Incorporating social-ecological complexities into conservation policy

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ABSTRACT

In the process of developing new conservation policies, policymakers must have access to information which will inform their decisions. Evidence rarely considers the complexities of social-ecological systems. The Social-Ecological Systems Framework (SESF) is an adaptable yet structured approach for understanding the processes that lead to changes in natural resources, using a systems-based approach that aims to treat ecological and social components equally. Few conservation planning and policy initiatives have implemented the SESF to assess the interlinked social and ecological consequences of conservation policies. We apply the SESF to explore the barriers to the potential implementation of a policy of consumptive utilisation of wildlife in Kenya, a policy regarded as successful in several southern African countries. Using secondary data and expert review we developed a conceptual model of the social-ecological system associated with consumptive utilisation of wildlife in Kenya. We then analysed how different combinations of first and second-tier variables interacted to create focal action situations, and subsequently identified seven barriers to this policy. Our analysis revealed that game ranching would require large-scale investment in effective monitoring systems, new regulations, training, market development and research, considerations about equity, and devolved ownership of wildlife. The least barriers existed for game farming. The SESF appears to be a useful framework for this purpose. In particular, it can help to reveal potential social and ecological barriers which conservation policies might face in attempting to meet intended goals. The information required to implement the SESF are necessarily cross-disciplinary, which can make it challenging to synthesise.

1. Introduction

Conservation policymaking should be evidence-informed (Adams and Sandbrook, 2013; Sutherland et al., 2004). When developing a new conservation policy, information from a range of sources can be integrated to improve the quality of evidence provided for policy decisions, including quantitative and qualitative data; local and indigenous knowledge; and local and international perceptions (Adams and Sandbrook, 2013; Bennett, 2016). Evidence can also be collated from the successes or failures of previous and current policies both within the administrative regions of interest (state, country, etc.) and regions external to the process. However, when implementing or importing new conservation policies, it is vital to acknowledge that regional heterogeneity exists in both the ecological and social realms, with differences across space, time and organisational units all influencing the potential

for social-ecological sustainability (Liu et al., 2007). The evidence used to enlighten policymaking should, therefore, be relevant to the context in which it will be applied rather than assuming that 'one-size-fits-all' (Adams et al., 2019; Dressel et al., 2018).

Frameworks for analysing policies, such as the "Eightfold Path to Policy Analysis" (Bardach, 2000) provide useful ways to think systematically about how to achieve the intended policy objective. However, when policies pertain to conservation, it is critical to acknowledge that natural resources, including wildlife, are part of multi-faceted and complex social-ecological systems (SES; Ostrom, 2007). Failing to explore the full suite of social and ecological factors determining the success or failure of conservation policies (Brehony et al., 2018), and preparing for unexpected interactions between factors, and their associated challenges can result in failure to achieve the intended conservation goals (Liu et al., 2007).

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1.1 Social-Ecological Systems Framework for conservation policy analysis

Ostrom's (2009) Social-Ecological Systems Framework (SESF) is an adaptable yet structured approach to understanding the processes that lead to the deterioration or improvement of natural resources, using a systems-based approach that aims to treat ecological and social components equally. SESF originates in the discipline of political science (social science) and is based on theories such as collective choice, common-pool resources, and natural resource management (Binder et al., 2013). It aims to move beyond simple panaceas, and towards a diagnosis of "the source, and possible amelioration, of poor outcomes for ecological and human systems" (Ostrom and Cox, 2010).

Establishing a diagnosis when faced with a problem in an SES requires careful study of complex, multi-variable, non-linear, and cross-scale interactions, and how these are changing through time (Liu et al., 2007). The SESF achieves this by analysing the attributes and interactions of four main subsystems: resource users, governance system, resource system, resource units (McGinnis and Ostrom, 2014; Fig. 2). The crucial part of this analysis is in the "Focal Action Situations", where interactions between the subsystems lead to outcomes and feedback to each core subsystem. While examples primarily at the local scale were used to develop the SESF, its design ensures it is applicable at regional, national, and international scales, and importantly, it incorporates interactions across these scales (Cumming et al., 2015). Based on these criteria, the SESF facilitates the selection of variables in relation to case studies and can be used to understand changes in interactions and outcomes when the system changes.

Despite the growing frequency of use of SES language within conservation planning and policy initiatives, and continued calls for further integration of the social sciences into conservation science (Bennett, 2016), few examples exist where the SESF has been implemented to enlighten the development of conservation policies (Ban et al., 2015; Bennett et al., 2018; Guerrero and Wilson, 2017). Importantly the SESF explicitly accounts for the influence of social factors on the system, including individuals, institutions, governance structures and existing policy, which many conservation planning processes inadequately address (Ban et al., 2015). In this paper, we show how the SESF can be used to enlighten conservation decision making, by applying it to a multifaceted conservation policy option, which has been implemented in other, apparently similar SES.

1.2 Implementing the Social-Ecological Systems Framework in context

Conservation of savannah ecosystems in Africa often encourages the sharing of practices and policies adopted in southern Africa countries with countries in East Africa. For instance, Beale et al. (2013) provided a set of ten valuable lessons for conservation in East Africa from experiences over the last few decades in southern Africa. Beale et al.'s (2013) lessons are ranked without consideration to local variations in social and ecological systems, and as the authors acknowledge, these local conditions are important considerations when implementing new policies or interventions.

The trade of wildlife and the cropping of wildlife for sale as game meat is common in southern Africa, and both are cited as ecologically and economically successful strategies for conservation (Lindsey, 2011). It is perhaps unsurprising, therefore, that there have been public, governmental, and academic calls to implement a policy of consumptive use of wildlife in Kenya (Norton-Griffiths, 2007).

In Kenya, 60% of wildlife exists outside protected areas (Western et al., 2009). Overcoming opportunity costs to landowners, which are created by forgoing land options which are not compatible with wildlife conservation, and incurring costs from wildlife - such as competition for grazing, human-wildlife conflict, and disease transmission - are major challenges to conservation sustainability (du Toit et al., 2017). In recent years, Kenya has developed policies and legislation which, under the mandate of Kenya's 2010 constitution, aim to increase the economic value of wildlife across the country, accrued at multiple levels including landowners, county government, and the national treasury (Government of Kenya, 2013). The resulting revenue generation has primarily occurred through ecotourism, but there are concerns that this is not enough to counter the continued widespread losses of wildlife (Ogutu et al., 2016).

