UNIVERSITEIT VAN PRETORIA
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Wildness as a science-policy bridge for a better world

by

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Submitted in partial fulfilment of the requirements for the degree

Doctor of Philosophy (Wildlife Management)

In the Faculty of Natural & Agricultural Sciences

University of Pretoria

Pretoria

27 March 2024

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Thus freedom always came nibbling my thought,
just as—often, in light, on the open hills—
you can pass an antelope and not know
and look back, and then—even before you see—
there is something wrong about the grass.
And then you see.

That's the way everything in the world is waiting.

William E. Stafford (1999), *A Message from the Wanderer*

Declaration

I, Matthew Child, declare that the thesis, which I hereby submit for the degree Doctor of Philosophy (Wildlife Management) at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution.

SIGNATURE:

X

DATE:

27 March 2024

Acknowledgements

I firstly want to thank my supervisors, Prof. Michael Somers and Dr Harriet Davies-Mostert. Both have greatly assisted in the original project design, advised on data analysis, manuscript preparation and have been immensely patient with me over the years as I have slowly brough this to completion. The topic of this thesis has changed significantly since it was first registered and your encouragement in allowing me to explore the topics that fascinate me and morph the ideas into what they are now is immensely appreciated. This thesis probably wouldn't have happened without Dr Hayley Clements and Dr Alta de Vos for 'sorting out my PhD' (!) by generously sharing their experiences and advice on how to get this done, as well as the many hours spent discussing the wildlife economy – in workshops, in the field, over a glass of wine – that shaped many of the ideas presented here. I am grateful for the dynamic working group we have put together and look forward to manifesting these ideas over the years to come. A huge thank you also to Dr Andy Taylor, firstly for his pioneering survey on the wildlife ranching sector, on which many of the analyses in this these are based, and which inspired the Sustainable Wildlife Economies Project that was subsequently launched; and also to his crystal-clear thinking and matter-of-factness that has served as a perfect counterpoint to my wild all-over-the-place thinking and led to enriching discussions and manuscripts on the wildlife economy. Thanks for being such a champion throughout the ebbs and flows of documenting these phenomena. This thesis also would not have been possible without the support of the Endangered Wildlife Trust and Dr Harriet Davies-Mostert for allowing me to access the wildlife ranching dataset.

To Sarchen, I will always be so deeply grateful for your unwavering support and love during this time. It has not been an easy road, and the stress has been difficult to deal with at times but your calmness, understanding and advice have meant so much to me and helped me stay the course. To my son Harley, thank you for being such a joy in my life and reminding me every day why I went down this path and what's at stake. To my parents and my brother, thank you for enduring this crazy son and sibling – one false start and one staggered finish. I appreciate all the love and support you've always provided.

For Chapter 2, I thank Alex Child and David Williams for significantly improving the initial manuscript through their insightful comments. I thank three anonymous reviewers for their equally constructive suggestions. I am grateful to Sarchen Hough for her help in designing the

figures. The University of Pretoria and the South African National Biodiversity Institute provided financial support that helped me to write this manuscript.

For Chapter 3, I thank the landowners for participating in the survey and acknowledge the South African National Biodiversity Institute, the Department of Forestry, Fisheries and Environment, E Oppenheimer & Son and De Beers Group of Companies, and the Endangered Wildlife Trust for funding the national Mammal Red List project. I thank Michael Somers and Peter Goodman for comments on an initial draft. Michele Pfab, Dean Peinke, Brent Coverdale and Daan Buijs participated in the expert workshops. Sam Ferreira also provided helpful suggestions on the framework. We thank Flippie Cloete for allowing us to use the game auction price data. I also greatly appreciate five anonymous reviewers for their insightful comments and suggestions, which significantly improved the manuscript.

For Chapter 4, I thank the landowners for taking their time to complete the survey and to Dr Conrad Katz from the Bontebok Breeders Association for coordinating the questionnaires submission with their members. I thank Sinozuko Silanda for her help with data preparation.

For Chapter 5, Norma Malatji kindly helped extract the cadastral data for the farms. Fahiema Daniels extracted potential communal rangelands from the former homeland data. We thank the landowners for participating in the survey.

For Chapter 6, I thank the Presidential Employment Stimulus Programme for funding this project and providing the opportunity to train and hire young scientists. I thank the land reform beneficiaries and community representatives for their time conducting the interviews. Special thanks go to team Aloes and Meerkats of the Sustainable Wildlife Economies Project, who interviewed the beneficiaries and CPAs and digitised the data.

Data-sharing

I used two main datasets for the study. The primary dataset was a wildlife ranch questionnaire survey conducted by DrI Andrew Taylor at the Endangered Wildlife Trust (EWT) (details in Taylor et al. 2015), where authorisation was granted through a Research Associate Agreement. The bontebok owner questionnaire survey was conducted by Dr. Jeanetta Selier at the South African National Biodiversity Institute (SANBI) (with support from me and Dr. Andrew Taylor) in conjunction with the Bontebok Breeders Association, where authorisation was granted from SANBI (see *Ethics*). Dr Hayley Clements kindly provided supplementary data to Chapter 5. Additionally, survey data for new market entrants contained in Chapter 6 are from the Sustainable Wildlife Economies Project (SWEP) coordinated and curated by SANBI and originated by myself, Dr Alta de Vos and Dr Hayley Clements, who personally conducted many of the interviews, along with a number of other collaborators. All remotely sensed datasets are freely available and were harvested by Dr Zander Venter from Google Earth Engine.

Ethics

There were no methodological procedures involving animal contact. The use of the bontebok survey data was approved by the University of Pretoria Research Ethics Committee under the project number (NAS120/2020).

Funding and publications

This study was partially funded by the University of Pretoria Postgraduate Bursary (2016, 2023), and South African National Biodiversity Institute (SANBI) Further Studies Bursaries (2017, 2018, 2023). SANBI provided special leave to assist me in writing this thesis.

The following manuscripts have been published or submitted for review. Published manuscripts have not been included in this thesis verbatim and have been edited and formatted to fit with the rest of the thesis.

Child, Matthew F. "Wildness, Infinity and Freedom." *Ecological Economics* 186 (2021): 107055. https://doi.org/10.1016/j.ecolecon.2021.107055. [based on Chapter 2]

Child, Matthew F., SA Jeanetta Selier, Frans GT Radloff, W. Andrew Taylor, Michael Hoffmann, Lizanne Nel, R. John Power, Coral Birss, Nicola C. Okes, and Michael J. Peel. "A Framework to Measure the Wildness of Managed Large Vertebrate

Populations.” *Conservation Biology* 33, no. 5 (2019): 1106–19. https://doi.org/10.1111/cobi.13299. [based on Chapter 3]

Child, Matthew F., Siviwe Shwababa, Alta de Vos, Naledi Mneno, Hayley S. Clements. “Unlocking opportunities for meaningful participation of land reform beneficiaries in the wildlife economy.” *Land Use Policy*. In review. [based on Chapter 6]

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Overall abstract

The world is losing its wildness – in the form of both wilderness areas and the ecological functioning of wildlife populations. While most literature investigates this loss as the negative consequences for biodiversity conservation and global conservation targets, especially through area-based conservation interventions, a more productive framing asks how the loss of wildness impacts socio-economic systems and how ‘rewilding’ can enhance society beyond the obvious benefits to biodiversity. In this thesis, I explore the concept of wildness as a ‘boundary object’ that can connect different stakeholders under a common conceptual paradigm and operate as a key science-policy-practice interface.

Wildness does not exclude human influence or activity but positions the autonomy of ecological systems and human domination of systems on opposite ends of a spectrum. However, conservation science, and public perception of conservation, is trapped by a false dichotomy between intrinsic and instrumental value (largely mirroring non-human and human value), which has produced perverse policy and socio-economic feedback. Conservation policy and assessments, mostly originating in the global north, are narrowly centred on intrinsic values of biodiversity and present negative framings through messages of ‘extinction risk’ and ‘wildlife decline’ that presents wildlife and wildness as fragile entities not to be disturbed by humans. Such framings hinder socio-economic value creation and alienate conservationists from the other major counterfactual land-use – agriculture. Wildness transcends this dichotomy by integrating the innate ecological processes of wildlife and functioning ecosystems (intrinsic value) into effects and services that generate both individual well-being and socio-economic returns (instrumental values). I map potential pathways of how wildness and rewilding connect conservation value to these various dimensions of societal value across scales, using both expert-elicitation data and empirical data from surveys of private wildlife managers in South Africa, and contextualise these pathways within the wildlife economy.

Beginning at the broadest scale, I discuss the various definitions of wildness and rewilding and construct a definition that articulates their boundary dimensions between conservation and policy, psychology and spirituality, agriculture and economics. I then construct a conceptual model, based on a literature review of nature’s contributions to people and capability theory from the social sciences, which demonstrates the infinite value of wildness to individual well-being and psychological functioning, focussing on wildness as a process rather than a pattern

or place. I then translate wildness as a process into potential policy design and decision-making tools. Using the South African policy context as a case study, whilst drawing on my experiences in compiling the 2016 national Red List of Mammals of South Africa, I design a framework to measure the wildness of privately managed wildlife populations as a step towards mainstreaming local-scale assessments of conservation value into national assessments and market-based incentives; and use this framework to inform a potential wildlife economy certification scheme. I then apply this framework to privately managed Bontebok *Damaliscus pygargus pygargus* subpopulations to assess the prevalence of ecologically functioning herds on private lands and the potential for these assessments to unlock economic value through lowering the technical barriers needed to demonstrate conservation enhancement under trophy hunting regulations. Collectively, these results demonstrate the extent of wildness of private ‘working lands’ in South Africa, driven by socioeconomic conditions, where biodiversity conservation is an outcome rather than a primary goal. Simultaneously, they demonstrate the limitation of current conservation legislation and regulations based on Global North aspirations and paradigms. As such, I then assess the potential for wildness and rewilding to interface with multilateral agreements that focus on land restoration and sustainable land management as well as national agricultural policies that seek to improve land productivity and ecosystem condition. Specifically, I assess the impacts of converting from cattle farming to wildlife-based land-uses (WBLUs) on vegetation productivity dynamics at a national scale compared to counterfactual land-uses (commercial livestock farming, communal rangelands and formally protected areas), where I find that the rewilding of indigenous herbivore species significantly improves residual grassy productivity over time and slows the rate of woody plant growth, demonstrating the efficacy of rewilding in combatting bush encroachment in African rangelands and enabling the development of inclusive, production-based enterprises. I also find that increases in vegetation productivity correlate with increased profitability of WBLUs as does the use of fire, which underscores the importance of rewilding programmes to include restoration of natural disturbance regimes. Considering that climate change is making many rangeland areas less and less suitable for livestock farming, these results corroborate previous studies and opinions that WBLUs are an ecosystem-based adaptation to climate change. Importantly, my results also suggest that WBLUs function as a nature-based solution for climate change in that restoring ecosystem functioning is likely to actively sequester carbon dioxide into soils, which I explore further in a literature review of the impacts of rewilding on soil carbon sequestration. Finally, using survey data from new market entrants to the wildlife economy, I investigate the barriers to wildlife economy enterprise development and find there is conflict between the environmental and agricultural mandates, both conceptually and operationally, and argue that a pragmatic view of wildness and wildlife is needed to start ‘rewilding the commons’. Taken together, I conclude that rewilding can be reframed as a tool to improve agricultural productivity and enhance the resilience of rural production landscapes by diversifying revenue streams, but that mixed farms (combining wildlife and cattle or cultivation) should be seen as a strategic novel ecosystem and not just a transitionary land-use to wildlife-only protected areas.

Considering this study's results, several themes emerge from viewing wildness as a boundary object with implications for wildlife economy development in African rangelands. Firstly, I suggest conservationists move away from a paradigm of ‘protecting and preserving wildlife’ towards a recognition of wildness and wildlife as assets in working lands that should be deployed to achieve positive socio-economic and social-ecological outcomes. This requires a shift in thinking from the vertical, top-down approach (current normative species-level assessments and regulations) to a horizontal, bottom-up landscape approach based on rewilding as restoration in novel ecosystems and an understanding of socio-economic systems as drivers of rewilding. Secondly, linked to this, is the need to transcend assessments of conservation value through Global North tools like the Red List, OECMs and non-detriment findings (under the Convention on International Trade in Endangered Species of Wild Fauna and Flora) to more proactive and locally relevant evaluations of wildness. The extension of ‘protected area thinking’ (species-level assessments, focus on static biodiversity features, emphasis on long-term legal guarantees) into working lands and commons is counterproductive in not understanding or recognising the drivers that lead to biodiversity conservation outcomes and feeds into public misinformation that in turn influences policy that negatively impacts sustainable use. Rather, seeing wildness and rewilding as inherently creative processes in novel ecosystems will free decision-makers to make more forward-looking policies and design market-based incentives that speak to impact investors. As such, thirdly, we must develop appropriate monitoring and evaluation frameworks to mobilise policy-relevant data. There is little evidence for the long-term impacts of rewilding on ecosystem and socio-economic system dynamics, and we lack fundamental information on the wildlife economy such as ‘what is it?’ (identification of viable business models and their impacts) and ‘where are they?’ (mapping the spatial extent of the sector and landscape-scale impacts), which leads to poor policy design as we cannot articulate the trade-offs or synergies between biodiversity, job creation and land productivity at multiple scales. Partnerships with private landowners are needed to document long-term dynamics, which must be facilitated through unlocking economic opportunities of wildlife. While the world is a significantly less wild place today than before the advent of neo-liberal market capitalism, restoring wildness, as a creative and unending process, provides hope that we can transition to ecological mindfulness and sustainable economic systems. “A ghost wilderness […] hovers around the entire planet,” wrote the poet Gary Snyder. It’s time for that ghost to guide us through the Anthropocene.

Chapter 1 Introduction – wildness as a unifying concept in the Anthropocene

“A howling wilderness does not howl, it is the imagination of the traveller that does the howling.”

Henry David Thoreau [The Maine Woods, 1864]

“An atom at large in the biota is too free to know freedom; an atom back in the sea has forgotten it. For every atom lost to the sea, the prairie pulls another out of the decaying rocks. The only certain truth is that its creatures must suck hard, live fast, and die often, lest its losses exceed its gains.”

Aldo Leopold (1968 p. 107)

1.1 A brief history of wildness

Henry David Thoreau famously stated, “in wildness is the preservation of the world” (Thoreau 1862 p. 665). While this adage has become an inspiration for today’s environmental movement, it suffers from two conceptual problems: firstly, it conflates wildness with wilderness through the word ‘preservation’ rather than conservation and the allusion to an ‘out there’ devoid of human influence. This sentiment has fuelled the philosophy of many influential conservation groups, such as the Sierra Club, that aim to preserve wilderness areas in their ‘pristine’ form and focus on the preservation of charismatic fauna (Haila 1997; Watson et al. 2018; Di Marco et al. 2019; Lele 2021; Pascual et al. 2021); and has also led to the interchangeable use of wildness and wilderness in the scientific literature (for example, see Mallon & Stanley Price 2013; Watson et al. 2018; Pérez-Hämmerle et al. 2022). This has significant impacts on both international and national policy levels as the primary conservation intervention focus has been on protected area expansion, with the recently completed 2030 Biodiversity Framework including the headline goal of protecting 30% of the earth’s terrestrial habitats, with even more ambitious calls of protecting half the earth (Dinerstein et al. 2017).

While well-intended, there are critical questions around protected area effectiveness (Coad et al. 2015; Geldmann et al. 2018, 2019; Clements et al. 2023), protected area financing, especially if relying on ecotourism (Coad et al. 2019; IUCN ESARO 2020; Lindsey et al. 2020, 2021), and the social justice of setting aside land in the global south while communities surrounding the reserves struggle to eke out a living and risk their livelihoods co-habiting with wildlife (Hulme & Murphree 1999; Agrawal & Redford 2009). On the other side of the spectrum, 'the new conservation', spearheaded by The Nature Conservancy, seeks to mainstream biodiversity into business through financial instruments such as payment for ecosystem services, carbon markets and green bonds (Kareiva & Marvier 2012; Doak et al. 2014; Marvier & Kareiva 2014). Critics of this approach argue that conservation cannot be achieved using 'the master's tools' – i.e. the economic system that has led us to ecological collapse cannot be relied upon to self-correct. For example, Spash (2022) argues that "a core problem with the 'pricing Nature' approach is recognising the implications of corporate power, backed by government, for how actual markets operate and the role prices play….[and] a major ethical problem is allying with those who maintain power and wealth through exploitative practices". On a more technical level, it is difficult to compartmentalise, package and sell aspects of biodiversity, which is innately complex and multi-scale, and ensure that there is no net loss in biodiversity. For these reasons, many argue that neoliberal economics will accelerate the biodiversity crisis. For example, Meyer (2006 p. 76) writes, "economists tell us to wait for the Kuznets curve, which does little more than assure us that everyone will lament the end of the wild hundreds of years from now". One of the reasons why conservation has been so easily allied with the system that is creating the problems is that we do not have an alternative vision, or coherent concepts that can inspire a different view of living. However, anti-neoliberal philosophies, such as convivial conservation (Büscher & Fletcher 2019), fail to gain traction in economic decision-making and so will not be mainstreamed until there is a tipping point in societal values. So, where does that leave the conservation sector?

The new frontier of ideological conflict in conservation has morphed into a simple dichotomy between intrinsic and instrumental value. It is now being battled along the faultline of whether conservation can be achieved through neoliberal capitalism or whether the social and economic values for biodiversity to flourish must be axiomatically separated from economic growth (Lele 2021; Moranta et al. 2022; Spash 2022). Conservation interventions are thus seemingly caught between a rock (intrinsic value) and a hard place (instrumental value) (Pascual et al. 2021), between romantic preconceptions and illusions of 'green growth'. However, arguably, the core problem of biodiversity loss is the conception of society as a 'capital accumulating machine' where environmental costs are externalised and shifted to voiceless ecosystems and communities (Otero et al. 2020; Spash 2022), and which results in steady homogenising of the diversity that is generated by wild ecosystems as the material output of human industry now exceeds the sum of all living biomass on earth (Elhacham et al. 2020). This requires a more nuanced understanding of the homogenising machinery and multifaceted valuation of biodiversity so that we can mainstream different values into different processes to effect overall change (Pascual et al. 2021).

Given the current polemic, how should policy-makers in Africa approach conservation and sustainable development in the context of such divergent and entrenched views from the global north? The answer may lie in what Thoreau realised later in his life, which brings us to the second conceptual problem with "in wildness lies the preservation of the world". Thoreau himself struggled to define wildness, ultimately realising that wildness is more of process than a place (Chapman 2004). As such, policy-makers should stop seeing wildness and rewilding as restoring romantic preconceptions of pristine nature, the '*everything that we are not'*, and see these as processes that can bring back life to the land, bring back prosperity and creativity, and ultimately deepen the different dimensions of ourselves. To do this, biodiversity should be seen as fundamental to restoring and sustaining human well-being, both psychologically and materially (Pascual et al. 2017, 2021; Díaz et al. 2018). However, bridging concepts are needed to counterbalance market-based values with relational and non-market instrumental values (Jepson 2022; Pascual et al. 2023). On the instrumental side, what is missing from mainstreaming the economic value of biodiversity is the recognition that it is the intrinsic values of nature that drive its instrumental value. In this thesis, I will argue that wildness is the concept that can unify these two views. I will develop conceptual models of wildness at both societal and population levels and provide evidence for the effects of rewilding on ecosystem scales. I will show that the mantra "in wildness is the preservation of the world" can be true if we define what we mean by wildness and what worlds we are trying to conserve.

1.2 Wildness as a boundary object

The most intuitive way to understand wildness as a concept is to assess how the understanding of rewilding has evolved. While rewilding was originally defined as the reintroduction of large mammals, particularly apex predators, and was originally conceptualised in North America (reviewed in Lorimer et al. 2015; Pettorelli et al. 2018; Carver et al. 2021), it has since been re-defined to encompass ecological processes and disturbances that operate at ecosystem and landscape scales. Perino et al. (2019) define rewilding as the restoration of trophic complexity, stochastic disturbances and dispersal. This definition draws on both resilience and complexity theory. Similarly, Pettorelli et al. (2018) define rewilding as the "reorganisation of biota and ecosystem processes to set an identified social-ecological system on a preferred trajectory, leading to the self-sustaining provision of ecosystem services with minimal ongoing management". Both definitions view rewilding as process-orientated and dynamic, emphasising system-level functioning and sustainability rather than the composition of the species in that system. While these definitions of rewilding are contested by some who prefer to keep rewilding and biodiversity conservation as primary goals with ecosystem service provision as secondary (Genes et al. 2019), the consensus that rewilding is a form of creative ecosystem restoration remains the strongest thread (Sandom et al. 2013b; Torres et al. 2018; du Toit & Pettorelli 2019; Carver et al. 2021; Jepson 2022; Schulte to Bühne et al. 2022a). Restoration in the context of rewilding does not correspond to the traditional definition of restoration, which aims to restore the historically correct 'pieces' of an ecosystem based on benchmarks (Chapman 2006; Jackson & Hobbs 2009; du Toit & Pettorelli 2019), but refers to the restoration of functional diversity and ecosystem processes that enable the ecosystem overall to operate autonomously and of the component populations to adapt to changing conditions. Rewilding thus has less concern for restoring the exact pieces of the system and focuses instead on the ecological functions of reintroductions and management interventions. The difference between restoration and rewilding resonates with the difference between 'naturalness' and 'wildness', where naturalness evokes historical patterns and processes congruent with narrow conceptions of biodiversity preservation but wildness is taken to mean process-oriented values that enabling nature's autonomy in novel ecosystems (Ridder 2007). Rewilding and wildness are thus conceptually interoperable with the 'working lands conservation' paradigm, defined as productive landscapes, such as rangelands, managed primarily for economic gain or subsistence but supporting significant biodiversity especially when managed in ways that mimic natural disturbances (as per the definition of rewilding by Perino et al. [2019]) and maintain habitats (Kremen & Merenlender 2018), and with nature-based solutions that manage novel ecosystems, defined as ecosystems that have been significantly altered in structure and function by human activity but which sustain high levels of ecosystem services and biodiversity and can be more resilient than historical ecosystems

(Hobbs et al. 2009; Desjardins et al. 2019). For example, rewilding can improve soil fertility and erosion control through greater grass productivity and plant species richness, enhance soil carbon sequestration, restore water retention and regulation, contribute to pollination and seed dispersal and add economic opportunities such as ecotourism, wildlife hunting, game meat production and silvo-pastoralism (Smit et al. 2015; Root-Bernstein et al. 2017; zu Ermgassen et al. 2018; Hall 2018; Lehmann 2021; Schou et al. 2021; Corson et al. 2022; Schulte to Bühne et al. 2022b; Wang et al. 2023). As such, rewilding in working landscapes can create novel ecosystems for win-win scenarios at the intersection of development and conservation.

The current definition of rewilding that centres on process and function rather than pattern and place resonates with the concept of wildness itself. Wildness is simultaneously an expression of independence and interdependence – a relational web of 'aliveness' (Ingold 2011; Van Horn & Hausdoerffer 2017; Vannini & Vannini 2019), which is inclusive of – but not determined by – humans (Deckers 2021), and exists on a spectrum from zero (in totally human-controlled landscapes where species are domesticated) to wilderness (where all ecological processes occur unimpeded and there is absolute ecological autonomy) (Aplet et al. 2000). Wilderness is often criticised as being a colonial construct to exclude indigenous people or as a form of elitist recreation (Cronon 1996). However, wilderness is simply the logical end-point on a spectrum of wildness and represents a scale-effect rather than deliberate human exclusion. When there are large, interconnected areas of mostly untransformed landscapes, ecological processes will dominate over anthropocentric ones and you will have wilderness. In this lens, it does not exclude human dwelling but assumes the scale of ecological process and pattern is sufficient to be largely unimpeded. Wildness is an emergent property of dynamic and reciprocal ecological processes between entities and between entities and their environment, a process of becoming rather than any state of being. It is the inimitable property of biodiversity that binds pattern to process in feedback loops of unending creativity. As such, wildness may be the umbrella concept under which both conservation value and eudaimonic well-being value converge (which I explore in Chapter 2). Because wildness emerges from interaction and reciprocity, it engenders relational values: how people identify with, derive meaning from, or attach responsibilities to nature or its components in a cycle where people and nature co-create and give form to each other (Chan et al. 2016). Relational occupy the spectrum between instrumental values (where nature has value because it provides tangible benefits) and intrinsic values (where nature is valuable in and of itself). Relational values rely on wild features as a substrate for 'effective information' – that which leads to action (Farnsworth et al. 2012) –

which are revealed as one moves through an unfolding landscape (Raymond et al. 2018; Lev et al. 2020). Thus, the intrinsic value of the features can only be discovered and converted into relational values within functioning ecosystems that provide a heterogenous patch-mosaic landscape to *move through* and *discover*. Relational values are thus key to redressing the duality between intrinsic-instrumental and preference-based versus principle-based values by embodying the *actual meaning* of human-environment engagements. In this context, wildness represents an ecosystem's transient and shifting manifestation of meaning through the cumulative self-expressions of its features through time and across space.

Moreso, wildness transcends the idea of cultural ecosystem services (CES) associated with the nonmaterial dimension of Nature's Contributions to People (NCP) paradigm. As Hirons et al. (2016) point out, "the challenge facing researchers and practitioners is balancing the promotion of valuing CES to improve decisions and decision-making processes with the risk that the CES agenda promotes an increasingly globalized and technocratic governance tool in the [ecosystem service] framework that could undermine the very values it aspires to protect". This echoes observations that CES and "nonmaterial benefits" winnow away the complexity and multiplicity of ecosystem values to package static and discrete bundles of services for a largely Western, disembodied and affluent consumer group (Kosoy & Corbera 2010; Norgaard 2010; Maund et al. 2020). However, arguably the most important feature of wildness is not the things we know we get from nature, but the things we *don't know* we could get. The concept of *unknownness* is an irreproducible feature of wild landscapes (explored further in Chapter 2) and serves as a 'reservoir of otherness' and mystery, which challenges our preconceptions and opens up our minds to new moral and epistemological dimensions (Child 2011; Naor & Mayseless 2017; Dereniowska & Meinard 2021). Encountering otherness disrupts the coherent 'totalised knowledge' of the world that we use to construct our egos (Dereniowska & Meinard 2021). Being challenged by chance encounters and peak experiences can open hearts and minds to new perspectives. "Nature emerged as the vehicle that revealed and enabled an as yet unknown, unconscious, and even contradictory aspect of self to emerge through concrete embodied experience" (Naor & Mayseless 2017). Knowledge does not have a finite ontology – the more we know does not mean the less there is to know. Rather, the more you learn and know, the more the world broadens out and deepens, the more there is to know, as edges become new dimensions. Thriving, complex ecosystems offer endless edges and fractals for us to follow. *Unknownness* facilitates the freedom *to become*, and to become better by updating our mental models and refreshing our relational values in continuously dynamic and novel ways. The more complex an ecosystem, the more *unknownness* there will be (Dereniowska & Meinard 2021), which means the moral significance of biodiversity is its infinite source of diversity through evolutionary and ecological processes, from which we can continually create ourselves through encountering otherness. Once more, wildness unifies a polemic (similar to the intrinsic-instrumental value false dichotomy) in conservation ethics, being the current dichotomy of values-based ethics (a natural entity is valuable on its own) and virtue-based ethics (humans should have morals *a priori* to protect nature). Conservationists must move away from bemoaning ‘extinctions of experience’ towards a transformation of experience with new natures (Clayton et al. 2017), through rewilding. Wildness is based on phenomenological experience and perception and therefore integrates values and virtues by using the relational values in nature to improve our psychological and moral functioning in a causal loop. Wildness, then, through *unknownness* is the space where culture is created.

Wilderness is a concept now crowded into small corners (Watson et al. 2018), but because wildness can exist at all scales (Schulte to Bühne et al. 2022a), it makes an ideal boundary object for linking systems across scales in nested complex adaptive cycles (Holling 2001). Additionally, as wildness and rewilding cut across all dimensions of NCP, integrating intrinsic, instrumental and relational values, it can be considered a boundary object at the science-policy-practice interface. Boundary objects are concepts or information products that connect and translate ideas between disparate groups of stakeholders (Star & Griesemer 1989). They mediate action across disciplines by interlinking different understandings of an issue under a common scaffold. Boundary objects thus provide space for different interpretations by not over-defining an issue but seeking commonalities in structure that can be redefined and more finely articulated as collaboration deepens. Boundary objects help us understand how different actors (such as scientists, managers and policy makers) with varied interests, expertise and objectives can cooperate on an intervention. Successful boundary objects begin simply but facilitate co-evolution of science and policy through practice. For example, the integration of systematic conservation plans into regulatory and decision-making processes in South Africa required, amongst other factors, shifting away from ecologically-based planning domains in favour of aligning with administrative boundaries and balancing standardisation (of terminology and methodology) with innovation through institutional-level application and incorporation of local knowledge (Botts et al. 2019). Co-production of implementation-focussed plans between scientists and practitioners over the past decade has resulted in a robust community of practice and highly policy-relevant end products (Botts et al. 2019), which has ensured that demand for the underlying science base has been sustained. Similarly, the role of rewilding is to create a policy and practice hook through which to normalise the consideration of wildness in socio-economic and social-ecological systems. This view of wildness echoes the idea that it is a cultural space, stitched together from many disciplines, to revitalise conservation as a social movement or philosophy (Jepson 2022).

Redford et al. (2015) define mainstreaming as "[changing] those policies and practices that influence land uses outside of protected areas as well as to change economic and development decision making by demonstrating the importance of conserving biodiversity for achieving development outcomes". Successful mainstreaming through boundary objects requires, amongst many other factors, establishing the credibility (scientific robustness), saliency (importance to policy-makers) and legitimacy (non-partisan co-production) of the boundary objects (Cash et al. 2003), where increasing one component (i.e. credibility) may diminish the effectiveness of the others. Creating a boundary object will thus often require distilling complex science into relatively simple concepts and targets, information products or frameworks through which demand for information can be sustained and embedded in policy cycles. For example, Dicks et al. (2015) describe how the complexity of pollinator science was distilled into a simple and understandable agri-environment option for farmers to qualify for financial incentives under the Countryside Stewardship scheme: maintain 2% flower-rich habitat and 1 km flowering hedgerow. This option, they calculate, would be sufficient to sustain six common pollinator species. However, they also outline all the sources of uncertainty and improvements to the intervention that could be made once the 'entry point' is consolidated in stakeholder practice and more policy-relevant evidence is produced. Wildness and rewilding could function similarly as conceptual interfaces between disciplines, decision-making bodies and stakeholders at first, which then become more effective at delivering impacts as evidence for the effectiveness of the interventions is generated through co-designed monitoring and evaluation frameworks.

The most important feature of defining wildness as a boundary object is its emphasis on process over pattern through the restoration of ecological processes (Perino et al. 2019), which leads to functional diversity that improves the provision of goods and services, maintains biodiversity and imbues resilience into socio-economic systems. This method of ecosystem management can be contrasted to the 'command-and-control' style of highly intensive management and inputs that can erode system resilience and lead to negative consequences for society (Holling & Meffe 1996). This concept of process over composition and productivity over preservation is also reflected in the agricultural and land management policy space through the Land Degradation Neutrality (LDN) framework, where indicators are designed to measure positive land-use shifts and ecosystem productivity (Cowie et al. 2018). The NCP concept embedded within the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) framework (Díaz et al. 2018), and the LDN framework embedded within the United Nations Convention to Combat Desertification (UNCCD), provide the foundation for rewilding as a boundary object to connect biodiversity to working lands. However, how this links to individual landowner and community enterprises, and ultimately to socio-economic outcomes and values, is missing from current conceptualisations. For example, Torres et al. (2018), while producing a comprehensive framework to assess the impact of management actions to increase ecosystem integrity, do not provide the necessary outcomes of such actions for enterprise viability in working landscapes, and thus do not answer the question “what’s in it for the manager?” In Figure 1.1, I delineate how rewilding might act as a boundary object by integrating the difference categories of NCP and connecting different policy spheres. I identify possible management interventions (though not exhaustively) on a local scale that lead to NCP outcomes, highlighting the pyric-herbivory model of rewilding that is best suited for African rangelands (McGranahan 2008; Fuhlendorf et al. 2009; Capozzelli et al. 2020). This theory of change demonstrates how different stakeholder groups and decision-makers can work together through the lens of rewilding to achieve their goals and provides interface points for knowledge product development and development of appropriate incentives.

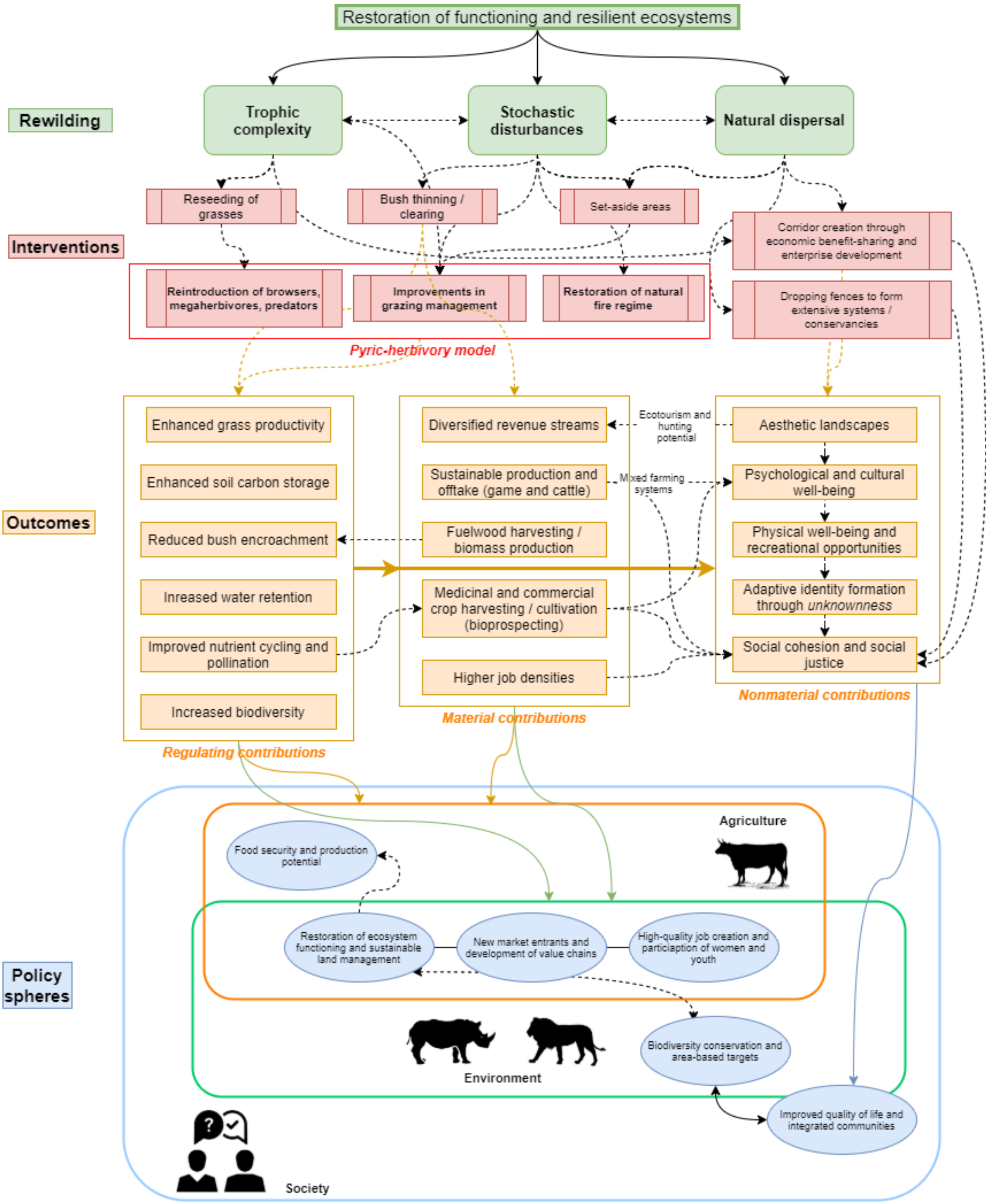

Figure 1.1. Recasting rewilding as a boundary object between stakeholders at different scales. Using the three elements of rewilding as an ecosystem restoration and management paradigm, relevant examples of management interventions at local scales can be mapped to these elements. For example, the reintroduction of indigenous grazers, browsers, megaherbivores and predators, alongside improvements in grazing management (ecological stocking rates, use of kraals, etc.) and the reintroduction of fire as a management tool all correspond to the pyric-herbivory model of rangeland management in Africa. These interventions produce outcomes that can be bundled into the different categories of the IPBES Nature's Contributions to People framework. For example, the pyric-herbivory cluster of interventions will act to increase grass biomass and palatability, reduce woody encroachment, enhance nutrient cycling and water retention, sequester soil carbon and improve soil fertility, as well as enhance biodiversity. These regulating contributions produce material contributions in the form of increased wildlife and/or cattle production and plant products and associated revenue streams, which leads to higher job densities.

Regulating contributions, as well as other primary interventions, such as dropping fences to form conservancies where community members are involved in enterprise development and value chain creation, will create nonmaterial values through landscape aesthetics ('wilderness aesthetic') with the various forms of psychological well-being. Importantly, the types of material outcomes, such as those produced by mixed farming enterprises, medicinal plant harvesting and fuelwood harvesting, promote cultural continuity with local communities and provide resources that contribute to social cohesion and community integration. The different contributions are relevant to different policy spheres. For example, increased food production and improved landscape productivity correspond to the mandate of agricultural institutions. However, through rewilding, there are discrete interfaces between agricultural and environmental institutions, including the production of jobs and new market enterprises through the wildlife economy and the restoration of ecosystem functioning.

1.3 The rewilding phenomenon in South Africa

The history of rewilding in South Africa is ostensibly an incredible success. Across most of Africa, cattle have replaced wildlife, both outside protected areas and within (Craigie et al. 2010; Hempson et al. 2017; Scholte et al. 2022), leading to degradation through overgrazing and the loss of key ecological functions such as nutrient dispersal and cycling. Similarly, prior to the growth of South Africa's commercial wildlife industry, most landscapes had been severely disturbed, and wildlife populations were greatly depleted (Boshoff et al. 2015). However, since the 1960s, wildlife has increasingly gained ascendency over livestock on private land for several mutually reinforcing reasons: more frequent drought conditions from the 1960s combined with the withdrawal of agricultural subsidies in the 1990s has made conventional livestock ranching less viable; wildlife management was decentralised to provincial governments in the1980s allowing private landowners to acquire permits for wildlife use, followed by a pivotal piece of legislation (Game Theft Act of 1991) that fully devolved user rights over wildlife to individual landowners (provided fencing could adequately enclose these populations); and the improved recognition by private landowners that wildlife could be financially viable through the growth in the trophy hunting market and wildlife-based tourism, which then generated associated industries to breed and sell wildlife between landowners (reviewed in Carruthers 2008; Taylor et al. 2015; Hoogendoorn et al. 2019; Bunn et al. 2023).

The outcome of these factors has been that since the 1960s, when wildlife was a minor part of the agricultural and conservation landscape, wildlife ranches and wildlife populations have increased to the extent that land under private wildlife ownership now constitutes 14–17% of South Africa's extent (almost double the formal protected area estate) and hosts an estimated 4.6-7.3 million herbivores (Taylor et al. 2015, 2020, 2021). This is a more than twelvefold increase since 1966, which was estimated at 575,000 herbivores (Du Toit 2007), with higher species richness than formally protected areas on average (Taylor et al. 2021). Comparatively, the total number of cattle in South Africa declined by 13% between 1996 and 2020[1], with non-dairy commercial cattle numbers (i.e. excluding subsistence or communal cattle production) being estimated at 2.5 million head in 2021, which represents a 19% decrease in commercial cattle production since 1985 (DALRRD 2022 p. 58). Similarly, the number of wildlife ranches themselves has grown from negligible numbers in the 1960s to at least 6,734 ranches in 2014 based on registered exemption permits from the provincial management authorities (Taylor et al. 2015), to an estimated 9,000-14,000 wildlife ranches in 2021 (based on updates to the exemption database and including an estimate for non-exempt farms; SANBI unpubl. data). However, the number of commercial livestock farms has stayed largely static over at least a decade, estimated at 13,414 farms in 2007 and 13,639 in 2017 (Statistics South Africa 2020). These data suggest that privately owned wildlife in South Africa now exceed commercial cattle numbers with a steadily increasing conversion of livestock to wildlife ranching farms, although the recent impacts of the COVID19 pandemic may halt or even reverse this trend (Clements et al. 2022), abetted by key policy constraints (explained further below).

