers to stop poachers
5	Poaching occurs: Near park borders
6	… Near roads
7	… Near water/salt licks
8	… Near human settlements
9	… Near offices and gates
10	… Near lodges
11	Poachers know who we are
12	We know who the poachers are
13	We have sufficient transport and equipment
14	The communities know who the poachers are
15	We have contact with the communities
16	Poaching was worse in the past
17	Poaching will get worse in the future
18	We are able to stop poachers
19	We patrol in the night
20	We patrol during holidays
21	Poaching is done by individuals
22	Our patrolling strategy is predictable
23	We meet poachers sometimes

Table 5: Open questions for communities.

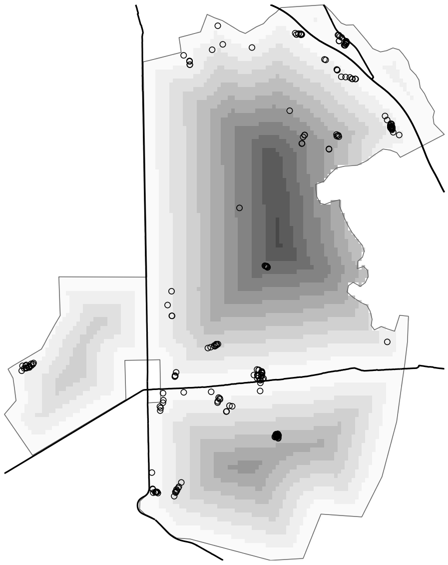
Nr	Question
1	What is the role of wildlife in your community?
2	What is needed to improve the situation if required?
3	Do you have contact with Soysambu, do they do projects here?
4	Have you been consulted for the LEWSE management plan?
5	Other – as appearing during the interview

Table 4: Open questions for rangers.

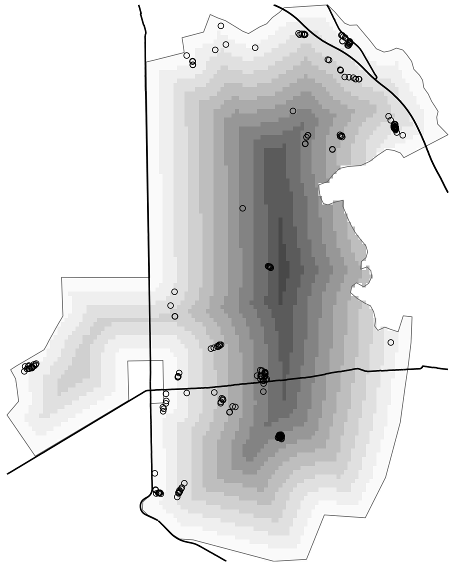
Nr	Question
1	What is the best time for poaching?
2	What is the best place for poaching?
3	Why do they (the poachers) poach?
4	Who are they?
5	If you would be in charge and have unlimited resources, what would you do to stop poaching?
6	If you would one place to find a poacher tomorrow, where would you go, and when?
7	In what sort of terrains do you find poachers?
8	How do they hunt?
9	Did this change over the years?
10	What are the current bottlenecks to find poachers?

Appendix E: Snare distances to spatial features

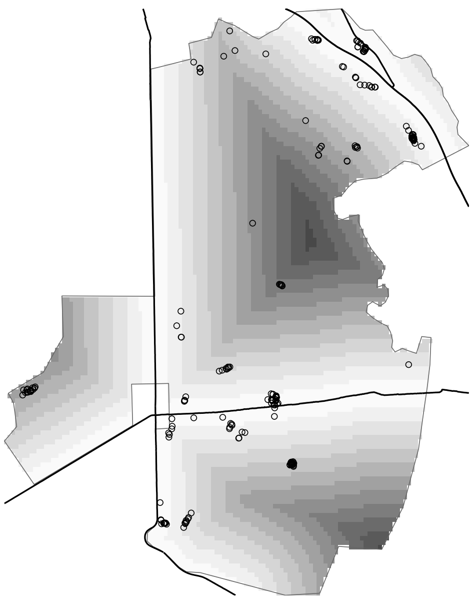
Distance to roads and boundaries



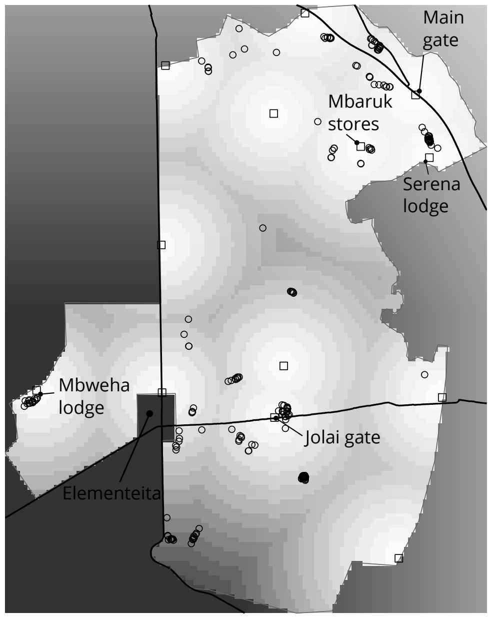
(a) Roads and boundaries



(b) Boundaries only



(c) Roads only



(d) Park infrastructure

Figure E1: Snare positions relative to conservancy boundaries, roads and park infrastructure (gates, lodges, and staff quarters).