In April 2018, Kenya's Ministry of Tourism and Wildlife launched a task force to investigate the potential for the sustainable consumptive utilisation of wildlife, through the cropping of wildlife for the sale of game meat, and the trade of live wildlife. The notice also explicitly stated that Kenya has banned sport hunting and has no intention of opening this debate. Both were to be examined through extensive "game ranching" and intensive "game farming" frameworks. Kenya's Wildlife Act (2013) describes "game farming" as the rearing of wildlife in an enclosed and controlled environment for wildlife conservation, trade and recreation; and "game ranching" as the keeping of wildlife under extensive natural conditions with the intention of engaging in wildlife conservation, recreation and trade. In addition to the appointed task force, there was a request for expert and public participation to provide evidence on the potential implementation and impact of this policy for consideration by the Ministry. We, the authors of this paper, used this opportunity to investigate how the SESF can be used as a tool to provide evidence to policy-makers (Toomey et al., 2017), by highlighting potential barriers to achieving intended policy outcomes. Two pieces of Kenyan policy defined the outcomes: firstly, the 2013 Wildlife Act, which includes a provision for a wide range of both consumptive and non-consumptive wildlife users' rights devolved to landowners (Government of Kenya, 2013); secondly, the National Wildlife Strategy which sets a clear vision for the conservation of wildlife in Kenya over the next decade, namely, that "Kenya's wildlife is healthy, resilient and valued by Kenyans" (Ministry of Tourism and Wildlife 2018). We used the SESF to explore the impact that potential policy changes might have on social and ecological subsystems when aggregated in different ways (Bodin and Tengö, 2012) through the following two guiding research questions:

1. Will the consumptive utilisation of wildlife in Kenya lead to improved benefits for landowners with wildlife, all Kenyans (including through increased food security), and the national government? (O1)
2. Will the consumptive utilisation of wildlife in Kenya lead to the sustainable conservation of wildlife and ecosystems? (O2)

   This allowed us to then ask:

3. What are potential ecological, social, and economic barriers to implementing a policy of sustainable consumptive utilisation of wildlife in Kenya?

The outcomes of this investigation were subsequently submitted as a report to the task force (Tyrrell et al., 2018).

2. Methods

The framework for our analysis is laid out in Fig. 1, and was guided by the updated SESF described in McGinnis and Ostrom, 2014 (Fig. 2) and the diagnostic framework of Hinkel et al. (2015). The under-recognized potential to contribute to the understanding of conservation problems through the integrated experience of individuals, or what Sanderson et al. (2002, p. 71) calls 'practical wisdom', guided the composition of this authorship team. We, the authors of this paper, have knowledge and experience from a variety of backgrounds, namely: conservation policymaking at a national level; representation of community landowners at a sub-national level; representation of private and community landowners at a national scale; rangeland and wildlife
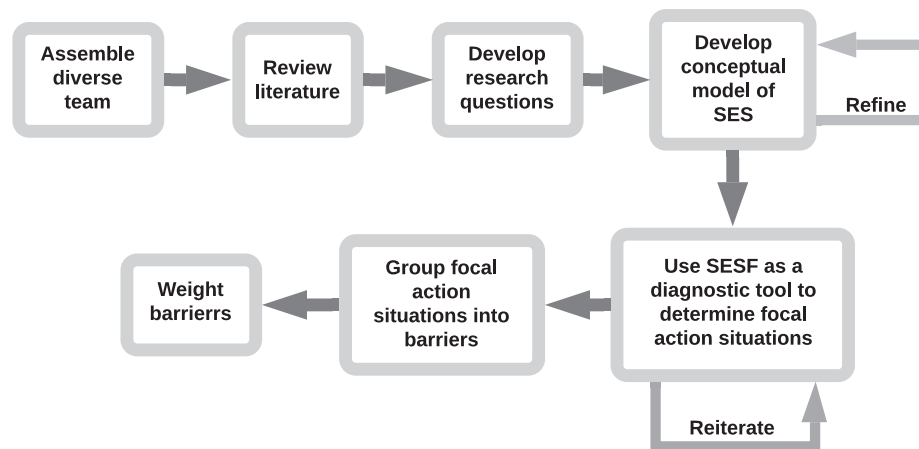
Fig. 1. The methodological framework used to develop the list of barriers to achieving desired outcomes from the implementation of the conservation policy.

ecology; geography. Importantly, the team included people with the insights and perspectives of key stakeholders (Bennett et al., 2018) and incorporated knowledge and experience from different academic disciplines.

2.1. Constructing a conceptual model

Our analysis began with a literature review on the consumptive use of wildlife in Kenya, East Africa, Southern Africa, and globally, conducted by PT and PB (Fig. 1). We then held a workshop with all the authors to feedback the findings from the literature review, and to develop our understanding of the SESF in this context, including the social, economic, and political setting. All our explorations were guided by the recommendations from McGinnis and Ostrom, 2014 (Fig. 2) and the diagnostic framework of Hinkel et al. (2015; Fig. 4, Supplementary Table 1) and were based on our research questions, which were determined by the scope of the task force, as laid out by the Ministry of Tourism and Wildlife. We discussed and defined the first-tier subsystems; the resource system, resource units, actors and governance systems (Fig. 2). We listed all the pertinent second-tier variables which were relevant to our research questions and the aforementioned policy objectives (Fig. 2). These are given in Fig. 2, where they have been assigned a code, which we refer to, in brackets, in our results. Although we do not report third-tier variables, we used Vogt et al.'s (2015) framework to improve our deductive inquiry by ensuring that ecological attributes in the resource system and resource units were explicitly incorporated into the SESF. The transdisciplinary composition of our authorship team was critical to identifying the non-exhaustive list of focal action situations that would result in this SES.

The resultant conceptual model includes the possibilities of both extensive game ranching and intensive game farming as resource systems (Figs. 3 & 4). In our analysis, we assumed these are the two types of resource systems (RS) that "create the conditions for the existence of a stock of resource units" (Ostrom et al., 1994). Our conceptual models also included cropping and live trade of wildlife as different types of consumptive use (Fig. 4).

Wildlife is a public resource in Kenya (ROK, 2010) but user rights of wildlife (both consumptive and non-consumptive) can be granted by the County Wildlife Conservation and Compensation Committees (Government of Kenya, 2013). In its current form, the 2013 Wildlife Conservation and Management Act (ROK 2013) allows for the keeping of some species on game farms (see Tenth Schedule for full list). Therefore, in our conceptual model, wildlife is defined as the resource unit (RU), separated into large mammals (e.g. zebra, kudu, eland etc.) and other wildlife (those listed in the Tenth Schedule of ROK 2013; Fig. 4). Wildlife is a public resource in Kenya; therefore, all wildlife is treated like a collective good in all focal action situations (Hinkel et al.,

2015).