The socioeconomic impacts of this large-scale conversion have also been immense. The number of jobs in the commercial agricultural sector has been declining over time (Meissner et al. 2013b; DALRRD 2022). The reasons for this are due to increased minimum wages, consolidation of properties into larger commercial enterprises (Meissner et al. 2013b), rural exodus and abandonment of farmland (Shackleton et al. 2019), the failings of the land reform programme (Rusenga 2020), and the impact of climate change making traditional agricultural activities less viable (Meissner et al. 2013a). In comparison, the wildlife ranching sector has been expanding, often having converted from cattle or sheep farming enterprises, and employs significantly more people both per farm unit and per hectare. For example, Meissner et al. (2013b) estimated that wildlife ranches had almost double the average number of employees per farm unit (7 compared to 4 and 5 employees per farm unit for cattle and sheep farmers, respectively) and average monthly wages (ZAR 30,228 compared to 15,625 and 16,377 for cattle and sheep farmers respectively) in 2010; and Taylor et al. (2020) similarly found that wildlife ranches employed people at double the density of commercial livestock farms (0.008 compared to 0.004 jobs / ha); which has been corroborated in an independent survey led by the

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[1] Livestock census numbers, Department of Agriculture, Land Reform and Rural Development

South African National Biodiversity Institute that showed employment patterns vary significantly by business model[2]. These national estimates are corroborated by smaller scale studies. For example, in the Eastern Cape, it was estimated that converting from agriculture to wildlife increased job numbers by a factor of 3.5 and increased average salaries by a factor of 20 (from USD 715 to USD 4064 per year per employee) (Sims-Castley et al. 2005), which has been corroborated in a more recent study at Amakhala Game Reserve (Achieng et al. 2020). Furthermore, several studies have concluded that wildlife ranching is more profitable and provides higher returns on investment than traditional livestock ranching (Cloete et al. 2007; Chiyangwa 2018; Taylor et al. 2020). On a food security level, many wildlife ranches have also been demonstrated to produce comparable levels of game meat to extensive livestock farms in most biomes (Taylor et al. 2020), thereby contributing to local food sovereignty and, on a national scale, through activities such as trophy hunting and ecotourism. This sector pulls in significant amounts of revenue from international clientele, thereby boosting the general economy of South Africa (Van der Merwe & Saayman 2003; Van der Merwe et al. 2014; Saayman et al. 2018).

The wildlife ranching sector has recently been subsumed under the umbrella term of the ‘wildlife economy’, which is itself subsumed under the banner of the ‘biodiversity economy’. The biodiversity economy in South Africa is defined as “the businesses and economic activities that either directly depend on biodiversity for their core business or that contribute to conservation of biodiversity through their activities” (Department of Environmental Affairs 2016). Similarly, the wildlife economy can be defined as any system that uses wild plants and animals as economic assets to create socioeconomic value that aligns with conservation objectives (adapted from Snyman et al. 2021). Wildlife economies are complex and multi-faceted agro-ecological farming systems, where the primary objective is to run a profitable enterprise across a range of four broad types of activities: 1) animal husbandry of game animals (breeding and live sales), including livestock in mixed farms; 2) legal hunting (trophy and meat); 3) game products (including game meat production) and 4) ecotourism (Taylor et al. 2020) (where they are often referred to as private protected areas (Child et al. 2013; Clements & Cumming 2017b; De Vos et al. 2019). Wildlife ranches are managed along a continuum from intensive to extensive systems, with some c. 50% ranches in South Africa also including

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[2] Denner, C., H. Clements, M. Child, A. de Vos. “Understanding the socioeconomic contributions of wildlife ranching business models in South Africa”. In review, *Conservation Science and Practice*

other agricultural practices (such as livestock farming and/or cultivation). Wildlife assets and economic activities are combined in novel ways to create a diversity of business models (Clements et al. 2016a, 2022). It is this opportunity to combine revenue generating activities, as well as wildlife themselves being more adapted to environmental conditions and less reliant on management, that can give wildlife economy enterprises a competitive advantage.

Yet it is precisely the commercial success of the wildlife economy that has generated criticism, with major spotlights on the 'too-good-to-be-true' success of the phenomenon. The major trade-off with this spectacular increase in wildlife numbers is the steady erosion of the distinction between wild and domesticated. To own and make use of wildlife in South Africa, landowners must obtain a 'certificate of adequate enclosure' to ensure that the animals can rightfully be ascribed to one's land (Blackmore 2020), meaning that almost all commercial wildlife-based enterprises in South Africa are fenced in (Somers & Hayward 2012). Fencing has resulted in more intensive management of populations, including veterinary care, supplementary feeding, protection from predators, and selective breeding for colour variant traits or phenotypes desired for hunting (Carruthers 2008; Taylor et al. 2015; Pitman et al. 2017; Blackmore 2017a; Selier et al. 2018; Russo et al. 2019; Phukuntsi et al. 2022). There is concern that such interventions are 'domesticating' wildlife populations. The traditional definition of domestication involves genetic changes over generations that result from selective breeding to accentuate traits beneficial to humans. While there is some selective breeding of wildlife on private lands, describing these populations as "domesticated" might be an oversimplification (but see Somers et al. 2020) as these breeding programmes generally have not been demonstrated to affect the capacity of wildlife to be reintroduced into other ecosystems and the genetic diversity of selectively bred populations is similar to wild reference populations or higher (Jager et al. 2020; Miller et al. 2023). As such, it is probably more accurately described as 'intensive management' (see Chapter 3), the level of intensity – again – varying by business model (Taylor et al. 2020). Nevertheless, the key property of *wildness* in wildlife has been questioned.

Using the rewilding paradigm of Perino et al. (2019) and the conceptual schematic of Figure 1.1, one can examine whether the claims of 'rewilding' are valid. The first component, trophic rewilding, has ostensibly been achieved with massive increases in species richness of herbivores and megaherbivores and managed metapopulations of lion (*Panthera leo*), cheetah (*Acinonyx jubatus*) and wild dog (*Lycaon pictus*) (Miller et al. 2013; Davies-Mostert et al. 2015; Clements et al. 2018, 2020; Buk et al. 2018; Taylor et al. 2021). However, when examined more closely, many of these populations are kept in enclosures or breeding camps – and thus not integrated into the broader landscape to express their ecological functions – and many are selectively bred or have their reproduction controlled in other ways, such as through contraception (Taylor et al. 2015; Blackmore 2017a; Selier et al. 2018). Additionally, because of the presence of high-value game species on many properties, ranchers often eradicate meso-carnivores and large carnivores to protect game stocks (Taylor et al. 2015; Minnie et al. 2016; Pitman et al. 2017; Pirie et al. 2017b). As such, the trophic levels integrated into ecosystems may be limited or compromised in many ranches. The impact of this throttled management of trophic diversity on ecosystem functioning and ecosystem services is unknown. The fencing issue also impacts the second component of rewilding 'natural dispersal' significantly, as non-permeable fences prevent most large herbivores and some predators from dispersing from the property or into other landscapes. This limitation may curtail one of the key ecological functions of vagile wildlife populations, nutrient dispersal (Hempson et al. 2017), the impact of which is likely to vary with size of property and home range size of the resident wildlife populations (see Chapter 3). Fencing may also hinder species from adapting to climate change (Blackmore 2020; Melville et al. 2021), although the effects of this could be mitigated through management and creation of metapopulations. While interventions exist to increase permeability of game fencing for wildlife species (Weise et al. 2014; Pirie et al. 2017a), these have not been widely adopted, probably due to the lack of economic incentive to do so. Finally, when it comes to the last component of rewilding, stochastic disturbance regimes, the evidence is mixed. Disturbance regimes in African rangelands are driven by herbivory and fire and their interaction (the 'pyric-herbivory' model) (McGranahan 2008; Fuhlendorf et al. 2009; Donaldson et al. 2018). While rewilding presents opportunities for disturbance via herbivory, which crucially includes browsing functions and disturbance by megaherbivores (Roques et al. 2001; Coverdale et al. 2016; Cromsigt et al. 2018; Guyton et al. 2020), the use of fire by wildlife ranchers is still limited. Only 25-38% of managers make use of fire as a management tool (Pienaar et al. 2017) (see Chapter 5 and *Supporting Information 5.4*). Taken together, rewilding in South Africa clearly has its caveats, and full rewilding has not been achieved. However, the outcomes of this, especially in light of possible trade-offs or synergies with socio-economic development, biodiversity conservation and ecosystem restoration, have not been assessed, which presents an opportunity for researchers and decision-makers to collaborate in how the aspects of rewilding can be mainstreamed into different policy spheres at different scales (Figure 1.1).

1.4 South Africa's wildlife economy policy context

Despite the limited understanding of how the 'managed' version of rewilding in South Africa impacts biodiversity and ecosystem functioning, the wildlife economy has become the focus for national-level development targets. Given the positive socio-economic and restoration outcomes of the rewilding, and the potential for wildlife ranches to be considered Other Effective Area-based Conservation Measures (OECMs) and thus contribute to Target 2 and 3 of the Global Biodiversity Framework[3], the Department of Forestry, Fisheries and the Environment (DFFE) has mainstreamed the concept of the wildlife economy into several aspirational policies in an attempt to expand these benefits to previously disadvantaged individuals and communities and stimulate rural development. Starting in 2016, a 'wildlife economy lab', under the umbrella of Operation Phakisa, convened a multi-stakeholder expert group to identify barriers to the wildlife economy and to develop solutions and action plans to unlock growth, job creation and transformation in the wildlife economy (DFFE 2016a). The Lab's participants were drawn from a wide range of public, private, research and conservation organisations. The Lab developed the following vision for the wildlife economy: "A thriving, inclusive and sustainable wildlife economy for the well-being of all South Africans". The Lab also developed a specific aspiration: an inclusive, sustainable and responsive wildlife economy that grows at 15% per year until at least 2030, while providing a foundation for social well-being and maintaining the ecological resource base (DFFE 2016a).

From this, the National Biodiversity Economy Strategy (NBES) was developed with the vision to "optimise economic benefits from the sustainable use of South Africa's biodiversity" (Department of Environmental Affairs 2016 p. 31), including lofty high-level goals of expanding the wildlife economy estate by 10 million hectares and creating 100,000 new jobs by 2030. The NBES also designated Biodiversity Economy Nodes across the country, where

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[3] Target 2 – Restore 30% of all Degraded Ecosystems: https://www.cbd.int/gbf/targets/2; Target 3 – Conserve 30% of Land, Waters and Seas: https://www.cbd.int/gbf/targets/3

investment into the biodiversity economy will be channelled. One such investment stream is expanding the game meat sector, where DFFE has developed a national game meat strategy that highlighted the lack of information as a key barrier to the expansion of the sector and a low-carbon meat industry (DFFE 2021a). The vision of this strategy is “to create an inclusive, industry-driven and government-enabled game meat industry that assures food safety and product quality – dynamically contributing to food security, sustainability and economy growth” (DFFE 2021a p. 11).

In parallel, in an attempt to resolve the issues around ‘canned’ hunting, rhino horn trade and intensive breeding, and the potential biodiversity and socio-economic impacts of intensive management of wildlife in general, along with potentially unethical hunting practices, a high-level panel of experts was established in 2019 by the Minister of DFFE (hereafter the ‘High-Level Panel’ or HLP) to “review policies, legislation and practices on matters related to the management, breeding, hunting, trade and handling of elephant, lion, leopard and rhinoceros”. The resulting HLP report generated 16 consensus goals and 2 additional majority goals (keeping of rhinos in captivity and captive lion breeding and keeping), with a vision of “secured, restored, and rewilded natural landscapes with thriving populations of Elephant, Lion, Rhino, and Leopard, as indicators for a vibrant, responsible, inclusive, transformed, and sustainable wildlife sector” (DFFE 2020b). Importantly, this HLP enshrined wildness as a guiding principle stating that “wildness, near natural areas and wilderness are the foundation of the ecosystem goods and services that sustain human health, fuel the economy, prevent environmental degradation, promote conservation of our wildlife heritage, and provide a competitive advantage for wildlife-based tourism and the wildlife economy” (DFFE 2020b p. 279) and that “domestication of wildlife poses a direct risk to the conservation of wildlife and its above-mentioned value propositions”. Specifically, Goal 3 “*Thriving populations of the five iconic species with localised wildlife economic value chains based on conservation and sustainable use of the five species, linked to state, private, and community wildlife areas, with innovations that can transform rural economies*” aims to increase the ‘wildness’ of the five iconic species and facilitate reintroduction (DFFE 2020b p. 287). Thus, the concept of wildness is being mainstreamed into South African conservation policy. However, the document seems to contradict both the NBES and the National Game Meat strategy in rejecting agricultural practices and agro-ecological rangeland management because of its perceived effect of reducing wildness, labelling practices such as water provision, fire management, bush clearing, mowing, supplementary feeding, lick-blocks, game introductions, contraception, reproductive suppression, translocation, culling, collaring and veterinary intervention as "over-management" to "achieve narrow reserve objectives [which] negates the perception and reality of the sense of place of wildness" (DFFE 2020b p. 303). This contradicts empirical evidence that such management practices, enabled by the Game Theft Act and CAEs, have been responsible for the surge in wildlife numbers and expansion of wildlife habitat (Carruthers 2008; Cromsigt et al. 2018; Taylor et al. 2021). Later on, the document states, "to protect the sense of wildness of South African wildlife landscapes and the iconic species, captive facilities for the five iconic species should be reviewed, with a view to phasing these out; and the domestication and intensive and selective breeding of the five iconic species should be prevented and restricted in legislation and regulation" (DFFE 2020b p. 306). However, again this lacks a systems-level view of the link between rewilding and socio-economic outcomes because a small amount of 'intensive management' may be an economic engine to enable more extensive wildlife areas. For example, intensive breeding camps cover only about 6% of the total sampled wildlife ranch extent (Taylor et al. 2015).

These initiatives represent policy incoherence regarding the wildlife economy, potentially undermining each other through the counter-current drivers they espouse and the goals they want to achieve. For example, regulating for increasing wildness at the expense of the economic value of wildlife breeding and utilisation (Taylor et al. 2020); endorsing ecotourism as a model for new market entrants despite its high capital costs, low viability and high vulnerability to external shocks (Sims-Castley et al. 2005; Clements & Cumming 2018; Clements et al. 2022); and promoting the expansion of the game meat market, and potential for intensive management (Klein et al. 2002), despite animal welfare and wellbeing being unclearly defined in policy. Policy incoherence is driven by a lack of common understanding of wildness and rewilding and how these concepts relate to socioeconomic, biodiversity and land productivity outcomes (Figure 1.1). Despite the documented benefits of the wildlife economy, very little evidence exists on the biodiversity, restoration and socio-economic contributions of the sector in different contexts, hindering effective policy design and implementation. For example, policy-makers lack basic information on the viability and sustainability of different business models within the wildlife economy (Clements & Cumming 2018; Clements et al. 2022). As such, national policies are under-delivering on socio-economic benefits (Mokotjomela & Nombewu 2019; Taylor et al. 2020), not effectively incentivising pro-biodiversity management practices (Clements et al. 2016c; Pitman et al. 2017; Child et al. 2019), and failing to connect with urban markets (both domestic and international) to sustainably produce wildlife products. This undermines the rights and well-being of local communities (Madzwamuse et al. 2020; DFFE 2021a).

The benefit of multiple policies aimed at expanding the wildlife economy is that it presents multiple entry points for mainstreaming wildness and rewilding knowledge products (Table 1.1). Of these, three may be particularly conducive to resolving incoherence: the putative wildlife economy certification scheme, the Biodiversity Economy Investment Platform, and the recapitalisation and development fund from the land reform programme. DFFE is developing a certification scheme for sustainable wildlife economy products in partnership with the wildlife industry, which is one of the solutions identified from Operation Phakisa and is incorporated into South Africa's Biodiversity Finance Plan (DFFE 2018). It provides the opportunity to interface wildness and rewilding with agricultural management paradigms and provides a market-based vehicle to incentivise true rewilding with socioeconomic outcomes. Aligned to this initiative, DFFE is also developing a Biodiversity Economy Investment Platform to expand wildlife economy enterprises and support new market entrants by providing information about the sector and investment opportunities in various projects to financiers and investors (DFFE 2021b). This platform will enable an impact evaluation framework based on the theory of change in Figure 1.1 to guide impact investors looking to invest in rewilding projects. From within the agricultural policy space, while land reform currently focuses on livestock and crop agricultural land-uses, it is increasingly looking to develop agro-ecological systems within the wildlife economy through its Recapitalisation and Development Programme (RADP)[4]. By linking rewilding and the wildlife economy to the generation of ecosystem services, RADP could be deployed to create multi-functional working lands (Clements et al. 2021). The integration of rewilding knowledge products into policy entry points can be structured according to a 'decision to data pathway' (DDP) (Figure 1.2), which outlines the process of identifying decision-making processes and structuring science to support these information flows adaptively and iteratively. This is tantamount to reverse engineering the usual top-down value chain by tailoring the decision support tools, knowledge products and ultimately the datasets specifically around informing the decision-making process. This DDP concept provides a general framework for how to establish wildness and rewilding as boundary objects in science-policy interfaces.

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[4] People and Parks programme interfaces with RECAP

Table 1.1. Policy entry points and potential knowledge products that could be developed to mainstream rewilding into decision-making processes.

Policy entry point	**Implementation path**	**Knowledge products**	**Impacts**
High-level Panel Report recommendations	SANBI-DFFE – stakeholders	Develop business model typologies and associated biodiversity impacts Spatial data products Decision-support tools for new market entrants	Contribute to Goal 9.3.1 (National Policy on Biodiversity and Sustainable Use) Contribute to Goal 9.3.2 (Transformation of sector)
Biodiversity economy node implementation	SANBI-DFFE – stakeholders	Spatial data products	Efficient and effective investment into infrastructure to support inclusive growth within nodes
National Game Meat Strategy	SANBI – DFFE/DALRRD – stakeholders	Socioeconomic and social-ecological profile of hunted species	Incorporate into business model decision-support tools for enterprises to optimize game meat production
Wildlife economy certification scheme	DFFE-private sector-DALRRD	Agro-ecological sustainability standards, combining sustainable land management, wild population management and socioeconomics.	Economic incentives aligned to ecosystem restoration and increasing wildness of managed populations
Recapitalisation funding within Land Reform programme	SANBI – DALRRD – stakeholders	Business profile and investment guidelines	Systemic increase in development of viable wildlife economy enterprises for new entrants
Biodiversity economy investment portal	SANBI – DFFE – private sector	‘Investment ready’ checklist and profile based on empirically derived business models	Efficient and effective investment into individual enterprises to enhance viability
Biodiversity economy satellite account	SANBI to Statistics South Africa	Quantification of economic profiles and value chains	Contributions of wildlife economy systematically mainstreamed into national economic decision-making
2030 Biodiversity Framework	National government to CBD	Other effective conservation measure (OECM) mapping of the wildlife ranching sector	High biodiversity value ranches mapped and incorporated into national biodiversity targets
Framework Strategy for SADC Wildlife-based Economy	DFFE - SADC	Socioeconomic and social-ecological profiles of wildlife economy	Coordinated regional approach to policy formation and investment
IPBES Africa assessment on sustainable use	SANBI/DFFE – IPBES	Quantification of impacts of sustainable use of wildlife	Evidence-based assessments of sustainable use integrated into multilateral policies

In this thesis, by casting rewilding as a boundary object to set a theory of change (described in Figure 1.1), I explore how wildness and rewilding can resolve policy incoherence at multiple scales. At the broadest scale, I present research summarising the effects of wildness on human well-being and capabilities, using Sen's (2013) capability approach, and provide policy recommendations for using wilder landscapes as a tool for societal transformation. I then explore how to measure the wildness of managed populations to provide consistency and standardisation in defining conservation value as a foundation from which to assess possible trade-offs between various policy objectives and land-use types. Specifically, I assess how wildness assessments can be used to unlock economic value from sustainable use and interlink with policy processes at different levels. I then assess the effects of rewilding landscapes on ecosystem functioning and enterprise profitability to find common ground with agricultural policy and the Land Degradation Neutrality framework under UNCCD. I also assess the barriers to rewilding agricultural land through the land reform programme. I use these results to advise on designing new market entrant programmes for expanding the wildlife economy and interface with the agricultural sector under a rangeland ecological infrastructure paradigm. Finally, I develop a theory of change to expand the wildlife economy across Africa by synthesising the results of this thesis. Throughout this thesis, I attempt to answer the questions of why wildness is valuable, what can be considered rewilding, what evidence are we missing, and how does rewilding enhance ecosystem functioning and well-being to make it a viable land-use option in Africa.

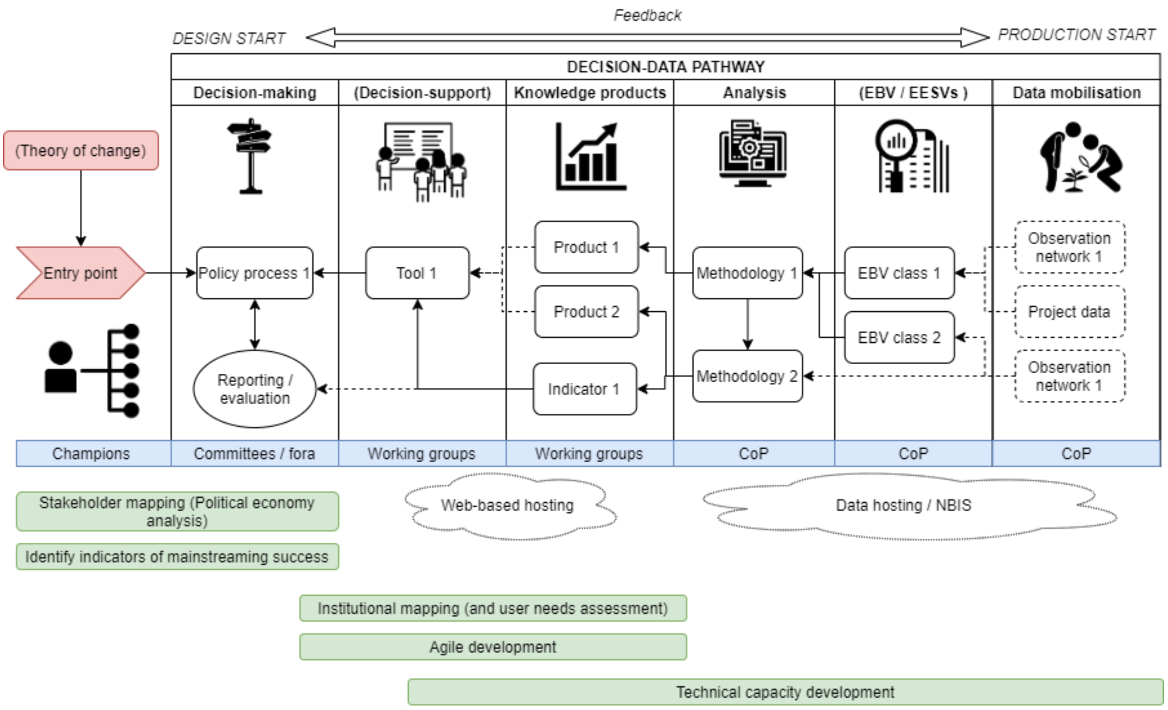

Figure 1.2. A decision-data pathway template to conduct mainstreaming work. It explicitly designs solutions to policy problems by identifying the decision-making process first and retroactively building knowledge products, decision support tools and observation networks that can support that decision-making function most effectively. In this way, the supply of biodiversity data is not ‘pushed’ into policy but rather the demand for data is used to source its supply and design its packaging. This format creates a dialogue between government, academia and civil society. The shaded blue bar indicates the relevant stakeholder group, where “CoP” refers to community of practice. EBV / EESVs refer to the Essential Biodiversity Variable and Essential Ecosystem Service Variables data cube frameworks for standardising analysis-ready datasets.

Chapter 2 Wildness as an active ingredient in societal transformation

I feel like a wet seed wild in the hot blind earth.

(Faulkner, [1930] (2007), p. 58)

1.1 Abstract

Biodiversity risks losing relevance in an increasingly urbanised, unequal and disembodied world. Beyond basic material needs, we might gain the greatest well-being from eudaimonia – the freedom to flourish and live meaningfully. Immersion in nature improve the fundamentals of eudaimonia: psychological, emotional and social health. This presents an opportunity to re-frame biodiversity from a passive entity needing to be saved by 'good people' to a catalyst in the quest to become good. Drawing on the capability approach, I propose that wild landscapes – defined as self-willed, ecologically complex communities comprising functioning ecosystems – are mediums that facilitate individuals' search for meaning. Features of wild landscapes (organisms, habitats, structures) stimulate unique perceptions and experiences that afford the elements of self-meaning (ideas, narratives, memories). Ecological processes (succession, disturbance, dispersal) generate dynamic perceptual experiences, which enhances our ability to comprehend meaning by restoring cognitive functions and relational values. Functioning ecosystems continually create and permute features in space and time, instantiating ever-varying patterns from which to adapt meaning as our contexts and aspirations change. Wild landscapes thus provide infinite value for our freedom to become. As widening income inequality amplifies asymmetric power structures; increasing the agency of those who seek to improve society is one pathway to a sustainable future.

2.1 The search for meaning matters most

Urbanised and affluent individuals drive global consumption, causing waves of environmental degradation, biodiversity loss and socioeconomic inequality that wash around the world (for example, Weinzettel et al. 2013). Although consumerism has colonised the idea of 'the good life' as a linear progression from poor to rich, fuelled by slash and burn production, the effects of material wealth on well-being saturate rapidly after basic needs are met (for example, Barrington-Leigh and Galbraith, 2019; Kahneman and Deaton, 2010). Rather, the freedom to realise one's potential and find meaning in life (eudaimonia) corresponds to lifelong psychological well-being and health (Baumeister et al., 2013; Frankl, [1946] 2004; Park et al., 2010; Ryff, 2017). Self-meaning comprises two primary dimensions: purpose – the over-arching aspirations and life mission that orchestrate one's daily activities and decision-making; and comprehension – the ability to interpret information from your environment and integrate it into the understanding of one's life (Steger, 2012). Purpose and comprehension interact and change adaptively over time in response to shifting contexts and personal circumstances (Child, 2011; Frankl, 2004; Heft, 2013; Kegan, 1982). As such, self-meaning is not a fixed outcome or end state but the process of *becoming* who one is or needs to be. Thus, rather than (over)supplying static (often lucrative) conceptions of 'needs', where individuals are cast as passive consumers or patients to be looked after, we should promote policies that create environments wherein individuals are free to search for unique self-meaning and become 'agents who can do effective things' (Sen, 2013). Individuals who create value and not merely consume it.

Reframing well-being from passive resource accumulation to the meaningful lives that resources may enable is a central tenet of the capability approach (Sen, 1999). Capabilities are the fundamental freedoms that enable us to find and manifest self-meaning, such as being nourished and healthy; being able to think, reason and imagine; and participating in decisions that affect one's life (Sen, 1999; Nussbaum, 2011). Capabilities are constrained or facilitated by personal, social and environmental (dis)enabling conversion factors that influence what individuals are actually able to do and become with their freedoms (their "functionings") (for example, Ballet et al., 2013, 2018). A meaningful life is continually constituted by the set of current functionings that individuals have reason to value, such as 'having self-respect', 'learning a new skill', or more materialist ambitions such as 'buying luxury vehicles'.

Shifting from materialist to 'mindful' functionings that result in prosocial and pro-environmental outcomes is a necessary condition for strong sustainability (Mabsout, 2015). This requires integrating the evaluative spaces of both the capability approach and the sustainable development paradigm (Anand and Sen, 2000; Ballet et al., 2013; Schultz et al., 2013; Pelenc and Dubois, 2020). Capabilities are increasingly being linked to ecosystem services as both resources and conversion factors, where provisioning services are always positive but regulating and cultural services (CES) act as both negative and positive conversion factors (reviewed in Ballet et al., 2018; Polishchuk and Rauschmayer, 2012). However, it remains unclear how capabilities can be functionally connected to the ecological condition of ecosystems such that achieved functionings consistently feedback into conserving the resources from which freedoms are made possible. For example, although Ballet et al. (2018) link cultural ecosystem services to the personal identity capability, they note that different aspirations may result in positive or negative choices concerning nature because people "do not have the appropriate cultural background to 'spontaneously' apprehend the cultural services a natural area can deliver" (Pelenc and Dubois, 2020, p. 36).

Indifference towards pro-environmental behaviour is at least partially due to the inert framing of nature as a service provider consisting of "spatially bound, temporally stable" stocks and flows to supply static user needs (Bekessey et al., 2018; Norgaard, 2010; Pröpper and Haupts, 2014, p. 29). This may create perverse incentives to commodify a particular service that best serves the interests of a dominant user group (Bateman and Mace, 2020; Hirons et al., 2016; Polishchuk and Rauschmayer, 2012; Pröpper and Haupts, 2014), leading to domesticated and ecologically barren landscapes that merely retain the veneer of biodiversity (Gobster et al., 2007; Lev et al., 2020; Truong and Clayton, 2020). Culture is a fluid concept, constituted by the collective search for meaning of individuals who "endlessly spin metaphors […] to weave labyrinthine and ever-expanding networks of symbolic equivalence" (Ingold, 2003, p. 330). Thus, rather than attempt to categorise current cultural configurations as end-points, a more progressive approach might ask what are the features and qualities of landscapes that create the meaningful experiences from which culture is continuously derived.

Sustaining biodiversity could be tantamount to sustaining everyone's capability to find self-meaning. Empirical evidence shows that immersion in nature improves multiple dimensions of eudaimonic well-being, including personal growth, vitality, positive affect, autonomy, cognitive functioning, positive relations with others, morality, and life satisfaction (reviewed in, for example, Bowler et al., 2010; Bratman et al., 2019, 2012; Hartig et al., 2014; Mygind et al., 2020; Russell et al., 2013; Sandifer et al., 2015, Pritchard et al., 2020). Seeking out a meaningful life has been hypothesised as a pathway that mediates the flow of eudaimonia from natural habitats (reviewed in Cleary et al., 2017; Hinds and Sparks, 2011; Lumber et al., 2017), which is supported by meta-analyses that found a significant positive relationship between nature connectedness and eudaimonic well-being (Capaldi et al., 2014; McMahan and Estes, 2015; Pritchard et al., 2020). Greater nature connectedness also increases pro-environmental and prosocial behaviour (Cleary et al., 2017; Lumber et al., 2017; Whitburn et al., 2020). An immutable message emerges from this research: we don't need to be 'good people' to value nature, we need nature to become good people.

Here, I develop a capability model linking the search for self-meaning to biodiversity through the concept of wildness. I define wildness as the self-willed, spontaneous and creative properties that emerge from functioning ecosystems (Cookson, 2011; Perino et al., 2019; Prior and Brady, 2017). Wildness exists on a spectrum of ecological autonomy from domesticated to wilderness and generates dynamic landscape patterns and species assemblages. A self-meaning capability, similarly, is the dynamic interplay between the possibility of purpose and the capacity for its comprehension, instantiated through internalised symbols (such as ideas, memories and stories) and bound together through the making and remaking of relationships (Child, 2011; Heft, 2013; Prior and Brady, 2017; Steger, 2012). Several studies have found that wilder landscapes increase eudaimonia relative to domesticated 'natural' spaces (Hinds and Sparks, 2011; Lev et al., 2020; Wood et al., 2018). The more we search for meaning in wild landscapes, the more sustainable our chosen functionings may become as we practice the "etiquette of freedom" (Snyder, 1990, p. 25) – to understand that our flourishing is interdependent with the flourishing of others; that we can be more without having more.

2.2 Wildness as a medium for self-meaning

Searching for meaning is rooted in evolutionary and ecological processes as all organisms strive to find solutions to changing environments. Cognition, consciousness and environment have thus become entwined through embodied interactions (Garbarini and Adenzato, 2004; Heft, 2013; Varela et al., 2017), extending the boundaries of the self into one's surroundings. As bodies with minds rather than minds attached to bodies, knowledge emerges by "discovering structure" in the phenomenal field of "dynamic, animal-environment reciprocity" instead of the "mind imposing structure on a malleable world" (Heft 2013:163-166). Self-meaning might thus be 'felt' first before filtering into consciousness, as our minds incorporate subjective experiences (Gibson, 1986; Heft, 2013; Merleau-Ponty, [1945] 2013). Embodied cognition implies that the landscapes through which we move (or sit) are an inextricable and fundamental constituent of our minds, with the corollaries being that landscapes with more diverse features to perceive may provide more opportunity to find self-meaning; and landscapes with more dynamic features may sustain our fascination and continuously provide new insights into self-meaning as one's context and aspirations change (Figure 2.1).

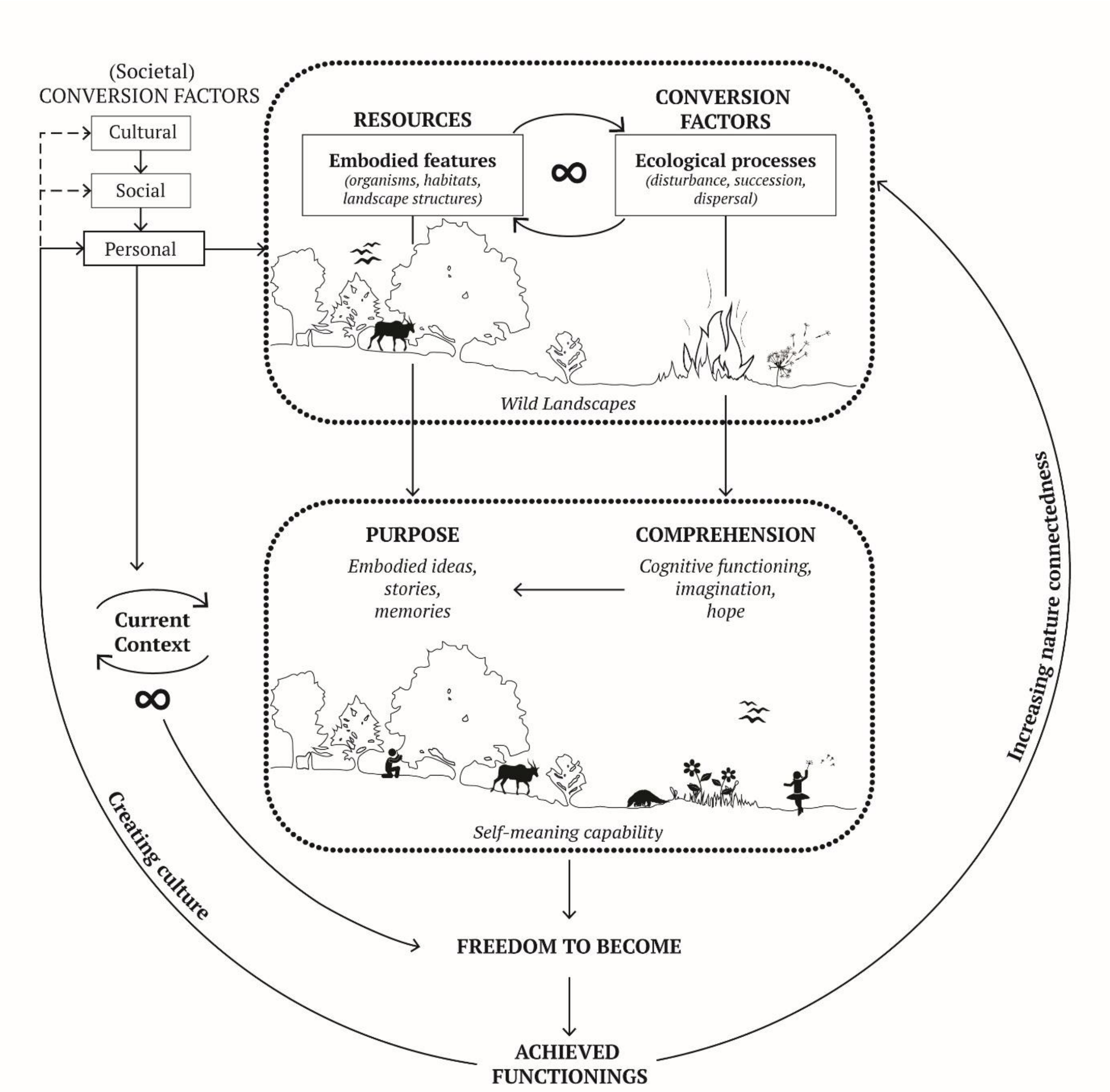

Figure 2.1. Conceptual model (adapted from Ballet et al., 2018) showing the proposed interlinkages between wildness and the self-meaning capability. Within wild landscapes, embodied features (e.g., organisms, habitats, and structures) afford unique perceptions and experiences that can be converted into purpose through embodied ideas, memories and narratives that improve relationships with people and places. Ecological interactions within and between species, as well as landscape-level processes (e.g., disturbance, succession, dispersal), generate dynamic and spontaneous variation that restores attention, sustains imagination and deepens relational values through intersubjectivity. Functioning ecosystems are thus positive conversion factors for comprehending self-meaning. By instantiating self-meaning, wild landscapes enable adaptive 'functionings' necessary for one's current context, such as developing a stronger relationship with your family, gaining the inspiration and vitality to find a more rewarding career, or quietening one's mind to make an important decision. Achieved functionings ultimately feed back into the cultural and social context that act as filters for personal conversion factors (e.g., what is deemed acceptable to do or become), and therefore, wild landscapes play an active role in increasing nature connectedness and shifting societal values towards the prosocial and pro-environmental. Wild landscapes

continually both create (through evolution and environmental change) affordances and permute existing affordances in space and time (e.g., chance occurrences, seasonal flowerings, natural disturbances and patch dynamics). Such ‘unknownness’ provides infinite opportunity for unique self-meaning to be found by each perceiver and drives the search for meaning cycle. As purpose changes throughout one’s life, self-meaning can thus be adapted from within the same wild landscape over time, which means wildness is infinitely valuable for eudaimonic well-being.

2.2.1 Wild features as resources for purpose

Within wild landscapes, the variety of perceptible features are resources that can be incorporated into one’s purpose (Gobster et al., 2007; Bratman et al., 2019; Lev et al., 2020), and include organisms of different species, habitat mosaics and ecotones, and structural elements such as variation in vegetation height and density (e.g., from old-growth trees to scrub). Higher species and habitat diversity positively correlates with aspects of purpose, including reflection (thinking and gaining perspective), identity (the degree of feeling differently in particular places), attachment (degree of emotional connection to places) (Fuller et al., 2007; Passmore and Holder, 2017); vitality (feeling alive, energetic and capable), positive affect (such as joy, interest, alertness) (Methorst et al. 2021); and aesthetic value (reviewed in Tribot et al., 2018). As such, almost all significantly meaningful interactions in nature occur in wild habitats (Lev et al. 2020) – those that are relatively unmanaged and have greater biodiversity.

Wild features embody multiple ‘affordances’ for developing purposes that are unique to the observer and change dynamically in space and time. Affordances are the properties of a feature that enable an individual to perceive and act (Chemero, 2009; Gibson, 1986; Heft, 2013; Ingold, 2002; Raymond et al., 2018). For example, a smile may afford an invitation to engage but also the opportunity to back away, and a fallen log may afford a place to rest for a human, a home for a vole, and a substrate to grow for fungi. Affordances are thus innately relational, defined by each organism’s current contexts and capacities. They generate both *affects* (direct emotional or intellectual responses) and *effects* (transformative experiences). Affects and effects can be converted into symbolic self-meaning by challenging beliefs and behaviours and enabling one to discover hidden aspects of the self (Clayton et al., 2017; DeMares and Krycka, 1998; Naor and Mayseless, 2017), which has been shown to significantly and consistently improve eudaimonic well-being (Pritchard et al., 2020).