Distance to park infrastructure

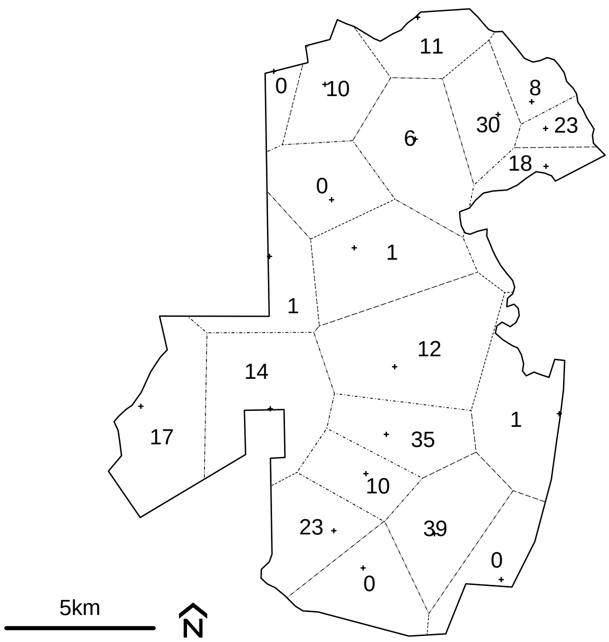


Figure E2: Snare count per park infrastructure tessellation tile

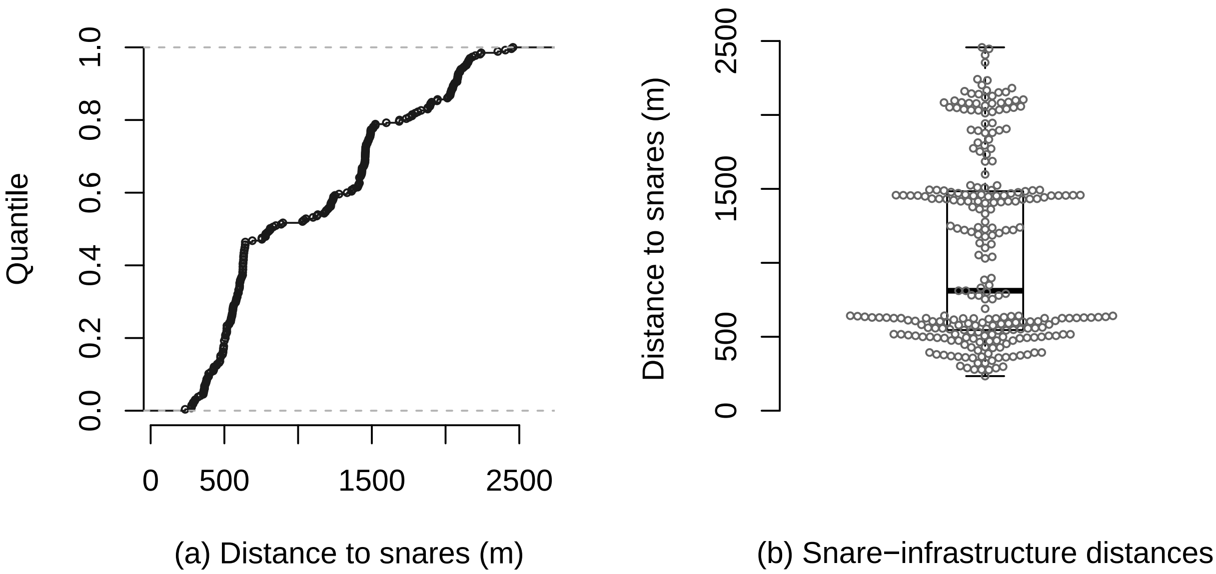


Figure E3: Empirical cumulative frequency distribution (ECDF) plot (a) and box plot (b) for snare-infrastructure distances.