Land in Kenya is considered to be either public, private, or community land (ROK 2010), and conservation as a land use may happen under all of these land tenure options. Our conceptual model assumes that private landowners (an individual, family or body corporate) and community landowners (defined by Kenya's 2016 Community Land Act as a group of individuals) will be the primary actors (and beneficiaries). Their income benefits would be from the sale of wildlife and their products, alive or dead (Figs. 3 & 4). The intensive nature of game farming makes it akin to agricultural production, even within areas of community land tenure, and the management of this resource system may act as if it has de facto private ownership.

2.2. Using the SESF as a diagnostic tool

Following this process of producing our conceptual model, in two subsequent meetings, the authors used the SESF as a diagnostic tool (Ostrom, 2007; Ostrom, 2009; McGinnis and Ostrom, 2014; Hinkel et al., 2015) to analyse how the interactions, outcomes, and feedbacks of the variables defined in the conceptual model, revealed focal action situations (when interactions become outcomes with feedback; this is the basis of Fig. 2). We specifically analysed how different resource systems, resource units, actors, governance systems, and types of consumptive use affected the focal action situations, and how this, in turn, affected the first tier variables (Fig. 4).

These focal action situations were subsequently compiled and analysed by PT and PB, who organised them into groups of focal action situations, based on similarity. This initial set of barriers was then presented back to all the authors over another meeting, where they were refined, based on our knowledge, experience, and collective understanding of the SES. These represented the barriers to the sustainability and positive social-ecological outcomes of consumptive utilisation of wildlife in Kenya (Nagendra and Ostrom, 2014). Finally, through consensus, we assigned a weight to each barrier– low, medium or high – for each combination of primary actor, benefit, type of consumptive use, resource unit, and resource system. We did not explicitly operationalise the SESF with quantitative and qualitative data in a spatially explicit manner (Leslie et al., 2015) as these data were not available within the time frame required for policy action. This entire process took 3 months.

3. Results

3.1. Barriers to effective outcomes

Using the SESF and our conceptual model, we have identified seven barriers to the successful implementation of consumptive wildlife
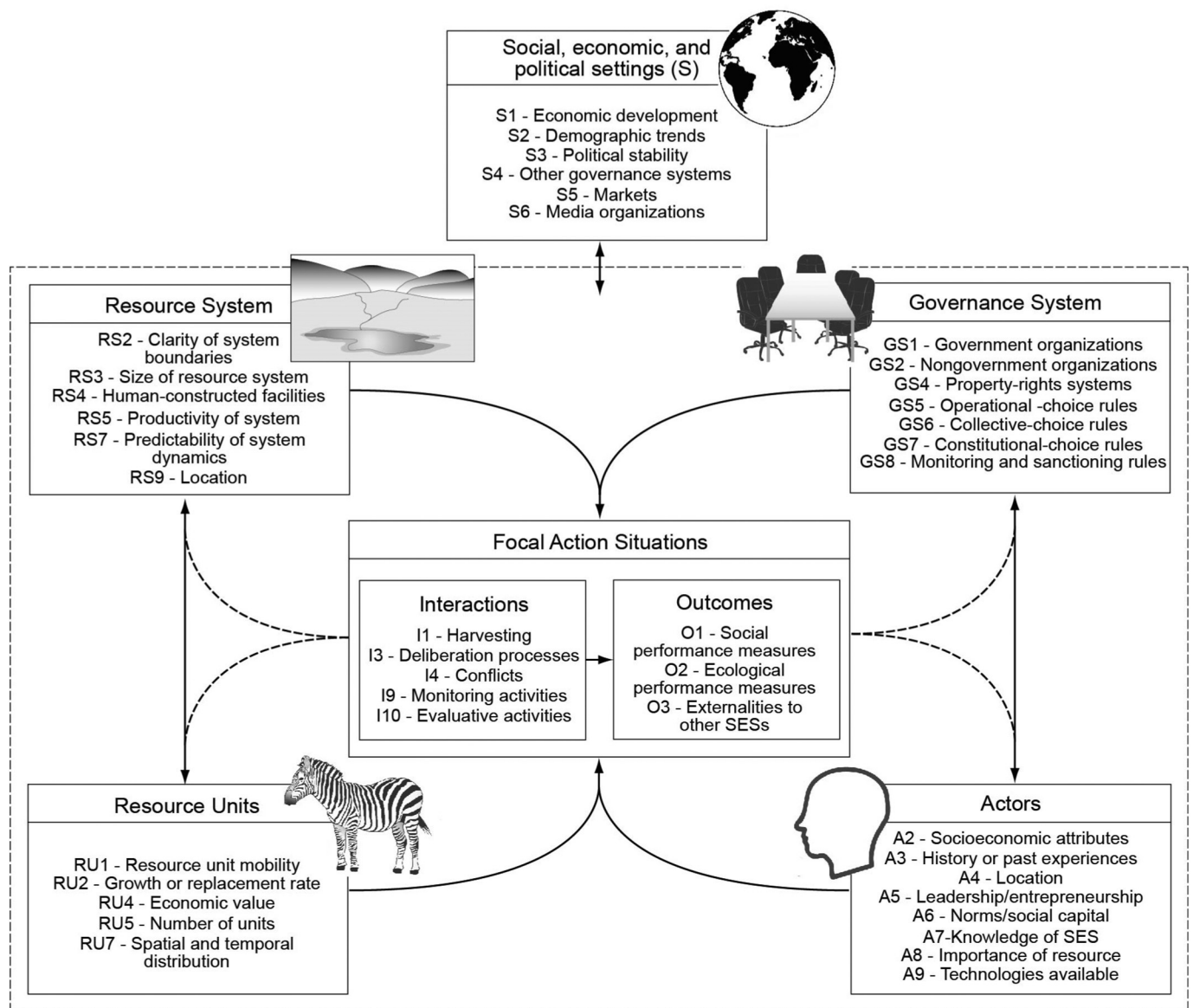
Fig. 2. Social-ecological system framework (SESF) with first-tier subsystems adapted from McGinnis and Ostrom (2014). Solid boxes denote first-tier subsystems: Resource System, Resource Units, Governance Systems, and Actors. Each subsystem contains the second-tier variables which were identified in our analysis. Focal Action Situations are where interactions between the subsystems are transformed into outcomes. Solid arrows denote direct connections. Dashed arrows denote feedback from action situations to each of the subsystems. The dotted-and-dashed line that surrounds the interior elements of the figure indicates that the focal SES can be considered as a logical whole, but that exogenous influences from related ecological or social-economic-political settings at larger or smaller scales, can affect any subsystem of the SES.

utilisation in Kenya. We present each of these barriers and elaborate on how the SESF and our conceptual model aided our analysis. Throughout our results, we refer to the relevant second-tier variables in brackets, based on the codes given in Fig. 2.