One of the most meaningful affects is ‘encountering wildlife’ (Lev et al., 2020). For example, watching an eagle swoop to catch a fish inspired one participant to “feel strong and […] empower[ed] to move forward and open a private practice” (Lev et al., 2020: 7). Every feature in wild landscapes has had to prove itself, and thus every feature has embodied meaning because “the perspective of a challenged and self-affirming organism lays a new grid over the world: a ubiquitous scale of value” (Weber and Varela, 2002, p. 118). Some ideas live in hard places: field mice, lizards, insects – eking out existence on metabolic water, reminders of true grit. Some ideas drift in the wind – seeds, spiders, spores; “a ghost wilderness [that] hovers around the entire planet” (Snyder, 1990, p. 16). Wild landscapes thus comprise a vibrant “communicative matrix woven through with signs and wonders” (van Dooren et al., 2016, p. 2). It’s not that you learn everything about life from observing wild features – a bird in itself isn’t a manifestation of morality – but it may be a trigger that ripples across a web of memories, experiences and internal symbolic meaning. As such, each affordance embodies potential significance, ideas that can structure and expand our understanding of ourselves and of the world, expanding the horizon of potential self-meaning. As Robert Macfarlane (2007, p. 100) noted, “it is valuable and disturbing to know that grand oak trees can take three hundred years to grow, three hundred years to live and three hundred years to die. Such knowledge, seriously considered, changes the grain of the mind”.

Wild affordances act as waypoints that magnetise movement through continual perception-action loops, drawing one into the landscape where new opportunities for experiences and ideas continually unfold (Gibson, 1986; Greaves, 2019; Heft, 2013; Lev et al., 2020). For example, “sitting by a [wetland]” with your child enables other affordances to enter the perceptual realm, leading to subsequent experiences like “watching ducks”, “observing insects” and “hearing owls” (Lev et al., 2020). Perception-action loops depend on the perceiver’s aspirations and abilities. A wetland might be perceived as a place to find a particular medicinal plant for one person, but a site to experience seclusion and contemplation by another. Perceiving leads to acting: while searching for the plant, one might come across spoor imprinted in the mud, laying a mental marker as a place for hunting or perhaps evoking an aesthetic response – ghostly creatures fading into the veld. Or, while contemplating, a strange birdcall might prompt exploring a nearby thicket, maybe eliciting a memory from childhood that helps cohere one’s self-identity. The movement of self-willed features alone drives perception-action loops as it is “both expressive and responsive, and thus open and indeterminate” (Greaves, 2019, p. 16). Wild landscapes thus directly generate knowledge along perception-action pathways where the terrain, being “infinitely variegated” (Ingold, 2010, p. 135), enables manifold and multisensory ways of knowing (Clayton et al., 2017; Ingold, 2010; Merleau-Ponty, 2013). Every wild feature is a degree of our own existential freedom (Collar, 2003), and diminishing wildness diminishes our capability to find self-meaning. “Thought, like memory, inhabits external things as much as the inner regions of the human brain. When the physical correspondents of thought disappear, then thought, or its possibility, is also lost” (Macfarlane, 2007, p. 100).

Effects from affordances are perception-action loops that we incorporate as stories and memories, enhancing both our “internal clarity” as individuals (Cookson, 2011) and our relationships with others, which is a significant factor in the relationship between eudaimonia and nature connectedness (Pritchard et al., 2020). A quarter of all experiences recorded by Lev et al. (2020) were relational, ranging from generating new relationships through perception-action pathways (for example, teaching someone how to forage mushrooms), to deepening existing relationships through conversation that “simply does not happen in everyday life” (Lev et al., 2020, p. 7), to storing memories that protect relationships (for example, one participant associated memories of her mother with a “high meadow that overlooks the bay”; Lev et al., 2020, p. 8). Wild landscapes “require that we learn the terrain, nod to all the plants and animals and birds, ford the streams and cross the ridges, and tell a good story when we get back home” (Snyder, 1990, p. 26). These stories seep into our relationships through the retelling and regaling, while the features themselves soak up and store the memories, releasing them slowly over one’s life to enrich self-meaning.

New experiences of nature, and thus new stories, are made possible when the landscape possesses ecological complexity and unpredictability (Clayton et al., 2017). The number of potential stories increases with the number of affordances in a landscape, and so wilder landscapes with more features are more strongly related to ‘continuity with the past’ (Fuller et al., 2007; Lev et al., 2020). Wild affordances produce immediately perceived place meanings that interweave individual narratives at landscape scales and thus cohere communities through distinct social-ecological systems (Colley and Craig, 2019; Ingold, 1993; Hartig et al., 2014; Raymond et al., 2017). For example, a tree growing in an agricultural field may afford shade, fruit and vantage, which precedes social construction of cultural ecosystem services such as picnicking (relationship building), picking (subsistence harvesting) or preying (sacred groves), and thus “the place was not there before the tree, but came into being with it” (Ingold, 1993:167). Wild affordances do not merely reflect cultural preferences; they create cultures.

As such, ecologically complex landscapes will be essential in cultivating the sense of place and relational values necessary for societies to adapt to the Anthropocene (Chan et al., 2016).

2.2.2 Conversion factors for comprehension

The practice of searching for self-meaning in wild landscapes through perception-action loops feeds back into one's ability to find it (Fig 1.). Comparative and experimental evidence shows that immersion in nature restores attention and mindfulness, reduces stress, enhances creativity and increases task performance (e.g., Atchley et al., 2012; reviewed in Bratman et al., 2012; 2019; Hartig et al., 2014; McMahan and Estes, 2015; Russell et al., 2013). Attention restoration theory describes how directed attention – the process of deploying cognitive energy to focus on selected stimuli while avoiding distraction – is restored in natural landscapes because the affordances themselves are inherently fascinating and elicit involuntary attention, thus allowing neural pathways a chance to recover and improve working memory (reviewed in Bratman et al., 2012, 2019; Mygind et al., 2021). By helping us to be present in our own lives and remember more of it, we might extract and sustain more self-meaning from our experiences.

The wilder a landscape, the more likely perception-action loops will draw you into the field and reveal the asymmetrical and intricate affordances that restore one's attention through softer, effortless perception. While we wander, we experience different scales of perception: from the drift of clouds across a mountain to the rustling of leaves in the canopy to the flicked shadow of a bird. The patterns never exactly repeat themselves; they meander and morph like "the calligraphy of rivers" (Snyder, 1990, p. 71), drawing our attention into unexpected and unpredictable directions. This fractal-like ecological complexity is more restorative to observe than built scenes because of the interwoven layers of subtle information (Van den Berg et al., 2016). Similarly, Wood et al. (2018) and Schebella et al. (2019) found a significant positive relationship between the wildness of urban parks and subsequent improved attention, reduced stress and general positive affect. Furthermore, because wilder landscapes are more likely to generate different habitats, they will increase the likelihood of affording complementary cognitive benefits. For example, the interior of a forest promotes stress recovery most effectively, but exploring the forest edges (which, as an ecotone, generally has higher species richness) best restores attention (Chiang et al., 2017).

Patterns are always changing in wild landscapes because their features are dynamic. Flowers bloom and blow away, animals dip in and disperse, and the seasons have their say. In functioning ecosystems, ecological and environmental processes will always reshuffle the affordances available in a landscape. For example, species disperse seeds to create new habitats, and animals seek new habitats to settle; fire creates clearings where different plants can germinate and gradually 'succeed' one another. Both dispersal and 'stochastic' disturbances (that afford opportunities for establishment) are fundamental to wildness because they enhance ecosystems' ecological complexity, autonomy and diversity (reviewed in Perino et al., 2019). Wildness is thus a process of becoming over-being, of flux over stasis, of dynamism over balance; a source of continual vitality (Vannini and Vannini, 2019). This explains the consistently strong vitalising effects of nature experience (Capaldi et al., 2014; Pritchard et al., 2020), which are mediated by the presence of wild features (Ryan et al., 2010). As patterns shift, emerge and evolve, there is more to notice and, thus, more opportunity for cognitive restoration and gentle sensory stimulation. Because wilder landscapes are more likely to generate perceptible changes through time, our connection to nature will strengthen as we stay fascinated and continue to find meaning (Schebella et al., 2019).

While we experience these dynamics themselves, we also experience others experiencing them. Such 'intersubjectivity' emerges from the collective unfolding of all organisms' perception-action loops in the landscape. Intersubjectivity creates a phenomenal field of multisensory experiences that situates the perceiver as a node within a "domain of entanglement" where relationships are "continually ravelling here and unravelling there" (Ingold, 2011, p. 71). Developing relational values facilitates capabilities concerned with self-identity and self-meaning (Chan et al., 2016; Child, 2011; Ballet et al., 2018; Mygind et al., 2021). For example, a review found that nature helps children's abilities to form positive relationships, social competencies, emotional management and self-expression (Mygind et al., 2021). Intersubjectivity deepens our sense of ourselves by widening our sense of others and otherness (Cleary et al., 2017; Naor and Mayseless, 2017), which leads to 'unselfing' – where one's ego dissipates into the landscape and receptivity to affordances increase as we become more mindful and observant (Zhang et al., 2014). As Macfarlane (2012) described in his re-tracing of the old paths of England, "I felt a sensation of candour and amplitude, of the body and mind opened up, of thought diffusing at the body's edges rather than ending at the skin".

In this state, one's intrinsic aspirations (such as personal development, intimacy, kindness, empathy, love and care) improve and extrinsic aspirations (such as money, image or fame) decrease (reviewed in Cleary et al., 2017). Currently, we are framed as 'stewards' responsible for saving the planet, but this is ineffective because we do not act on values; we develop values from action, and the potential to perceive appropriate action is a function of our environment (Berger and Beckmann, 2010; Ingold, 2002; Lumber et al., 2017). Incentivising a shift to sustainable functionings will thus require strengthening people's perceptive abilities and not simply admonishing why nature *should* be valued (Schebella et al., 2019).

Perhaps most importantly, as much as wildness provides the affordances to know, it also creates the space to *unknow* (Dereniowska & Meinard 2021). In wild landscapes, you're never quite sure what you'll find, and sometimes not sure what you've seen. Shapes morph in and out of messy banks, re-manifest as a sound from somewhere, or a sudden stillness before a snapped-twig twitch. Wildness is always equivocating. Such "[i]llusions [that] drive home the truth that our habitual vision of things is not necessarily right: it is only one of an infinite number, and to glimpse an unfamiliar one, even for a moment, unmakes us, but steadies us again" (Shepherd, [1977] 2019). Through their dynamic and self-willed natures, the affordances embodied within wild landscapes continuously create opportunities for us to 'unmake' and 'steady ourselves', providing windows for new meaning to be incorporated. Furthermore, because each organism chooses which affordances to act on based on its aspirations or abilities, each organism 'creates' its reality as it responds to triggers tucked into the unfolding landscape. This "allows […] one existential reality to reside alongside innumerable, created realities" (Birkin and Polesie, 2013, p. 151; Gibson, 1986). These co-occurring realities are refugia for our imaginations because they show us we are not alone out there – the edge of existence is a nexus. By creating unknownness, wild landscapes resist the homogenisation of experience and sustain existential freedom for all (Clayton et al., 2017). Unknownness cannot be replicated by technology or domesticated green spaces as both serve to diminish dynamic sensory stimulation and standardise ("optimise") experience (Truong and Clayton, 2020),

Because the mystery of what *might be* renews us, unknownness gives us hope. It reminds us that change is always possible, and that we are free to change. Unknownness has the power of breaking our current conceptualisations and thus alter our worldviews, values and outlooks (Dereniowska & Meinard 2021). Monbiot (2013) recounts a story about kayaking off the Yorkshire coast when a storm-swell threatened to push him onto the rocks. At his lowest ebb, muscles exhausted, when he had nearly given up, he saw a flash of chestnut and movement on the shore – a corncrake –rare and out of its usual range. Seeing this frail bird battling the same elements, he felt solidarity, and "as the bird receded up the beach, [..] felt [his] energy surging back”. Wildness is an indeterminate process that sustains possibility through its “immanent potential” (Vannini and Vannini, 2019). Each time we enter a wild landscape, it is a new domain, enabling us to discover increasingly “fine-grained specifications” to self-meaning (Lev et al., 2020). To know that some things are unknown, to know there are landscapes where serendipity, spontaneity and creativity can spark something in you, gives a sense of limitlessness to one’s life. *Unknownness* thus drives the search for meaning. By stimulating us to carry on searching, the presence of meaning itself becomes more significant as we become primed to “elevate the significance of meaning-relevant information” (Steger et al., 2011, p. 179), feeding back on itself in a virtuous cycle. “One idea above all emerges: that the self-willed forms of wild nature can call out fresh correspondences of spirit in a person. Wildness […] is an energy which blows through one’s being, causing the self to shift into new patterns, opening up alternative perceptions of life” (Macfarlane, 2007, p. 209).

2.3 Wildness affords infinite value

Wildness is limitless for all generations as long as the sun shines because functioning ecosystems continually permute matter in response to environmental change. Landscape patterns are “perceptible instantiations of interrelated, interdependent, environmental phenomena” (Gobster et al., 2007, p. 963), where ecological and evolutionary processes present a palette of ever-morphing affordances – Darwin’s ‘endless forms’ – for our perception (Ingold, 2002). Ultimately, evolution creates new wild affordances. Proximally, affordances are dynamic in space and time: a bird seen transiently in a wetland along its migratory route, a geophyte germinating after the right fire intensity; asynchronous cicada swarms. On landscape scales, disturbance creates phase diversity and thus habitat mosaics, ecological succession creates vegetation structural diversity, herbivory creates patch dynamics, predators create ecotones through landscapes of fear. Wild features themselves grow, transform and die, reflecting the vitality of the landscape. For example, trees embody relations with pollinators, seed dispersers, soil conditions, droughts and other events within the landscape (Ingold, 1993). Wild affordances thus express “patterns of being and becoming, and thus ways of affecting and being affected (van Dooren et al., 2016)”. Between organisms “issuing forth along the lines of their relationships” (Ingold, 2011, p. 71), and landscapes subject to flux, wildness continually opens up the seams of existence, thus always offering opportunities for finding self-meaning.

As such, wild landscapes possess infinite value for individuals because, for each ‘interaction pattern’ (from ‘walking along the edge of a river’ to ‘foraging for mushrooms’), “countless different embodied versions […] can be uniquely realised given different types of nature, people and purposes” (Lev et al., 2020).

Affordances are also dynamic for a particular feature-perceiver relationship over time and thus have the potential to provide different self-meaning from the same feature at different points in one’s life (Chemero, 2009). As our personal context changes, we will ‘see’ different affordances in the same feature. A wetland may provide a place to swim when young, a place to read and write when older, and a place to fish with your children when older still. Coming across a porcupine quill on a dusty trail at one point in your life might conjure an aesthetic response in imagining the creature lurking in the landscape at dusk; and at another time might become a token of your time with a loved one – a shared memory to lengthen your life. Over one’s lifetime, wildness becomes an asset that does not suffer from diminishing marginal returns like manufactured consumer products. Rather, one’s experiences within wild landscapes provide additive value in the personal growth they afford over time, leading to greater eudaimonic well-being (Pritchard et al., 2020). For this reason, ecosystem service valuations are always underestimates of embodied landscapes that continually reconstitute and reimagine themselves (Norgaard, 2010; Bateman and Mace, 2019).

Thus, for each potential affordance – there are two levels of dynamic change – one from the feature itself as it exhibits its own life history, and one from the perceiver depending on her circumstances and receptivity to a particular affordance. This, combined with the continual permuting of features in wild systems, generates an infinitely re-arranging array of affordances from which to potentially draw self-meaning and achieve our chosen ‘functionings’, such as deepening friendships through story-telling, restoring mental energy to make an important decision; or perhaps finding inspiration to finish your thesis. Thoreau declared, ‘in wildness is the preservation of *the* world’. Wildness is really the conservation of *your* world.

2.4 Self-meaning and sustainable economies

As globalisation quickens, we risk becoming ever more homogenised and manipulated by atomised content and misinformation. Instant information is like eutrophication of meaning, creating toxic algal blooms in our minds that suppress the rarer, more considered thoughts. By providing a medium for one's unique ideas to surface and for external information to percolate through finely textured layers, wild landscapes incubate effective agents (Sen, 2013). For example, Atchley et al. (2012) showed that being immersed in nature for four days without access to technology increased creative problem-solving by 50%, which helps counteract consumer culture's instant gratification and impulsivity (reviewed in Bratman et al., 2012). From a public health perspective, wild landscapes function as 'equigenic environments' that mitigate the negative conversion factors of socioeconomic inequality into poor mental health by enabling everyone to equitably improve their eudaimonic well-being (for example, Mitchell et al., 2015). Wild landscapes can also be considered 'heterotopia', which Foucault (1986), for example, called spaces that simultaneously represent, disrupt and transform – that are *other* to the establishment. By facilitating self-actualisation on an individual level, wild 'heterotopia' may facilitate adaptation on a societal level by acting as "transgressive spaces where it is possible to think differently, be differently, and engage in the practices of freedom" (Beckett et al., 2017, p. 10; Vannini and Vannini, 2019). For this reason, spatially and temporally dynamic wild landscapes are increasingly recognised as key assets for urban spatial planning for the unique experiences they offer (Threlfall and Kendal, 2018).

Mainstreaming the value of wild heterotopia into economic policy-making might be most effective through Experimental Ecosystem Accounting (EEA), which has been developed to shift the System of National Accounts beyond gross domestic product (Hein et al., 2020). EEA includes four account types that measure the spatial extent of various ecosystem types, the ecological condition of ecosystems, the physical flows of ecosystem services, and (if needed) the monetary value of ecosystem assets and services (Hein et al., 2020; Figure 2.2). Ecosystem assets are defined as different ecosystem types (landscape units likely to share broadly similar ecological characteristics and functioning, such as forests, grasslands, wetlands or estuaries) permuted into various themes and scales according to the particular policy decision (such as enhancing psychological well-being from green urban infrastructure at city scale, or increasing pollination services from ecological corridors within agricultural landscapes at regional or national scales). The wildness of a particular landscape can be measured as the ecological condition of the ecosystems comprising the landscape, which can then be aggregated into the condition account for each ecosystem type for the policy focus areas overall (Figure 2.2). Ecological condition can be measured through various ecological complexity indicators, such as the biodiversity intactness index (Scholes and Biggs, 2005).

Integrating EEA and the capability approach (CA) helps to solve their respective shortcomings. The CA currently lacks a feedback process between the condition of the resources and the functionings that can (or should) be achieved (Ballet et al., 2013; Schultz et al., 2013; Pelenc and Dubois, 2020). Making capabilities spatially-explicit by integrating with EEA would help policy makers more clearly assess the trade-offs between human, natural and manufactured capital at specific scales. Similarly, neither natural capital accounting nor the ecosystem services concept captures the intrinsic value of ecological processes and species diversity or fully encompasses human well-being (Bratman et al., 2019; Bateman and Mace, 2020; Hein et al., 2020), and thus would become more holistic by linking ecological complexity more fully to human capabilities and functionings. For example, integrating the self-meaning capability into EEA makes a direct connection to the extent of ecosystem types themselves as fundamental mediums available to search for self-meaning, as well as generating the affordances to convert into purpose. Similarly, the wilder a landscape (the more ecologically functional it is), the more likely it is to hold our fascination and increase the comprehension necessary to find meaning. Thus, both ecosystem extent and condition are important to human well-being in their own right, in addition to the selected ecosystem services that are ultimately evaluated (Bratman et al., 2019). This integration may help to reduce the entropic loss of value when quantifying ecosystem services alone (Bratman and Mace, 2020). By functionally interlinking the capability of both ecosystems and humans to manifest their own ‘functionings’, EEA-CA can directly incorporate the existential freedom of humans and non-human entities into economic decision-making.

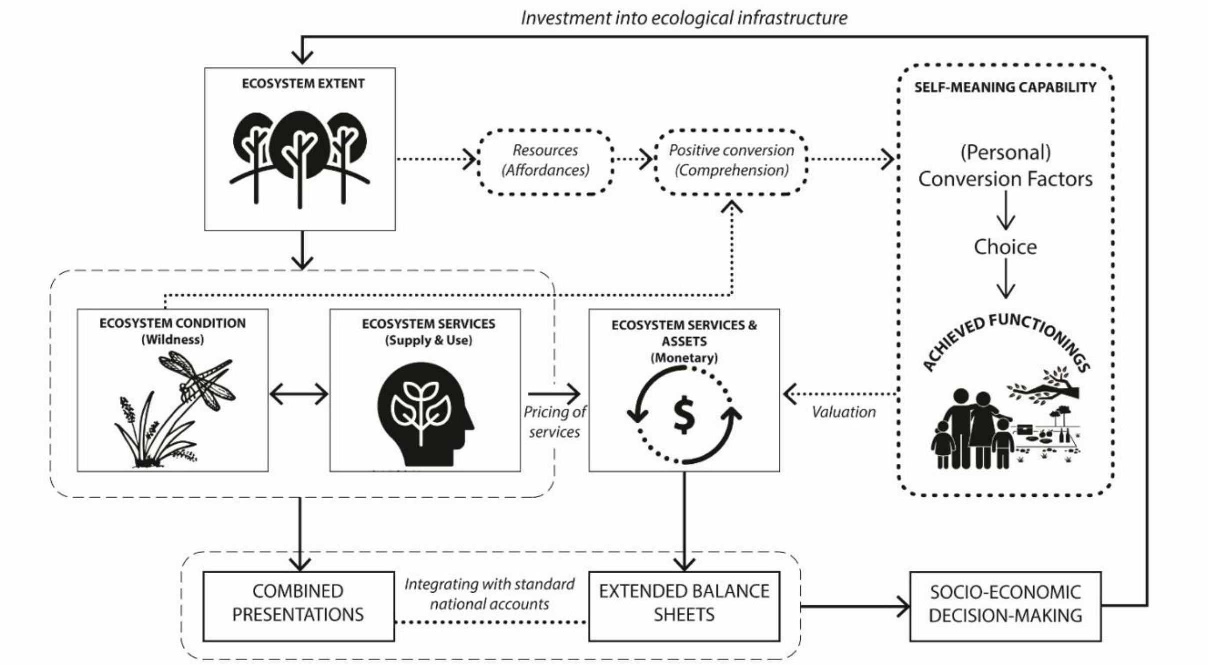

Figure 2.2. Conceptual diagram for integrating wildness and the self-meaning capability into Experimental Ecosystem Accounts (EEA) (adapted from UN, 2019). Round rectangles represent EEA accounts and processes; square rectangles represent the system of national accounts and economic decision-making processes; and dotted rectangles represent capabilities. Ecosystem assets are mapped according to the relevant policy question (such as optimising public health benefits through a network of urban green spaces; increasing pollination services from ecological corridors within agricultural landscapes; or understanding potential ecotourism value from protected area networks at national scales). Wildness – the degree to which ecosystems function autonomously – can be integrated into the ecosystem condition accounts through methods such as the biodiversity intactness index (Scholes and Biggs, 2005) and other indicators that measure ecological complexity. The size of the remaining ecosystems (extent accounts) and their relative wildness (condition accounts) within a particular landscape will influence the overall capacity to deliver ecosystem services. Under the self-meaning capability pathway, such services include reduced anxiety and depression, improved cognitive capacities and physical health (sensu Bratman et al., 2019). Ecosystem assets can be integrated into standard national accounts through both combined presentations (which do not require monetary evaluation) and extended balance sheets (which incorporate the monetary costs of using ecosystem capital) (UN, 2019). Integrating the self-meaning capability into the EEA emphasises that it is not simply the final ecosystem services that are important for freedom: the ecosystems and their wildness levels are important independently in generating the affordances for purpose (resources) and dynamic change for comprehension (positive conversation factor) that help manifest an individual's realised freedoms (functionings). Achieved functionings then feed back into identifying what ecosystem services are relevant and how they are valued in society. Capabilities thus help to explicitly define the ultimate goal of natural capital accounting as sustaining intergenerational and existential freedom rather than passively reflecting and responding to systems of linear growth. By expanding the ecological infrastructure available to enhance our self-meaning capability, individuals may become more connected to nature and shift towards prosocial and pro-environmental functionings that increase the willingness to pay for ecosystem services and conservation, which may prompt greater investment into restoration and conservation by decision-makers in a virtuous cycle. (All vector images downloaded from the Noun Project https://thenounproject.com/).

2.5 Conclusions

Wildness is an infinite resource for human flourishing and freedom. The ecological and evolutionary processes that generate biodiversity also contribute to the individual's lifelong capability to find self-meaning, helping one to create culture rather than simply respond to it. Integrating the self-meaning capability into a natural capital accounting framework switches the focus from categorising what we can get *from* the environment to what is *within* the environment that enables us to (re-)create value. Mainstreaming interlinked models of wildness and well-being will make clear to decision-makers that any development option that simplifies a landscape to maximise manufactured capital or commodities (such as approving agricultural monocultures or open-cast mining) will ultimately reduce human freedom to find self-meaning.

Rewilding, in its broader sense of restoring ecological complexity and functioning (Perino et al., 2019), can thus guide development programmes that seek to improve prosperity, public health and socioeconomic resilience simultaneously. This does not mean we have to abandon all land back to wilderness – not all landscapes ought to contain apex predators – but rather that our policies should promote the fullest expression of wildness achievable in each landscape, given context-specific opportunities and constraints. "Biodiversity is an expression of the possible" (Collar, 2003, p. 268), where wild landscapes, replete with self-willed features and messy relations, sustain the infinitely possible meanings of being alive. As much as rewilding is a process of restoring ecosystem functioning, so does it restore our capability *to become* and do 'effective things'. Wildness should be at the forefront of policies that resist economic hegemony and cultural homogenisation.

Chapter 3 A framework to measure the wildness of managed populations

3.1 Abstract

As landscapes continue to fall under human influence through habitat loss, fragmentation, and settlement expansion, fencing is increasingly being used to mitigate anthropogenic threats or enhance the commercial value of wildlife. Subsequent intensification of management potentially erodes wildness by disembodying populations from landscape-level processes, thereby disconnecting species from natural selection. Decision-makers thus require tools to measure the degree to which populations of large vertebrate species within formally protected areas and other wildlife-based land-uses are self-sustaining and free to adapt. I present a framework comprising six attributes relating to the evolutionary and ecological dynamics of vertebrates. For each attribute, I set empirical, species-specific thresholds between five wildness states using quantifiable management interventions. The tool was piloted on six herbivore species with a range of Red List conservation statuses and commercial values using a comprehensive dataset of 205 private wildlife properties with management objectives spanning ecotourism to consumptive utilization. Wildness scores were significantly different between species, and the proportion of populations identified as wild ranged from 12% to 84%, which indicates the utility of the tool to detect site-scale differences between populations of different species and populations of the same species under different management regimes. By quantifying wildness, this foundational framework provides practitioners with standardised measurement units that interlink biodiversity with the sustainable use of wildlife. Applications include informing species management plans at local scales; standardising the inclusion of managed populations in Red List assessments; and providing a platform for certification and regulation of wildlife-based economies. I hope that applying this framework will assist in embedding wildness as a normative value in policy, thereby mitigating the shifting baseline of what it means to truly conserve a species.

3.2 Introduction

Fragmentation from road construction, human settlement expansion and a myriad of associated anthropogenic pressures is bringing wildlife species under human influence (Peterson et al. 2005; Laurance et al. 2014; Jones et al. 2018). Many protected area managers across the world, most notably in southern Africa, Australia, New Zealand, and the USA, are increasingly using fencing to respond to these threats (Hayward & Kerley 2009; Packer et al. 2013; Ringma et al. 2017), but there are concerns that such confinement undermines conservation value by stabilising abundance at the expense of broader landscape connectivity (Woodroffe et al. 2014). Private landowners also use fences to reduce risks and manage the commercial utilisation of wildlife (Butler et al. 2005; Carruthers 2008; Mysterud 2010), which includes activities such as trophy hunting, selective breeding for live sales, meat production and ecotourism (reviewed in Taylor et al. 2015). Both conservation- and commerce-oriented paradigms can thus result in the intensification of management. Management practices may convert selective pressures from natural to artificial by controlling breeding (for example, mate pairing), mortality (for example, disease control, hunting or predator removal), access to food and water (supplementary feeding and artificial water-point construction) and patterns of space use (including dispersal barriers and the installation of enclosures) (von Brandis & Reilly 2007; Hetem et al. 2009; Mysterud 2010; Taylor et al. 2015; Pitman et al. 2017), which undermines the fitness of the managed animals (Jule et al. 2008; Willoughby et al. 2017). Such practices may ultimately reduce natural variability in pattern and process and thus homogenise ecological communities (Dalerum & Miranda 2016; Clements & Cumming 2017b). As management strategies exist along a spectrum from captive-breeding to landscape-scale management, conservationists must determine at what point wildlife ceases to be wild so that biodiversity conservation and sustainable development can be balanced. Conservationists must measure wildness to evaluate the true success of interventions towards the ideal of flourishing populations in functioning ecosystems (Redford et al. 2011), while policy-makers should foster multifunctional landscapes that provide economic opportunities but also retain biodiversity. Developing tools that help to quantify and visualise the potential trade-offs and synergies between these two goals will be crucial in bridging the gap between science and policy.

Wildness concerns the degree to which individuals exist autonomously in evolutionarily and ecologically functioning populations where genetic and phenotypic diversity enables natural selection to produce adaptation (Moritz et al. 2002; Redford et al. 2011; Mallon & Stanley Price 2013). The dynamic functional relationships between and within species sustain biodiversity by creating niches and generating landscape heterogeneity, thus establishing feedback loops between ecological and evolutionary processes (Erwin 2008; Laland & Boogert 2010; Odling-Smee et al. 2013). Cumulatively, these emergent properties of flux, dynamism and autonomy can be called "wildness" (Evanoff 2005; Mallon & Stanley Price 2013; Pickett 2013), where interactive processing between organisms and their environment produces resilient systems (Cookson 2011). Thus wildness is an integral property of ecosystem functioning and potentially ecosystem service delivery. Wildness, however, does not necessarily correspond to "pristineness". Rather, they can be seen as orthogonal qualities where the apex of both is wilderness (Aplet et al. 2000). Specifically, Aplet's et al. (2000) continuum of wildness distinguishes between 'naturalness', which describes the composition and structure of an ecosystem, and "freedom from human control", which describes the degree of biodiversity being 'self-willed'. It is this latter quality, as applied to wildlife populations, which we aim to describe here. Selective pressures may be different in human-modified landscapes ("novel ecosystems", Hobbs et al. 2013), but degrees of wildness can still occur if species are provided with the opportunity to adapt to these pressures through natural selection and fulfil their functional roles within the landscape. Management that enables interaction between all components of the ecosystem will work to "produce wild things" (Cookson 2011 p. 191) even within novel environments.

Biodiversity assessments should thus incorporate the capacity of populations (which we define as geographically distinct groups between which there is little demographic or genetic exchange), to be self-organised, self-sustaining and integrated into an ecosystem. Currently, there is no standardised, measurable definition of wildness of a population. For example, the Red List criteria of the International Union for Conservation of Nature (IUCN) define managed populations as wild if management aims to counteract human-induced threats or manage the overall habitat for the long-term persistence of the population. Conversely, populations dependent on direct intervention, where they would become locally extinct within ten years without management, are not considered wild (IUCN Standards and Petitions Subcommittee 2022). However, these guidelines lack comprehensive empirical thresholds that can be used to standardise wildness evaluations. The vagueness of wildness as a concept prevents decision-makers from establishing clear interventions and standards relating to species and land management and may lead to inflated estimates of conservation success. Given the global push to expand protected areas, and the simultaneous demands of conservation areas to contribute to sustainable development (Watson et al. 2014; Taylor et al. 2015), evaluating the effectiveness of these multifunctional landscapes in retaining conservation value is becoming a key policy issue.

Decision-makers need objective, standardised and fine-scale frameworks to both measure wildness and determine at what point management intensity may negate wildness. The framework must be able to evaluate wildness at a local population scale, corresponding to the extent of the management regime or habitat "island" imposed by artificial barriers; and must identify wildness equitably across species, management regimes and land-use types. This requires defining wildness states, mapping the relevant management attributes and actions applicable to each state, and delineating quantifiable thresholds between each state. Previous frameworks have categorised attributes fundamental to the wildness of populations but without assigning quantitative thresholds. Those developed by Leader-Williams et al. (1997) and Mysterud (2010) distinguish between wild and non-wild (called ”captive breeding” and ”domestic,” respectively) populations and are congruent in their identification of breeding manipulation, space requirement, harvest selectivity, resource provision and predation as key management interventions. However, the frameworks are based on binary responses and arbitrarily defined thresholds that lack fully quantitative and standardised species-specific thresholds, which makes inconsistent interpretation probable. More recently, Redford et al. (2011) defined five states of conservation success along a wildness spectrum. However, this classification also cannot be operationalized as a decision-making tool because: 1) the attributes are qualitative and do not provide species-specific measurable thresholds to objectively distinguish between states and, 2) they apply to the species overall and thus do not provide a platform for assessing the conservation value of local populations. In this study, I adapt the framework of Redford et al. (2011) to create a tool that both articulates and measures the wildness of populations by quantifying management interventions that impact on the evolutionary and ecological dynamics of species. My desired outcome is to integrate successful large vertebrate conservation (*sensu* Redford et al. 2011) into regulation and reporting, such that wildness becomes a normative value in management, assessment and policy.

3.3 Methods

3.3.1 Building the framework

To lay the foundation for a wildness framework, two expert workshops were convened by the South African National Biodiversity Institute (SANBI) at the Pretoria National Botanical Gardens (10th of December 2014 and 24th February 2015). Thirty experts were invited, of whom 13 participated in one or more workshop and three others commented on draft versions of the framework. The participants had expertise across a broad spectrum of relevant wildlife management fields including population biology, conservation science, resource economics, evolutionary biology, natural resource management and spatial ecology. Participants were drawn from organisations representative of wildlife management and policy development in South Africa. Iterative discussions at the first workshop produced the prototype framework by:

1. Identifying attributes that influence both short-term survival of populations as well as long-term implications for the adaptive potential of the population overall (reflecting functioning evolutionary processes).
2. Defining states along the wildness spectrum by adapting the Redford et al. (2011) classification to local-scale context and justifying the boundary between wild and non-wild states.
3. Listing the potential management actions or characteristics that influence each attribute. These were drawn from field surveys (for example, Taylor et al. 2015) and from the experience of the experts.
4. Developing measurable thresholds for each attribute to discern between states. Species-specific threshold values (home range size, social group size and composition) in each habitat type were gleaned from the literature (*Supporting Information 3.1*).

The prototype framework was then validated at the second workshop using a training dataset from a 2014 survey sent out to private landowners to support the revision of the Red List of Mammals of South Africa (M.F. Child unpubl. data). Additional indicator variables for some attributes were identified to give further empirical power in determining wildness states and the quantitative thresholds were recalibrated.

3.3.2 Piloting the framework

We then piloted the revised framework on six herbivore species that are both of conservation concern and have high value in the South African wildlife industry (breeding for live sale, trophy hunting and ecotourism), with values ranging from USD 1,200 to USD 38,000 at game auctions in 2014 (Dalerum & Miranda 2016): white rhinoceros *Ceratotherium simum*; tsessebe *Damaliscus lunatus*; bontebok *Damaliscus pygargus pygargus*; mountain zebra *Equus zebra*; roan antelope *Hippotragus equinus*; and sable antelope *Hippotragus niger*. The potential trade-off between conservation and commercial goals for these species thus provided an opportunity to test the efficacy of the framework in identifying wild populations across a range of management goals. We used a comprehensive dataset on the management systems of 205 private wildlife areas (hereafter 'properties') comprising structured interviews conducted between 2014 and 2015 across South Africa (Taylor et al. 2015). These properties pertain to landowners utilising wildlife on a commercial basis, with management regimes ranging from intensive breeding to extensive ecotourism and range in size from 0.9 to 1,030 km^2. Many properties have mixed economic portfolios, with management regimes that vary according to the species (Taylor et al. 2015). As all properties in the dataset are fenced, we consider the property boundary to define a population of each species as there is limited movement between properties aside from deliberate translocation. The dataset included information relevant to all identified attributes, including property variables (size, location, land use type and fencing patterns); herbivore species composition and abundance; predator species composition; and management interventions that include veterinary care, supplementary feeding and water provision, predator control, intensive breeding, hunting and habitat management practices.

3.3.3 Applying the framework

Once we developed the framework, we applied the data from Taylor et al. (2015) to assess the wildness of populations belonging to the focal species. For each population, the attributes were scored by evaluating the data against the thresholds between wildness states. For each attribute, a score was assigned on an ordinal scale, with the least wild state scoring 1. The final wildness score for each population was calculated as the median value across attribute scores (see

Supporting Information 3.1, *Supporting Information 3.2* and *Supporting Information 3.3* for more detail). Interquartile ranges (IQR) were used to express the variation around wildness scores, both on a population and species level. We then tested whether the distribution of wildness scores across populations was significantly different between species using Mood's median test. The explanatory power of both population size and property size in determining the wildness status of a population was tested using ordered logistic regression. Species identity was included as a factor in the model to determine species-specific effects (see *Supporting Information 3.2* and *3.3* for more detail). All analyses were performed in R 3.4.2 (R Core Team 2023).

3.4 Results

3.4.1 The Framework

Six interlinked attributes relating to evolutionary and ecological dynamics were identified as contributing to the wildness of a population (Table 3.1). The attributes were then used to characterize five states [Captive Managed (CM), Intensively Managed (IM), Simulated Natural (SN), Near Natural (NN) and Self-sustaining (SS)] along the wildness spectrum (defined in Table 3.2). These wildness states can be visualised as existing on a spectrum within a matrix where management activities increasingly dominate resource provision and vital rates (births and deaths) (Figure 3.1). The quantifiable variables for each attribute from Table 3.1 were then converted into empirical thresholds (both binary and continuous) to delineate between states (see framework summary in Table 3.3). The division between non-wild and wild states was drawn between IM and SN (Table 3.2), meaning that CM and IM states were non-wild and received a wildness of 1 and 2 respectively; while SN, NN and SS were defined as wild states and received scores of 3, 4 and 5 respectively. Thus, a population is considered wild if the median score across attributes was ≥3.

Table 3.1. Definition of key identified attributes relating to the evolutionary and ecological dynamics of managed populations and their key quantifiable indicator variables used to set threshold values between wildness states.

Attribute	Definition	Supporting references*	Key indicator variables
Space	Facilitates co-existence and niche differentiation / adaptation through microhabitat utilisation and habitat partitioning. Allows populations to meet nutritional requirements across seasons. Enables intraspecific interactions between social units (e.g. breeding and competition), interspecific interactions (e.g. predator-prey dynamics), and interactions with abiotic components of the landscape (e.g. ecological engineering).	Walker et al. (1987) Jule et al. (2008) Hayward & Kerley (2009) Jackson et al. (2014)	Home range size of species in specific biome or habitat. Dispersal capacity of species deduced by fence type and surrounding land use compatibility.
Disease and parasite resistance	Plays a major role in regulating and creating biodiversity through co-evolution. Periodic disease outbreaks are important population control mechanisms. Conversely, biodiversity loss can exacerbate the spread of infectious diseases.	Altizer et al. (2003) Fincher & Thornhill (2008) Pongsiri et al. (2009)	Frequency, extent and purpose of veterinary care (preventing all diseases versus pre-emptive vaccination against non-native diseases).
Exposure to natural predation	Predation plays a top-down role in sustaining biodiversity. Predator-prey relationships are important drivers of evolution, creating trait diversity and new species, and enhance overall biodiversity through the creation of landscapes of fear. Intra-guild competition within the predator community has important consequences for predator population dynamics and sustainability.	Linnell & Strand (2000) Creel (2001) Ripple et al. (2001) Yoshida et al. (2003) Thomson et al. (2006) Creel et al. (2007) Oro et al. (2013) Sandom et al. (2013a) McArthur et al. (2014) Terborgh (2015) Owen-Smith (2015)	Presence/absence of predators. Functional composition of predator community. Frequency of exposure to predators.
Exposure to natural food limitations and fluctuations	Being exposed to fluctuations in food availability, or resource pulses, influences evolution by driving diversity of life history traits, and thus facilitates the coexistence of ecological communities, especially when synergising with the effects of predation. Limited food availability regulates population sizes and enhances community diversity.	Walker et al. (1987) Bond & Loffell (2001)) Chesson et al. (2004) Yang et al. (2008) Schmidt & Hoi (2002) Peterson et al. (2005) Blanchong et al. (2006) Bishop et al. (2009)	Presence/absence of food provisioning. Frequency of food provision. Presence / absence of habitat modifications for production or ecosystem restoration. Inside or outside native range

Attribute	Definition	Supporting references*	Key indicator variables
Exposure to natural water limitations and fluctuations	Migrations and dispersals forced by water fluctuations are critical for ecosystem functioning as individuals will transport nutrients, energy and other organisms between locations and enable ecological interactions between species in both space and time. Subsequent range expansions can feed back into evolutionary processes. Limited water availability regulates population sizes and enhances community diversity.	Walker et al. (1987) Owen-Smith (1996) Gaylard et al. (2003) Peterson et al. (2005) Smit et al. (2007) Bauer & Hoye (2014) Fronhofer & Altermatt (2015) Selebatso et al. (2018)	Even versus clumped distribution of water points, average inter-point distance. Frequency of water provision at artificial water-points (pumped year-round or collects water seasonally).
Reproduction	Competition for mates determines what alleles are passed onto the next generation and at what frequencies, thus influencing evolutionary trajectories. Spatial and temporal variability in habitat and climate helps to conserve genetic diversity where natural selection ensures that the individuals with the best chance to survive and reproduce in a particular setting will do so most successfully. This engenders adaptive capacity within the population and resilience to the population overall.	Jarman (1974) Price (1984) Allendorf et al. (2001) McPhee (2004) Allendorf et al. (2008) Hetem et al. (2009) Jule et al. (2008) Olden et al. (2004) von Brandis & Reilly (2007) Mysterud et al. (2008) Champagnon et al. (2012) Willoughby et al. (2017)	Degree of breeding competition control. Degree of mate selection control. Off-take / augmentation strategy selective or non-selective.