Distance to communities

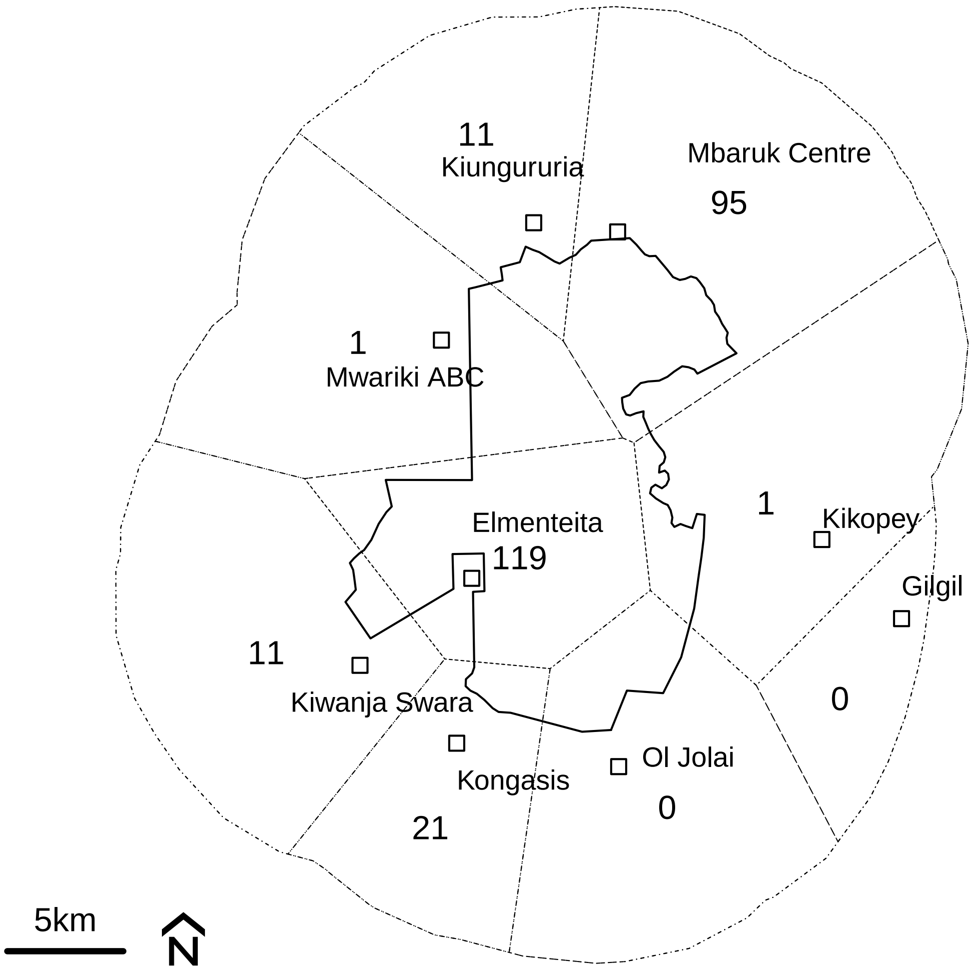


Figure E4: Snare count per community via Dirichlet tessellation.

Appendix F: Exact binomial tests ranger interviews

All statistical tests included in the tables below are two-sided exact binomial tests with $H_0 = 0.5, \alpha = 0.05$.

Table F1: Exact binomial tests for questions regarding temporal occurrence of poaching.

Question	agree	N	p	s
Poaching was more severe in the past	23	27	0.000	***
Poaching is currently severe	17	18	0.000	***
Poaching will become more severe in the future	10	25	0.424	

Table F2: Exact binomial tests for questions regarding occurrence of poaching.

Question	agree	N	p	s
Poaching occurs in hotspots	27	28	0.000	***
Poaching occurs throughout the year	4	27	0.000	***
Poaching is carried out by individuals	2	27	0.000	***

Table F3: Exact binomial tests for questions regarding deterrence capability of rangers.

Question	agree	N	p	s
We sometimes meet poachers	24	26	0.000	***
Our patrolling strategy is predictable	19	25	0.015	*
We have sufficient rangers to stop poachers	10	28	0.185	
We are able to stop poachers	6	26	0.009	**
We have sufficient transport and equipment	2	26	0.000	***

Table F5: Exact binomial tests for ranger opinions on likely snare positions.

Question	agree	N	p	s
Snares are found near park borders	25	26	0.000	***
Snares are found near human settlements	24	28	0.000	***
Snares are found near roads	18	26	0.076	
Snares are found near water/salt licks	14	22	0.286	
Snares are found near lodges	12	24	1.000	
Snares are found near offices and gates	2	26	0.000	***

Table F4: Exact binomial tests for familiarity between poachers, rangers and communities.