3.1.1. Ownership and wildlife movement

The ownership and mobility of wildlife pose a significant barrier to the successful implementation of consumptive utilisation of wildlife for game ranching, particularly on community-owned land. The semi-arid and arid nature of most of Kenya's wildlife areas lead to high spatial and temporal variability in rainfall and, therefore, pasture quality and quantity (Fynn et al., 2014; RS7, RS5). Consequently, wildlife moves to capitalise on this underlying functional heterogeneity (RU1, RU7; Tyrrell et al., 2017). The scale of movement can be orders of magnitude larger than individual land parcels, representing a mismatch between ecological and anthropogenic boundaries of resource systems (Hobbs

et al., 2008; RS3). The movement of wildlife across landscapes blurs the lines over wildlife user rights, whether for trade or cropping. In fenced, private lands, the mobility of wildlife is restricted (RU1), the resource system is bounded (RS2), and the landowner becomes the de facto 'owner' of wildlife (GS4). In many landscapes without fencing, primarily community areas, but also private land in some regions, wildlife ownership is less clear. Considerable consensus-building between landowners (I3) will be required to ensure equitable sharing of resources to avoid conflict (I4). A lack of consensus could potentially lead to an increase in fencing (RS4) to claim resource ownership (Løvschal et al., 2017). Restrictions on mobility by fencing dryland ecosystems can result in: a reduction in the size of the resource system (RS3); a mismatch between the ecological and the user-defined boundaries of the resource system (RS2; Western and Gichohi, 1993); limited access to spatial and temporal heterogeneity for people and wildlife (RS7, Western et al., 2020); lower productivity and carrying capacity for
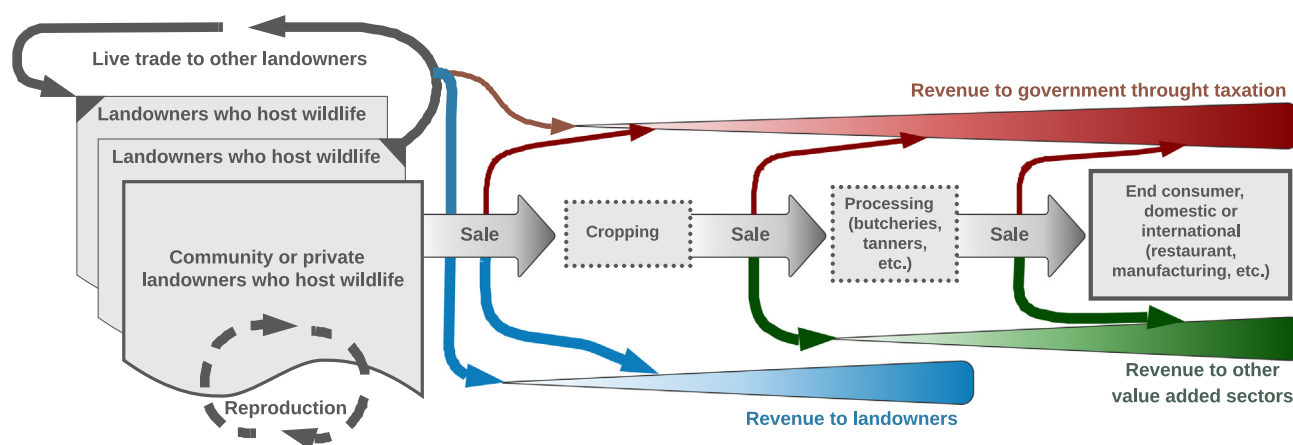
Fig. 3. Simple conceptual model and value chain of potential game ranching and game farming operations in Kenya. This value chain does not include non-consumptive values of wildlife, including, and not limited to cultural, ecotourism, or research (Western et al., 2019).

livestock and wildlife (RS5; Fynn et al., 2014). This conflicts with wildlife and other national policies that aim to maintain mobility (O1, O2, O3).

In South Africa and Namibia, wildlife ownership rights are devolved to landowners. This clarifies resource ownership, and theoretically, provides incentives for improved management. Such a solution could reduce conflict between landowners, and between landowners and the government (I4), especially regarding the setting of harvesting quotas (I1). In such a system, landowners decide which species they will manage, and these decisions are often driven by market forces (S5). Landowners may also choose to work collectively to manage larger ecosystems, thereby overcoming constraints to wildlife mobility (Lindsey, 2011). However, this is no panacea, as similar situations can exacerbate the fencing of rangelands and encourage the removal of non-profitable wildlife (Pitman et al., 2017; O2). Furthermore, devolved ownership could create confusion (I4) over who is responsible for human-wildlife conflicts, such as crop-raiding and predation (O2).

In Kenya, game farming is already based on an intensively managed system, with user rights and ownership devolved to the landowner.

At present, when considering live national trade, the low economic value of the majority of live wildlife (RU4) and the lower frequency of trade (RU5; I1) means that there is less likely to be significant conflict (I4) between resources users with unclear ownership.

3.1.2. Market-based challenges

The consumptive use of wildlife results in different wildlife products that can be sold to consumers. These products include primary products, such as game meat, horns, skin, and bone, but further processing can also provide secondary products such as treated hides and ornaments (Fig. 3).

A successful economic model of the consumptive use of wildlife relies on the assumption that there exists a large market that values these products (S5, RU4; Department of Environmental Affairs 2018). Without this economic value passed on to landowners, the consumptive utilisation of wildlife might not be viable (O1), opportunity costs of living with wildlife would not be met, and when landowners are solely dependent on consumptive utilisation (A8), this can result in land conversion to alternative non-wildlife uses (Cousins et al., 2008; O2). The lack of economic value was a significant issue in the previous wildlife cropping program in Kenya (Kock, 1995; Tasha Bioservices, 2001). For both game ranching and game farming, considerable thought, investment, and research are required to ensure a sustainable and profitable game meat market (A7, A4) with input required from nongovernment (GS2) and government agencies (e.g. Ministry of Agriculture, GS1).