*The supporting references are not exhaustive but emblematic of the research supporting the importance of the listed attributes.

3.4.2 Framework application

In testing the framework, I found that the wildness scores varied considerably for each focal species across the sampled properties. The distribution of wildness states between species yielded significant differences (Mood’s median test, X-squared = 89.7, df = 5, p-value < 0.05; Figure 3.2), with three species having median scores of ≥3 (wild) and three species <3 (non-wild). At the population level, 186 populations were analysed across the six focal species, where 63 (34%) populations were wild. Most populations (102; 55%) exhibited low variation across attribute scores (IQR < 1) where 134 (72%) populations possessed a wildness score and IQR that fell entirely within either wild or non-wild states. The proportion of wild populations among species ranged from 12% (*Hippotragus equinus*) to 84% (*Ceratotherium simum*) (Figure 3.2, Supporting Information 3.2). Wildness states of species were not entrained by property identity: of 23 properties where three or more of the focal species co-occurred, 74% (N = 17) of the properties contained both wild and non-wild populations for different species, meaning the same property contained some species that were considered wild and some that were not. Wildness scores did not correlate with population size (ordered logistic regression model $p = 0.21$) but did correlate with property size across species ($p < 0.01$) where smaller areas generally had lower wildness scores, but the effect was species-dependent (*Supporting Information 3.2*).

Wildness scores for high value game species are mostly limited by space (most often maintained in small fenced areas) and exposure to natural ecological fluctuations. For example, the least wild species, sable and roan, exist almost exclusively in breeding camps with an average camp size (taking the maximum when a range is given) for sable being 2.2±3.6 km^2 (N = 50 properties). Breeding manipulation is one of the lowest-scoring attributes for ranches. For sable, while a typical herd is 10-30 individuals (Skinner & Chimimba 2005), 29% of all properties contain herds of fewer than 10 and 55% fewer than 30.

Table 3.2. A description of the wildness states adapted from Redford et al. (2011) during the expert workshops with a summary of the predicted effects on both the short-term survival and long-term resilience of populations.

Wildness state	**Definition**	**Effects on short-term survival**	**Effects on long-term resilience**
Captive Managed (CM)	Total control over the individual and population in breeding camps. Animals will die at this location without continual management. Social dynamics and resource fluctuations negated by management.	Completely dependent on provisioning and veterinary care. Will die within days without intervention.	Selective breeding negates adaptation and undermines the adaptive capacity of the population.
Intensively Managed (IM)	Direct human intervention at the individual and/or population levels. Social dynamics and resource requirements actively manipulated and thus mate selection occurs in an artificial setting with limited opportunity for adaptation to the natural environment. Resource fluctuation negated by provisioning in times of nutritional stress. These populations may exist in semi-extensive systems (as opposed to breeding camps) but with conditions controlled to benefit the focal species. This category includes captive breeding for conservation.	More individuals may be present than can naturally be supported. Veterinary care provided continuously and non-selectively in landscape. Population may be non-viable without provisioning and thus may become locally extinct within ten years without human intervention.	Only selected ecological interactions allowed, typically to maximise production of specific traits. Selective breeding or mate selection under non-natural conditions dominates so population may not become adapted to the environment. Adaptation / adaptive capacity thus severely limited.
Simulated Natural (SN)	Limited but specific set of interventions to sustain populations and mitigate extrinsic factors (for example, metapopulation management). Management is aimed at reducing the impact of humans (i.e. habitat fragmentation, fences and illegal trade) at population level, rather than focusing on the individual. Inability to maintain viable/self-sustaining populations without long-	No resource provisioning to individuals, unless in severe conditions where ordinarily animals would disperse. *Ad hoc* veterinary care in response to non-native diseases. Number of individuals is close to what	Most ecological interactions are functional but links may be missing due to absence of certain species or habitats. Limited movement occurs across the landscape and there is limited dispersal between populations.

Wildness state	**Definition**	**Effects on short-term survival**	**Effects on long-term resilience**
	term, periodic management of habitat and extrinsic factors. Social and resource requirements thus need punctuated intervention. No deliberate interference with mate selection although indirectly affected through harvesting or hunting of breeding individuals. Management is aimed at simulating natural processes through hunting, harvesting and translocation.	can be supported naturally (without intervention). Population likely to become extinct over time.	
Near Natural (NN)	Very few interventions, all of which are directed at long-term ecosystem process management and not at either specific individuals or populations. Social requirements of the population are met, but resource requirements might be altered in response to anthropogenically induced limitations. No deliberate interference with mate choice as management is aimed at sustaining long-term ecosystem processes.	Very occasional food provisioning. Space is sufficient for the species to survive amidst environmental fluctuations (through die-offs if necessary). Major unnatural disturbances are mitigated periodically.	Evolutionary process functioning in a near natural setting with mate choice unimpeded by human artefact. However, long-term resilience may still need assistance through periodic translocation between areas to ensure gene flow.
Self-sustaining (SS)	No deliberate human interference to sustain or grow the population. However, there may be, or may have been, indirect human influence to which the population has adapted (for example, black-backed jackals *Canis mesomelas* on farmland in South Africa). Social and resource requirements are met.	No direct provisioning. Space is sufficient for the species to survive amidst environmental fluctuations (through die-offs if necessary). Population self-sustaining under current conditions.	Ecological and evolutionary dynamics unimpeded. Dispersal/migration is possible such that natural selection is operating and adaptive capacity is sustained in the population.

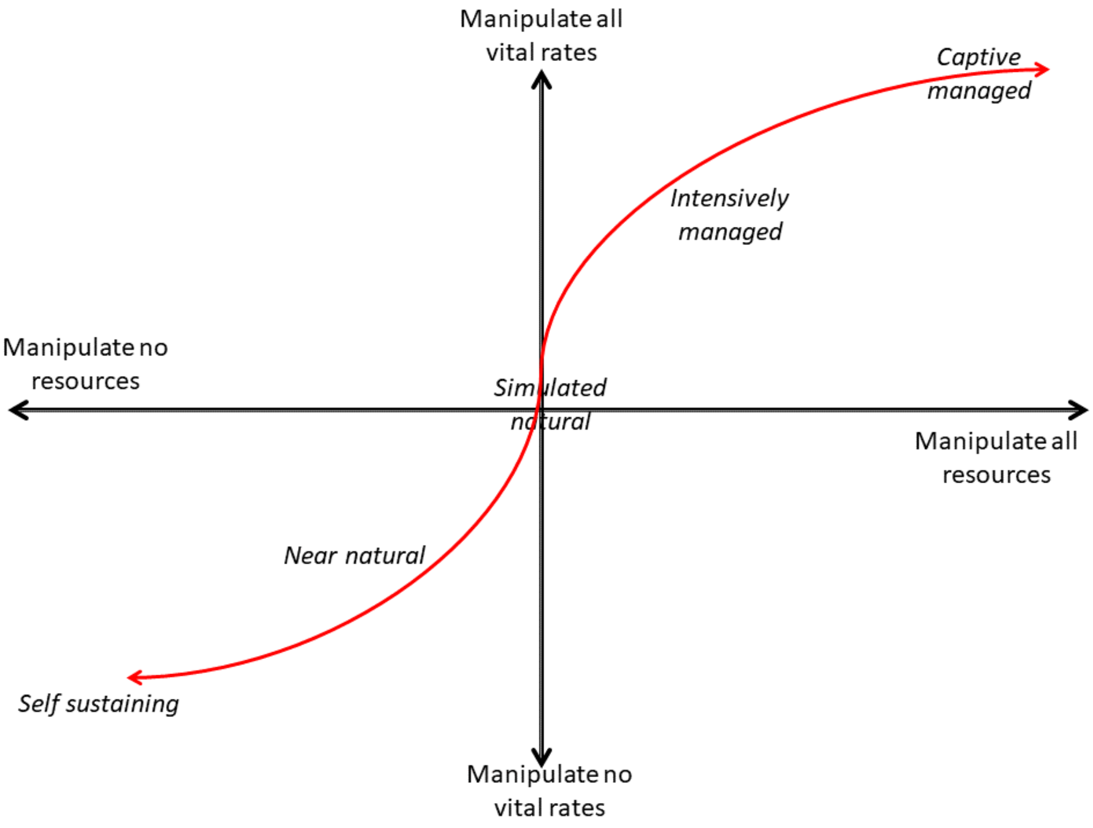

Figure 3.1. Conceptual representation of the wildness spectrum as delineated here along two axes of human management intensity – resource manipulation and vitality engineering. Wildness states can be visualised as existing on a spectrum within a matrix where management activities increasingly dominate availability of resources (supplementary feeding and artificial water provision) and vital rates (selective breeding and demographic control and prevention of death through natural selection). Where both intervention types are intensive, the managed subpopulations would be classified as captive managed, as they are entirely dependent on human intervention for survival. At the opposite end of the spectrum are subpopulations that experience natural fluctuations in resource availability and full freedom in demographic fluctuations, including mate selection and deaths through natural factors.

Table 3.3. Summary framework to determine the wildness state of managed populations, displaying the empirical thresholds between each state. For each population, scores are assigned to each attribute using the thresholds, where the score corresponds to the wildness state on an ordinal scale (Captive Managed scores 1 and Self-sustaining scores 5). The net wildness score of the focal population is calculated as the median of the attribute scores. *****The division between wild and nonwild populations is drawn between simulated natural and intensively managed, respectively. For each population, scores are assigned to each attribute based on the thresholds. The score corresponds to the wildness state on an ordinal scale (captive managed—1 to self-sustaining—5). Net wildness score of the focal population is calculated as the median of the attribute scores.

Attributes	**Thresholds***				
	Captive Managed (CM)	**Intensively Managed (IM)**	**Simulated Natural (SN)**	**Near Natural (NN)**	**Self-sustaining(SS)**
Space	Single species camps	Area < 1 home range unit	Area => 1 home range unit	Area => 2 home range unit	Home range units of area > no. social groups present
	Camp (internal) fence: electrified / impermeable	Perimeter fence: electrified game fence.	Perimeter fence: meshed or stranded with artificial passageways installed	Perimeter fence: cattle fence with artificial passageways installed	Perimeter fence: no fence or cattle fence with artificial passageways installed
Disease and parasite resistance	Veterinary care: continuous direct to all individuals (including antibiotics) to mitigate native and non-native diseases	Veterinary care: permanent preventative measures in landscape (e.g. Duncan applicators and dips) to mitigate native and non-native diseases	Veterinary care: *ad hoc* preventative vaccination against native and non-native diseases	Veterinary care: *ad hoc* preventative vaccination against non-native diseases	Veterinary care: no disease control
Exposure to natural predation	Small predators: 0 species (excluded or removed).	Small predators: ≥ 1 species; continual exposure	Small predators: ≥ 3 species continual; exposure	Small predators: ≥3 species continual; exposure	Small predators: ≥ 3 species; continual exposure
	Mesopredators: 0 species (excluded or removed)	Mesopredators: ≥ 1 species; occasional exposure (removed)	Mesopredators: ≥ 2 species; continual exposure (removed *ad hoc*)	Mesopredators: ≥ 2 species; continual exposure	Mesopredators: ≥ 2 species; continual exposure
	Apex predators: 0 species (excluded or removed)	Apex predators: 0 species (excluded or removed or absent)	Apex predators: ≥ 1 species; occasional exposure (removed *ad hoc,* controlled or absent)	Apex predators: ≥ 1 species; continual exposure (removed *ad hoc*)	Apex predators: ≥ 2 species; continual exposure

Exposure to natural food limitations and fluctuations	Continuous food provision to all individuals in enclosures	> 1 supplementary feeding event per year on average; salt licks	= 1 supplementary feeding event per year on average	< 1 supplementary feeding event per year on average	No supplementary feeding events
	Habitat management: no access to natural habitat	Habitat management: ≥ 1 habitat modification for production	Habitat management: 1 habitat restoration intervention	Habitat management: 2 habitat restoration interventions	Habitat management: ≥ 3 habitat restoration interventions
	Indigenous habitat: outside range	Indigenous habitat: outside range	Indigenous habitat: inside or outside range	Indigenous habitat: inside range	Indigenous habitat: inside range
Exposure to natural water and limitations and fluctuations	Water-point distribution: ≥1 water-points / encamped animal group	Water-point distribution: ≥1 water-point / home range unit, even spacing	Water-point distribution: < 1 water-point / home range unit, even spacing	Water-point distribution: < 0.5 water-point / home range unit, asymmetrical spacing	Water-point distribution: < 0.25 water-point / home range unit, asymmetrical spacing
	Seasonality: 100% artificial water-points, continuous availability	Seasonality: ≥ 50% artificial water-points, continuous availability	Seasonality: < 50% artificial water-points, mixed availability	Seasonality: < 25% artificial water-points, seasonal availability	Seasonality: 100% natural water-points, seasonal availability
Reproduction	Breeding competition: 1 breeding male / enclosure	Breeding competition: population size < 1 social unit (= 1 breeding male)	Breeding competition: population size = 1 social unit (≥ 2 breeding males)	Breeding competition: population size ≥ 2 social units (multiple breeding males)	Breeding competition: population size ≥ 3 social units (multiple social groups)
	Selection: individuals matched and selected for specific traits (controlled breeding); presence of non-native subspecies or ecotypes	Selection: intensive breeding for production, periodically replacing breeding stock; presence of non-native subspecies or ecotypes	Selection: individuals not matched or selected but limited mate choice *de facto* from small population size; absence of non-native subspecies or ecotypes	Selection: no breeding manipulation, mate choice uninhibited but some demographic processes may be lacking; absence of non-native subspecies or ecotypes	Selection: no breeding manipulation, mate choice uninhibited, all demographic processes functioning, absence of non-native subspecies or ecotypes
	Off-take / augmentation: individuals selected for genotypes (based on stud book)	Off-take / augmentation: individuals selected for specific traits	Off-take / augmentation: individuals selected to simulate dispersal as part of metapopulation strategy	Off-take / augmentation: non-selective (based on post-reproductive age where appropriate)	Off-take / augmentation: non-selective (based on post-reproductive age where appropriate); no hybridisation; no augmentation following initial reintroduction

3.5 Discussion

I present a framework to measure the wildness of large vertebrate populations by quantifying management intervention thresholds that potentially impact the evolutionary and ecological dynamics of species. Captive Managed and Intensively Managed states are non-wild because management influences the reproduction, mortality and resource requirements of all individuals directly. Conversely, Simulated Natural, Near Natural and Self-sustaining states are considered wild and characterised by management at the population or landscape scale. The division thus marks the difference between ensuring short-term survival of a population versus facilitating its long-term resilience. For natural selection to be the primary driver in managed ecosystems, animals must be allowed to die and thrive in spatially and temporally explicit cycles linked to non-equilibrium landscape-level processes (*sensu* Pickett 2013). The attributes relate to the potential of a population to experience fluxes in landscape-level patterns and processes relating to resource distribution, intra- and interspecific competition, and environmental conditions. Management regimes in the wild states employ holistic land management and thus are likely to sustain functionally diverse populations contributing to local ecosystem functioning (for example, Gagic et al. 2015). Wild states thus embody the properties of biodiversity we seek to protect.

While previous conceptual frameworks for categorizing the wildness of populations exist (Leader-Williams et al. 1997; Mysterud 2010; Redford et al. 2011), this is the first that sets comprehensive empirical thresholds between wildness states. I have taken these foundational frameworks one step further by testing whether their theoretical underpinnings have efficacy as a regulatory tool. I found significant differences in the median wildness scores of the six pilot species, possibly co-varying negatively with commercial value (*sensu* Dalerum & Miranda 2016, *Supporting Information 3.2*), which demonstrates the ability of the tool to delineate broad patterns between species under different management regimes. Importantly, each species exhibited both wild and non-wild populations (varying from 12% to 84% wildness) across a range of management systems, indicating that wildness can be identified for each species. Similarly, populations of different species co-occurring on the same property often spanned wild and non-wild states. These patterns indicate that wildness would be underestimated if deduced from the commercial value of species or top-down land-use classifications. Conversely, wildness would be overestimated if population size was used as a proxy, as our preliminary results show that local abundance does not correlate with wildness, which may be due to managers using intensive management to increase numbers for commercial or conservation goals. This framework thus enables a bottom-up quantification of wildness, avoiding the pitfalls of qualitative classifications, and can detect differences in wildness patterns between species overall; between populations on properties under different management regimes; and between populations of different species on the same property. This will enable policy-makers to produce more meaningful national assessments and provide a fine-scale species management planning and auditing tool.

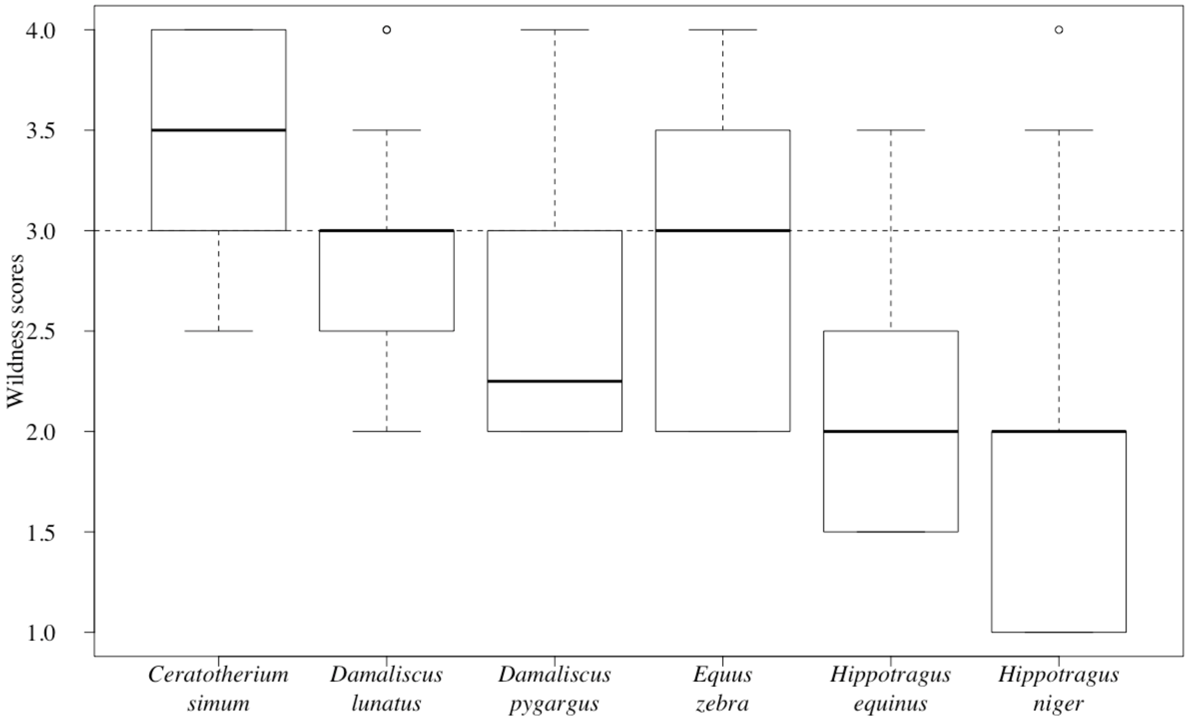

Figure 3.2. Distribution of wildness scores across properties for each pilot species where the threshold for net wild populations is a median score of ≥3 (represented by the horizontal dotted line). Boxes represent first quartile, median (bold line), and third quartile while the dotted lines represent minima and maxima. The median wildness scores and interquartile ranges of each species are: *Ceratotherium simum* 3.5 (3-4) (N = 25); *Damaliscus lunatus* 3 (2.5-3) (N = 23)*; Damaliscus pygargus pygargus* 2.3 (2-3) (N = 18); *Equus zebra* 3 (2.1-3.5) (N = 18); *Hippotragus equinus* 2 (1.5-2.5) (N = 26); and *Hippotragus niger* 2 (1-2) (N = 76).

In line with species conservation guidelines (IUCN Standards and Petitions Subcommittee 2022), we consider wild populations within their indigenous range as possessing conservation value. The framework can thus be used to objectively identify populations that contribute to the conservation of the species and thus included in IUCN Red List assessments, thereby mitigating the often subjective interpretation of the guidelines by different assessors (Hayward et al. 2015). Captive breeding programmes for threatened species or populations managed outside their indigenous range (for example, due to security threats or lack of natural habitat) might also have conservation value and here the framework can be applied to ensure the population remains as wild as possible to facilitate successful reintroduction. Populations outside their natural range, which are not considered of conservation value, can still benefit from the framework by using it to facilitate ecological land management for broader biodiversity benefits. Similarly, as this framework measures the viability of populations, it may also have utility in the newly-developed IUCN Green List of species (Akçakaya et al. 2018), particularly in quantifying and standardising the ecological functionality parameter. Finally, this framework may have relevance in facilitating identification of Other Effective Area-based Conservation Measures (OECMs) to achieve Target 3 of the Global Biodiversity Framework (30% of areas of biodiversity importance under protection) (*Supporting Information 3.5*).

Discerning between wild and non-wild populations will allow policy-makers to create multifunctional landscapes where wildlife can both provide socio-economic opportunity and sustain ecological processes. For example, evaluating wildness will also contribute to the green economy as the framework provides a mechanism to deliver market information to consumers of ecotourism or trophy hunting who are concerned about the sustainability and authenticity of their experience. For example, there is increasing pressure on the hunting industry to demonstrate that the quarry is wild and free-roaming and that hunting contributes to maintenance of wild populations of indigenous species and their habitats, which has resulted in the proposal of a certification scheme for informing consumer choice (Wanger et al. 2017). Additionally, non-wild populations provide economic value in their contribution to the rural economy and food security through game meat markets and associated services (Mysterud 2010; Taylor et al. 2015). The framework thus provides a tool to evaluate multifunctional landscapes based on species wildness patterns and can assist with designing incentives and regulating landowners under green certification schemes. For example, while a property may be specialising in intensive breeding for a certain species, the rest of the property may be extensive and provide conservation benefits for other species. While my framework does not explicitly link to indices of natural habitat, intactness or productivity, the wildness scores can be ultimately aggregated for each property or protected area (if a standardised set of species is assessed) and incorporated into broader biodiversity assessments at landscape scales. For example, the wildness scores can be incorporated into landscape-scale indicators that measure wilderness characteristics (Carver et al. 2013) to prioritise areas for protected area expansion or corridor creation.

This framework is currently most applicable to populations of large vertebrate species that may be directly impacted by management activities (smaller species with high mobility and small home ranges would likely be classified as Self-sustaining). Large vertebrates possess economic value (both consumptive and non-consumptive use) and are thus most often the focal points of management plans and conservation strategies. The way in which they are managed is thus likely to have ramifications for other species and the ecosystem as a whole ("umbrella" species). The attribute scores provide a diagnostic to design appropriate conservation-oriented management plans. For example, protected area managers might use the framework to modify management effectiveness templates so that the data more accurately incorporate the effects of management on species. While our dataset includes private protected areas, future work will survey statutory protected areas to provide baseline wildness evaluations and thus management effectiveness indicators.

I encourage modification of the framework to suit user needs. For the framework to be widely applied across geographic regions and land management systems around the world, it must become less data intensive. Once a larger sample size has been obtained, we can identify attributes that co-vary and the redundant variables can be removed in favour of the covariate that is easier to measure to produce a data-light version of the framework. For example, intensive breeding and veterinary care may co-vary as both are used by managers to produce disease-free Cape buffalo *Syncerus caffer caffer* (Laubscher & Hoffman 2012), meaning data on either reproduction or veterinary care might be used as a proxy for the other. Similar to reducing the attribute load of the framework, the relative explanatory power of each management variable should be explored through statistical modelling and weighted accordingly, as some may be more important in determining wildness. For example, as one of the main mechanisms of natural selection is competition for scarce resources, supplementary feeding may more directly influence the evolutionary dynamics of species than other attributes (reviewed in Oro et al. 2013). Space is also likely to be more influential as wildness scores are negatively correlated with decreasing property size, which is expected as smaller areas require more intensive management. Determining property size thresholds for species of varying body sizes, below which all populations of a particular species can be considered non-wild, will reduce processing time in applying the framework.

A major theme for future research must focus on ground-truthing the wildness states predicted by the framework (*Supporting Information 3.4*). The current evidence demonstrates that captive-bred animals have reduced fitness in unmanaged landscapes (Elsbeth McPhee 2004; Jule et al. 2008; Willoughby et al. 2017), but much work remains to measure the long-term effects of various management intensities on the survival and adaptive capacity of populations across species. One approach is measuring population-level indicators of evolutionary and ecological functioning, such as genetic and trait diversity, and the persistence probability of the population when management interventions are removed or when the animals originating from various wildness states are reintroduced into unmanaged areas. I expect animals at the lower end of the wildness spectrum to have lower chance of long-term persistence, whereas animals at the higher end should have increasingly higher probabilities of survival and persistence over time as these populations should have retained relatively more adaptive capacity. Collecting these data would enable me to calibrate the threshold values, which may lead to collapsing or expanding the number of wildness states.

As wildlife is increasingly brought under human influence, embedding an empirical evaluation of wildness into regulatory processes becomes paramount to counteract the shifting baseline syndrome of the conservation ideal: evolutionary and ecologically dynamic species integrated into functioning ecosystems. My foundational framework standardises the measurement of the wildness of managed large vertebrate populations at the property scale and conceptually aligns management with the overarching goal of sustaining biodiversity and ecosystem functioning. The quantification of wildness also has importance beyond technical measurement for policy and assessment purposes because it represents a more positive and creative conservation agenda. If we fail to articulate, measure and mainstream our conservation ideals, the world will be composed of little more than megalopolises, techno-gardens and zoos, bereft of the wildness needed to sustain human imagination.

Supporting Information 3.1 Explanation of wildness scoring system

The information below provides an explanation of how wildness scores were assigned for each attribute using the empirical thresholds. For each species, the sample of properties containing populations of the species was analysed by reviewing the management data from Taylor et al. (2015) and determining a score for each attribute. Scores were assigned on an ordinal scale, with Captive Managed (CM) receiving a score of 1 and Self-sustaining (SS) receiving a score of 5 (Table S3.4). Only one score per attribute was given, even if there were multiple indicator variables (additional variables were used to corroborate the score given for the attribute).

Species-specific threshold values were used for home range size and social unit size for each biome. Home range and social group unit size for each species in each region were compiled from Skinner and Chimimba (2005), Jones et al. (2009) and Child et al. (2016). Where available, the home range or social unit specific to a particular habitat type was used to calibrate the thresholds. We define a social unit as per Jones et al. (2009) as "the number of individuals in a group that spends the majority of their time in a 24 hour cycle together where there is some indication that these individuals form a social cohesive unit using non-captive populations", which mostly corresponds to the size of a typical breeding herd (*sensu* Skinner & Chimimba 2005) but we construe the presence of multiple social units on a property as comprising breeding herds as well as other social groups, such as bachelor herds, coalitions or disperser groups (*sensu* Skinner & Chimimba 2005).

The space attribute was measured by two variables: home range size of the species in relation to property size and, secondarily, fence type (indicating dispersal capacity), which was part of the Taylor et al. (2015) dataset. Home range size of the focal species is used to assess whether the property is large enough to accommodate the ecological processes of at least one single social unit, with increasing area available inferred to mean the possibility of establishing multiple territories and dynamic demographical processes operating within the population. Dispersal capacity is also key to demographical processes and was inferred from the type of internal or perimeter fencing around the property where wildlife-friendly or cattle fencing have the lowest probability of hindering movement and electrified game fences the highest (Taylor et al. 2015). Artificial passageways refer to any installed gap in a fence (such as tyres) that may assist dispersal (Weise et al. 2014). Fence type (both external and for breeding camps) is part of the Taylor et al. (2015) dataset. Artificial passageway presence was not possible to quantify at present but will be included in future surveys. Analysing dispersal capacity from the perspective of surrounding land-use compatibility will only be possible once the property cadastre can be identified and fine-scale land-cover data can be generated, which is the subject of ongoing work. Here, the proportion of wildlife-friendly land-uses surrounding the focal property will be quantified to assess dispersal capacity. Alternatively, if the surrounding properties have been evaluated using the framework, the median wildness scores of surrounding properties can also be used to standardise the threshold values for dispersal capacity.

Disease and parasite resistance was measured by the level and frequency of veterinary care given to the population. Veterinary intervention data is part of the Taylor et al. (2015) dataset. The difference between wild and non-wild states corresponded to permanent ongoing veterinary care (through antibiotics, de-worming, cattle dips, etc.) as opposed to periodic vaccinations at population level to mitigate disease outbreaks (i.e. if the population is a threatened species, action is taken, but if a single individual has problems it is left). Pre-emptive vaccinations against all native and non-native diseases, versus periodic reactive vaccinations against non-native diseases only, such as rabies during wild dog *Lycaon pictus* translocations (Vial et al. 2006), were construed as the difference between Simulated Natural (SM) and Near Natural (NN) respectively.SM also includes legislated requirements, such as buffalo vaccinations against foot and mouth disease (Laubscher & Hoffman 2012).

Exposure to natural predation is measured by assessing the probability of exposure to predators, the duration of that exposure and the richness of the predator guild present. We split the predator guild into small carnivores (e.g. mongooses), mesopredators (e.g. jackals and caracals) and apex predators, and assessed whether each functional group was present in the landscape, as indicated by the level of control by the landowner (for example, lethal or live removal), and the frequency of exposure, as indicated by whether the functional type was resident or only occasionally present (these data were available in the Taylor et al. 2015 dataset). We assumed that the presence of multi-species predator guilds has more influence on the evolutionary and ecological dynamics of species than single-species guilds or the absence of some guilds (Linnell & Strand 2000). Here, the difference between “occasional” and “continuous” exposure refers to the assumed frequency of exposure where occasional is infrequent exposure based on active removal or absence from the area, and continuous refers to frequent exposure due to a resident predator population being present with no removal or *ad hoc* removal only (e.g. damage causing animals) by managers. “Exclusion” is through predator proof fencing and landscape level removal. “Removal” is where the manager does not totally exclude predators but they are removed when encountered. “Ad hoc removal” is removal only of damage-causing animals and not all predators when encountered. For NN, there is no predator removal but it may include controlling predator numbers (e.g. through contraception) or mitigating predation impact (e.g. through the use of livestock guarding dogs) may be in place. In the next iteration of the tool, the specific relationships between the focal species and its key predators should be quantified and built into the species-specific parameters of the tool. Similarly, the baseline predator guild of each biome or habitat type should be quantified and converted into a % of the total predator community present (to avoid biases using absolute species number in naturally predator rich versus predator poor areas).

Exposure to natural food limitations and fluctuations was measured primarily by the frequency of supplementary food provision whereby direct continuous food provision was considered CM and periodic provision of food such as hay bales and lucerne *Medicago sativa* pellets at feeding troughs in the landscape was considered IM to NN depending on the frequency of the provision. Specifically, the provision of permanent salt licks and other nutritional supplements in the landscape was considered IM. Providing supplementary food over more than one period per year (e.g. dry spell of winter or a drought) is also IM as its aim is to boost production of the population, whereas an average of once per year is considered SM and assumed to correlate with the dry season when forage shortages are experienced on an annual basis in response to limited areas and the inability for the population to disperse. Food provision only during extreme droughts corresponds to NN. This attribute also includes habitat management techniques that may indirectly influence resource provisioning for the population. These were categorised as ‘production orientated’ or ‘restoration orientated’. The latter was assumed to influence resource availability positively through practices such as alien invasive vegetation removal, erosion control, bush encroachment control (as a result of previous overgrazing in many cases), and the existence of mosaic fire management plans (but see Parr & Andersen 2006). The former (production-orientated) is related to managers using planted food crops such as lucerne, grass pastures or oats to negate fluctuations in food availability, which, together with a block burning regime, may lead to a loss of landscape heterogeneity. Additionally, we assumed that if a population was outside of its natural distribution range that resource quality would be lower, outbreeding depression could occur and the population could negatively impact the habitat for native species. Conserving species inside their natural range is also in line with the IUCN guidelines (IUCN/SSC 2013; IUCN Standards and Petitions Subcommittee 2022). The natural ranges of the species were determined using the maps produced by Birss et al. (2015). If a population exists outside its natural range, it can only correspond to CM, IM or SN for this attribute (regardless of the values of the other variables for the attribute), where SN accommodates situations of “benign introductions” (IUCN/SSC 2013; IUCN Standards and Petitions Subcommittee 2022). Near Natural and SS populations must exist within the natural range for this attribute. In future revisions of the framework, this variable may be weighted more strongly as the framework becomes refined for a planning tool. We consider a wild population within its natural range as possessing conservation value (see Discussion).

Exposure to natural water and limitations and fluctuations was measured primarily by the number of water-points in the landscapes calibrated by the number of home range units of the species. The migrations and dispersals forced by water fluctuations are critical in ecosystem functioning as individuals will transport nutrients, energy and other organisms between locations and enable ecological interactions between species in both space and time (Bauer & Hoye 2014). Subsequent range expansions can feed back into evolutionary processes (Fronhofer & Altermatt 2015). A density of more water-points than the number of home range units was considered IM due to the population not being restricted by water availability. Fewer water points than the number of home range units was considered to limit availability and stimulate movement of animals. Ideally, the spatial configuration of water-points, calibrated by the average distances each species travels for water, should also be considered because even spacing will negate natural movements and possibly decrease habitat heterogeneity overall through habitat degradation or ecological community homogenisation due to making broader areas of the landscape accessible to generalist herbivore and predator species (for example, Owen-Smith 1996; Smit et al. 2007; Cain et al. 2012). However, we do not have detailed geo-spatial data on water-point distribution. Similarly, for the degree of seasonality of the water source (dictated by natural hydrology and local rainfall), higher proportions of artificial to natural water-points were inferred to mean increased water availability throughout the year as artificial water-points are often pumped all year round (Taylor et al. 2015), whereas higher proportions of natural water-points dry out during the dry season (for example, building pans and letting water collect there naturally), allowing for natural fluctuation in water availability and facilitating competition for available ephemeral water sources during the dry season. The effect of rivers in the landscape was not considered. Large vertebrates also vary in their dependence on water. Such factors will be considered in the next iteration of the framework once more data are available.

Reproduction was measured by estimating breeding competition, intraspecific processes and the degree of artificial selection. We looked at two categories of indicator variables: breeding competition and selection specificity. The former was measured by the number of social units that could potentially be present in the population (population size divided by social unit size), as a proxy for breeding competition (multiple males) and intraspecific processes through the presence of different social units (i.e. bachelor or disperser groups). The more social units present in the population, the more we assumed self-sustaining demographic processes could occur (*sensu* Redford et al. 2011). For the latter, selection specificity measures the degree of artificial selection being imposed, as indicated by whether the focal species is the subject of intensive breeding for a specific trait (such as horn length or colour variant; Taylor et al. 2015 and references therein) where mate selection is controlled, or whether natural mate selection is allowed to take place. The presence of non-native ecotypes or subspecies of the focal species was assumed to lead to hybridisation and thus weaken the adaptive capacity of the population (for example, Allendorf et al. 2001). Trophy hunting and live game sales were similarly considered 'selective' off-takes that could disrupt social structures and demographics (and thus decrease sexual selection pressure) due to certain individuals being actively introduced or removed from the population, whereas 'subsistence' hunting and culling (for overall ecosystem management) were considered 'non-selective' off-take and considered less influential on population dynamics. For augmentation, where additional animals are reintroduced into the system, it was considered selective if alien subspecies or ecotypes have been introduced and / or stud males are introduced for breeding, but non-selective if introductions are performed to enhance the genetic diversity of the population, such as metapopulation management where translocation that follows reintroduction guidelines (IUCN/SSC 2013), or when no continued supplementation is necessary after the initial founder event.

In the current analysis, only one score was given per attribute. Where there are multiple indicator variables per attribute, they were used to corroborate the final attribute score. If the scores reflected by the indicator variables in the attribute were not synonymous, the lower score was used to determine the final attribute score. We also note that some indicator variables in the attributes were not possible to fully quantify at present or require further accumulation of baseline data. These include the spatial orientation of water-points, baseline predator communities in each biome and the dispersal capacity of the focal species given the surrounding land-use of the property. For the latter, analysing dispersal capacity in context of the surrounding land-use compatibility will only be possible once the property cadastre can be identified and fine-scale land-cover data can be generated, which is the subject of ongoing work. Here, the proportion of wildlife-friendly land-uses surrounding the focal property will be quantified to assess dispersal capacity. Alternatively, if the surrounding properties have been evaluated using the framework, the median wildness scores of surrounding properties can also be used to standardise the threshold values for dispersal probability. Once these additional data layers are available, scores should be assigned for each variable across the attributes, thereby giving the framework even finer predictive power. While we have used the South African context as a pilot study, the framework has global application and future work should test its efficacy in other geographical regions exhibiting different land-tenure patterns.

We also tested the impact of the accuracy in interpreting the information. Each attribute for each population was scored twice: the first score (the default used in the analyses) represented the best estimate while the second score reflected the alternate possibility given uncertainty in the dataset. There were low levels of possible inaccuracy (93% of differences between score 1 and score 2 were <1; which is less than the distance between two states). The two sets of scores are not significantly on a species level (Mood’s median test, X-squared = 2.5367, df = 1, p-value = 0.11), which shows that the method is robust and not sensitive to low levels of uncertainty in the underlying data.

Table S3.4. Specific explanations of the data used to infer between wildness state thresholds and thus assign wildness scores for each attribute.