Question	agree	N	p	s
The communities know the poachers	28	28	0.000	***
Poachers know who we are	25	26	0.000	***
We have contact with the communities	10	23	0.678	
We know who the poachers are	7	19	0.359	

Appendix G: Snare density estimations by rangers

Table G1: Snare density estimations by rangers.

i[a]	n[b]	x[c]	$\bar{x}$[d]	PD[e]	IE[f]	CE[g]	r[h]
1	6	4	4.333	30.222	30.333	0.111	0.004*
2	6	3	10.000	24.333	73.333	49.000	0.668
3	6	2	3.833	1.472	4.833	3.361	0.695
4	7	12	3.143	7.837	86.286	78.449	0.909
5	7	7	4.000	1.714	10.714	9.000	0.840
6	6	0	7.167	41.806	93.167	51.361	0.551
7	5	0	2.600	8.240	15.000	6.760	0.451
8	6	1	1.000	1.333	1.333	0.000	0.000*
9	7	0	1.857	2.980	6.429	3.449	0.537
10	6	18	4.500	4.250	186.500	182.250	0.977
11	4	7	5.750	3.188	4.750	1.562	0.329
12	7	10	4.143	4.122	38.429	34.306	0.893
13	5	0	1.200	3.760	5.200	1.440	0.277
14	13	7	13.462	33.479	75.231	41.751	0.555
15	6	1	2.333	6.222	8.000	1.778	0.222
16	6	0	4.167	15.139	32.500	17.361	0.534
17	5	9	3.600	9.040	38.200	29.160	0.763
18	6	1	7.833	36.806	83.500	46.694	0.559
19	17	1	12.059	122.997	245.294	122.298	0.499
20	5	1	2.200	2.960	4.400	1.440	0.327
21	5	2	1.800	2.160	2.200	0.040	0.018*
22	6	1	2.667	11.889	14.667	2.778	0.189
23	5	3	4.800	1.360	4.600	3.240	0.704
24	5	74	4.400	4.240	4848.400	4844.160	0.999
25	8	35	16.500	99.750	442.000	342.250	0.774
26	6	20	6.667	20.222	198.000	177.778	0.898
27	7	7	5.714	4.490	6.143	1.653	0.269

[a] Snaring transect index number.
[b] Number of ranger estimations for transect i.
[c] Number of snares found during transect i.
[d] Group average estimation for transect i.
[e] Prediction Diversity.
[f] Individual Error.
[g] Collective Error.
[h] CE/IE. Values smaller than 0.05 are marked with an asterisk.

Annex H: Improved desnaring strategies

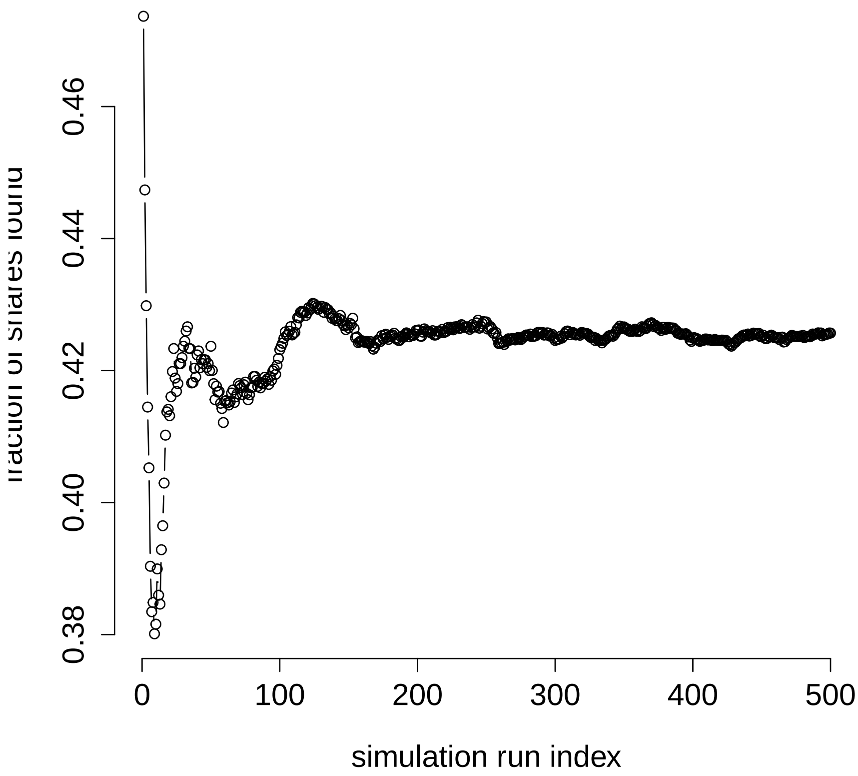


Figure H1: Cumulative moving average of snares found for hotspot search strategy.