To find a market (S5) the game meat industry could target a) the 1.4 million tourists arriving in Kenya each year (Turner, 2017); b) the broader Kenyan population; or c) an international market. In South Africa, for instance, revenue is generated from tourists in the domestic market, and international export sales, with over 160,000 carcasses sold to Europe in 2005 (Hoffman and Wiklund, 2006). However, Kenya cannot assume that these same markets will exist as, for example, Kenyan game meat may not be accepted in the European market due to disease control regulations (Naziri et al., 2015). Other markets might be found, for example by targeting legal game meat markets in the Middle and the Far East, and by promoting the potential health benefits of some types of game meat (Hoffman and Wiklund, 2006). However, following the outbreak of SARS-CoV-2, these markets are likely to be very carefully controlled and regulated, thereby increasing costs all along the value chain (Fig. 3) and decreasing economic value to the

Primary Actors (A)	Benefit	Type of consumptive use	Stock of Resource Units (RU)	Resource Systems (RS)
Private landowners or Community landowners	Income from consumptive wildlife utilization	Sell wildlife products through cropping — Sell wildlife through live trade	Large mammals* — Other wildlife**	Extensive game ranch — Intensive game farm

* Large mammals include zebra, kudu, eland, etc.
** Other wildilfe are listed in the Tenth Schedule of the 2013 Wildlife Conservation and Management Act: reptiles, snails, frogs, butterflies, ostrich, pigeons, doves, ducks, guinea fowl, quelea (for all animals groups, those listed in the Sixth Schedule are excluded).

Fig. 4. Appropriating action situations for the system studied (adapted from Hinkel et al., 2015).

producers. A longer-term problem is presented by the changing values of some global (and local) consumers, who are increasingly averse to the consumption of any animal products, including wildlife (O1), perhaps increasingly so after the SARS-CoV-2 outbreak. The market for secondary products also needs consideration. Previously in Kenya, due to a lack of infrastructure (RS4), animal hides were sent to South Africa for processing, significantly reducing the value of the product to both the landowner and harvesters (RU4; Tasha Bioservices, 2001). Increasing the economic value of these products, for the benefit of the landowners and the government, requires significant infrastructural development and improved knowledge of landowners (RS4, A7, A9; Department of Environmental Affairs 2018). The trade of wildlife raises similar concerns.

Finally, as long as wildlife is a common (GS4), ungoverned resource without monitoring (I9, GS5), legal markets (RU4) could make wildlife protection on game farms and game ranches increasingly difficult, and rather than decreasing poaching, consumptive utilisation could encourage poaching syndicates to form and exploit this resource (I4, A8; Schneider, 1990; Macnab, 1991).

3.1.3. Unintended conservation consequences

Game ranching can potentially ensure that landscapes remain in a state which benefits conservation (Cousins et al., 2008). However, there may be uneven importance assigned to some species with greater market value (Cousins et al., 2008; Pitman et al., 2017). As the market for wildlife products grows (S5), there could be a drive to decrease the number of species that serve no economic purpose as cropped wildlife and are instead seen as an additional 'economic cost' due to competition or predation (RU4; Geist and Veleius, 1988; Macnab, 1991; Cousins et al., 2008). For instance, carnivores, many of which are of high conservation concern, may be eradicated or removed from game ranching operations if they are seen as an economic threat (Cousins et al., 2008; Geist and Veleius, 1988). There could also be incentives to introduce non-native herbivores with a greater economic value (RU4) as has happened in South Africa (Castley et al., 2001). Furthermore, market incentives could result in the intensification of even extensive ranching, through hormones, sterilization, and breeding (S5, RU2, RU5; Knox et al., 1991; Mulley et al., 1996). All of these would have widespread impacts on ecosystem functionality and biodiversity conservation (O2; Richardson, 1998; van Kooten et al., 1997). These focal action situations do not necessarily fit the ecosystem-based approach to conservation that Kenya currently aspires to in the Wildlife Act. 2013 (O1, O2).

The intensive nature of game farming, with single species and high productivity, makes it unclear how this would contribute to broader conservation goals (Macnab, 1991). Game farms could cause a decrease in non-farmed wildlife without regulatory frameworks (GS8), devolved property rights (GS4), and enforcement (I9,I10; Brooks et al., 2010; Lyons, 2012; Tensen, 2016). In Vietnam, for example, commercial farming of the Southeast Asian porcupine (Hystrix brachyura) lead to the exploitation of wild porcupines, caused by high demand from farms and consumers. This may be the cause of the massive decline in wild populations across the region (Brooks et al., 2010). Game farming will likely require a species by species approach, with appropriate regulation and monitoring of both wild and farmed stock (GS8, I9), to ensure the sustainability of wild populations (O2).

Cultural perceptions of wildlife may change when they are given an instrumental monetary value. Changing wildlife to a consumable commodity, like livestock, could alter the care that humans provide wildlife when the economic value of wildlife is emphasized over other, non-monetary values (Pitman et al., 2017). This is of particular importance in community land, where cultural values that allowed for coexistence between wildlife and livestock are rapidly being eroded (A6), and with it reduced tolerance for wildlife, including carnivores (Western et al., 2019). Nevertheless, for landowners where wildlife does not have an intrinsic or cultural value, their instrumental value

may be necessary for continued conservation (A3, A4, A2, A6).

3.1.4. Equity and conflict between actors

The previous wildlife cropping program was seen to only really benefit large-scale private landowners (Tasha Bioservices, 2001). Private landowners are likely to benefit more from consumptive utilisation, due to higher productivity lands (RS5, see below); established knowledge of requirements and regulations (A7, GS8; Norton-Griffiths, 2007); demarcated land, often fenced, to effectively own wildlife (GS4); infrastructure for wildlife cropping (A9); closer proximity (in general) to established markets and transport (A4); and greater financial capital to successfully implement and market wildlife products (A2). Additionally, without regulation and support (GS8), many communal lands have historically been susceptible to poor governance (A3, A5) and elite capture of resources, which can result in unscrupulous practices and exploitation by harvesters (A7; Bond and Mkutu, 2018; Tasha Bioservices, 2001). Models such as the Kenya Wildlife Conservancies Association and the new Community Land Act (2016) can play an important role in supporting devolved management and equitable revenue sharing (A5, A6, I3, GS6, GS5, O2).

In Kenya, the largest and most stable remaining wildlife populations (RU2, RU5) are in areas of high rainfall and forage productivity with low variability (RS7, RS5). These areas are generally under private ownership and would likely have the largest quotas (I4) and potentially the most profitable wildlife cropping business. However, much of Kenya is semi-arid and arid, and these areas are generally under community ownership. They have much lower wildlife densities, with populations that move across large landscapes (RU1, RU2, RU5; Ogutu et al., 2016). This reality could limit the scope for financially and ecologically sustainable wildlife cropping in community lands.