Attribute	***Threshold calculations***				
	Captive Managed (CM)	**Intensively Managed (IM)**	**Simulated Natural (SN)**	**Near Natural (NN)**	**Self-sustaining(SS)**
Space	Population exists in exclusionary breeding camps within the property. Non-permeable (predator-proof) fencing around camp. Ecological and demographical processes not possible.	Population not restricted to enclosures or breeding camps but size of property is smaller than the average home range size of the species in the biome. Perimeter fencing impermeable to dispersal by being electrified. Ecological and demographical processes assumed to be severely restricted.	Property size large enough to accommodate the home range of at least 1 social unit, but all social units may not be present (i.e. bachelor or disperser groups) and thus demographical processes may be limited. Porous or non-permeable perimeter fencing allows limited dispersal for some species.	Property size is sufficient to accommodate at least 2 social units**.** Social interactions between groups enabled. Size allows for full suite of ecological interactions. A degree of dispersal and the establishment of new social groups allowed for. Permeable fences or no fences (dependent on size – for example, Kruger National Park is large enough for dispersal needs of most species even though there are boundary fences).	Size of property is sufficient for there to be more home range units available than there are social units present. There is thus always sufficient space for multiple social units where both evolutionary and ecological processes proceed uninhibited. Social units able to track seasonal changes in landscape. Both density dependent and density independent population regulation occurring. Permeable fences or no fences. Dispersal unassisted.
Disease and parasite resistance	Direct provision of antibiotics and to all individuals or direct treatment of injured animals.	Existence of permanent structures in the landscape intended to prevent tick- or parasite-borne diseases (such as Duncan applicators, tick-off machines and livestock dips).	No individual veterinary care but founder groups receive pre-emptive vaccinations against all native and non-native diseases.	No individual veterinary care but founder groups receive reactive vaccinations against non-native diseases only, such as rabies during wild dog *Lycaon pictus* translocations.	No veterinary interventions

Exposure to natural predation	All predators excluded through lethal control (non-selective) or predator-proof fencing.	Apex predators absent, excluded or controlled. Limited exposure to small and mesocarnivores where mesocarnivores actively managed (through hunting and culling).	Occasional exposure to apex predators and continuous exposure to other predator guilds. Apex predators may be subject to contraception control. No active predator control but damage-causing individuals removed non-lethally (live capture and translocation).	All predator functional guilds present. No predator control except occasional live capture and release of damage-causing apex predators.	Full complement of predators present. No predator control.
Exposure to natural food limitations and fluctuations	Continuous supplementary food provision. Whole feeding / nutrient supplements. (No natural forage available).	Lucerne or other forage provided in landscape more than once / annum. Permanent salt licks and other nutritional supplements in landscape. Presence of planted crops to boost on-site forage production.	Lucerne or other forage provided in landscape on average once / annum. No permanent nutritional supplements. At least 1 habitat management technique to restore ecological functions (e.g. alien invasive removal, erosion control, natural fire regimes).	Lucerne or other forage provided in landscape on average less than once / annum (only during severe droughts). No permanent nutritional supplements. At least 2 habitat management techniques to restore ecological functions (e.g. alien invasive removal, erosion control, natural fire regimes).	No supplementary feeding. No permanent nutritional supplements. At least 3 habitat management techniques to restore ecological functions (e.g. alien invasive removal, erosion control, natural fire regimes).
Exposure to natural water and limitations and fluctuations	Continuous water provision in camps. Water available directly to all individuals.	≥1 water-point / home range unit of the species (i.e. each social unit has access to a water point and does not need	<1 water-point / home range unit of the species (i.e. some social units do not have access to water and must	<0.5 water-points / home range unit of the species < 25% water-points are artificial (dams and	<0.25 water-point / home range unit of the species. 100% natural water-points, thus water sources are all seasonal.

		to disperse). ≥ 50% water-points are artificial (dams and boreholes), thus limited seasonality in water availability.	disperse). <50% water-points are artificial (dams and boreholes), thus water sources are predominantly seasonal.	boreholes), thus water sources are predominantly seasonal.	
Reproduction	Presence of ‘stud’ male in camps (1 breeding male / enclosure) (Selective breeding through deliberate mate pairing). Selective breeding for colour variants or specific trait Presence of alien / extra-limital species. Trophy hunting or live sales activities.	Population size < 1 social unit – no natural breeding competition =. Selective breeding for colour variants or specific traits. Presence of alien / extra-limital species. Trophy hunting or live sales activities.	Population size comprises at least 1 social unit (i.e. natural competition for mates). No selective breeding in place. No alien / extra-limital species. Subsistence hunting or culling for habitat management.	Population size ≥ 2 social units. No selective breeding in place. No alien / extra-limital species. Subsistence hunting or culling for habitat management.	Population size ≥ 3 social units . No selective breeding in place. No alien / extra-limital species. Subsistence hunting or culling for habitat management.

Supporting Information 3.2 Wildness scores: full results and summary tables

The individual attribute score for each population of each species is displayed in Table S3.5. Wildness scores are calculated as the median and interquartile range (IQR) of the attribute scores. As the IQR can be symmetrical or asymmetrical around the median, it is reported as a range (Quartile 1 – Quartile 3), rather than a single value, to help interpret the confidence in the wildness status of each population. The scoring process was as follows:

1. We identified evolutionary and ecological attributes of populations that may be affected by management interventions and derived a set of indicator variables that could be used to measure the potential impacts of management on the attributes (Table 3.1).
2. We identified wildness states (or nodes) along the wildness spectrum (adapted from Redford et al. 2011) and defined these Table 3.2. Identifying discrete states is necessary to develop quantitative thresholds to make the framework measurable.
3. We used the indicator variables of each attribute to set empirical thresholds between wildness states (Table 3.2).
4. The thresholds were used to assign a score for each attribute for a given population (see *Supporting Information 3.1*), which were ordinal values corresponding to the identified wildness states. For example, if a population was kept in breeding camps on a particular property, it would score 1 for the Space attribute (corresponding to the Captive Managed state) whereas if the space available to the population was more than one home range unit per herd (i.e. there is enough space for normal demographical and ecological processes to occur) then the population would score a 5 on the space attribute corresponding to the Self-sustaining wildness state (see Table S3.5).

Across all species, there are 63 (34%) wild populations. Of these, 39 (62%) populations had wildness scores and IQR ≥3, while the remaining 24 (38%) populations have at least one attribute ≤2 (in all attributes besides Space). Of the 123 non-wild populations, 95 (77%) had a wildness score and IQR of <3. The Space attribute was an anchor score as no population scoring a 1 (Captive Managed) or 2 (Intensively Managed) had a wildness score of ≥3 on net.

Table S3.5. The attributes scores for each population of each focal species used to pilot the framework. Attribute scores were assigned using the empirical thresholds between wildness states (Table 3), where Captive Managed = 1; Intensively Managed = 2; Simulated Natural = 3, Near Natural = 4; and Self-sustaining = 5. The overall wildness score of each population is the median of the attribute scores. The interquartile range of the wildness scores is shown through quartile 1 (Q1) and quartile 3 (Q3).

POPULATION CHARACTERISTICS				**ATTRIBUTE SCORES**						**WILDNESS SCORE**			
Species	**Property ID**	**Property size (km²)**	**Population size**	**Space**	**Disease**	**Predator**	**Food**	**Water**	**Breeding**	**Wildness score**	**Q1**	**Q3**	**Wildness State**
Ceratotherium simum	SP41	30	5	5	4	3	4	4	3	4	3.25	4	Near Natural
Ceratotherium simum	ALR41	150	36	5	4	4	4	2	4	4	4	4	Near Natural
Ceratotherium simum	ALR43	90	21	5	4	4	4	2	4	4	4	4	Near Natural
Ceratotherium simum	ALR49	142	46	4	4	4	4	2	4	4	4	4	Near Natural
Ceratotherium simum	ALR31	24	4	5	4	4	2	4	2	4	2.5	4	Near Natural
Ceratotherium simum	SP17	20	4	4	4	4	2	4	2	4	2.5	4	Near Natural
Ceratotherium simum	SP32	38	24	4	4	4	4	4	4	4	4	4	Near Natural
Ceratotherium simum	AT04	1 030	75	5	4	4	3	3	4	4	3.25	4	Near Natural
Ceratotherium simum	JM03	540	12	3	4	4	3	4	4	4	3.25	4	Near Natural
Ceratotherium simum	JM51	120	6	3	4	4	5	2	3	3.5	3	4	Simulated Natural
Ceratotherium simum	JM54	75	3	3	4	4	4	3	2	3.5	3	4	Simulated Natural
Ceratotherium simum	SP21	190	43	3	3	3	4	4	4	3.5	3	4	Simulated Natural
Ceratotherium simum	AT01	330	21	5	4	3	3	2	4	3.5	3	4	Simulated Natural
Ceratotherium simum	SP06	55	5	3	4	3	2	2	3	3	2.25	3	Simulated Natural
Ceratotherium simum	ALR6	17	9	4	2	3	3	3	4	3	3	3.75	Simulated Natural
Ceratotherium simum	ALR28	32	3	5	2	5	2	4	2	3	2	4.75	Simulated Natural
Ceratotherium simum	ALR36	35	12	5	2	4	2	2	4	3	2	4	Simulated Natural
Ceratotherium simum	SP12	14	6	3	4	4	3	3	3	3	3	3.75	Simulated Natural
Ceratotherium simum	AT05	11	13	3	2	4	2	3	4	3	2.25	3.75	Simulated Natural
Ceratotherium simum	JM10	200	5	5	4	3	3	2	2	3	2.25	3.75	Simulated Natural
Ceratotherium simum	AT03	167	150	4	4	3	2	2	3	3	2.25	3.75	Simulated Natural
Ceratotherium simum	ALR39	363	130	4	2	3	2	2	4	2.5	2	3.75	Intensively Managed

Ceratotherium simum	SP25	25	5	4	2	3	2	3	2	2.5	2	3	Intensively Managed
Ceratotherium simum	JM09	30	2	5	4	2	2	3	2	2.5	2	3.75	Intensively Managed
Ceratotherium simum	JM15	110	5	5	2	2	3	2	3	2.5	2	3	Intensively Managed
Damaliscus lunatus	ALR41	150	4	5	4	4	4	2	2	4	2.5	4	Near Natural
Damaliscus lunatus	AT04	1 030	98	5	4	4	4	3	4	4	4	4	Near Natural
Damaliscus lunatus	ALR28	32	5	4	3	4	3	4	2	3.5	3	4	Simulated Natural
Damaliscus lunatus	ALR39	363	130	5	2	4	3	3	4	3.5	3	4	Simulated Natural
Damaliscus lunatus	JM67	65	21	5	4	3	3	4	3	3.5	3	4	Simulated Natural
Damaliscus lunatus	ALR10	45	12	5	3	3	3	2	4	3	3	3.75	Simulated Natural
Damaliscus lunatus	ALR11	13	6	4	3	3	3	3	2	3	3	3	Simulated Natural
Damaliscus lunatus	ALR46	150	3	5	3	4	3	2	1	3	2.25	3.75	Simulated Natural
Damaliscus lunatus	ALR50	18	35	4	3	3	3	2	3	3	3	3	Simulated Natural
Damaliscus lunatus	JM10	200	3	5	4	2	4	2	1	3	2	4	Simulated Natural
Damaliscus lunatus	JM15	110	30	5	3	2	3	2	3	3	2.25	3	Simulated Natural
Damaliscus lunatus	AT03	167	200	4	3	3	3	2	3	3	3	3	Simulated Natural
Damaliscus lunatus	AT05	11	30	3	2	4	2	3	3	3	2.25	3	Simulated Natural
Damaliscus lunatus	AT31	43	60	4	3	3	3	2	3	3	3	3	Simulated Natural
Damaliscus lunatus	AT32	50	28	5	3	2	3	2	3	3	2.25	3	Simulated Natural
Damaliscus lunatus	ALR12	12	40	4	2	2	3	2	4	2.5	2	3.75	Intensively Managed
Damaliscus lunatus	ALR23	16	17	1	2	3	3	3	1	2.5	1.25	3	Intensively Managed
Damaliscus lunatus	JM11	52	8	5	4	2	3	2	2	2.5	2	3.75	Intensively Managed
Damaliscus lunatus	JM16	35	12	5	4	2	2	2	3	2.5	2	3.75	Intensively Managed
Damaliscus lunatus	VvdM01	150	3	5	2	4	3	2	1	2.5	2	3.75	Intensively Managed
Damaliscus lunatus	AT10	16	5	3	2	3	2	3	2	2.5	2	3	Intensively Managed
Damaliscus lunatus	AT15	4	20	3	3	4	2	2	2	2.5	2	3	Intensively Managed
Damaliscus lunatus	AT16	120	50	1	3	2	2	2	1	2	1.25	2	Intensively Managed
Damaliscus pygargus pygargus	JM02	9	27	4	4	3	3	4	4	4	3.25	4	Near Natural
Damaliscus pygargus pygargus	JM36	250	30	4	4	4	3	2	4	4	3.25	4	Near Natural

Damaliscus pygargus pygargus	JM23	240	60	4	3	2	4	2	4	3.5	2.25	4	Simulated Natural
Damaliscus pygargus pygargus	JM01	2	15	4	3	3	2	4	3	3	3	3.75	Simulated Natural
Damaliscus pygargus pygargus	JM47	200	100	4	3	2	3	3	4	3	3	3.75	Simulated Natural
Damaliscus pygargus pygargus	JM61	60	25	4	4	3	2	2	3	3	2.25	3.75	Simulated Natural
Damaliscus pygargus pygargus	JM11	52	30	1	3	2	2	3	4	2.5	2	3	Intensively Managed
Damaliscus pygargus pygargus	JM15	110	30	3	2	2	3	2	3	2.5	2	3	Intensively Managed
Damaliscus pygargus pygargus	JM66	105	100	4	4	2	2	2	3	2.5	2	3.75	Intensively Managed
Damaliscus pygargus pygargus	JM10	200	9	2	3	2	3	2	1	2	2	2.75	Intensively Managed
Damaliscus pygargus pygargus	JM26	12	15	4	2	2	2	2	3	2	2	2.75	Intensively Managed
Damaliscus pygargus pygargus	JM34	28	42	1	2	2	3	2	2	2	2	2	Intensively Managed
Damaliscus pygargus pygargus	JM45	170	12	4	2	2	2	2	2	2	2	2	Intensively Managed
Damaliscus pygargus pygargus	SM03	130	46	2	4	2	2	3	2	2	2	2.75	Intensively Managed
Damaliscus pygargus pygargus	SP08	1	8	4	2	2	2	2	2	2	2	2	Intensively Managed
Damaliscus pygargus pygargus	AT03	167	150	4	2	2	3	2	2	2	2	2.75	Intensively Managed
Damaliscus pygargus pygargus	AT16	120	10	2	3	2	2	2	2	2	2	2	Intensively Managed
Damaliscus pygargus pygargus	AT21	20	39	2	2	2	2	2	2	2	2	2	Intensively Managed
Equus zebra	JM36	250	61	5	4	3	4	3	4	4	3.25	4	Near Natural
Equus zebra	JM51	120	71	5	4	3	4	3	4	4	3.25	4	Near Natural
Equus zebra	AT04	1 030	199	4	3	4	4	3	5	4	3.25	4	Near Natural
Equus zebra	JM03	540	3	5	3	3	4	4	2	3.5	3	4	Simulated Natural
Equus zebra	JM61	60	30	4	4	3	2	3	4	3.5	3	4	Simulated Natural
Equus zebra	JM67	65	7	4	4	3	2	4	3	3.5	3	4	Simulated Natural
Equus zebra	JM23	240	35	5	5	2	3	2	3	3	2.25	4.5	Simulated Natural
Equus zebra	JM66	105	45	4	4	2	2	2	4	3	2	4	Simulated Natural
Equus zebra	AT03	167	40	4	3	3	3	2	4	3	3	3.75	Simulated Natural
Equus zebra	AT16	25	60	3	3	3	2	2	4	3	2.25	3	Simulated Natural
Equus zebra	JM10	200	8	5	5	2	2	2	3	2.5	2	4.5	Intensively Managed
Equus zebra	JM11	52	30	4	4	2	2	2	3	2.5	2	3.75	Intensively Managed

Equus zebra	JM19	40	40	3	4	2	2	2	3	2.5	2	3	Intensively Managed
Equus zebra	JM15	110	40	5	2	2	2	2	4	2	2	3.5	Intensively Managed
Equus zebra	JM26	12	5	3	2	2	2	2	3	2	2	2.75	Intensively Managed
Equus zebra	JM35	20	10	4	2	2	2	2	4	2	2	3.5	Intensively Managed
Equus zebra	AT15	4	5	2	2	3	2	2	3	2	2	2.75	Intensively Managed
Equus zebra	AT16	120	20	2	3	2	2	3	2	2	2	2.75	Intensively Managed
Hippotragus equinus	JM46	280	5	5	4	5	3	3	2	3.5	3	4.75	Simulated Natural
Hippotragus equinus	JM67	65	37	3	4	3	3	5	4	3.5	3	4	Simulated Natural
Hippotragus equinus	JM11	52	10	3	3	2	3	2	3	3	2.25	3	Simulated Natural
Hippotragus equinus	ALR12	12	49	1	3	1	2	3	3	2.5	1.25	3	Intensively Managed
Hippotragus equinus	AT03	167	350	2	3	3	3	2	1	2.5	2	3	Intensively Managed
Hippotragus equinus	AT04	1 030	31	2	3	3	2	2	3	2.5	2	3	Intensively Managed
Hippotragus equinus	AT32	50	18	2	3	3	3	2	2	2.5	2	3	Intensively Managed
Hippotragus equinus	JM61	60	8	3	4	2	3	2	2	2.5	2	3	Intensively Managed
Hippotragus equinus	ALR36	35	20	2	2	2	2	2	2	2	2	2	Intensively Managed
Hippotragus equinus	ALR6	17	14	2	2	2	2	2	3	2	2	2	Intensively Managed
Hippotragus equinus	AT19	3	40	2	2	3	2	2	2	2	2	2	Intensively Managed
Hippotragus equinus	AT21	20	2	2	2	2	2	2	1	2	2	2	Intensively Managed
Hippotragus equinus	AT28	10	2	2	3	3	2	2	1	2	2	2.75	Intensively Managed
Hippotragus equinus	SP21	190	20	1	2	1	2	3	2	2	1.25	2	Intensively Managed
Hippotragus equinus	SP30	5	1	2	2	2	4	3	1	2	2	2.75	Intensively Managed
Hippotragus equinus	ALR10	45	45	1	2	1	3	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	ALR39	363	200	1	3	1	2	3	1	1.5	1	2.75	Captive Managed
Hippotragus equinus	ALR40	15	85	1	3	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	AT06	11	30	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	AT08	9	10	1	2	1	3	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	AT16	120	50	1	4	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	AT18	10	30	1	2	1	2	2	1	1.5	1	2	Captive Managed

Hippotragus equinus	AT37	20	80	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	JM10	200	34	1	2	1	3	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	JM15	110	70	1	2	1	3	2	1	1.5	1	2	Captive Managed
Hippotragus equinus	JM45	170	10	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	JM36	250	7	4	4	4	4	3	2	4	3.25	4	Near Natural
Hippotragus niger	SP19	21	1	3	4	4	4	2	1	3.5	2.25	4	Simulated Natural
Hippotragus niger	AT34	950	24	5	3	3	5	4	3	3.5	3	4.75	Simulated Natural
Hippotragus niger	AT17	23	9	3	4	2	3	3	1	3	2.25	3	Simulated Natural
Hippotragus niger	AT31	43	60	4	3	3	3	2	3	3	3	3	Simulated Natural
Hippotragus niger	ALR3	46	5	5	2	4	3	3	1	3	2.25	3.75	Simulated Natural
Hippotragus niger	AT32	50	40	4	2	3	3	2	3	3	2.25	3	Simulated Natural
Hippotragus niger	JM67	65	124	3	4	3	2	4	3	3	3	3.75	Simulated Natural
Hippotragus niger	JM32	17	22	3	4	3	2	2	2	2.5	2	3	Intensively Managed
Hippotragus niger	SP13	35	5	2	3	2	4	4	1	2.5	2	3.75	Intensively Managed
Hippotragus niger	JM66	105	80	4	4	2	2	2	3	2.5	2	3.75	Intensively Managed
Hippotragus niger	AT15	4	6	1	2	2	2	2	1	2	1.25	2	Intensively Managed
Hippotragus niger	SP30	5	109	1	2	2	4	4	1	2	1.25	3.5	Intensively Managed
Hippotragus niger	ALR22	5	5	2	2	2	3	2	1	2	2	2	Intensively Managed
Hippotragus niger	AT21	6	30	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	AT33	6	3	2	2	2	2	2	1	2	2	2	Intensively Managed
Hippotragus niger	AT36	8	50	2	2	2	2	2	2	2	2	2	Intensively Managed
Hippotragus niger	JM02	9	3	2	3	3	2	2	1	2	2	2.75	Intensively Managed
Hippotragus niger	JM39	9	5	2	4	2	3	2	2	2	2	2.75	Intensively Managed
Hippotragus niger	AT05	11	5	2	2	2	2	4	1	2	2	2	Intensively Managed
Hippotragus niger	ALR12	12	98	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	JM26	12	3	2	2	2	2	2	1	2	2	2	Intensively Managed
Hippotragus niger	ALR11	13	100	1	2	1	3	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	AT35	14	12	3	2	2	2	2	1	2	2	2	Intensively Managed

Hippotragus niger	AT14	16	5	2	3	3	2	2	1	2	2	2.75	Intensively Managed
Hippotragus niger	AT27	20	60	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	AT37	20	200	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	ALR25	23	7	3	3	2	2	2	1	2	2	2.75	Intensively Managed
Hippotragus niger	AT34a	24	10	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	SP37	32	80	1	2	2	3	2	1	2	1.25	2	Intensively Managed
Hippotragus niger	JM16	35	2	2	3	2	2	2	1	2	2	2	Intensively Managed
Hippotragus niger	ALR10	45	60	1	2	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	JM11	52	25	2	3	2	2	2	4	2	2	2.75	Intensively Managed
Hippotragus niger	AT42	55	6	1	4	2	2	2	1	2	1.25	2	Intensively Managed
Hippotragus niger	SP06	55	2	3	2	3	2	2	1	2	2	2.75	Intensively Managed
Hippotragus niger	JM61	60	25	1	4	2	3	2	2	2	2	2.75	Intensively Managed
Hippotragus niger	JM15	110	70	1	2	1	3	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	SM03	130	25	1	4	1	2	3	2	2	1.25	2.75	Intensively Managed
Hippotragus niger	ALR46	150	105	1	3	1	2	2	2	2	1.25	2	Intensively Managed
Hippotragus niger	AT03	167	500	1	3	1	3	2	2	2	1.25	2.75	Intensively Managed
Hippotragus niger	SP21	190	30	1	2	2	2	2	2	2	2	2	Intensively Managed
Hippotragus niger	AT04	1 030	88	1	4	2	2	2	2	2	2	2	Intensively Managed
Hippotragus niger	AT22	2	10	2	4	1	2	1	1	1.5	1	2	Captive Managed
Hippotragus niger	ALR8	5	12	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	ALR5	9	80	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	AT12	10	40	1	2	1	2	1	2	1.5	1	2	Captive Managed
Hippotragus niger	AT28	10	70	1	2	2	1	1	2	1.5	1	2	Captive Managed
Hippotragus niger	AT31	14	100	1	3	1	3	2	1	1.5	1	2.75	Captive Managed
Hippotragus niger	SP12	14	6	1	2	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	ALR40	15	120	2	2	1	1	1	2	1.5	1	2	Captive Managed
Hippotragus niger	JM20	16	16	1	3	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	ALR50	18	60	2	2	1	1	1	2	1.5	1	2	Captive Managed

Hippotragus niger	AT16	120	65	1	4	2	2	1	1	1.5	1	2	Captive Managed
Hippotragus niger	AT19	180	52	1	2	1	1	2	2	1.5	1	2	Captive Managed
Hippotragus niger	JM10	200	60	1	2	1	1	2	2	1.5	1	2	Captive Managed
Hippotragus niger	JM46	280	100	1	4	1	2	2	1	1.5	1	2	Captive Managed
Hippotragus niger	ALR14	4	2	1	2	1	1	2	1	1	1	1.75	Captive Managed
Hippotragus niger	AT41	5	7	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	AT08	9	25	1	2	1	3	1	1	1	1	1.75	Captive Managed
Hippotragus niger	ALR34	10	20	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	AT18	10	15	1	4	1	1	2	1	1	1	1.75	Captive Managed
Hippotragus niger	AT06	11	100	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	ALR23	16	2	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	ALR6	17	14	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	ALR35	19	60	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	AT21	20	11	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	SP25	25	168	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	JM24	25	7	1	1	1	2	2	1	1	1	1.75	Captive Managed
Hippotragus niger	ALR36	35	35	1	2	1	1	1	2	1	1	1.75	Captive Managed
Hippotragus niger	ALR7	36	27	1	2	1	1	1	1	1	1	1	Captive Managed
Hippotragus niger	ALR30	40	36	1	2	1	1	1	2	1	1	1.75	Captive Managed
Hippotragus niger	ALR32	50	65	1	2	1	1	1	2	1	1	1.75	Captive Managed
Hippotragus niger	JM57	80	31	1	1	1	3	1	1	1	1	1	Captive Managed
Hippotragus niger	JM51	120	25	1	4	1	2	1	1	1	1	1.75	Captive Managed
Hippotragus niger	JM45	170	160	1	1	2	2	1	1	1	1	1.75	Captive Managed
Hippotragus niger	ALR39	363	700	1	2	1	1	1	2	1	1	1.75	Captive Managed

The median wildness scores across populations for each species are summarised in Table S3.6. The framework was applied to all populations of the six pilot species. The commercial value of the species is taken from the average game auction prices (114 auctions) in 2014 (Dalerum & Miranda 2016). "Populations" refers to the number of properties in the dataset in which the species occurred.

Table S3.6. Summary table of the number of properties on which each focal species occurred (populations) and median wildness scores with interquartile range (IQR), proportion of wild populations and average commercial value (ordered from highest to lowest).

Species	**Population sample (N)**	**Median wildness score (IQR)**	**Wild populations (%)**	**Commercial value 2014 (USD)**
Hippotragus equinus	26	2 (1.6-2)	12	37,943
Hippotragus niger	76	2 (1-2)	11	36,529
Ceratotherium simum	25	3.5 (3-4)	84	28,969
Damaliscus pygargus pygargus	18	2.3 (2-3)	33	2,804
Equus zebra	18	3 (2.1-3.5)	56	1,288
Damaliscus lunatus	23	3 (2.5-3)	65	1,270

Supporting Information 3.3 Ordered logistic regression results

Ordered logistic regression was used to determine whether population size and / or property size had significant explanatory power in determining the wildness statuses of populations across species. Ordered logistic regression is an appropriate model as the wildness statuses were ranked from least to most wild and the independent variables were both continuous. As such, the information contained in the wildness state ranks is retained (unlike multinomial logistic regression that assumes no order). Model fitting was done using the *polr* function in the MASS library of SPLUS (Venables & Ripley 2002). The final model included both main effect terms, plus an interaction term. Statistical significance was assessed using likelihood-ratio tests of nested model subsets (Table S3.7). Similarly, the effect size of different species contributing to the model was assessed by comparing the t-values for each species against the standard normal distribution. Species effects for the best fit model (property size alone) are shown under intercept coefficients (Table S3.7).

The model shows that wildness is not influenced by population size for any species (overall $p = 0.21$, Figure S3.3) but that wildness tends to be higher with increasing property size ($p < 0.01$, Figure S3.4). However, species identity is an important covariate, both as a main effect and in interaction with property size where only the wildness scores of the following species are significantly correlated with property size: *Damaliscus pygargus pygargus* ($p < 0.01$), *Equus zebra* (p < 0.001) and *Hippotragus niger* (p < 0.0001) (Figure S3.4). However, we need a larger sample size for each species to test these patterns comprehensively. The best fit model (lowest AIC value) is the relationship between wildness and property size alone. Overall, populations existing on larger properties are likely to be wilder, but we need a higher sample size to test this relationship and the explanatory size of property size is insufficient to predict local wildness status, as wildness is determined by many other management factors and species being restricted to breeding camps is a confounding variable that negates overall property size.

Table S3.7. Best fit model results of the ordered logistic regression between wildness scores, population size and property size, including the values of regression coefficients and intercepts.

Model coefficients	**Value**	**Standard error**
Population size	0.0087	0.00082
Property size	0.0033	0.0075
Intercept coefficients	**Value**	**Standard error**
1 |1.5	-0.8521	0.4604
1.5 | 2	0.3313	0.4444
2 |2.5	1.9426	0.4706
2.5 | 3	2.5611	0.4827
3.5|4	3.6806	0.5134
4 | 4.5	4.4696	0.5521
Species-specific coefficients		
Damaliscus lunatus	0.434630	0.5233557
Damaliscus pygargus pygargus	1.572751	0.5461268
Equus zebra	2.116281	0.5573004
Hippotragus equinus	-0.663235	0.4813800
Hippotragus niger	1.714122	0.4382172

Term removed	**Df**	**AIC**	**Residual deviance**	**Chi Sq Statistic**
Full model		647	621.6	
Without property size	1	664	640.1	< 0.01
Without population size	1	646	622.98	0.24

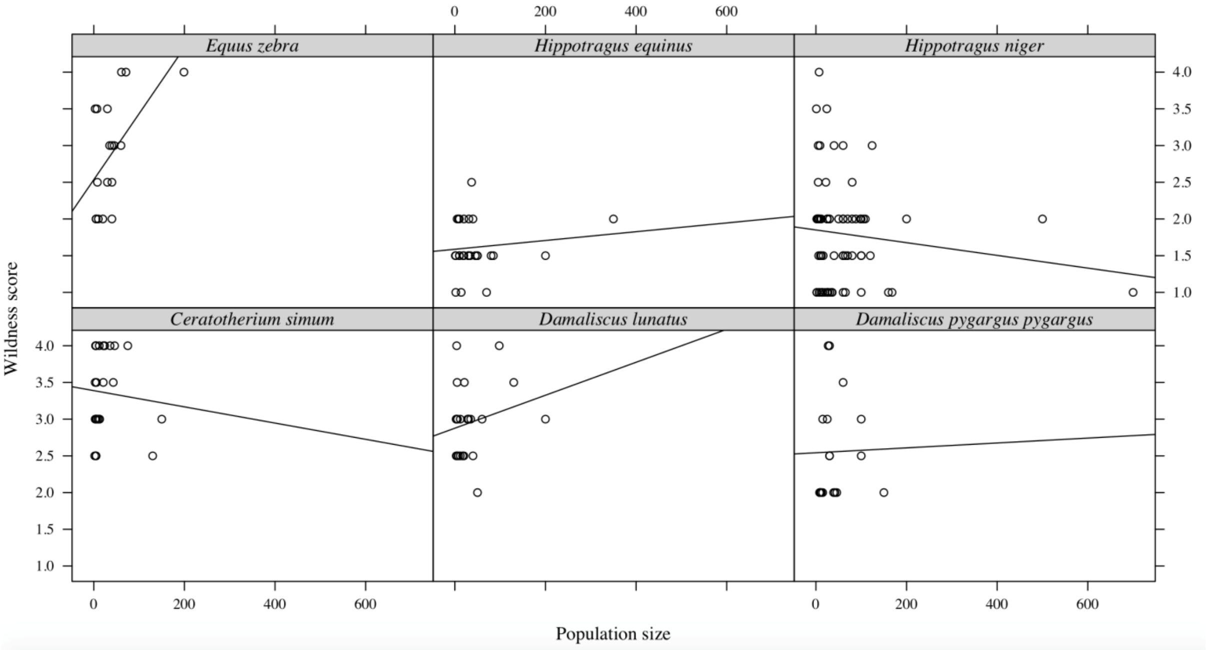

Figure S3.3. Wildness scores against population size for the six focal species where ordered logistic regression detected no significant relationship for any species (all $p > 0.05$).

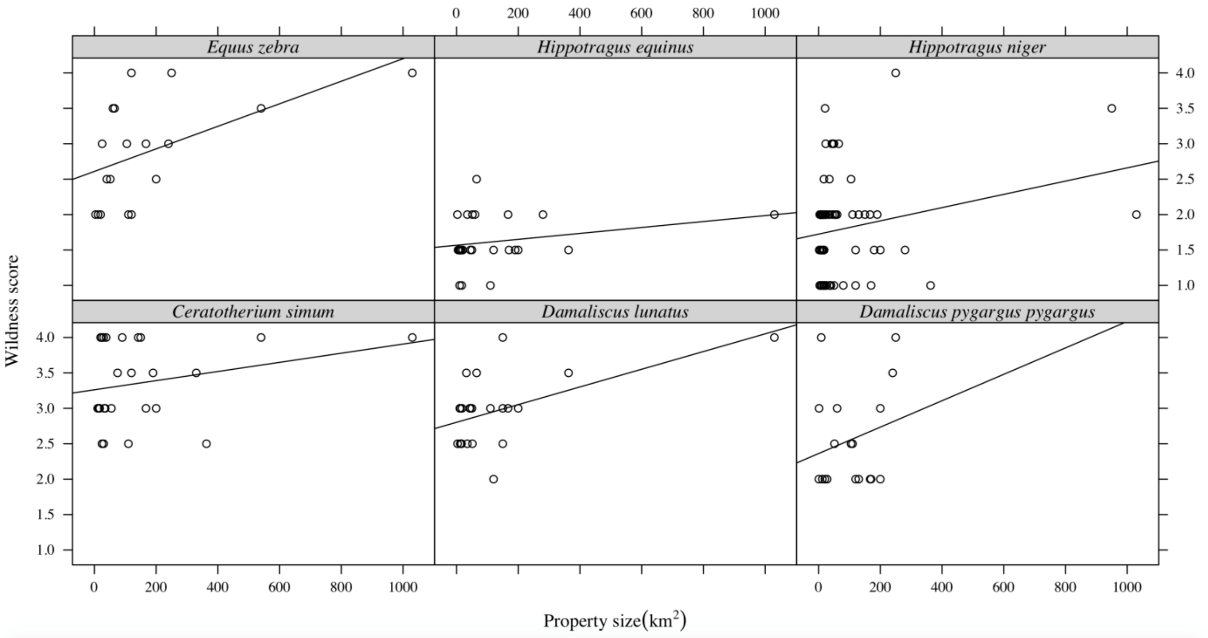

Figure S3.4. Wildness scores plotted against property size for the 6 focal species where ordered logistic regression revealed *D. p. pygargus*, *E. zebra* and *H. equinus* had significantly positive relationships between wildness scores and property size ($p < 0.01$).

Supporting Information 3.4 The potential mainstreaming value of the framework

The future of many wildlife populations appears to hang delicately between local extinction and domestication. While conservationists pin their hopes on other effective area-based conservation measures (OECMs) to plug the gaps between protected areas and fulfil ambitious post-2020 targets (Dudley et al. 2018), there is a growing realisation that these must comprise "working lands" that both support biodiversity and provide socio-economic benefits (Kremen & Merenlender 2018). It will be difficult for land managers, communities and policy-makers to reconcile these potentially conflicting mandates and create nuanced sustainable use regulations at landscape scales without undermining ecosystem functioning, human well-being, or both. The wildness framework can help navigate between these two poles by developing a standardised measurement of the potential wildness of populations under different management objectives. This provides a practical, quantitative tool that unlocks economic value from wildlife whilst preventing the erosion of natural variation and thus social-ecological resilience (*sensu* Holling & Meffe 1996).

In South Africa, the framework can be applied as a response to the perverse incentive to intensively breed wildlife. The potential impacts of management activities (such as supplementary feeding, mate selection, fencing, predator persecution and veterinary care) on the wildness of herbivore populations were set to measurable thresholds in Table 3.3. While the South African government is investing in the wildlife economy to support socio-economic development (Department of Environmental Affairs 2016), no guidelines exist to link management activities to biodiversity outcomes on a population level. Simple assessments based on the wildness framework, however, could help managers adapt management activities to improve the conservation benefits of their economic activities, and assist state authorities monitor progress towards biodiversity and sustainability targets on working lands and help managers adapt management activities to qualify for financial incentives under biodiversity stewardship schemes (Figure S3.5). Frameworks based on ordinal scoring systems have precedent in other policy processes, such as the non-detriment findings conducted by the Scientific Authority to assess the extinction risk of species under international trade regulated by CITES[5]. As such, a property-level wildness assessment would be congruent with, and aggregate into, national-scale risk assessments for species.

The simplicity of the framework's scoring and thresholds is necessary to establish it as a 'boundary object' at the science-policy interface (see Chapter 1). One of the strengths of the wildness framework is to mitigate conflation of abundance with wildness, as was done in Cromsigt et al. (2018). Population size is not a proxy for wildness. In many cases, increased population sizes are the result of anthropogenic influence either directly (by managing specifically for production in the South African game ranching case) or indirectly through ecosystem degradation and disturbance where generalist species tend to dominate (for example, Oro et al. 2013). The wildness framework explicitly counters the conflation of artificially increased abundance with 'conservation success' by measuring the degree to which the population is integrated into its ecosystem. While the framework does not explicitly incorporate minimum viable population (MVP) size, it does use social unit size as a threshold to determine the freedom of mate choice and surpassing the minimum social group needed to maintain mate selection in our framework and be classified as wild will be synonymous with surpassing MVP.

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[5] Example of non-detriment finding from the Scientific Authority of South Africa

For example, when assessing the difference in size between those populations classified as wild and the MVP sizes estimated by Hilbers et al. (2017) for the six focal species, 32% of populations were between 2 and 8 individuals lower than the modelled viability threshold and the remainder were considerably higher than the viability threshold (Figure S3.6). This indicates that the assessed wild populations are mostly viable or close to being viable and demonstrates how the framework can be used to prioritise wild populations that need conservation efforts (such as increasing connectivity or reintroduction schemes).

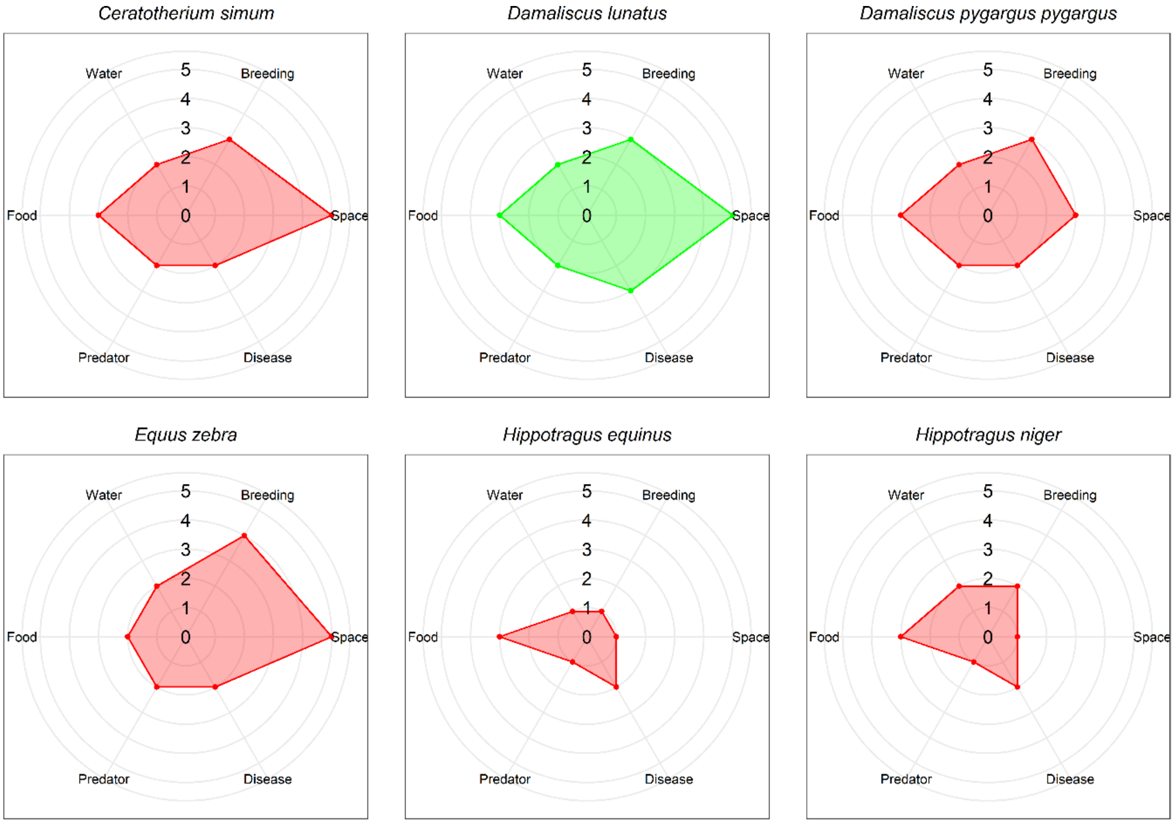

Figure S3.5. An example of a hypothetical wildness dashboard for 6 managed herbivore species for a specific property. The visualisation of wildness space can assist provincial authorities in rapidly determining which populations are wild (green) and thus meet the requirements of policies that regulate sustainable use and harvesting of wildlife populations, as well as which properties qualify for incentives such as tax deductions and payment from ecosystem services schemes.