Differences in historical perceptions between actors (A3), are also likely to have an impact on the support for the consumptive utilisation of wildlife in Kenya. Private landowners saw the greatest benefit under the previous wildlife cropping program (Tasha Bioservices, 2001), and many are in favour of the return of wildlife cropping and trade (Kaelo, pers. coms.). Many community landowners, however, report that they saw little benefit from the program, and are averse to the re-adoption of a system that has historical failures, and that can clash with cultural values attached to wildlife (RU4, O2; Tasha Bioservices, 2001; Western et al., 2019).

Tourists are an additional important actor in this SESF who must be considered in this context. Tourism is a major revenue earner for Kenya, contributing 9–11% of the total gross domestic product (Turner, 2017). In particular, tourism plays an essential role in wildlife conservation by giving wildlife additional economic value (RU4). The greater Maasai Mara ecosystem is the best example of this in Kenya, where tourism generates revenue for 106,102 households, covering 170,131 Ha of land, and hosting 25% of Kenya's wildlife (KWCA, 2016). In areas like this, where non-consumptive ecotourism is profitable, consumptive utilisation of wildlife could be a competing interest. For instance, in the greater Maasai Mara, community and private land surround the National Reserve and wildlife utilizes public, private and community land (RU7, RU1 I4; Ogutu et al., 2016). Conflicts may arise between actors, including landowners, protected area authorities, and tourism operators regarding wildlife ownership, cropping, and trade. Such conflicts may restrict the potential of wildlife cropping to areas away from National Parks and Reserves, and into private, and community lands far from government-protected areas (O1).

Equality between actors is also an important consideration where tensions might arise between community landowners (I4) who are directly dependent on their land and also wildlife, for their income (A8), and the more productive land of large scale private ranchers with often more profitable ecotourism, livestock, and agricultural businesses. Failing to consider such tensions will ultimately hinder success (O1; Pasmans and Hebinck, 2017), and could result in reduced tolerance or even resentment towards wildlife in community land (O2; Josefsson,

2014). On the other hand, due to their higher productivity, private lands are also the most vulnerable to conversion to other non-wildlife compatible land use options (S1, RS5, O3). Therefore, denying the opportunity to offset wildlife-related costs by allowing consumptive utilisation may hinder broader conservation goals in some of these areas (du Toit et al., 2017). Careful prioritization and evaluations of these trade-offs are, therefore, a significant consideration.

Game farming typically induces fewer conflicts between actors because of its large production yields, similar to those of agricultural, and because they are managed and owned by individuals in much smaller areas where perceived ownership of the resource unit is clearer.

Many of the same equity issues apply to the trade in wildlife. Unlike wildlife cropping, however, the high infrastructural costs related to meat and wildlife products which may act as a barrier for community landowners, are not required, which lowers the barriers to wildlife trade (RS4, A2).

3.1.5. Contribution towards national goals
3.1.5.1. Food security.
The argument that wildlife cropping could be adopted in Africa to increase food security goes back to the 1960s and 1970s (Macnab, 1991). Much of the argument for wildlife cropping as a form of food security emphasizes the economic and ecological viability of wildlife over livestock in semi-arid and arid areas (RU2, RU5). However, livestock systems are much more efficient for meat production for a variety of ecological and economic reasons (RU2; RU5; see Macnab, 1991).

Additionally, the scope of game ranching may be limited spatially across the country, due to many of the reasons discussed in this article. In Kenya, where livestock already represent the vast majority of rangeland biomass (Ogutu et al., 2016), food security would be better addressed through provisions to increase the efficiencies of livestock systems, rather than the exploitation of game meat. Furthermore, for Kenyans to eat game meat, they would have to want to eat it (many traditions in Kenya forbid this, A6), and it would have to be affordable, which would drive down the economic value of wildlife to landowners (RU4), potentially resulting in land conversion to more profitable alternatives (see Section 3.1.2; O1, O2).

It is possible that game farms, with their focus on intensive production, could potentially support goals of food security and economic development (O1; Macnab, 1991). These would be input-heavy industries with other potentially harmful ecological impacts, in the same vein as intensive livestock production.

3.1.5.2. National economy.
If implemented successfully and monitored effectively, wildlife cropping and/or trading would result in increased revenue generation to landowners who host wildlife, other sectors which generate value from this sector, and the national government (Fig. 2; GS1, RU4, O1). However, evidence suggests that returns on consumptive utilisation can be highly variable (Barnes, 1998). Revenue generation would depend on taxation structures, legislation, markets for game products, and the scale of production available for consumptive wildlife utilisation systems (RU5; see above).

If consumptive utilisation of wildlife does not provide financial revenue to the national treasury (either through taxation, employment etc.; O1), it will likely be side-lined in national development plans (S1), impacting the long-term sustainability of these systems (O2).

3.1.6. Logistical, infrastructural, and regulatory sustainability
3.1.6.1. Setting offtake quotas.
A critical tenet of the wildlife cropping model on game ranches and game farms is that wildlife will be sustainably harvested (I1), to ensure the long-term viability of wildlife populations (O2). Despite evidence for sustainable harvesting of wildlife in other regions (such as South Africa), determining levels of wildlife offtake, and setting operational (GS5), and monitoring and sanctioning (GS8) rules is a contentious issue within game farming (Georgiadis et al., 2003). Long-term population monitoring programs

are one of the most informative approaches to provide baseline information against which any harvesting effects on game ranches and farms can be monitored, and quotas decided (Weinbaum et al., 2013). High-quality data are needed to inform quotas (GS8), which requires expertise and financial capital (A2, A7, A9, I9, I10). The closest equivalent is the Department for Resource Surveys and Remote Sensing (DRSRS) who have conducted aerial counts across Kenya for the last 50 years (see Ogutu et al., 2016). However, these data are patchy in both time and space; they are poor for monitoring smaller-bodied species (Buckland et al., 2001); they do not account for demographic parameters needed to set quotas (Ginsberg and Milner-Gulland, 1994); and their outputs do not always reflect the relevant spatial and temporal variation in wildlife abundance needed for quota setting (see below; O2).

During the previous wildlife cropping program in Kenya, quota setting was a major problem. Several methods were used to count wildlife; there were disputes about who should conduct the censuses (I4); there was a lack of scientific basis for setting and allocating quotas (I1); there was little enforcement of quotas (see below, van Kooten et al., 1997; Tasha Bioservices, 2001). Subsequent evaluations of the previous wildlife cropping program found that in Laikipia County, when incorporating environmental variability (RS7), wildlife counts (RU5) and demographic parameters (RU2), the approved quota (I1) of 15% of the population was not sustainable (Georgiadis et al., 2003).

Any quota setting should be based on rigorous and species-specific scientific data, build in uncertainty, allow for adaptive management capacity (GS5; Ling and Milner-Gulland, 2006), and acknowledge the complex dynamics of wildlife populations (Weinbaum et al., 2013).

Given the Kenyan context, implementing harvest quotas would be easiest on some private ranches with high-quality wildlife count data (e.g. Ogutu et al., 2017; A7) and with the financial resources to continue monitoring and quota setting (A2). This would be made easier by the fact that many private ranches are fenced, with clear boundaries (RS2).

Similar consideration has to be given to the trade of wildlife between areas in a game ranching context. However, the lower volume of live trade between landowners (RU5) over cropping, could ensure its sustainability.

Intensive game farming could be sustainable on a smaller scale, where wildlife effectively becomes a private good, with easier monitoring, management, and regulation. Such game farms effectively function in a similar way to commercial livestock farms (Bulte and Damania, 2005; Mockrin et al., 2005).

In Kenya's current policies, it is not clear which government institution should have the mandate to set quotas on wildlife cropping and regulate trade, both issues where clarity and consent need to be clear (GS1, GS2, GS8, I9, I10).

3.1.6.2. Harvesting process.
The previous attempt at wildlife cropping on game ranches in Kenya faced several harvesting issues which continue to act as barriers (Tasha Bioservices, 2001). First, harvesters cropped beyond quota (I1) to increase short-term gains for themselves (A7), leading to decreasing ecological sustainability in the long term (O2). This was due to a lack of oversight (GS8) or incorrect perceptions of sustainable levels of harvest without adequate knowledge of the SES (A7), as discussed above. Second, a lack of monitoring and oversight (GS8) meant that benefits accrued by the harvester were not always passed on to the landowners (mainly in community land; Fig. 3; O1). Third, the inefficient use of wildlife products meant that the full economic values of wildlife (RU4) were not realised by landowners (O1). Finally, abuse of user rights and unethical culling practices undermined the legitimacy of the process, primarily due to a lack of governance (GS8).

These barriers might be overcome with clear harvesting rules, regulations, and governance structures (GS5, GS6, GS8). In South Africa, practical harvesting techniques have been developed, which both

minimise stress to the animal without compromising efficiency (Hoffman and Wiklund, 2006; Lewis et al., 1997). These techniques could increase the economic return for landowners while improving the legitimacy of the process (O1). Certain species such as zebra, eland, impala, and kudu, are more costly to crop, often involving helicopters (A9), which can significantly increase costs for harvesters (A2), and decrease the economic gain from wildlife for landowners (O1; Hoffman and Wiklund, 2006).

Importantly, in addition to the harvesting procedures, careful thought must be given to the control of wildlife disease (Kock, 1995). Diseases that are not controlled and not monitored may have negative consequences for the health of other animals and humans, and knock-on effects to other SESs (O3). If this happens, the potential loss in the market value of meat could undermine its long-term sustainability (see Section 3.1.2 above). A key mechanism to prevent the spread of disease in South Africa is the requirement that all animals are chilled < 2 h after slaughter (Hoffman and Wiklund, 2006). In many areas of Kenya, especially those far from urban centres (RS9, A4), this would require significant investment in cold storage and transport facilities (RS4), increasing the cost of wildlife cropping for harvesters and landowners. Monitoring and harvesting rules (GS5, GS8) will also require conformity with several different legislations.

Overall, regulation of every step in the process (Fig. 3) will require resources, workforce, and processes. In South Africa, for example, each farm is registered, each harvester is licensed and regulated, the game depot is licensed, and each carcass is traced from the moment of cropping to its sale (Hoffman and Wiklund, 2006). Harvesters, farmers, and distributors are inspected for conformity with the rules and regulations. To ensure this level of conformity, and to ensure positive conservation outcomes, Kenya will require new or adapted legislation (GS6, GS7, GS8), new or modified government agencies (GS1), and finance and expertise to conduct these activities (I9, I10; Cousins et al., 2010). Even in the productive game ranches of South Africa, the regulatory environment is plagued with uncertainty (Kamuti, 2014).

Achieving this kind of systemic change is likely to be difficult under the current economic and political setting. In particular, corruption could hinder transparency at multiple points in the process and prevent the equitable sharing of revenue (S1, S4, S3).

3.1.7. Economic and demographic development

Kenya is in a stage of both rapid economic (S1) and demographic (S2) growth. Policy directives and decisions made now must be viewed in the light of future developments. Geographic areas with the highest opportunity cost of conservation to game farmers and game ranchers are usually those with the highest cultivation and development potential (du Toit et al., 2017). The potential for cultivation coincides with high rainfall and high ecosystem productivity (RS5, RS7). With increasing demands for food production, the cost of forgoing development may not outweigh the costs of utilizing wildlife (O1). Even if wildlife cropping and trade could increase the return on investment to landowners, landowners may not be able to generate enough revenue in the long run to offset the total opportunity costs without other socio-cultural or financial incentives (RU4). On the other-hand, game farming is a more intensive land use, with potentially higher financial returns per unit area, which lowers the cost of foregoing other development and land use options.

4. Discussion

Our analysis used the SESF to identify seven overarching barriers to the successful implementation of consumptive wildlife utilisation in Kenya. These barriers, unless appropriately addressed, could limit the ability for the consumptive wildlife utilisation to achieve its desired ecological and socio-economic goals (Fig. 5). Our analysis also revealed that of all the options that have been suggested, the option with the lowest barriers to implementation is game farming on private or community land (Fig. 5). Although this option might contribute to national goals of economic development and food security, our results show that it is unlikely to offset opportunity costs to landowners who host wildlife and does little to contribute towards Kenya's conservation goals of protecting wildlife populations and ecosystem services. More importantly, it is not clear how game farming's focus on high productivity of a single or few species can contribute to broader conservation goals (Macnab, 1991). Cropping of wildlife in game ranching, on both community and private lands, has a broad range of barriers to overcome if it is to be successful (Fig. 5). Overcoming these would require large scale investment in effective monitoring systems, new regulations, training, market development and research, considerations about equity, and devolved ownership of wildlife.