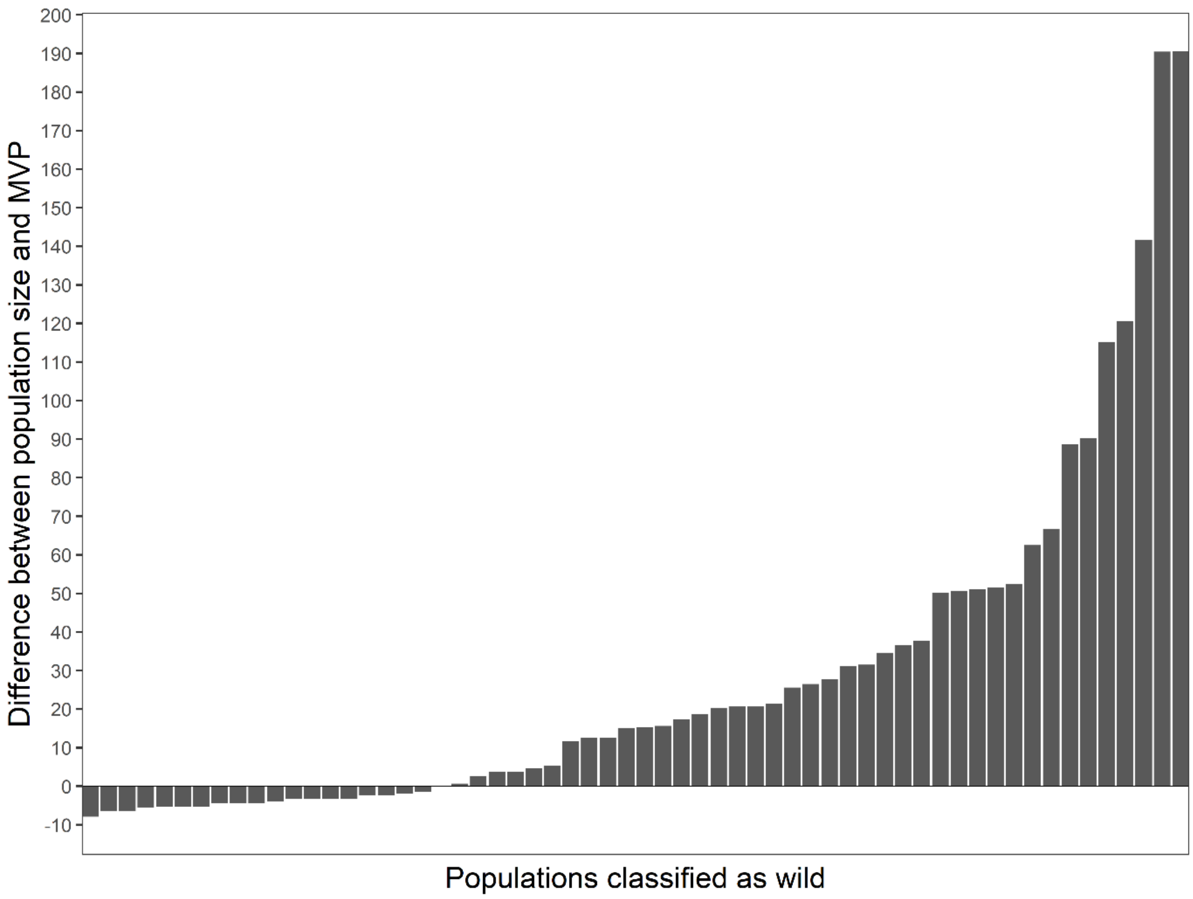

Figure S3.6. Difference between the size of populations classified as wild, and the minimum viable population (MVP) estimates for those species modelled by Hilbers et al. (2017). Negative values indicate populations that are less than the estimated MVP whereas positive values are sufficient for an MVP.

The framework will need to be revised over time with experimental evidence to calibrate thresholds between different wildness states. The wildness framework attempts to link wildness to adaptive capacity of populations and hypothesises that populations categorised as wild will have higher probabilities of persistence than non-wild populations if translocated to unmanaged landscapes or if management becomes less intensive. This hypothesis can and must be validated through experimental evidence. For example, Selebatso et al. (2018) document how blue wildebeests (*Connochaetes taurinus*) herds accustomed to artificial water provisioning died in the dry season when these water sources failed as they had lost adaptive foraging knowledge, whereas a herd without access to artificial water retained the behavioural trait to search for water-rich fruit bodies (e.g. tsamma melons *Citrullus lanatus*) and survived. This presents empirical evidence for the importance of the "exposure to natural water limitations and fluctuations" attribute. The framework can thus be used to design studies that generate evidence for the impact of varying management intensity when herds are translocated from one system to another or management activities change through adaptive management cycles; and to assess the relative importance of various attributes in retaining adaptive capacity. Collating this evidence would help quantify the uncertainty around the wildness scores as well as being able to weight the relative importance of different attributes in determining wildness (which are currently all weighted equally). Research should explore the links between interventions that may reduce wildness but improve ecosystem functioning in novel ecosystems. For example, while supplementary feeding in the wildness framework is expected to reduce adaptive capacity and thus wildness, it could be used as a tool to improve nutrient cycling and reduce bush encroachment. Mkhize et al. (2016) found that supplementing goat herds with a high-energy and high-protein food sources (maize and soybean meal) led to increased time spent browsing than herds that were not supplemented because the supplementation replaced the nutrients that are normally rendered indigestible by tannins in shrub species, which means that supplementary feeding could be used to increase the rate of browsing by mixed feeders, rendering them more effective at bush encroachment control. If these results were generalisable to wildlife species in simulated natural wildness states, perhaps supplementary feeding could be evaluated against improved ecosystem functioning depending on the goals of the management system. The wildness framework, in tandem with an impact evaluation framework for the social-ecological system overall, could help to measure the trade-offs and net gains in wildness and broader ecosystem restoration.

Supporting Information 3.5 Can rewilded rangelands be included in global conservation targets?

Rewilding rangelands may be one way of meeting global biodiversity targets, especially Target 2 on restoring 30% of ecosystems and Target 3 on protecting 30% of ecosystems. ‘Other Effective area-based Conservation Measures’ (OECMs) is a conservation designation for areas that are achieving the effective conservation of biodiversity outside of protected areas. It is a key component of protected area expansion plans (Visconti et al. 2019) and has been included in the 2030 Global Biodiversity Framework under Target 3 to achieve a 30% protected area target (Alves-Pinto et al. 2021; Gurney et al. 2021). From 2015 to 2018, IUCN provided technical advice to Parties on OECMs to the UN Convention on Biological Diversity (CBD) through a World Commission on Protected Areas (WCPA) Task Force on OECMs. In 2018, Parties to the CBD agreed to the guiding principles, common characteristics and criteria for the identification of OECMs (CBD Decision 14/8, 2018). An OECM is defined by the CBD as:

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, socio–economic, and other locally relevant values.*

With the definition and criteria for identifying OECMs (CBD, 2018) and IUCN guidelines for Recognizing and Reporting OECMs (IUCN-WCPA Task Force on OECMs 2019), there is a need to assess the potential extent of OECMs in South Africa and analyze whether the OECM designation strengthens the national conservation estate when applied to wildlife ranches. A global study that assessed the prevalence of potential OECMs in 740 terrestrial Key Biodiversity Areas (KBAs) outside protected areas across ten countries found that the majority of unprotected KBAs (77%) were at least partly covered by one or more potential OECMs (Donald et al. 2019). These results provide strong evidence that OECMs could complement protected areas to meet the 30% protection target of the 2030 Global Biodiversity Framework (Target 3) (Dudley et al. 2018; Gurney et al. 2021). The premise of providing win-win scenarios for conservation and development is founded on the fact that OECMs are located in ‘working lands’ that are agriculturally productive, yet not enough work has been done to test how the criteria variously enable or block various types of working landscapes from participating in OECMs.

In South Africa, Marnewick et al. (2020) assessed 9 pilot sites comprising private wildlife ranches and game reserves, state land and community areas. Of the nine sites assessed, two were state-owned and managed sites; three were communally owned properties governed and managed by respective traditional authorities (one site was managed in cooperation with an NGO); three were privately-owned game farms; and one was an academic research facility, which is governed and managed by a university. Six sites qualified as OECMs, while one of the privately-owned game farms remained a potential OECM pending improvements in management. Two of the communally owned sites did not qualify due to their governance and management not being strong enough.

I built on the assessment conducted by Marnewick et al. (2020) by conducting a deeper dive into the wildlife ranching sector. I produced a workflow to apply the OECM criteria to the sector (Table S3.8) and applied it to the 53 wildlife ranches in the Eastern Cape province for biogeographical consistency.

Table S3.8. Workflow for applying the OECM rapid screening tool criteria for the wildlife ranching sector with explanation of how the criteria were applied to data used in this thesis.

Action step	**OECM criterion**
Overlay with PA and PACA datasets to check not PA and/or conservation area double counting (1.1) Green if outside PA Red if not	1.1 Is the whole site, or the part being assessed as an OECM, outside of a protected area?]
Calculate the geographic area of biodiversity features (2.1) ● Calculate the areas of spatial features and area of free roaming pops if in 'natural' area as defined by land cover data ● Green if features can be clipped by one of those official datasets For this analysis I used the Red List of Ecosystems remnant vegetation layer to identify remaining natural habitats. This means that those portions of ecosystems that have been lost to anthropogenic activities such as mining, or croplands excluded and only the remnants are part of the dataset Green if natural habitats can be mapped and identified Red if not	2.1 Is the site (meaning the area of natural habitat) geographically defined, with agreed and delineated boundaries?
Governance of a specific entity (2.2). An entity exists and makes decisions about biodiversity features. Green if privately owned, the owner was interviewed, and the enterprise is locally owned. Orange if privately owned but under foreign ownership and the manager was interviewed. Red if land is leased or no governance structure in place.	2.2 Is the site under the governance authority/ies of a specified entity or an agreed upon combination of entities?
A management plan to address threats have been developed and is in place (2.3) Green if management plan exists and was developed by the manager/owner or a consultant Red if no management plan exists	2.3 Is the site subject to a management system with clear objectives and measures that achieve in-situ biodiversity conservation and manage threats?
Is the management likely to be sustained? (2.4). Use the length of time that the enterprise has existed, together with whether the enterprise is financially self-sustaining, to determine likelihood of the governance of the site being sustained into the future. Green if enterprise is ≥ 5 years old and self-sustaining	2.4 Is the governance and management 'sustained', i.e., expected to continue for the foreseeable future?

Action step	OECM criterion
Orange if enterprise is between 5 and 10 years old and not currently self-sustaining Red if enterprise is fewer than 5 years old (unclear what will happen) or if not financially viable between 10 and 15 years (if not financially viable after 15 years but still in existence, we assume a strong motivation to keep the enterprise going despite the loss incurred)	
Identify biodiversity features (3.1) Collate suite of biodiversity spatial layers to derive biodiversity features, including Critical Biodiversity Areas (CBAs), and Red List of ecosystems. Intersect with the spatial layers to get ecosystem-based feature proportions on each site. For the purpose of this analysis, I used the CBA layer to determine geographical extents of biodiversity features. Green if the site contains CBA1, CBA2, or threatened ecosystem (CR, EN, VU) patches equalling >5% of the area Orange if the site contains Ecological Support Areas (ESA) patches equalling >5% of the area Red if the site only contains Least Concern ecosystems	3.1 Is there a strong likelihood that the area contains important biodiversity values?
How likely is it that the biodiversity management will be sustained (3.2)? Assess whether the site has some form of legal agreement or market-based mechanism to ensure biodiversity conservation. For the purposes of this assessment, we use existing stewardship agreements or market-based incentives such as the Responsible Wool Standard. Green if legal agreement in place (any category) Red if no agreement is in place.	3.2 Is there a strong likelihood that the governance and management of the site achieves or is expected to achieve long-term positive and effective *in-situ* conservation of biodiversity, over the long-term, through legal or other effective means?
Is there a strong likelihood that the governance and management of the site support associated ecosystem functions and services (4.1)? Here we use the presence of sustainable land management techniques (SLM) that are likely to support natural ecosystem functioning, such as alien plant removal, soil erosion control and grazing management. We also look at whether the managers monitor changes over time, such that: Green if site practices multiple SLM techniques (covering at least 10% of the natural feature) and monitors vegetation and animal trends Orange if site practices multiple SLM techniques (covering at least 10% of the natural feature) but only monitors animal populations trends (vegetation more closely linked to ecosystem services) Red if site doesn't practice SLM techniques or doesn't monitor either animal or vegetation trends	4.1 Is there a strong likelihood that the governance and management of the site supports associated ecosystem functions and services, and that the enhancement of any of these services does not negatively impact the sites' biodiversity?
Quantify the socioeconomic values to assess whether the farm is a source of income/ job creation for the area, thereby denoting that working to maintain natural areas is a benefit to local communities. Green if >50% are permanent employees and women are employed Red if there are no permanent employees or no women employed	4.2 Is there a strong likelihood that governance and management measures identify, respect and uphold the associated cultural, spiritual, socioeconomic, and other locally relevant values of the

Action step	OECM criterion
	area, where such values exist, as well as respect and uphold the knowledge, practices and institutions that are fundamental for the in situ conservation of biodiversity.

Overall, only 1 of the sampled wildlife ranches scored green on all criteria (representing 2% of the sample) and thus qualified as a potential OECM (pOECM) (Table S3.9, Figure S3.7). The critically limiting criteria were criterion 3.1 (relating to legal or effective long-term commitment to biodiversity conservation) where only 15% of the sample scored green through either stewardship agreements or belonging to the Responsible Wool Standard; and criterion 2.3 (relating to the existence of management plans in place with biodiversity conservation goals) where only 51% of the sample even had formal management plans (see also Figure 2). However, wildlife ranches clearly possess biodiversity value in that 78% have significant biodiversity features, most notably the Critical Biodiversity Areas (CBA). In total, the sample overlapped with 286,261 ha of CBA1 and CBA2, which amounted to a total of 55% total sampled area covered by CBA (although individual sites varied from 0 – 93% overlap with CBA with a median value of 11% for CBA1 and 6% for CBA2). Thus, the OECM criteria may be overly restrictive when it comes to insistence on management agreements to protect biodiversity and this will lead to large swathes of potential biodiversity features being excluded from Target 3. To illustrate, if the hindrance associated with criterion 3.2 were to be lifted, then it would increase pOECMs to 17% within the sample. If the issues with criterion 2.3 were similarly resolved and all sites scored green, the proportion of pOECMs would increase to 34%. For example, switching to an outcomes-based assessment could provide empirical demonstration of sustained biodiversity management. This could include site-level changes in ecosystem extent and condition relating to changes in land-use, as well as metrics on vegetation productivity and soil carbon sequestration. If these data are plotted as a time series, then the trajectory of effective management could be estimated, and thresholds of potential concern set to flag sites that may become nearer to OECM eligibility or the reverse – sites that are in danger of losing OECM status. Alternatively, becoming certified through the putative wildlife economy certification scheme could be a legal, market-based mechanism through which to qualify as a pOECM.

Table S3.9. Summary of the desktop OECM analysis where the rapid screening tool was applied to a sample of 53 surveyed ranches from the Sustainable Wildlife Economies Project. Proportions correspond to the proportion of properties qualifying for green, orange or red status for each criterion.

Primary criterion	**Sub-criterion**	**% Green**	**% Orange**	**%Red**	**Additional information**
Criterion 1: The site is not a protected area (PA)	1.1. Is the whole site, or the part being assessed as an OECM, outside of a protected area?	83	0	17	The median size of the sample ranch was 2805 ha with a maximum size of 27574 ha and a minimum size of 755 ha.
Criterion 3: The site is a geographically defined area (meaning the area of natural habitat)	2.1. Is the site geographically defined, with agreed and delineated boundaries?	100	0	0	All wildlife ranches had remnant natural habitats across mapped ecosystem types, which can accurately delineate the potential biodiversity features on the sites. The average remaining natural vegetation was 92% and the minimum was 67%.
Criterion 4: The site is confirmed to support important biodiversity values	3.1. Is there a strong likelihood that the area contains important biodiversity values?	78	13	9	The average proportion of CBA1 and 2 across sites was 47±38%, for threatened ecosystems was 8±18%, and for SWSAs was 6±21%.
Criterion 5: Activities which threaten the important biodiversity values of the site are prevented or mitigated	2.2. Is the site under the governance authority/ies of a specified entity or an agreed upon combination of entities?	85	9	6	Unlike high-end ecotourism and trophy hunting models that are often foreign-owned (Clements et al. 2016a), which may hamper decision-making efficacy; most wildlife ranches are locally owned and managed by the owner.
	2.4. Is the governance and management ‘sustained’, i.e., expected to continue for the foreseeable future?	83	8	9	The median age of wildlife ranch enterprises is 19 years where 70% of enterprises were reported as being financially self-sustaining.
Criterion 6. Governance and management of the site results in the in situ	2.3. Is the site subject to a management system with clear objectives and measures that	51	0	49	The lack of formal management plans, as a precursor to even including biodiversity conservation goals, is a major barrier to wildlife ranchers being included in GBF targets. Our data are congruent with Pienaar et

Primary criterion	**Sub-criterion**	**% Green**	**% Orange**	**%Red**	**Additional information**
conservation of important biodiversity values	achieve in-situ biodiversity conservation and manage threats?				al. (2017) who found 64% of surveyed ranchers did not have a formal management plan.
	4.1. Is there a strong likelihood that the governance and management of the site supports associated ecosystem functions and services, and that the enhancement of any of these services do not negatively impact the sites' biodiversity?	92	0	8	Most properties practice multiple SLM measures on their natural habitat areas and most monitor both animal and vegetation variables, even if only informally.
Criterion 7. Governance and management arrangements are likely to be sustained	3.2. Is there a strong likelihood that the governance and management of the site achieves or is expected to achieve long-term positive and effective *in-situ* conservation of biodiversity, over the long-term, through legal or other effective means?	15	0	85	Only 5 sites (9%) of the sample were part of the biodiversity stewardship programme, and only 3 of the properties in the sample were signatories to the Responsible Wool Standard. No other existing legal or relevant market-based standards, or industry norms and standards were detected that could provide evidence for explicit long-term biodiversity conservation.
Criterion 8: Governance and management arrangements address equity considerations	4.2. Is there a strong likelihood that governance and management measures identify, respect and uphold the associated cultural, spiritual, socioeconomic, and other locally relevant values	66	0	34	Only 11 properties (21%) employed no permanent female workers, while 12 properties (23%) employed fewer than 50% permanent to seasonal workers.

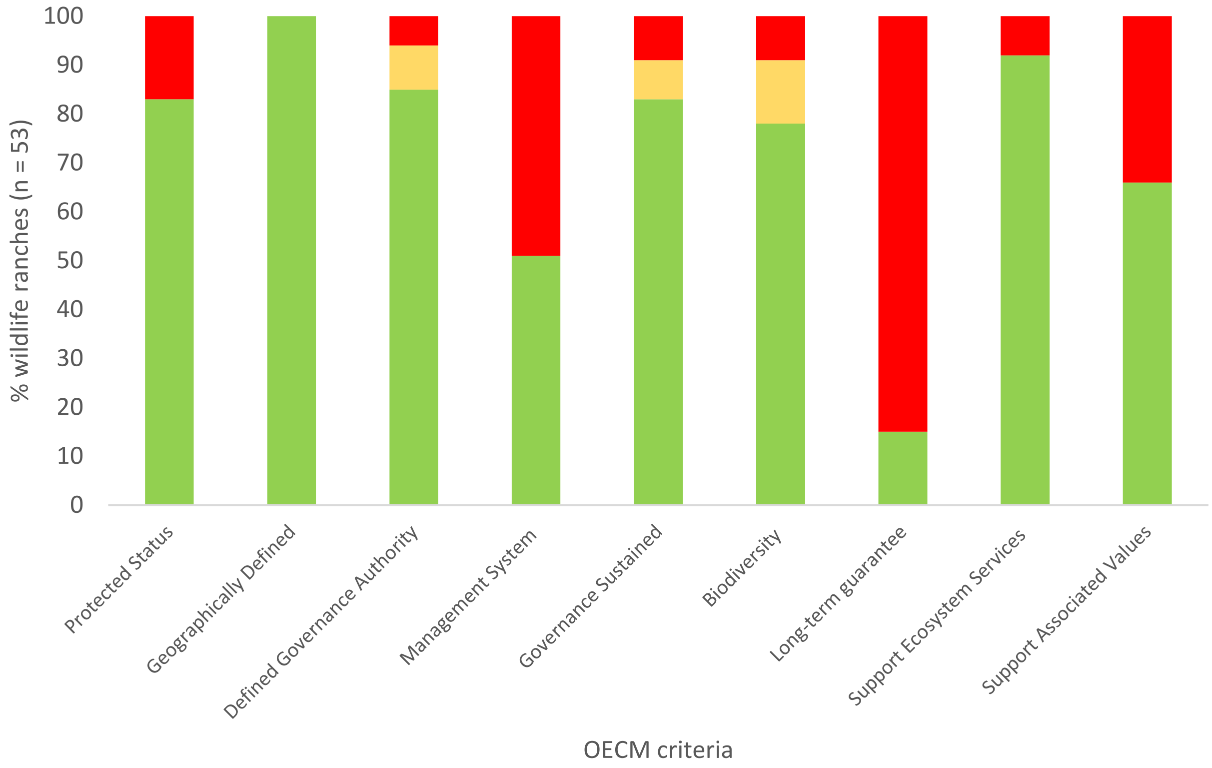

Figure S3.7. Summary of the desktop analysis of applying the OECM criteria to the wildlife ranching sector. For each criterion, green means the site is eligible, orange means it is partially eligible but may require specific interventions to fully qualify, and red means the site does not currently meet the criterion and will not in the foreseeable future. Overall, most sites fail on having a documented management plan in place and providing a long-term legal guarantee of biodiversity conservation.

‘Working lands conservation’ seeks to identify production-orientated land-uses that reverse biodiversity loss, mitigate climate change, and sustain ecosystem services (Kremen & Merenlender 2018). To reach the draft Global Biodiversity Framework goal of 30% terrestrial and marine area under protection (by 2030), OECM assessments should bring more players into the ‘conservation tent’ by valuing biodiversity more broadly and with a social-ecological systems perspective. However, the criteria, as they stand, appear counterproductive to this goal. At the very least, they are focused primarily on the ecotourism-based business models where governance structures and management plans explicitly mention biodiversity conservation in legal documents. However, legal status does not guarantee effective conservation (Shumba et al. 2020). As such, we suggest that this criterion is unreasonably restricting for working lands and a report should be drafted to the IUCN WCPA OECM Specialist Group for their consideration. Overall, we find OECM assessments are an insufficient tool to measure the biodiversity contribution value of wildlife ‘working lands’, as they do not sufficiently recognize broader ecosystem restoration and productivity gains made under an agro-ecological systems perspective.

The first conceptual stumbling block is the definition of ‘biodiversity value’. Currently, biodiversity value is considered any of the following (IUCN/WCPA 2022):

- rare, threatened or endangered species and ecosystems
- natural ecosystems which are under-represented in protected area networks
- high level of ecological integrity or intactness
- significant populations of range restricted species or ecosystems
- important species aggregations, such as spawning, breeding or feeding areas
- importance for ecological connectivity, as part of a network of sites in a landscape or seascape

Ecosystem services and local economic values are simply listed as ‘other values’ (IUCN/WCPA 2022). However, it is the provision of ecosystem services and subsequent economic values that will enable the site to be financially viable and thus continue to protect the biodiversity features that are the focus of the OECM assessment. Additionally, sustainable land management and restoration may promote biodiversity even in areas that do not qualify for the checklist above. For example, Site 1 does not qualify for an OECM and fails even on biodiversity value (amongst other criteria), but they have put significant capital into clearing bush encroachment and improving grass productivity to the extent that they are less reliant on importing fodder into the system. This is a form of regenerative agriculture and should be considered as a biodiversity value if we consider restoring ecosystem functioning as part of biodiversity.

Overall, similar to Biodiversity Stewardship, the practice of including only the area of the ‘protected’ biodiversity feature is problematic because it is the whole ‘system’ that drives the ability to protect a feature. By not recognising / rewarding the socioeconomic and social-ecological system that has demonstrated its effectiveness in conserving a biodiversity feature, the OECM mechanism is severely limited in being able to attract financing to the site, or designing market-based incentives, or producing policy advice that could enable the expansion of working lands assessed as OECMs. Conservation policy should shift to systems-thinking rather than feature-thinking to accumulate hectares under protection. As such, we need to collect information on wildlife economy business models to inform the certification scheme business case as a market-based incentive.

Restoration of ecosystems is also a key target under the Global Biodiversity Framework (Target 2: "*Ensure that at least 20% of degraded freshwater, marine and terrestrial ecosystems are under restoration, ensuring connectivity among them and focusing on priority ecosystems*"). Rewilding land can lead to restoration in reinstating trophic complexity, natural disturbances, and dispersal mechanisms, all of which act to improve ecosystem functioning and thus maintain biodiversity (Perino et al. 2019). As such, wildlife ranchers that 'rewild' their lands can contribute to biodiversity conservation even if the ecosystems themselves are 'novel' (i.e., differ from historical composition of species and take place on transformed lands or old fields). If restoration of ecological functioning in novel ecosystems is not valid, then it will be very difficult to reach the 30% target and will alienate many working lands managers who must balance production systems with biodiversity conservation. Similarly, there is a lack of clarity on ecosystem services: whether sustainable land management (SLM) in general is good for ecosystem service provision (i.e., even on transformed land) or whether the criterion should only apply to management of ecosystem services that occur on the remaining natural habitat or biodiversity feature itself. Improving ecosystem services even on old fields or cultivated lands is most likely good for biodiversity on the landscape scale (Perino et al. 2019) (and see Chapter 5).

The second conceptual stumbling block is the overreliance of the OECM criteria on management plans or systems and legal documents rather than outcomes-based assessments. Only half of the sampled wildlife ranchers had formal management plans. The lack of formal management plans, as a precursor to even include biodiversity conservation goals, is a major barrier to wildlife ranchers involved in GBF targets. It should be noted that the lack of management plans does not necessarily denote that there is no biodiversity conservation taking place, but simply that the farmers run the enterprise more informally using their experience in the field and still practice stewardship activities (such as alien plant clearing, soil erosion control, grazing and fire management) despite not having it written down (Pienaar et al. 2017). A better indicator would be to measure biodiversity values over time, such as the extent of natural habitat change over time, and some self-reporting mechanisms, such as perennial grass species abundance or fixed-point photography. Landowners can thereby demonstrate their contribution to OECM status or certification without strictly relying on a management plan.

However, this still does not guarantee conservation effectiveness. For example, Shumba et al. (2020) showed that Biodiversity Stewardship sites with legal protection of nature had similar natural land cover and biodiversity intactness to informal stewardship sites with no legal agreement.

Additionally, criteria relating to the explicit protection of biodiversity in a legal document are unreasonable as most working lands are not conservation areas so why would they have biodiversity protection as their primary mandate? The definition of 'long-term' is problematic in production landscapes, especially how long it means to ensure long-term effectiveness (Mwamidi et al. 2018). The proof should be in the system itself – if natural habitat or species are well-conserved through the operations of the enterprise then it makes business sense to maintain that. In particular, criterion 5 (3.2) is a major barrier to OECM identification as only 15% of sites have legal mechanisms for long-term protection. Criterion 5 guidance reads 'a site where legal means or other effective means as well as criterion 7 which calls on the governance and management to show evidence of being likely to be sustained through suggested schemes, special plans as well as land use plans. This is difficult to prove for the wildlife ranching sector, as they are agro-ecological enterprises within the market economy. A key gap would be through market schemes to favour the landowners addressing the in-situ biodiversity on their property.

To solve the hindrances associated with management plans and governance structures**,** we should see OECMs as effective if they have been proven to successfully protect biodiversity (as measured through remote sensing and /or in-field surveys), regardless of the legal agreements in place. This amounts to outcomes-based evidence rather than contractual or characteristic evidence. This idea is gaining traction in international conversations. "Policymakers need to agree on targets that are based on outcomes — not just coverage for both OECMs and protected areas" (Gurney et al. 2021). Furthermore, "a common currency of outcomes could alleviate concerns that there is an uneven burden of proof for the OECM and protected-area tools. It could also prevent the misuse of either to meet targets based on area without actually sustaining biodiversity" (Gurney et al. 2021). Evidence-based assessments are becoming more robust using remote sensing techniques. For example, indigenous territories in the Brazilian Amazon reduced forest cover loss between 5 and 17 times more effectively than matched control sites between 2005 and 2017 (Alves-Pinto et al. 2021); and in South Africa, between 1990 and 2013, private wildlife-based land-uses lost significantly less natural land cover and biodiversity intactness than matched unprotected areas (Shumba et al. 2020).

Evidence-based outcomes also align more strongly with the Land Degradation Neutrality (LDN) concept.

Lastly, the OECM criteria as they stand now may present a barrier to entry for new market entrants, especially where governance structures and management plans are concerned. While OECMs are after 'what is already there' and thus new market entrants could be assessed, some of the criteria may be tricky for local communities, especially as the opportunity cost for managing explicitly for biodiversity might prevent the enterprise from becoming viable. It may also clash with cultural and/or alternative economic activities. For example, if harvesting medicinal plants is perceived to negatively impact the plants then it cannot be considered OECMs. If the failure to achieve OECM status means reduced access to international finance for 'rewilding' projects, then OECMs become a barrier to entry for previously disadvantaged individuals looking to enter the wildlife economy.

In conclusion, a top-down approach to the biodiversity framework might need to be replaced with a sustainability framework where the emphasis on biodiversity targets as a primary goal is regulated to ensure sustainable land management and socioeconomic viability (Selinske et al. 2017; Wright et al. 2018). Working lands, such as wildlife ranches, might be better viewed as models of sustainable agriculture, where the emphasis is on landscape-scale processes, ecosystem functioning, wildness and not necessarily on historical species composition or pristine islands of habitat. Biodiversity policies currently view agriculture as a threat, but multifunctional landscapes under a land-sharing paradigm can produce mutually beneficial scenarios (Alkemade et al. 2022). Viewing all wildlife-based land-uses as contributing to area-based protection targets, might dissuade policy-makers from recognising other GBF targets that wildlife ranches, as agro-ecological systems, might be contributing towards (Table S3.10). Altering this view and recognising wildlife ranching as a spectrum of enterprises that vary in their agricultural and ecotourism products, might result in only a small proportion of these areas to qualify as OECMs, such as nature reserves under stewardship agreements and qualifying private protected areas.

Rather, a more promising global framework might be LDN under the United Nations Convention to Combat Desertification (UNCCD), where LDN is defined as "a state whereby the amount and quality of land resources necessary to support ecosystem functions and services to enhance food security remain stable, or increase, within specified temporal and spatial scales and ecosystems". LDN has been mooted as a framework to measure progress under Target 2 of the GBF. Monitoring achievement of neutrality is based on quantifying the baseline and then assessing the balance between the area of "gains" (significant positive changes/improvements) and area of "losses" (significant negative changes/degradation) relative to the baseline, within each land type, at the end of LDN implementation. Crucially, this conceptualisation views restoration as the result of socioeconomic and social-ecological systems that drive change (Cowie et al. 2018), rather than identifying set features and attempting to preserve them through legal agreements, which is the mechanism of protected area expansion through BDS and OECMs. This is a fundamental change in perspective because it assesses the outcome of a system rather than a single characteristic of that system. Contrary to expectations, effectiveness in conserving natural habitat and biodiversity intactness does not depend on legal protection in private lands in South Africa, where private lands with no formal conservation agreements had natural habitat protection outcomes comparable to declared nature reserves (Shumba et al. 2020). Shifting to a systems-level understanding of biodiversity conservation and associated policies is a major recommendation in the recent High-Level Panel report (DFFE 2020b).

Three indicators are used to measure LDN (UNCCD/Science-Policy Interface 2016; Cowie et al. 2018):

1. Land cover (land cover change);
2. Land productivity (net primary productivity, NPP);
3. Carbon stocks (soil organic carbon, SOC).

All three indicators are measured simultaneously in a system and LDN status is determined through the "one-out, all-out" principle where if either SOC or NPP decreases, or negative land-use change occurs then LDN has not been achieved (UNCCD/Science-Policy Interface 2016). The LDN indicators are complementary components of land-based natural capital because gains in one of these measures cannot compensate for losses in another. If one of the indicators/metrics shows a negative change, LDN is not achieved, even if the others are substantially positive. Each individual land unit is determined this way, and the results are aggregated to the level of the landscape.

The "one-out, all-out" principle is also critical to prevent perverse incentives. For example, bush encroachment and inappropriate planting of trees (as through the Bonn challenge) may increase above-ground carbon stocks but decrease biodiversity and grazing productivity, thus undermining sustainable development in the long run. Linking LDN to the 2030 GBF through an ecosystem type perspective will help to resolve these issues, as land-cover change and restoration strategies can then be contextualised through ecosystem type characteristics (such as the appropriate woody to grassy ratio, or the relevant indicator species). An ecosystem perspective, based on rewilding as a restoration process, can give land-use change analyses more significance and help identify scenarios where LDN is contributing to biodiversity, community livelihoods, as well as land productivity. Integration of biodiversity conservation outcomes and sustainable land management outcomes should form the basis of a certification scheme for the wildlife ranching sector that would enable congruence with the OECM criteria (Figure S3.8).

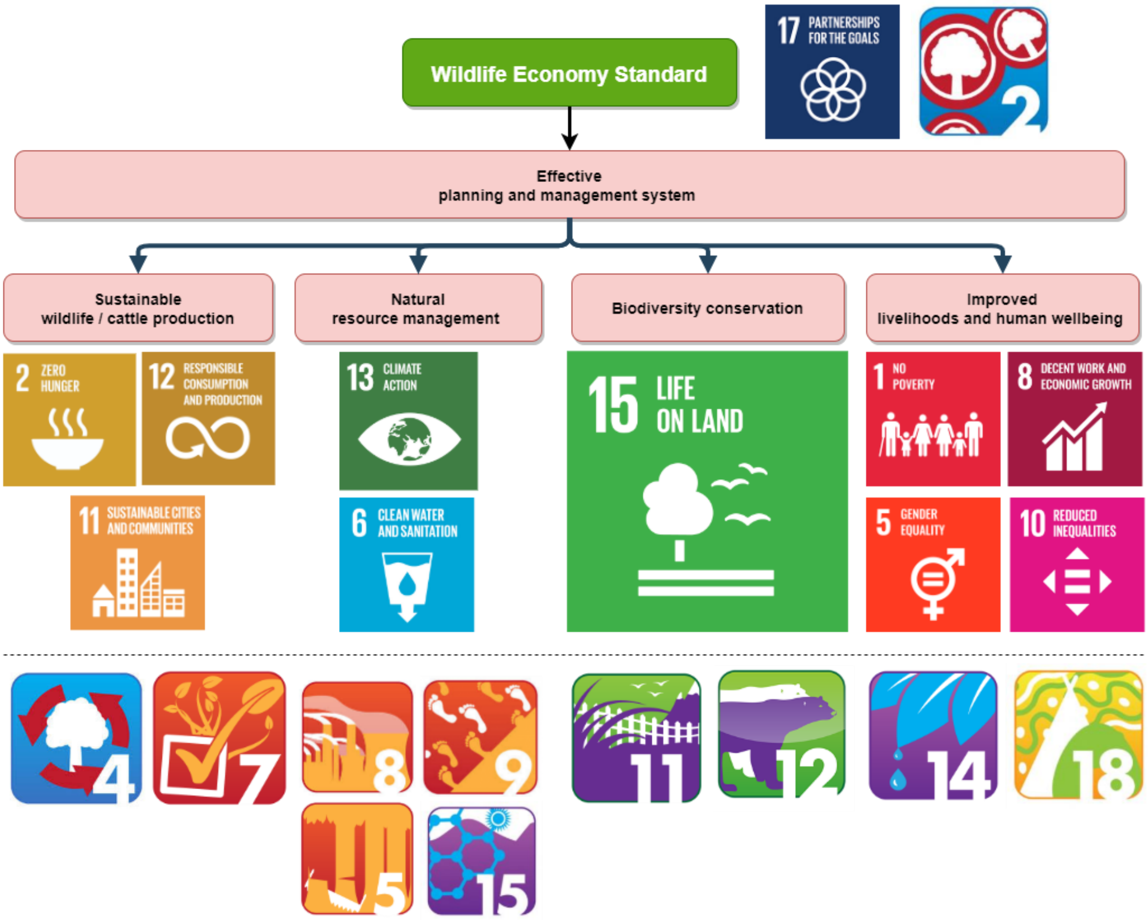

Figure S3.8. The relation of the proposed wildlife economy certification scheme to various Sustainable Development Goals. The wildlife economy interlinks with multiple Sustainable Development Goals centred on sustainable agro-ecological systems with consequences for socioeconomic development, restoration and habitat protection, as well as innovative public-private partnerships. Wildlife-based land-uses interface between production landscapes and wildlife conservation directly and thus will contribute to progress across multiple sectors. (Source: wildlife economy certification scheme).

Table S3.10. List of post 2020 Global Biodiversity Framework (GBF) targets that are directly relevant to the wildlife ranching sector and wildlife economy in general. Most of the relevant targets pertain to modes of sustainable harvesting, agriculture and land-use change, rather than protected area expansion. However, the means of calculating the indicators are not as advanced as they are for indicators based on terrestrial protection targets. These targets are taken from https://www.post-2020indicators.org/ and are current as of 10 March 2022. As most indicators are still under development, this presents an opportunity to have wildlife-based working lands mainstreamed into the GBF.

Target	**Target description**	**Indicator**	**Means of calculation**	**Opportunities**
Target 2	Ensure that at least 20% of degraded freshwater, marine and terrestrial ecosystems are under restoration, ensuring connectivity among them and focusing on priority ecosystems.	2.0.1. Percentage of degraded or converted ecosystems that are under restoration	*To be decided [under development] – but possibly:* *Land cover (land cover change); land productivity (net primary productivity, NPP); carbon stocks (soil organic carbon, SOC)*	Rewilding can improve ecosystem functioning, rangeland condition (multiple references), and soil carbon sequestration (for example, Sitters et al. 2020) Wildlife ranching enterprises cause land cover shifts from cultivated fields to old fields and near natural habitats (SWEP data). The majority of wildlife ranchers practice some form of sustainable land management (SWEP data). Using the working lands spatial database, the indicators under this target, if LDN metrics are used, could be populated through remote sensing to align the land-use and land-cover change histories of the properties.
Target 3	Ensure that at least 30% globally of land areas and of sea areas, especially areas of particular importance for biodiversity and its contributions to people, are conserved through effectively and equitably managed, ecologically representative, and well-connected systems of protected areas and other effective area-based conservation measures and integrated into the wider landscapes and seascapes.	3.0.1 Coverage of Protected areas and OECMS (by effectiveness)	OECM assessment criteria Identification of biodiversity features	Some wildlife ranches will contain priority biodiversity features (see preceding sections) and be willing to protect them through long-term agreements, particularly those focused on ecotourism.
Target 4	Ensure active management actions to enable the recovery and conservation of species and the genetic diversity of wild and domesticated species, including through ex situ conservation, and effectively manage human-wildlife interactions to avoid or reduce human-wildlife conflict.	4.0.1 Proportion of species populations that are affected by human wildlife conflict	*To be decided [under development]*	Wildlife ranching in South Africa has led to the recovery of wildlife populations (Taylor et al. 2021). Sustainable use should be recognised for increasing the abundance of species.
Target 5	Ensure that the harvesting, trade and use of wild species is sustainable, legal, and safe for human health.	5.0.1 Proportion of wildlife that is harvested legally and sustainably	*To be decided [under development]*	The sustainable use model, through the legal production, trade and hunting of wildlife species, has led to unprecedented 'rewilding' (Taylor et al. 2021). The volumes of animals traded and hunted for meat from these systems can

Target	Target description	Indicator	Means of calculation	Opportunities
				feed into this indicator. To produce a national verified dataset, the certification scheme could be used as an instrument to capture sustainable game meat production and sustainable trade, which would then link to the CITES trade database.
Target 8	Minimize the impact of climate change on biodiversity, contribute to mitigation and adaptation through ecosystem-based approaches, contributing at least 10 GtCO2e per year to global mitigation efforts, and ensure that all mitigation and adaptation efforts avoid negative impacts on biodiversity.	8.0.1 National greenhouse gas inventories from land use and land use change	*To be decided [under development]. However, it will draw on Chapter 5 of IPCC Guidelines for National Greenhouse Gas Inventories*	The wildlife-based land-uses and land-cover changes can be quantified and linked to changes in above-ground biomass and soil carbon content and can thus be aggregated into changes of GHG inventories. Such quantifications are made possible through programmes such as SWEP (for example, Figure 14). Wildlife ranches are also key ecosystem-based adaptations to climate change.
Target 9	Ensure benefits, including nutrition, food security, medicines, and livelihoods for people especially for the most vulnerable through sustainable management of wild terrestrial, freshwater and marine species and protecting customary sustainable use by indigenous peoples and local communities.	9.0.1 National environmental-economic accounts of benefits from the use of wild species	*To be decided [under development].*	Using the SEEA EA approach, the socio-economic benefits of sustainable use of wildlife can be aggregated into national natural capital accounts. The SWEP programme has already collected such socioeconomic data (Figure 15). Additionally, a biodiversity economy satellite account, disaggregating the employment numbers being generated by sustainable use of wild species, is currently under development at Statistics South Africa in collaboration with SANBI.
Target 10	Ensure all areas under agriculture, aquaculture and forestry are managed sustainably, in particular through the conservation and sustainable use of biodiversity, increasing the productivity and resilience of these production systems.	10.0.1 Proportion of agricultural area under productive and sustainable agriculture	FAO indicator synonymous with SDG indicator 2.4.1. Calculated as *Area under productive and sustainable agriculture /Agricultural land area* (see here for more details)	The majority of wildlife ranchers practice some form of sustainable land management (Figure 16) and around half of all wildlife ranches are mixed wildlife and livestock/crop farms (Figure 17). The SWEP survey form could easily be amended to capture all relevant information relating to this indicator in collaboration with DALRRD and StatisticsSA surveys that are regularly implemented.
Target 18	Redirect, repurpose, reform or eliminate incentives harmful for biodiversity, in a just and equitable way, reducing them by at least 500 billion per year, including all of the most harmful subsidies, and ensure that incentives, including public and private economic and regulatory incentives, are either positive or neutral for biodiversity.	18.0.1 Value of subsidies and other incentives harmful to biodiversity, that are redirected, repurposed or eliminated. Proposed indicator name: Positive incentives (by type) in place to promote biodiversity conservation and sustainable use.	OECD indicator: Data on positive incentives for biodiversity is collected via the OECD database on Policy Instruments for the Environment (PINE). Countries are requested to report on when the policy instrument was introduced, what it applies to, the geographical coverage, the environmental domain, the industries concerned; the revenues, costs or rates; whether the revenue is earmarked; and any exemptions.	The proposed voluntary certification scheme for the wildlife economy is a key market-based mechanism to promote triple bottom line models of sustainable wildlife use.

Chapter 4 Wildness in working lands – a case study of managed bontebok subpopulations

4.1 Abstract

The legal and sustainable use of wildlife on private land can incentivise the conservation of habitats and populations. However, mounting public pressure and conflation of illegal with legal hunting has precipitated restrictions on trophy imports from Africa where hunting operations and exporters require increasingly scientifically complex evidence that sustainable use enhances the conservation of species. Such impediments to unlocking the economic value of wildlife and subsequent expansions in range and population size, require consistent, robust, rapid and understandable assessments of the conservation value of managed subpopulations. Here, we apply a recently developed wildness assessment tool to a sample of South African private landowners managing bontebok (*Damaliscus pygargus pygargus*) subpopulations. Bontebok are endemic to the East Coast Renosterveld bioregion within the Cape Floristic Region of the Western Cape. It is currently listed as Vulnerable on the IUCN Red List due to its small population size and declining area of occupancy. However, many subpopulations exist on privately managed ranches both within and outside the natural range, where the conservation value is uncertain and thus not included in national conservation targets and restricted from earning hunting revenue due to regulatory conditions. Using a survey of private landowners who manage bontebok, I conducted a wildness assessment to determine the extent of privately owned subpopulations with conservation value. Overall, 35% of subpopulations qualify as wild (N = 57 subpopulations) compared to 100% of formally protected areas control sites. These wild subpopulations hold 64% of the total private bontebok population. Thus, there is scope to include these subpopulations in the Biodiversity Management Plan for the subspecies and to direct trophy hunters to enterprises that manage herds with conservation value. The results also demonstrate that the wildness of most non-wild subpopulations could easily be enhanced by increasing subpopulation size by an average of 5 adults and allowing herds to occupy entire properties rather than the enclosure system predominantly in place. A certification scheme could provide the vehicle to standardise and mainstream such wildness assessments, as well as create market-based incentives by reducing the administrative burden of hunting permits.

Additionally, I show that the Eastern Cape province (currently considered extra-limital) has significantly higher levels of wildness than either the natural range or the rest of the extra-limital range, which may demonstrate a case for benign introductions into this region and an expansion of positive market-based incentives for landowners participating in a certification scheme. These results demonstrate the need to assess the wildness of managed subpopulations on a local scale rather than rely on taxon-level assessments that can inhibit economic opportunity. The bontebok was brought back from the brink of extinction by private landowners, and policy-makers should consider positive incentives for this conservation success story to be sustained.

4.2 Introduction

The sustainable use of wildlife has helped secure much habitat across Africa, where private hunting estates and wildlife ranches contribute more land to conservation than formally protected areas (Lindsey et al. 2007b; Taylor et al. 2015, 2020; Snyman et al. 2021). Sustainable use of wildlife is defined as the consumptive use of components of biological diversity in a way and at a rate that does not lead to its long-term decline, thereby maintaining its potential to meet the needs and aspirations of present and future generations (Department of Environmental Affairs 2016). Hunting has been assessed as contributing to wildlife conservation and socioeconomic benefits in well-governed systems (IPBES 2022). Within the context of the wildlife economy, defined as “the uses of wildlife, plants and animals, as an economic asset to create value that aligns with conservation objectives and delivers sustainable growth and economic development” (ALU 2020a), sustainable use of wildlife primarily entails recreational hunting (local or international), game breeding and the sale of live animals, and game meat production (Taylor et al. 2020). Such wildlife-based land-uses (WBLUs) are viable in landscapes marginal for agriculture, often outcompeting equivalent livestock production models in job density and profitability (Cloete et al. 2007; Child et al. 2012b; Taylor et al. 2020), and can generate huge revenues from international clients and export markets that represent significant north-south funding flows for developing countries (Lindsey et al. 2007b; Taylor et al. 2015; Saayman et al. 2018).

The success of sustainable use as a conservation and economic model has brought about significant benefits to both communities and wildlife populations in Africa. For communities, sustainable use generates revenues, creates employment, reduces human-animal conflict and provides food sovereignty from game meat rations (Lindsey et al. 2007b; Taylor et al. 2015; Naidoo et al. 2016; Di Minin et al. 2021). Hunting also generates revenue flows to communities quicker than ecotourism models (3 years compared to 6 in Namibia) (Naidoo et al. 2016). A case study of these benefits comes from Botswana when the government banned trophy hunting in 2014 due to perceived wildlife declines based on one aerial survey by a conservation NGO (Elephants Without Borders). The survey results were criticised by local academics for being spurious. Additionally, local communities opposed the ban when the government embarked on public consultation (Mbaiwa 2018; Coe et al. 2023). The ban subsequently impacted communities through loss of revenue and jobs, and increased poaching rates, as they were forced to switch to ecotourism models that earn significantly less income in the region (Mbaiwa 2018) and are viable in large areas of the country; for example, only 22% of northern Botswana is suitable for ecotourism (Winterbach et al. 2015). Similarly, research shows the potential support for the return of sustainable use policies in Kenya, both to alleviate human-wildlife conflict and enhance economic opportunities in rural areas (Lindsey et al. 2006; Romañach et al. 2007; Brehony et al. 2020). In South Africa, trophy hunting bans would also have negative consequences for wildlife and communities, as many WBLUs believe the viability of their land-use would be compromised following the ban and subsequently, 63% of ranchers would transition from WBLUs to other land uses, such as cattle farming (Parker et al. 2020). In Namibia, 91% of rural residents of communal wildlife conservancies were opposed to a ban on trophy hunting, and only 11% would tolerate wildlife on the land if a ban was implemented (Angula et al. 2018). More than the material economic benefits of sustainable use, local communities view hunting as an instrumental value, enabling them to have agency over resource use, and as such, strongly disagree with humanising animals through welfare-based hunting bans while serving to dehumanise the people living with wildlife (Strong & Silva 2020; LaRocco 2020). Ultimately, hunting was reinstated in 2019 in Botswana due to local communities voicing their plight following the ban, and it has been documented as a successful instance of allowing ‘the subaltern to speak’ (Mbaiwa & Hambira 2021)

One of the main arguments against sustainable use is in the definition: is it sustainable? Detractors often conflate wildlife population declines caused by habitat loss and illegal poaching with the legal offtake of animals in hunting operations and other wildlife ranching enterprises (Lindsey et al. 2013a). This is exacerbated by international assessment tools, such as the IUCN Red List of species, which often do not distinguish between illegal hunting and legal hunting in the threat classification criteria (Challender et al. 2022; Marsh et al. 2022), which conflates the sustainable use of a species in local production systems with poaching across other parts of its range. Conversely, the available scientific evidence indicates that sustainable use of wildlife in well-managed WBLUs rewilds formerly degraded agricultural land and often increases wildlife populations (McRae et al. 2022). For example, Craigie et al. (2010) found that southern Africa, with the highest prevalence of WBLUs and legal hunting operations, was the only region in Africa where wildlife populations were increasing. On a national scale, Taylor et al. (2021) found that privately managed WBLUs in South Africa had significantly higher species richness and more threatened species (controlling for land area) than formally protected areas and that an estimated 4.66–7.25 million wild herbivores were conserved in these systems. Specifically, the sustainable use model has increased the population sizes and conservation value of the highly threatened white (*Ceratotherium simum*) and black (*Diceros* spp.) rhinos in South Africa and Namibia, as removing a small proportion of post-reproductive males can sustain ecological processes, promote genetic health of the population and encourage range expansion (Child 2012b; ’t Sas-Rolfes et al. 2022).

Opposition to sustainable use often takes on the form of misinformation on the impacts on wildlife populations (Hart et al. 2020) and the moral concern for an individual animal’s well-being (as opposed to the land ethic, King 1991; Batavia et al. 2019; Horowitz 2019), without having adequately incorporated local people’s perceptions, values and knowledge on these issues (Mkono 2019; van Houdt et al. 2021; Mbaiwa & Hambira 2021). Worryingly, there is a power asymmetry at play where voices from the global north, who are not saddled with the responsibility of conservation and community upliftment, support hunting bans, but African residents generally do not (van Houdt et al. 2021). Worryingly, there have been moves from the United Kingdom to ban the importation of hunting trophies, but this policy formulation is considered reactionary to public sentiment. It could amplify threats to species and create additional costs incurred by local communities who live with wildlife (Clark et al. 2023). At a global level, only CITES can restrict trade in animal products from threatened species, but at a national level, import bans may be imposed where there is evidence of population decline or mismanagement.

As such, a ‘smart’ ban should be pursued in policy to incentivise best practice, enhance benefits flowing to communities and disenfranchise bad practices, such as canned hunting operations or operations that do not monitor wildlife populations and thus demonstrate sustainable offtake (Webster et al. 2022). For example, despite raising concerns over wildlife trade system insufficiencies, the European Union effectively implements a smart ban by requiring hunting import permits to guarantee that the origin of a trophy is legal and sustainable. Specifically, alongside legal compliance with CITES, the EU requires evidence that the trophy has been derived from sustainable hunting practices that do not threaten the long-term survival of the species and that the activity contributes to conservation efforts or community development. Such evidence might comprise wildlife management plans, quotas, and conservation strategies related to the hunted species.

A smart ban is also in place in the USA, where the importation of hunting trophies is regulated through its enhancement findings, which is legislatively grounded in the Endangered Species Act (ESA) (Smith et al. 2018a; Noss et al. 2021). Enhancement findings require evidence that hunting a particular species in a specific country enhances the survival of that species in the wild. This essentially devolves decision-making on trophy imports to a case-by-case basis rather than at a sovereign level. For African species listed under the ESA, the impacts on revenue earned from hunting can be significant. For example, the 2014 ban on African Elephant (*Loxodonta africana*) trophy importation by the USA significantly decreased the hunting quota utilisation in Zimbabwe (dropping from 57% to 41%) as elephant hunts were cancelled by US clients (Nyamayedenga et al. 2021). Prior to the ban, most hunting clients were from the USA, followed by Europe and South Africa, respectively (Nyamayedenga et al. 2021), which supports other studies highlighting the importance of the USA hunting market for African revenue generation from hunting (Saayman et al. 2018; Johanisová & Mauerhofer 2023). In 2022, the USA lifted the blanket ban on elephant trophies in favour of the case-by-case approach based on sustainably managed systems.

Decision-making processes to implement smart bans suffer from a fundamental scale issue: blanket listings based on a whole-species assessment across its range without considering the context or conservation value of local wildlife populations. As such, the custodians of wildlife in private or communally managed WBLUs have no influence on trade policy, despite the profound implications of these decisions on local livelihoods (Wright et al. 2018). One of the main issues with CITES is that listings are based on simple aggregated biological and trade data that do not consider the impact of the listings on wildlife populations (Cooney et al. 2021). Such aggregate indices of population trends, such as the Living Planet Index, hides heterogeneous trends across regions and land-use systems (Leung et al. 2020), which, if considered in wildlife trade decisions, could enhance sustainable development. Central to fixing these problems is disentangling the complex impacts of different management systems on wildlife populations. The Red List criteria stipulate that only 'wild' populations should be included in assessments, but this lacks empirical thresholds, which can lead to inconsistency between assessments if not applied objectively and systematically (Hayward et al. 2015). The success of conserving managed populations depends on several ecological and evolutionary attributes (Redford et al. 2011), which was adapted by Child et al. (2019) [see Chapter 3], who developed a tool to measure wildness by setting empirical thresholds in management activities that are theoretically mapped onto the reduction of a population's ability to be self-sustaining and enact its ecological functions. This tool was designed to be implemented at a local population level, before aggregating into national and international assessments.

Here I apply the wildness assessment tool to a national survey of bontebok (*Damaliscus pygargus pygargus*) populations. Bontebok is an ideal case study because it is listed as Vulnerable on the IUCN Red List (Radloff et al. 2016) and Appendix II on CITES, meaning that international trade is permissible, but a threatened status triggers a non-detriment finding to be conducted. The bontebok is indigenous to (primarily) renosterveld remnants in the southern parts of the Western Cape (natural distribution range, NDR), but due to habitat loss, expanding the range of the species necessitated a range expansion project under a CapeNature policy. The keeping of this subspecies by private landowners outside of the natural distribution was therefore allowed by CapeNature, and a buffer population (extended natural distribution range, ENDR) was created from which populations within the natural distribution range could be augmented using a meta-population management approach. Bontebok have also been translocated extensively outside of its extended natural distribution range to the rest of the Western Cape and the Eastern Cape, Free State, Limpopo, Northern Cape, and North West provinces for a combination of commercial and conservation purposes.

Private landowners brought bontebok back from the edge of extinction as their indigenous habitat was ploughed up for agriculture and human settlements (Radloff et al. 2016; Cowell & Birss 2017). The dominant threat to this species remains lack of suitable habitat. However, there is a current paradox at play: bontebok conservation is largely driven by rewilding on private farms and protected areas, but there are few economic incentives for such landowners to invest in bontebok conservation. Increased restrictions on trophy imports constrain current incentives and could negatively impact the conservation of the species and its habitat. For example, bontebok are listed on the Endangered Species Act of the USA, necessitating an enhancement finding (demonstrating the conservation value of the hunt), to be able to import trophies. The number of Bontebok trophy exports from South Africa has steadily declined along with the average price of live animal sales at auctions (*Supporting Information 4.1*). This is despite the NDF[6] finding that "legal local and international trade in live animals and the export of hunting trophies at present poses a low risk to the survival of this subspecies in South Africa". I assess the wildness of managed bontebok populations on WBLUs to demonstrate whether import bans of this subspecies are warranted on the grounds that managed populations possess no conservation value.

4.3 Methods

I developed a survey for the Bontebok Breeders Association (BBA) to gather information on WBLUs with resident Bontebok populations. The BBA is a subgroup within Wildlife Ranching South Africa (WRA), a private wildlife industry association mandated to represent the interests of wildlife ranchers. This survey adapted the wildness framework of Child et al. (2019) (*Supporting information 4.2*), and also collected information on other wildlife species existing on the property. The survey focussed on how much area was available to Bontebok and the camp structure (if any) as well as the type of fencing on the perimeter or camps to measure dispersal potential (*sensu* Perino et al. 2019), Bontebok herd sizes and demographics (breeding males and females, as well as juveniles) across the camps and property overall, the frequency of replacing breeding males (to avoid inbreeding), provision of supplementary feeding and veterinary care density of permanent artificial water sources in the landscape, and exposure to predation. The survey was distributed electronically as an Excel spreadsheet to members of the BBA between 1 November 2019 and 29 February 2020 and was accompanied by a letter from

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[6] Unpublished non-detriment finding for *Damaliscus pygargus pygargus* (bontebok); Issued by the Scientific Authority of South Africa; October 2022

the South African National Biodiversity Institute explaining the purpose of the survey (*Supporting information 4.2*). Ethical clearance for analysis of the survey data was obtained by the University of Pretoria (ethics number: NAS120/2020).

Parameters used to score the wildness of bontebok populations included average home range size, male home range size, average herd size, minimum viable population size and the ratio of breeding males to ewes in a natural herd (see Table 4.1). The average breeding herd size (11 bontebok) based on ecological studies of the species compares well to the minimum viable population size (10–15 bontebok) based on allometric scaling models produced by Hilbers et al. (2017). To assess wildness, we adapted the Delphi technique for evaluating expert knowledge (Mukherjee et al. 2015), whereby three assessors (MC, JS, AT) independently scored each bontebok population using the wildness framework between 1 March and 1 April 2020. A workshop was then convened on 2 April 2020 to discuss issues encountered during the scoring. We then rescored based on that information, and the consensus scores were used where there was a difference. Where a property contained multiple encamped herds, the wildness of each herd was evaluated separately. To generate a property-level wildness score, the median score was taken across populations. We assessed bontebok wildness both as the number of ‘wild properties’ and the proportion of the overall bontebok population that can be considered wild and discuss the pros and cons of both evaluation methods. We compared wildness trends on private WBLUs between the NDR and ENDR, which we collectively call the ‘natural range’ and properties falling outside (the extra-limital range) (Figure 4.1) to that of formally protected areas who completed the same survey. As only three private populations in the Western Cape fell outside of the NDR and ENDR, we included them in the broader definition of ‘natural range’. We used the Wilcoxon rank-sum and Kruskal-Wallis tests to compare medians and interquartile ranges across groups as this is the most appropriate test for ordinal data. Finally, we assessed Bontebok population growth between 2017 and 2020 using surveys conducted by the Bontebok Breeders Association. We contextualised this in the context of the Bontebok Biodiversity Management Plan (Cowell & Birss 2017).

The BBA conducted surveys of bontebok numbers on private properties in 2016, 2017 and 2020. From a database of 206 farmers, the BBA received 166 replies in 2016, reporting 5 933 adult animals; 150 replies in 2017, reporting 5 797 adult animals; and 102 replies in 2020, reporting 4 468 adult animals. Subsets of these data representing the same landowners were used to compare overall population trends on private land to data collected through CapeNature on private land in the Western Cape. The CapeNature datasets represent data collected through the certificate of adequate enclosure permit applications and renewals and thus are more representative of the total population size than the BBA survey.

Survey respondents provided geographical location data only at district or municipality level. I used municipality-level property location data to generate a map of the median Bontebok population wildness in different parts of its range. All analyses were conducted using R (R Core Team 2023)

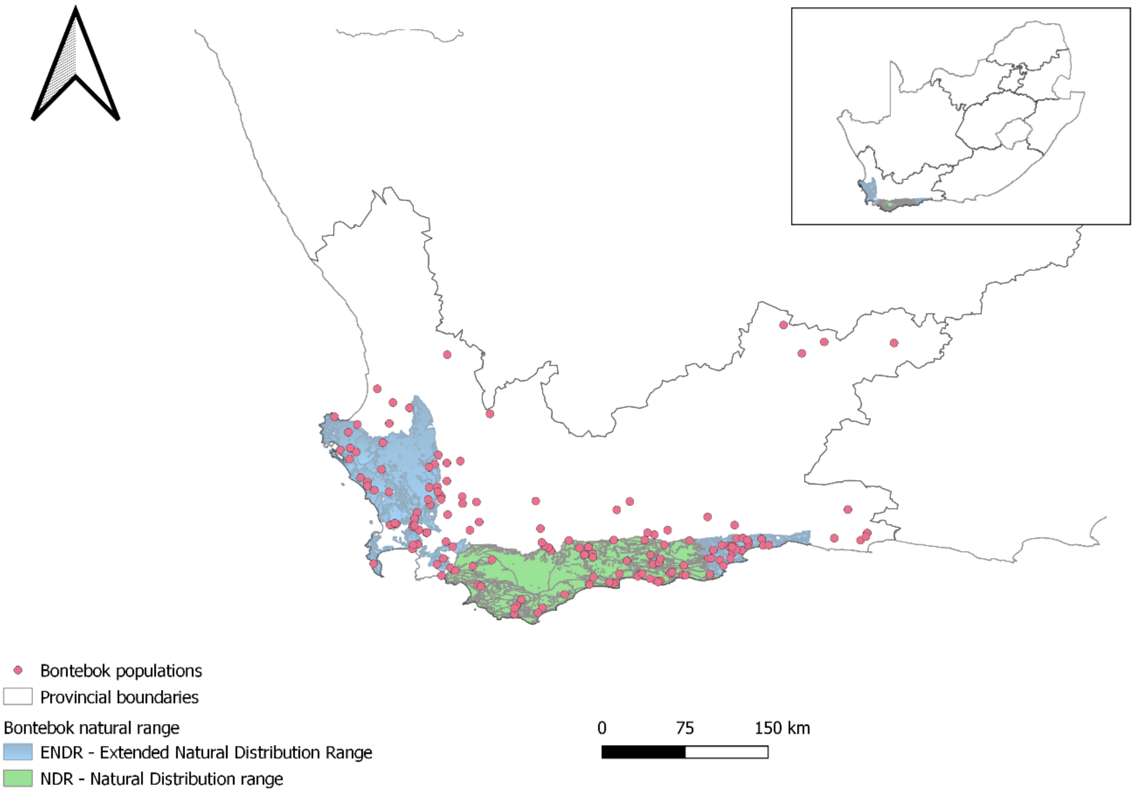

Figure 4.1. The natural distribution zones of Bontebok in the Western Cape Province of South Africa showing the occurrence of discrete Bontebok populations in 2021 (source: CapeNature).

Table 4.1. Bontebok home range and demographic parameters used to assess the wildness of managed populations in South Africa. Mean versus precautionary minimum viable population (MVP) size refers to the mean estimate from allometric scaling models compared to the estimate two standard deviations from the mean Hilbers et al. 2017).

Parameter and unit	**Value**	**Source**
Home range of typical herd (ha)	350-600	(Skinner & Chimimba 2005; Bothma & du Toit 2010)
Ram home range (ha)	30	(Bothma & du Toit 2010)
Breeding herd size	11	(Bothma & du Toit 2010)
Minimum viable population	10 (mean MVP) to 15 (precautionary MVP) at intrinsic growth rates	(Hilbers et al. 2017)
Breeding male to female ratio	1 ram to 10-25 ewes	(Bothma & du Toit 2010)

4.4 Results

We received 101 responses from landowners, representing 172 bontebok subpopulations. Of these, 94 responses were from private landowners, representing 163 subpopulations, and seven were from formally protected areas, representing nine subpopulations. Responses represent 52% of the total privately managed bontebok according to the property database but of the BBA (209 properties in total). Private WBLUs are significantly less wild than state protected populations (median ± IQR: 2.5 ± 1 compared to 5 ± 0.5; Wilcoxon rank-sum test: W = 26.5, $p < 0.01$; Figure 4.2), with 35% of populations qualifying as wild [median score ≥ 3] (N = 57 subpopulations) compared to 100% of formally protected areas (N = 9 subpopulations). However, when looking at bontebok abundance overall, the wilder subpopulations tend to be more abundant (Figure 4.2), meaning that, overall, 64% of the total sampled private bontebok population can be considered wild. For full distribution of wildness scores, see Figure 4.3. When comparing wildness patterns between the natural and extra-limital range to protected areas (Table 4.2), the extra-limital populations are overall wilder and have larger population sizes with more area available. Luyt (2005) suggested an ‘ecological stocking density’ of Bontebok of between 0.08 animals/ha based on data from Bontebok National Park. In the natural range, 39% (N = 24) of subpopulations were at or below this stocking density, with 72% of the ecologically stocked subpopulations corresponding to wild subpopulations. In the Eastern Cape province, 77% (N = 33) of subpopulations were at or below ecological stocking density, of which 61% corresponded to wild subpopulations. Finally, in the rest of the extra-limital range, 62% of subpopulations were at or below ecological stocking density, of which only 32% corresponded to wild subpopulations.

While most properties (70%) only managed one Bontebok subpopulation, some properties managed several subpopulations in different enclosures or camps. Of the private properties, the average was 1.7 ± 2.0 subpopulations / property with one outlier of 18 subpopulations on one property (due to multiple herds contained within separate camps), which, when removed, brought the average down to 1.5 ± 1.1 subpopulations / property. If we assess only the largest subpopulation on each property, the proportion of properties with wild subpopulations increases to 53%, but these properties only contain 40% of the total sampled population.

Table 4.2. Distribution of wildness scores amongst private populations within the natural range, extra-limital range, and formally protected populations. Populations within the natural range have lower average wildness scores, population sizes and area available to Bontebok. Where averages are reported, standard errors are displayed. Where medians are reported, interquartile ranges are reported. All differences in means between variables are statistically significant.

Variable	**Private - natural range**	**Private - extra-limital range**	**Formally protected areas**
Number of subpopulations	61	103	8
Number of wild populations	16	42	8
% of wild populations	26	41	100
Median wildness score	2±1	2.5±1	5±0.5
Average population size	14±2.7	33±4	80±31
Average area available (ha)	669±207	1186±356	11253±3942
Median area available (ha)	34±460	350±600	6875±13140
% subpopulations below ecological stocking rate	39	77 (Eastern Cape) 62 (elsewhere)	100
% tested genetic purity	79	86	No data

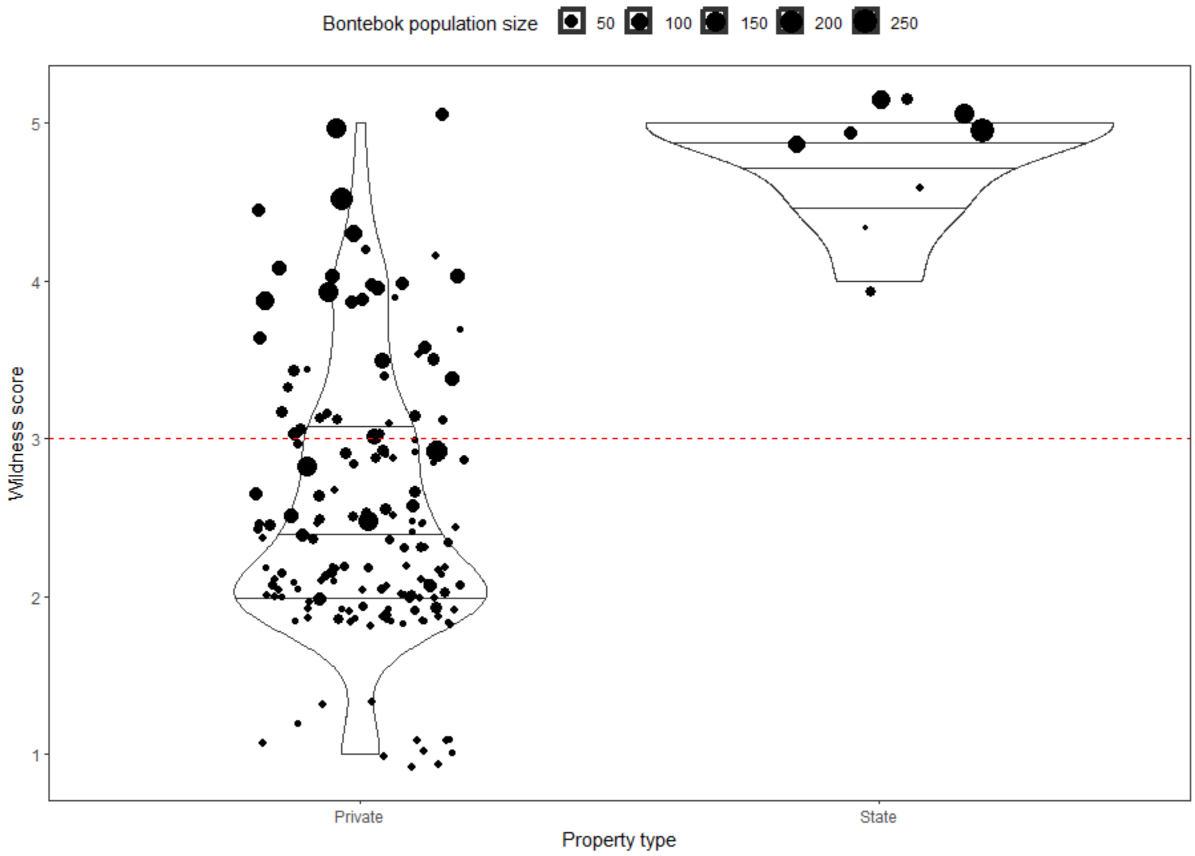

Figure 4.2. The difference in wildness between state (formally protected areas) and private subpopulations of bontebok. Median wildness scores are significantly higher in state populations. However, most of the private bontebok population is contained in wild properties (properties are scaled by population size), meaning that, overall, 64% of the private population can be considered wild although only 34% of the individual subpopulations are wild.

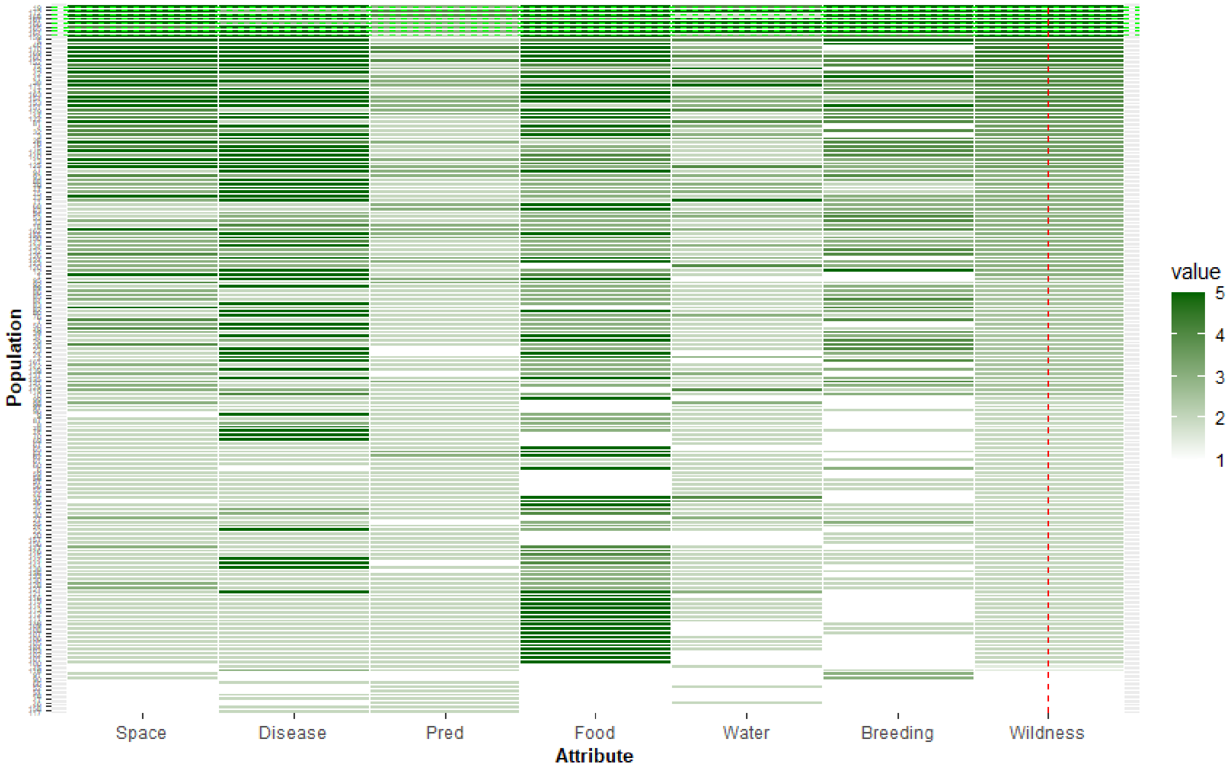

Figure 4.3. Summary of wildness scores for each attribute and overall (red dotted line) for each Bontebok population. Light green horizontal lines indicate state protected area populations. Formally protected subpopulations are indicated by the light green dotted lines and have the wildest subpopulations.

When looking at individual attribute scores across the private subpopulations (Figure 4.4), the distribution of scores across wildness attributes varies significantly across properties (Kruskal-Wallis test: K =126.6, df = 5, p < 0.01), and all pairwise attribute comparisons are significantly different (Dunn's test: Z scores -6.9 – 8.0, all p < 0.01). However, the available space for bontebok herds, the lack of predation pressure and the access to artificial water sources are more heavily clustered around "intensive management" than other attributes (Table 4.3, Figure 4.4), meaning that improvements in these attributes may substantially increase the wildness of the overall subspecies on private land. The total wildness space across populations for each region is shown in Figure 4.5, which reveals that subpopulations in the natural range are, on average, less wild than in other parts of the range. Subpopulations in the Eastern Cape generally display higher wildness across the attributes.

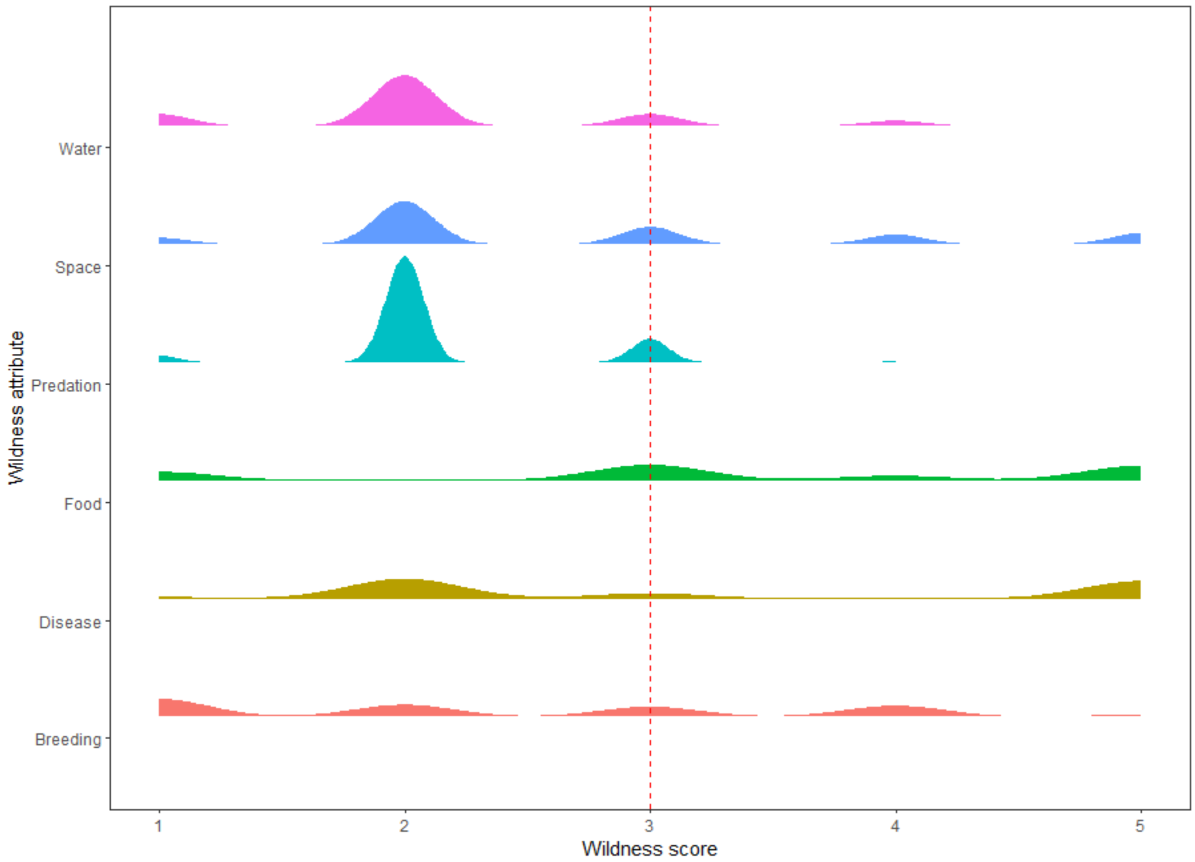

Figure 4.4. Frequency distribution of wildness scores across all attributes for the privately managed populations. Lack of adequate space, lack of predation pressure, and provisioning of artificial water appear to be the most low-scoring attributes, which may provide a road map for increasing the wildness of populations given appropriate incentives.

Table 4.3. Distribution of median wildness scores across attributes for different regions. Interquartile ranges are reported in square brackets. While most attribute scores vary widely across properties and regions, exposure to natural predation is consistently low.

Region	**Breeding**	**Disease**	**Food**	**Predation**	**Space**	**Water**	**Populations (N)**
Natural	1 [2]	2 [1]	5 [4]	2 [1]	2 [1]	2 [1]	61
Eastern Cape	3 [1.8]	5 [2]	4 [2]	2 [0]	3 [2]	2 [1]	42
Northern Cape	2 [1]	3 [3]	3 [0.8]	2 [0]	2.5 [1]	2 [1]	30
Free State	2 [3]	3 [3]	3 [2]	2 [0]	2 [1]	2 [0]	27
Protected areas	5 [2]	5 [2]	5 [0]	3 [1]	5 [0]	5 [2]	9

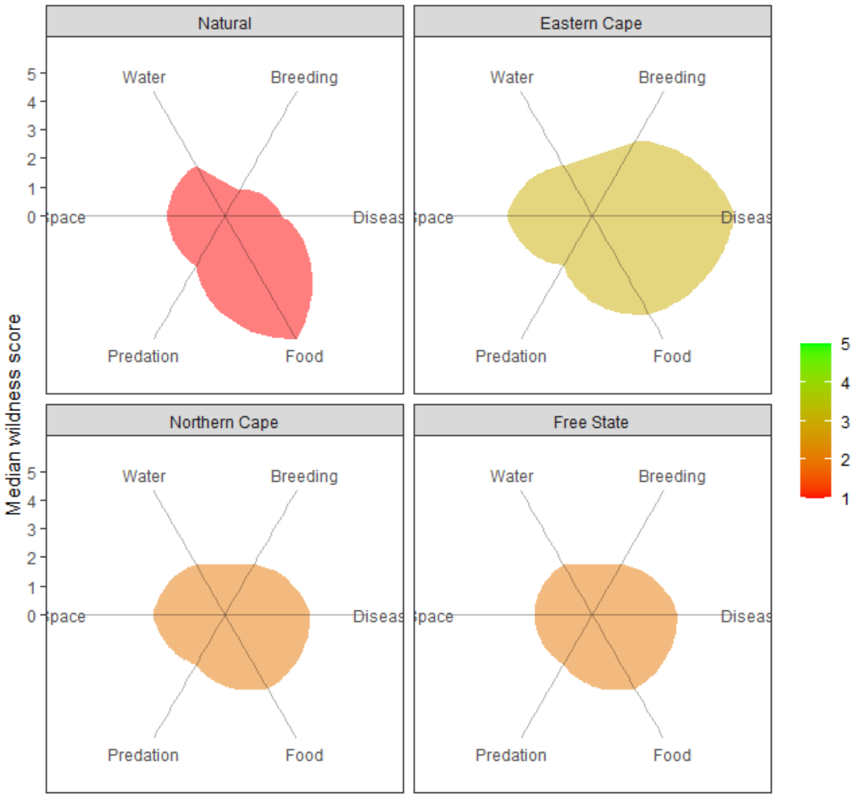

Figure 4.5. State space plots showing the median wildness scores for each attribute between different region, thus representing the total ‘wildness space’. Colours represent the overall wildness score of populations in the different regions where redder colours are lower wildness and greener are higher wildness.

When looking at wildness trends between different geographic regions, there were significant differences in median wildness (Kruskal-Wallis test: KW = 20.2, df = 5, $p < 0.01$; Table 4.4, Figure 4.6), where the Eastern Cape province had significantly higher median wildness scores than all other range areas and the natural range was not significantly different in wildness than the Northern Cape and Free State provinces (Dunn’s test: Z score = 2.56 -3.75 for Eastern Cape compared to other regions, $p < 0.05$; Table 4.4). When these wildness scores are mapped to municipality level, the natural range and Eastern Cape regions have the highest concentration of wild populations, often in contiguous municipalities (Figure 4.7).