Although we focus on Kenya, this approach can be applied in many other areas. In this article, we have shown its value when assessing the potential impacts of conservation policies pertaining to wildlife utilisation and the wildlife commons (Smith et al., 2019). The SESF can be useful when developing strategies for the implementation of conservation plans (Ban et al., 2015; Guerrero and Wilson, 2017), but as we have shown here, it also represents an important tool for untangling the complex impacts of conservation policy decisions on social and ecological components of these systems, and how these can ultimately impact intended goals. In so doing, the SESF allows conservation policymakers to avoid some of the pitfalls of using a one-size-fits-all approach to conservation policy formation and implementation.

Furthermore, it allows for a focus beyond the social *or* ecological component – such as the potential economic gain or ecological viability of offtake – which, when considered alone, would not allow for a comprehensive understanding of more complex systems with feedbacks (Brehony et al., 2018). The SESF allows conservation policymakers to conceptualise the interactions between subsystems at and across multiple scales, where markedly different factors can influence the sustainability of conservation actions (Cumming et al., 2015). For example, we demonstrate that for some private landowners the barriers to implementing consumptive utilisation may be low, yet at the scale of a national SES, cross-scale feedbacks between subsystems (such as market constraints and equity) may limit the success of this policy in fulfilling national goals. Indeed, previous attempts at wildlife cropping in Kenya were found to only really benefit large-scale private landowners, whereas most community landowners saw little benefit (Tasha Bioservices, 2001). Such inequality between actors have to be seriously considered as they can result in tensions which could ultimately lead to resentment towards wildlife in community land, and hinder overall success of the policy (Josefsson, 2014; Pasmans and Hebinck, 2017; Martin, 2017).

The SESF also allows conservation policymakers to consider variations between different resource systems and the type of consumptive use, which can help in identifying the factors which are key for a policy to achieve intended goals.

Policymakers are routinely faced with tight deadlines to make important decisions. In our analysis, our transdisciplinary team used evidence and information from numerous sources, interpreted through the SESF, to understand multiple focal action situations resulting from the suggested policy intervention, all under a tight deadline. As discussed, this process revealed important barriers to achieving intended goals, but it was also challenging and time-consuming. Transdisciplinary teams, as in the case of this paper, can help to provide high-quality information, knowledge from a broad set of scientific disciplines, and importantly, perspectives from stakeholder representatives (Bennett et al., 2018). Without this diversity and wealth of experience, the process of utilizing the SESF requires even greater effort (Leslie et al., 2015). However, incorporating more quantitative data relevant to our research questions, such as market surveys, wildlife numbers, and spatially explicit data would have enriched our analyses (Dressel et al., 2018). We suggest that further research is required to explore these individual barriers in greater depth and determine if these
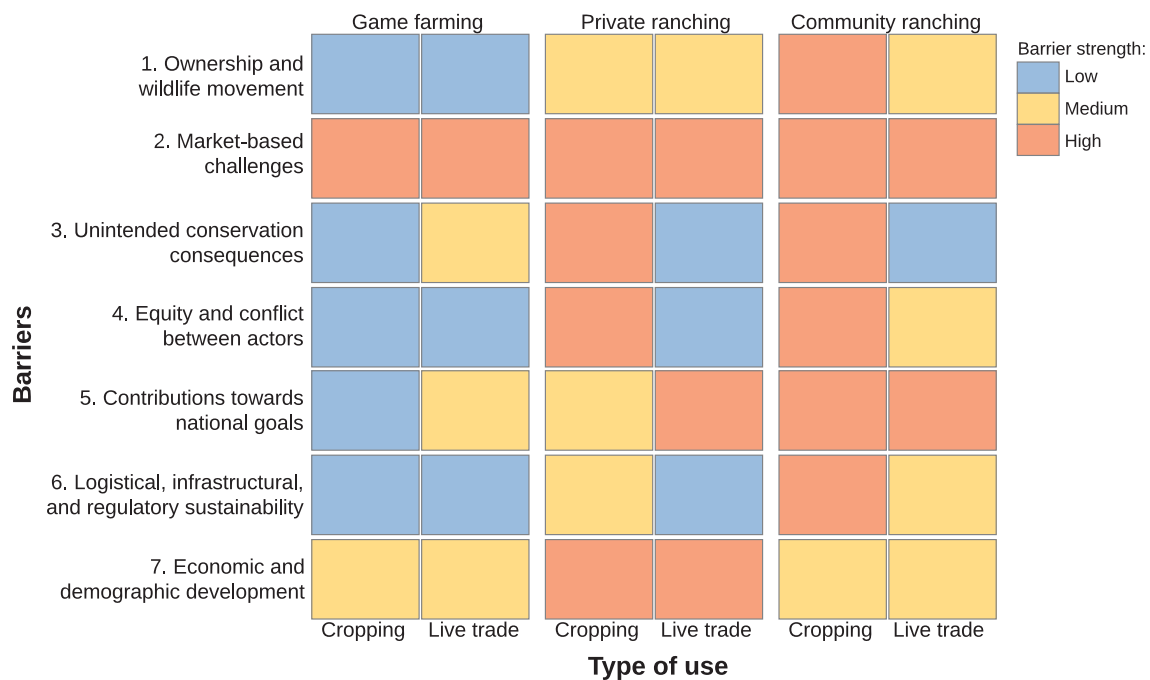
Fig. 5. Summary of the relative strengths of the barriers to consumptive use of wildlife, either trade or cropping, under different land-tenure arrangements.

barriers to sustainability can be overcome.

SES are complex, and our analysis focused on the system and scale that was most relevant to our research questions. However, as Larrosa and Carrasco (2016) point out, even considering a new conservation policy is a process embedded in the SES. The moment the policy is discussed, it becomes part of the SES, redefining it, and therefore affecting all four subsystems, with feedback loops (Fig. 2). Such complexity continues to be a challenge to understand, but building on approaches like ours will help us to untangle the complexity.

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CRediT authorship contribution statement

Peadar Brehony: Conceptualization, Methodology, Formal analysis, Investigation, Writing - review & editing. **Peter Tyrrell:**Conceptualization, Formal analysis, Investigation, Writing - original draft. **John Kamanga**: Conceptualization, Formal analysis, Writing - review & editing.**Lucy Waruingi:**Conceptualization, Formal analysis, Writing - review & editing.**Dickson Kaelo:**Conceptualization, Formal analysis, Writing - review & editing.

Declaration of competing interest

PT, JK, LW, and DK are engaged in conservation in the region. However, the authors have no affiliation with any organization with a direct or indirect financial interest in the subject matter discussed in the manuscript.

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