Table 4.4. Differences in wildness scores, population size and area available to Bontebok in different parts of its range. Median and interquartile range is reported for wildness score. Unless otherwise stated, values represent means and standard errors. Comparisons represent z values of pairwise wildness score differences through a Dunn's test, where significance values are * for p < 0.05 and ** for p < 0.01.

Region	**Subpopulations (N)**	**Wildness score**	**Population size**	**Average area available (ha)**	**Median area available (ha)**
Natural range	61	2±1	14±3	669±207	34±460
Eastern Cape	42	3±1.5	45±7	1930±811	514±850
Northern Cape	30	2.5±1	15±2	510±134	356±412
Free State	27	2±0.5	35±9	663±365	150±300
Significant Pairwise comparisons				*Z-score*	
Eastern Cape – Natural range				3.75**	
Eastern Cape – Northern Cape				2.56*	
Eastern Cape – Free State				3.50**	

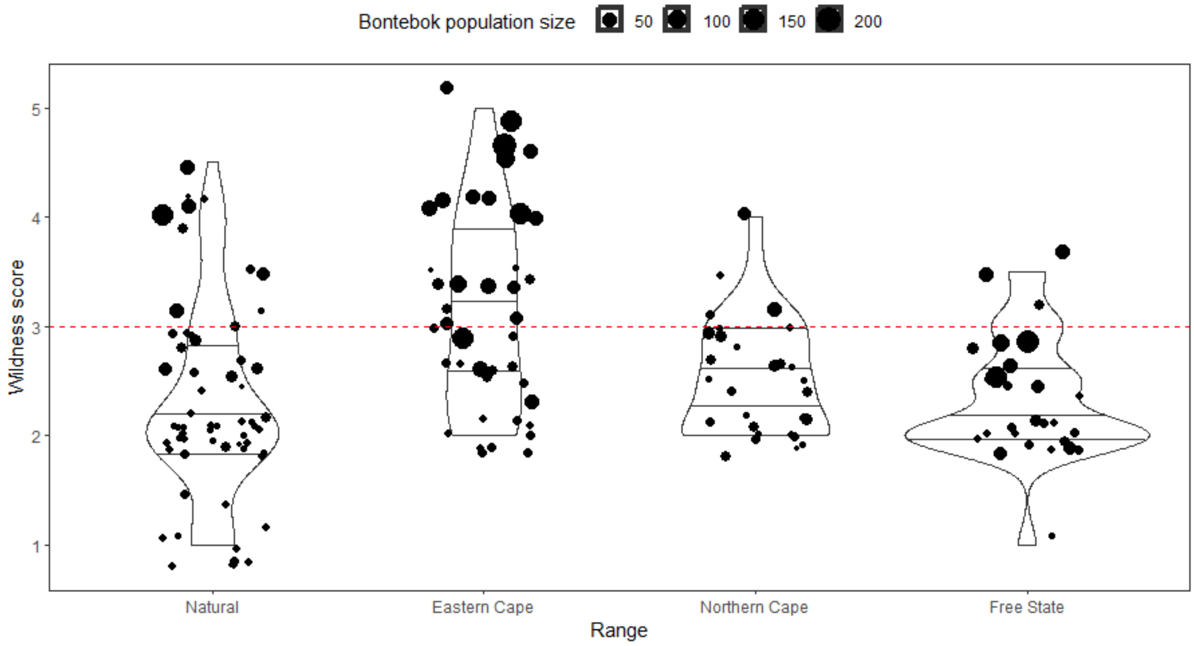

Figure 4.6. The difference in wildness between different geographical distributions comparing the natural range in the Western Cape to various extra-limital provinces. Limpopo and North West provinces are excluded from the graph due to low sample sizes. Median wildness scores are significantly in the Eastern Cape province that all other areas of the range, including the natural range.

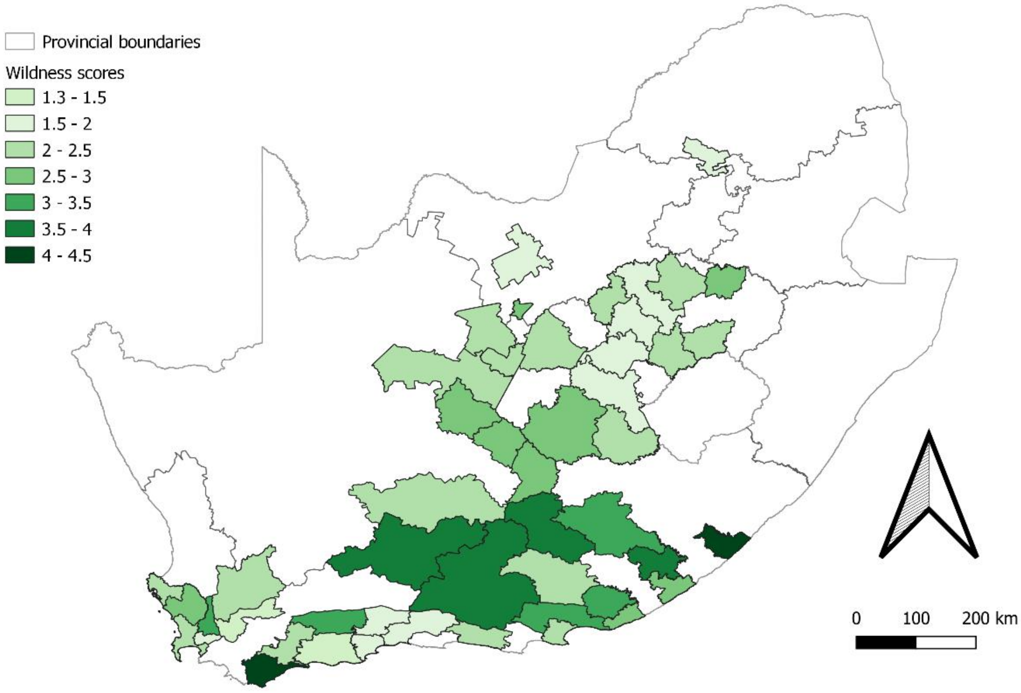

Figure 4.7. Distribution of Bontebok wildness scores summarised to municipality scale. Where there were multiple respondents per municipality, the median wildness scores of the populations were used. Darker green colours indicate wilder populations, and the two provinces with the most numerous wild Bontebok populations were Western Cape and Eastern Cape.

When considering the ‘breeding’ attribute, 56% (N = 91) of subpopulations were classified as ‘intensive’ or ‘captive’ due to insufficient herd sizes to be demographically self-sustaining or insufficient breeding males to ensure natural reproductive competition. Small subpopulation sizes were especially prevalent in the natural range (Figure 4.8), where only 26% (N =16) of populations surpassed the minimum viable population size of 10 adults (as per Table 4.1). However, while 53% of subpopulations overall did not meet the minimum viable population size, these subpopulations would only require increases of 5 ± 2 adults on average to become viable. Thus, in general, there is sufficient space available for subpopulation size to increase and thus enhance wildness.

The smaller subpopulation sizes in the natural range are most likely due to the lower space available to populations in the Western Cape (Figure 4.8). The area available to bontebok is highly variable both within and between regions, and the mean area available may be heavily influenced by outliers, which is why medians are also reported (Table 4.3, Table 4.5). Overall, 42% of populations (N = 69) pass the 350 ha minimum area requirement (as per Table 4.1, Figure 4.8), where the area available to bontebok subpopulations ranges from 2 ha (population size of 2 in the natural range) to 29,000 ha (population size of 150 in the Eastern Cape), with a median size of 250 ha. When looking at the area potentially available to bontebok subpopulations (total property size) compared to the summed area of camps that contain bontebok, (Figure 4.9) then most properties could easily expand the area available to bontebok and increase population size to surpass both minimum viable population and area sizes needed to help increase wildness scores. Average property sizes and average camp sizes were not significantly different between different parts of the range (ANOVA: $F_{3,87} = 0.33$, N = 93, p = 0.81 for property size; $F_{3,87} = 0.80$, N = 93, p = 0.50). For populations in the natural range, however, the average property size is lower when compared to all other ranges, except Free State, and has the lowest median property size (Table 4.5), which means that average camp sizes were smaller than that of the Eastern Cape and median camp size was also lowest of all the provinces. Overall, only 21% (N = 19) of the properties had no internal camp system, and so were fully open to bontebok subpopulations.

Table 4.5. Property sizes and area available to Bontebok populations within properties across different parts of the range. Means and standard errors are reported. Average property size and camp size were not significantly different between different parts of the range. However, the natural range has the lowest median property size and camp size.

Area variable	**Natural**	**Eastern Cape**	**Northern Cape**	**Free State**
Properties (N)	26	34	12	19
Average property size (ha)	3953±1763	4432±1154	4724±1826	2644±893
Median property size (ha)	875	2228	2822	1200
Sum camp area (ha)	1551±528	2385±989	1276±271	942±511
Mean camp area (ha)	1172±417	2290±994	874±301	856±515
Median camp size (ha)	215	642	462	250
Sum of camps (% of property)	63±8	61±7	49±12	42±6
Average of camps (% of property)	49±8	57±7	39±12	33±6

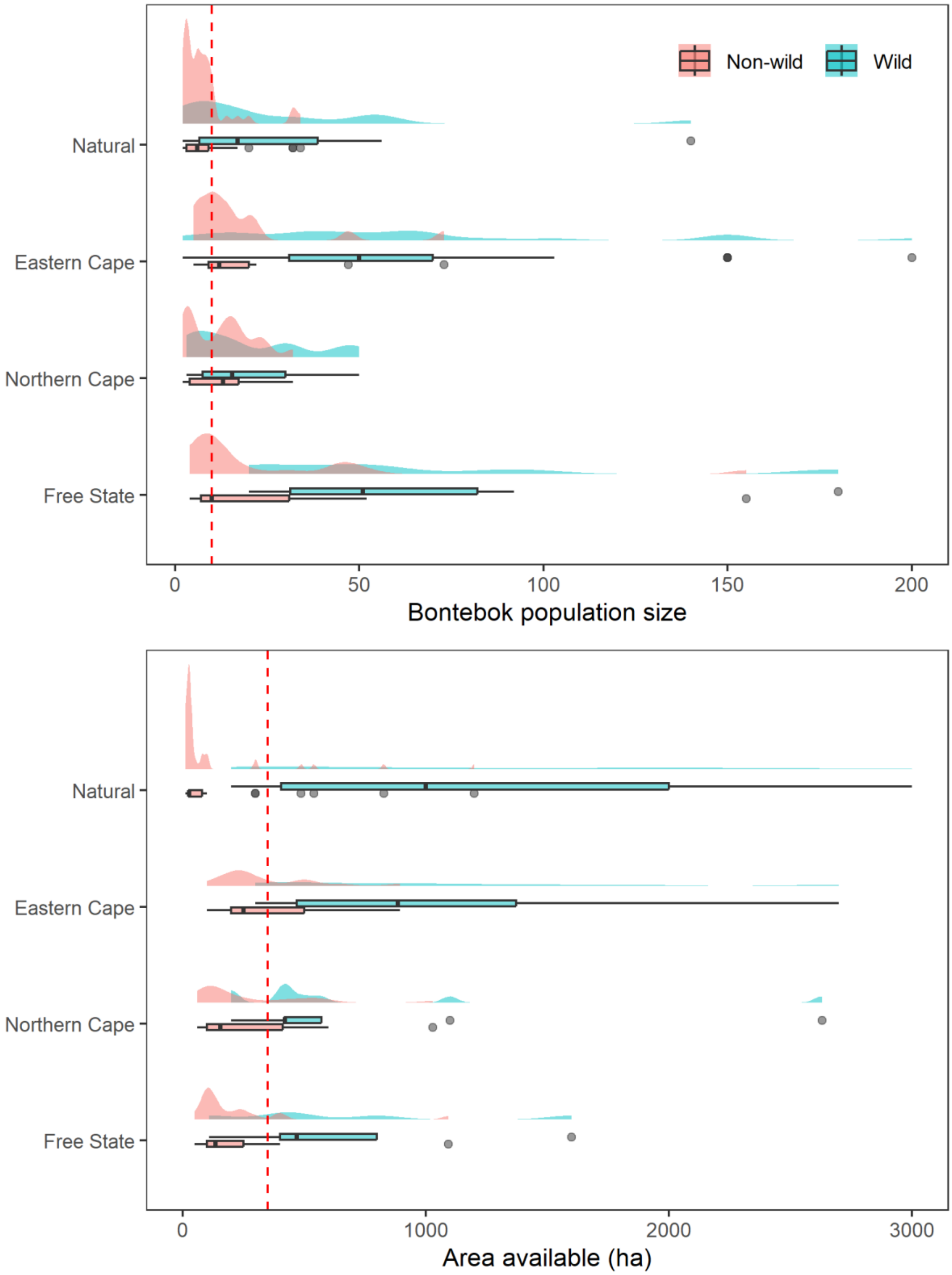

Figure 4.8. Rainfall plots showing the density distribution of Bontebok populations (top panel) and available area to Bontebok populations (bottom panel) between populations classified as wild and non-wild in different parts of the range. Red dashed lines indicate thresholds for minimum viable population size (10 adults) and minimum area requirement (350 ha) for Bontebok, respectively. Outliers were removed from the bottom plot (but not from the dataset) using winsorization to make the graph more readable. The majority of populations do not exceed the threshold for population or area, and most non-wild populations are highly clumped around a narrow range of

values, suggesting a bifurcation between most landowners managing small, non-viable populations and landowners managing wild populations over a wide range of sizes and areas.

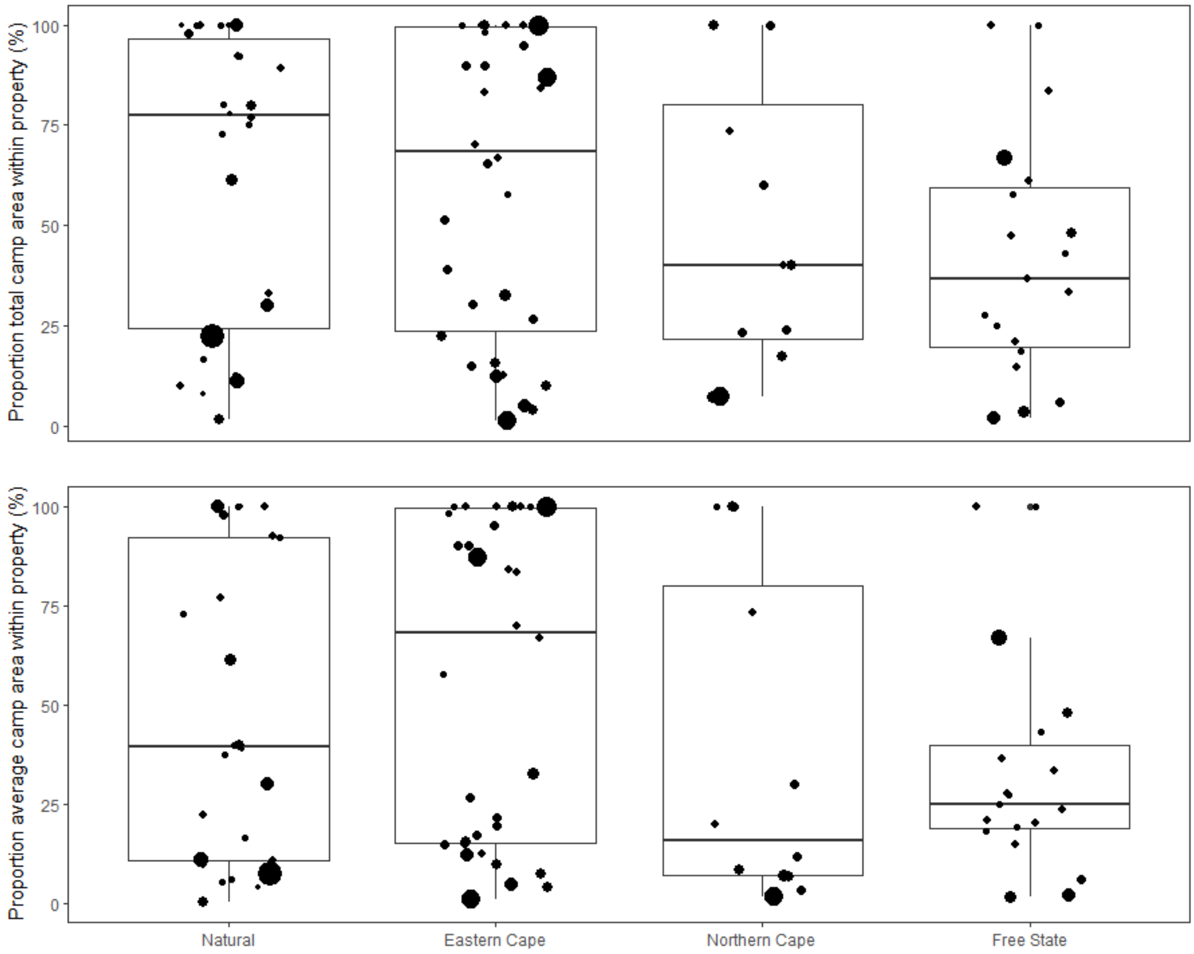

Figure 4.9. Area available to bontebok subpopulations expressed as a proportion of total property size, where properties are scaled by property size. The top panel shows the proportion of the total camp area (for properties where there are multiple encamped populations), while the bottom panel shows the average camp size relative to property size. There is a wide variation in the area available to bontebok subpopulations both within and between range areas. However, when looking at the sum of camp areas, the natural range has the highest median proportion, meaning that relatively more of the total area is being used for bontebok.

The average population sizes for the Western Cape are not significantly different between the CapeNature dataset and the BBA (T-test: T = 1.68, df = 149, p = 0.09; mean ± standard error 22 ± 4 and 14 ± 4 for the CapeNature and Bontebok Breeder Association data respectively), although the CapeNature survey has a slightly higher average population size but still substantially below the Eastern Cape subpopulations. This shows that the estimates of average population size are robust for the natural range. Overall, while the total estimated population sizes in 2014 and 2021 are very similar, the number of subpopulations is lower in 2021 (Table 4.6). Conversely, data from the Bontebok Breeders Association show consistent population growth rates of around 4% in two different periods using survey returns of the same sets of landowners. In 2016, the average number of bontebok per property was 36 individuals, 39 individuals in 2017 and 44 in 2020.

Table 4.6. Population size estimates for different regions from both Cape Nature and the Bontebok Breeders Association. CapeNature data are filtered for private landowners only. The total population size has stayed approximately the same in seven years in the natural range despite a general decrease in properties managing Bontebok.

Year	**Properties**	**Bontebok population**	**Average population size / property**	**Region**	**Source**
2014	123	1965	16	Natural	CapeNature
2017	102	3847	38	All	Bontebok Breeders Association
2019	102	4010	39	All	Bontebok Breeders Association
2021	90	1966	22	Natural	CapeNature

4.5 Discussion

Wildness is at the core of how we define conservation value and is thus essential in evaluating the impact of sustainable use in managed wildlife populations. Wildness is the degree of demographic and ecological autonomy and self-sufficiency of populations (Redford et al. 2011; Child et al. 2019), where the intensity of management is expected to inversely correlate with the capacity of the subpopulation to be viable if the management was withdrawn (IUCN Standards and Petitions Subcommittee 2022). The less intensively managed a subpopulation, the more integrated into the ecosystem it will be – dependent on natural foraging for food and water, exposed to selection pressure, and carrying out ecological processes such as nutrient dispersal, seed dispersal, herbivory and interspecific competition. As such, sustaining wild subpopulations will also improve ecosystem functioning and resilience. Considering that 39-77% (natural range to Eastern Cape) of landowners stock bontebok below the recommended ecological stocking rate, there is a tangible policy link to the long-term grazing capacity guidelines produced by the Department of Agriculture, Land Reform and Rural Development (DAFF 2018). My results show that there is currently significant conservation value of managed bontebok subpopulations in working landscapes and that the right economic incentives could be used to increase wildness even further as most non-wild subpopulations are close to key thresholds for being considered wild.

Privately managed Bontebok populations' wildness varies substantially between properties and provinces. Currently, 35% of the sampled subpopulations can be considered wild. Interestingly, these results are consistent with Child et al. (2019), who found that 33% of sampled Bontebok subpopulations could be considered wild. These analyses were conducted on two independently collected datasets, representing a robust estimation of the proportion of wild subpopulations. The wild subpopulations hold most of (64%) the total abundance of the private population, which means that these properties contribute most to Bontebok conservation and could meet the enhancement criteria of the ESA, thereby facilitating import permits for legal hunts. If wildness assessments were mainstreamed into export permit documentation by landowners, hunting outfitters and provincial permit officers, this could provide a consistent mechanism to demonstrate conservation enhancement and unlock economic value from bontebok. This might then help improve the live animal price that has been declining over the past decade and is now lower than the average plains game price (*Supporting Information 4.1*), which may explain the stagnation of population expansion in the natural range. Additionally, the 35% of wild subpopulations contain at least 2,500 adult bontebok. Considering the USA only imported 10 Bontebok trophies in 2020 (*Supporting Information 4.1*), the number of trophies could be increased by at least an order of magnitude and not cause population decline (based on an empirically observed 4% population growth rate over two years on private lands). In general, hunting-based WBLUs are inherently sustainable as harvesting is typically only 2-3% of the populations that can biologically increase at rates of 10-30% (Child et al. 2012b). On an ecosystem level, sustainable use models also entrain sustainable land management and restoration practices, where the majority of WBLU managers improve habitat quality and condition through removing alien invasive vegetation, reducing erosion, reinstituting fire as an ecological process and managing stocking densities to reduce pressure on forage species (Pienaar et al. 2017).

Wildness assessments could be linked to a certification scheme to provide a more robust vehicle for evaluation and deliver the extension support necessary that provincial management agencies often lack the capacity to conduct. For example, Wanger et al. (2017) proposed a certification scheme for trophy hunting based on criteria-based performance in population management, ethical practices and community benefits. Wildness assessments could form part of the population management criteria to certify the conservation value of the managed subpopulation. As trophy hunters are willing to pay significant premiums for proven conservation and community benefits (Lindsey et al. 2007b; Fischer et al. 2015), operating a certification scheme would be viable. In South Africa, such a certification scheme is currently being pursued by the Department of Forestry, Fisheries and Environment (DFFE), where a scoping study found that 64% of wildlife ranchers would support a certification scheme (Turpie & Letley 2018).

However, before such a scheme is implemented, stakeholders must decide which subpopulations would be eligible for certification, even if they are considered wild. The Red List guidelines define conservation value as “wild subpopulations inside the natural range of a taxon” (IUCN Standards and Petitions Subcommittee 2022), unless the translocation event was to reduce the extinction risk of the species, the introduced subpopulation is ‘geographically close’ to the natural range (for example, same ecoregion), or the introduced subpopulation has produced viable offspring and five years have passed since the introduction. My results show that the proportion of wild populations inside the natural range is lower than the extra-limital range, and specifically, the Eastern Cape province is the only range where subpopulations were considered wild on average with significantly larger subpopulation sizes than other parts of the range. Expansion of suitable wildlife habitat within the natural range of Bontebok is severely limited by extensive transformation of the landscape for crop agriculture and urban expansion, where 107 km^2 of land was converted to agriculture per year between 2006 and 2011, 31% occurred within Critical Biodiversity Areas; and there has been an 8.6% increase, from 1,029 km2 to 1,118 km^2, in urban expansion between 2000 and 2013 (Radloff et al. 2016). Results from this study support the findings from the 2016 Red List assessment that the population size in the natural range remained stable between 2004 and 2014 (Radloff et al. 2016). Thus, the population size has not changed significantly for about two decades, most likely because there are fewer habitats available where bontebok can reintroduced, combined with declining economic value. In an assessment of the population size of large herbivore species in the Western Cape, Kerley et al. (2003) note that before extensive agricultural transformation, the natural range could have supported around 6600 bontebok, but can now only support around 2500. While the current population size in the natural range is below that, it demonstrates the limited potential for additional bontebok conservation according to existing IUCN guidelines.

Landowners in the Western Cape, saddled with the opportunity cost of earning revenue from crop agriculture and viticulture, may be less inclined to continue to sustain wild populations of bontebok without the compensatory revenue stream from international hunters. As such, the Red List assessment may have the opposite effect of what it is intended to do.

While wild subpopulations within the natural range should be immediately eligible for expedited trophy hunting export permits, the Eastern Cape plays an important role in conserving wild bontebok subpopulations than the natural range. Bontebok select burnt veldt and kraal lawns (Watson et al. 2011), and thus can adapt easily to old fields that have been taken out of cultivation and managed as grazing lawns. The creation of such novel ecosystems (sensu Hobbs et al. 2009) may explains why Bontebok seem to thrive outside of their natural range. However, does the Eastern Cape province satisfy the IUCN’s conditions of firstly being ‘geographically close’ to the natural range and secondly ‘done to prevent extinction risk’? Arguably, neither condition is satisfied as the Eastern Cape – while being a neighbouring province to the Western Cape – contains distinct ecoregions (South African National Biodiversity Institute (2006- 2018), and delineating the ‘natural range’ of the bontebok, while subject to some debate, is generally thought to be limited to the Western Cape as a maximum extent (Birss et al. 2015; Radloff et al. 2016; Cowell & Birss 2017). Benign introductions into the Eastern Cape and other extra-limital areas have probably not been done with the explicit intention of reducing the extinction risk of bontebok. This is because wildlife ranches in South Africa are commercial enterprises focused on revenue generation (Carruthers 2008; Taylor et al. 2020), where farmers are conceptually more orientated towards the agricultural sector (Somers et al. 2020). As such, wildlife ranches are more aligned with a ‘working lands’ paradigm (Kremen & Merenlender 2018), where conservation should be seen as an outcome rather than a primary goal. These principles are reflected in the Biodiversity Management Plan for bontebok, where population increases both within and outside the natural range are recognised as important interventions. This is especially true if these subpopulations are free of hybridisation risk from blesbok (*Damaliscus pygargus phillipsi*) and connected through a meta-population to enhance genetic diversity (Cowell & Birss 2017). As such, wild subpopulations within the Eastern Cape could potentially contribute to broader bontebok conservation and be rewarded with a positive enhancement funding to facilitate revenue from trophy hunting. To encourage more reintroductions in the natural range and restoration of native bontebok habitat, a tiered certification structure could be implemented. This could include where hunting within the natural range is the highest level, followed by wild subpopulations within the Eastern Cape that do not overlap with the blesbok distribution (following Birss et al. 2015) and have not been hybridised with blesbok.

Wildness of bontbok subpopulations could be achieved easily in most non-wild subpopulations. Overall, 53% of subpopulations would only need to add five adult bontebok on average to pass the threshold for a minimum viable subpopulation size. Additionally, almost all properties could surpass the minimum threshold for space available to bontebok if camp fencing was removed and the whole property was made available to bontebok (currently only 21% of properties were fully open to bontebok subpopulations). Overall, most subpopulations are classified as intensively managed for both 'breeding' and 'space' (besides the Eastern Cape province). This indicates that increasing population size and area available will improve these scores as subpopulations will become more demographically self-sustaining with more natural breeding competition and be functionally integrated into natural home range areas. All properties averaged 'intensively managed' for predation pressure with the lowest interquartile range across properties, meaning that only limited exposure to mesocarnivores is possible. Increasing predation pressure would be substantially harder as apex predators are absent on most working lands besides those whose business model is ecotourism (Clements et al. 2018); and most wildlife ranchers actively persecute predators to protect high-value game stocks (Pitman et al. 2017). However, tolerance of predators could potentially be achieved if the returns from trophy hunting outweighed the opportunity cost of losing stock, especially seeing as trophy hunters are willing to pay a premium for hunting operations with demonstrable conservation benefits (Fischer et al. 2015). Finally, most properties also fared poorly on natural fluctuations in water provisioning. This may be difficult to change given the agro-ecological infrastructure of most properties and the barriers to the dispersal to natural water sources for most bontebok subpopulations. However, enabling seasonal water availability at artificial points could improve adaptive capacity. For example, the provision of season-long surface water has threshold effects on the survival and adaptability of wildebeest (Selebatso et al. 2018; Weeber et al. 2020). Overall, wildness could be significantly improved with moderate increases in subpopulation size and space available and more tolerance for predation. To achieve this, market-based incentives would have to be in place to overcome opportunity costs associated with such decreases in management intensity, and revenue from trophy hunting is key to achieving this.

Wildness is fast becoming a central tenet in South African conservation policy (see Chapter 1), with the recent High-level Panel of experts reviewing elephant, lion, leopard and rhinoceros management, breeding, hunting and trade using wildness as a unifying theme (DFFE 2020b). For example, Under goal 11 “South Africa repositioned and promoted as a destination of choice for legal, regulated and responsible hunting of the five iconic species, emphasising the authentic wildness experience, that is humane, grounded in universally-shared ethical principles, and that promotes conservation and rural livelihoods”, one of the recommendations is “the consideration of alternative approaches to the allocation of hunting permits that incentivises wildness of population and landscapes, such as preference to land associations that have green certification and drop fences” (DFFE 2020b p. 309). This provides a clear policy context for the role of wildness assessments in facilitating responsible, sustainable use of bontebok and other (sub)species. Interventions such as creating a commons-based approach to bontebok ‘wealth’ by dropping fences and establishing conservancy and community-based management structures could be directed at properties with high existing wildness scores and be integrated into social impact certification schemes to unlock overseas markets.

The necessity of redefining conservation value in policy also underscores the ineffectiveness of global top-down conservation assessments for wildlife in working lands. Due to the slow-moving nature of these global assessments and significant lags between synchronising updates across all multilateral frameworks, conservation decision-makers may be losing a key moment to capitalise on the investments by private landowners (sensu Clements et al. 2020). Our results corroborate this. The bontebok is still listed as Vulnerable on the IUCN Red List under D1 (<1000 adults) and B2ab(ii) (severely fragmented subpopulations with continuing decline in area of occupancy) (Radloff et al. 2016). This is arguably an overly negative framing, as these results, and those of Cowell and Birss (2017), show there are more than 1000 wild adult Bontebok if private sector data are included, and the definition of natural range is expanded to all of Western Cape (and even parts of Eastern Cape non-overlapping with blesbok range).

While the opportunity for further reintroduction in the natural range is threatened by ongoing agricultural transformation, a Red List assessment that triggers further restrictions on sustainable use does not ameliorate this situation as it makes it harder for landowners to earn revenue from trophy hunting and thus disincentivises reintroduction (as may be reflected in the declining value of live animal sales; *Supporting information 4.1*). As such, top-down regulations triggered by global assessments (*Supporting Information 4.3*) may not be appropriate for rapid change and innovation under the Anthropocene, especially to guide market-based incentives to unlock fine-scale economic and ecological value. Most landscapes outside of protected areas comprise 'working lands' that balance both agricultural production and conservation in novel ecosystems where historical species composition may have changed significantly but ecosystem functioning is sustained (Hobbs et al. 2009; Kremen & Merenlender 2018), which requires that conservation policies and tools can adapt and produce multifaceted value assessments.

Before mainstreaming wildness assessments into decision-making processes, one question that should be answered is whether to assess each subpopulation or aggregate the wildness score to the property level. In this sample, 30% of landowners managed more than one bontebok subpopulation. Assessing each subpopulation may give landowners flexibility in their business models, where wild subpopulations could be used for hunting and trophy exports and counted towards national conservation targets. In contrast, non-wild subpopulations used for breeding and live animal sales could be used to ensure a diversified revenue stream for the enterprise. However, assessing wildness at a whole-property level would enable easier monitoring, and reporting and integration into national indicators for biodiversity conservation and easier integration into market-based incentives such as certification schemes. Such operational issues should be discussed at stakeholder meetings and incorporated into the wildlife economy certification scheme currently under development (Turpie & Letley 2018).

4.5.1 Conclusions

My study demonstrates the considerable prevalence of wild bontebok subpopulations on 'working lands', meaning that the revenue from trophy hunting could be scaled up if a process was in place to ease the enhancement findings needed under US legislation. A survey conducted by SANBI in 2016 to support the development of the NDF indicated that around 30% of private landowners had reintroduced bontebok because of potential revenue from trophy hunting and that 75% of private landowners use revenue from trophy hunting in general to secure bontebok habitat and manage the land sustainably (SANBI, unpubl. data[7]). Increasing hunting could unlock larger land parcels bontebok, especially as the majority of revenue on trophy hunting WBLUs come from landscapes larger than 3000 ha (von Solms & Merwe 2020), which satisfies high-level policy ambitions for the wildlife economy (DFFE 2020b). Wildness assessments offer a nuanced approach to identifying subpopulations with conservation value and could be incorporated into permit applications and stewardship officer extension service audits. Qualifying enterprises might receive preferential government tax breaks for participating in the goals of the Biodiversity Management Plan (Cowell & Birss 2017) through tax incentives, as has been suggested for private rhino subpopulations (Clements et al. 2023), or be supported through habitat restoration initiatives and impact investment.

The Biodiversity Management Plan enshrines the principle of sustainable use of bontebok (Cowell & Birss 2017). However, such landowners are often inhibited from doing so because of increasingly complex and onerous trophy import regulations from the USA and other Global North countries. As such, the same blunt measures that are ostensibly put in place to protect threatened species are impeding the capacity of private landowners and communities to contribute to their conservation and thus improve their conservation status. This is an example of policy incoherence at international, national and sub-national scales (*Supporting Information 4.3*). Wildness assessments represent a more pragmatic assessment of rewilding, focussing on the ecological role subpopulations play in ecosystems and the resultant impacts on ecosystem functioning, even if these subpopulations exist in novel ecosystems. The potential for bontebok conservation is limited in the NDR due to extensive habitat transformation (Kerley et al. 2003; Clements et al. 2018), which our results corroborate and expand by sowing the limited wildness of private subpopulations in the Western Cape. However, the Eastern Cape has significant potential to protect wild subpopulations of bontebok as part of a managed metapopulation approach under the auspices of the BMP and linked to a market-based vehicle such as a wildlife economy certification scheme. These qualifying properties could receive market premiums from certification standards incorporating sustainable trophy hunting practices and contributing to landscape restoration and community

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[7] Unpublished non-detriment finding for *Damaliscus pygargus pygargus* (bontebok); Issued by the Scientific Authority of South Africa; October 2022..

beneficiation (Fischer et al. 2015; Wanger et al. 2017; Barchett 2021). Conducting wildness assessments in working lands provides a key tool for decision-makers to promote subpopulations and lands with conservation value and rapidly facilitate positive economic incentives to sustain biodiversity in a rapidly changing world.

Supporting Information 4.1 Trophy export and auction price trends

The CITES trade database was used to extract trophy export and import data for the last 10 years (2011-2020). All Artiodactyla exports from South Africa were extracted using only the term “trophy”. While all source codes for Bontebok were used, 81% were from source code W (wild), with 9% from source code C (captive-bred) and 7% from source code F (a variety of different management systems). The data were split into bontebok exports versus all other Artiodactyla exports (to measure comparative trends in other game hunt trends) and between the USA and all other countries as trophy importers (Figure S4.10). These data show the importance of the USA as a source of hunters for South African wildlife-based land uses as this country alone accounts for more game trophy imports than all other countries (5689 compared to 5309 trophies over the time period, including bontebok). Considering the importance of the USA hunting market to South Africa, the decline of bontebok exports to the USA may constitute a threat to the continued economic incentive of WBLUs to conserve this subspecies on private land.

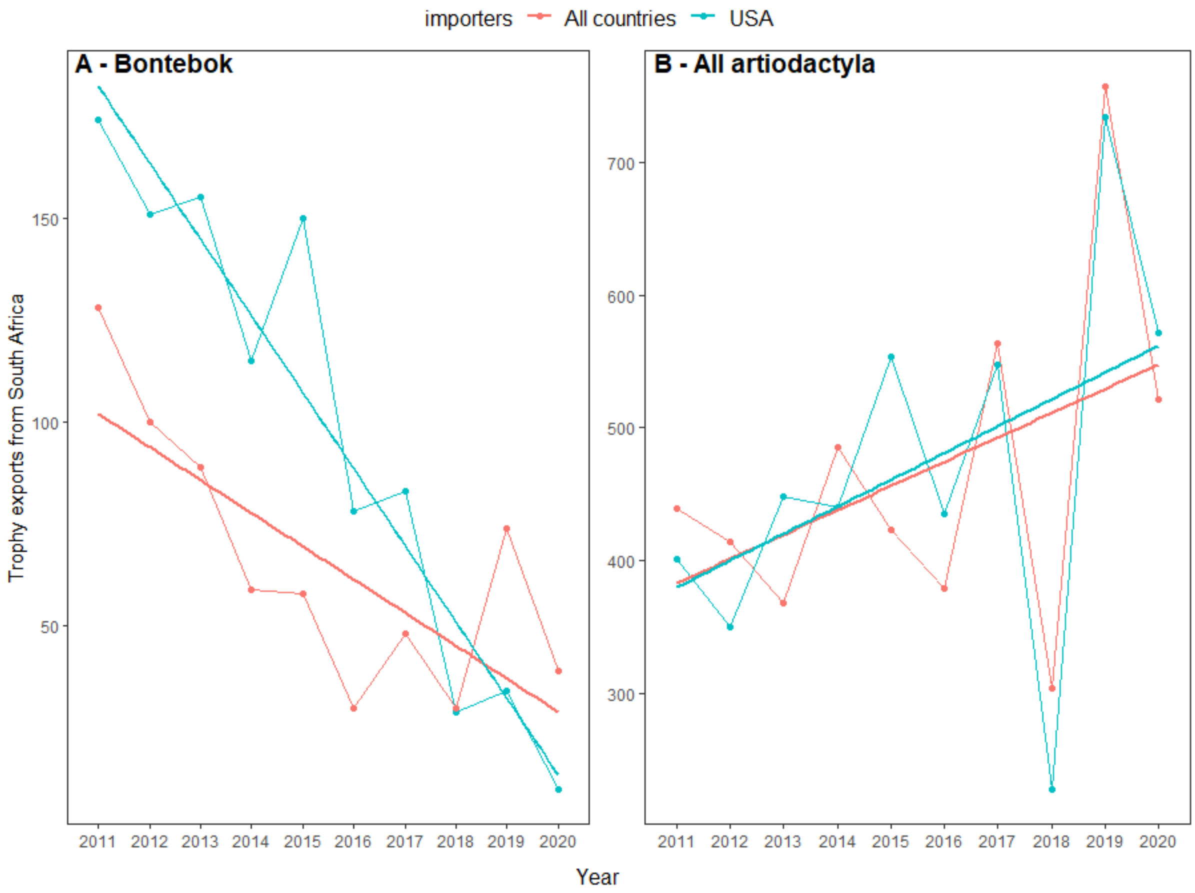

Figure S4.10. Trends in export quantities from South Africa to the USA and all other countries for (A) bontebok only and (B) all other importing countries. While bontebok exports to the USA have declined steadily between 2011 and 2020, exports to other importing countries have declined but have since stabilised. Comparatively, exports of all other artiodactyla species have increased over the same period to the USA and other importing countries.

I compiled game auction price data from seven different game auctions in the Eastern Cape and categorised them into high-value game species (those with average prices over an order of magnitude above other game), colour variant types (whose price is variable), plains game species and bontebok (Figure S4.11). Bontebok prices have decreased significantly since 2015 compared to plains game and colour variants, to the point that the 2021 average price was lower than both those groups. Average high-value species auction prices have also dropped significantly over time. In the Eastern Cape auction dataset, these include white rhino (*Ceratotherium simum*), buffalo, sable (*Hippotragus niger*), roan (*Hippotragus equinus*), hippo (*Hippopotamus amphibius*) and Livingstone's eland (*Taurotragus oryx livingstonei)*. The costs of purchasing and protecting these populations without a compensatory revenue stream through hunting may be causing active disinvestment by private landowners. For example, 28% of current White Rhino owners are disinvesting due to financial stress and thus white rhino populations may become concentrated into ever fewer properties with increasingly intensive management (Clements et al. 2020), which is counter to the goal of current policy that seeks to ‘rewild’ rhinos (DFFE 2020b). Plains species are less dependent on an international hunting clientele as they are the focus of local ‘biltong’ hunters, and auction prices are generally lower and more stable. This is probably why wildlife economy business models that incorporated local hunting operations or mixed livestock and hunting operations fared better during the COVID19 pandemic than the more specialised trophy hunting model, which consisted of an average of 87 ± 13% foreign clientele compared to 30 ± 21% for more locally-focussed business models (Clements et al. 2022).

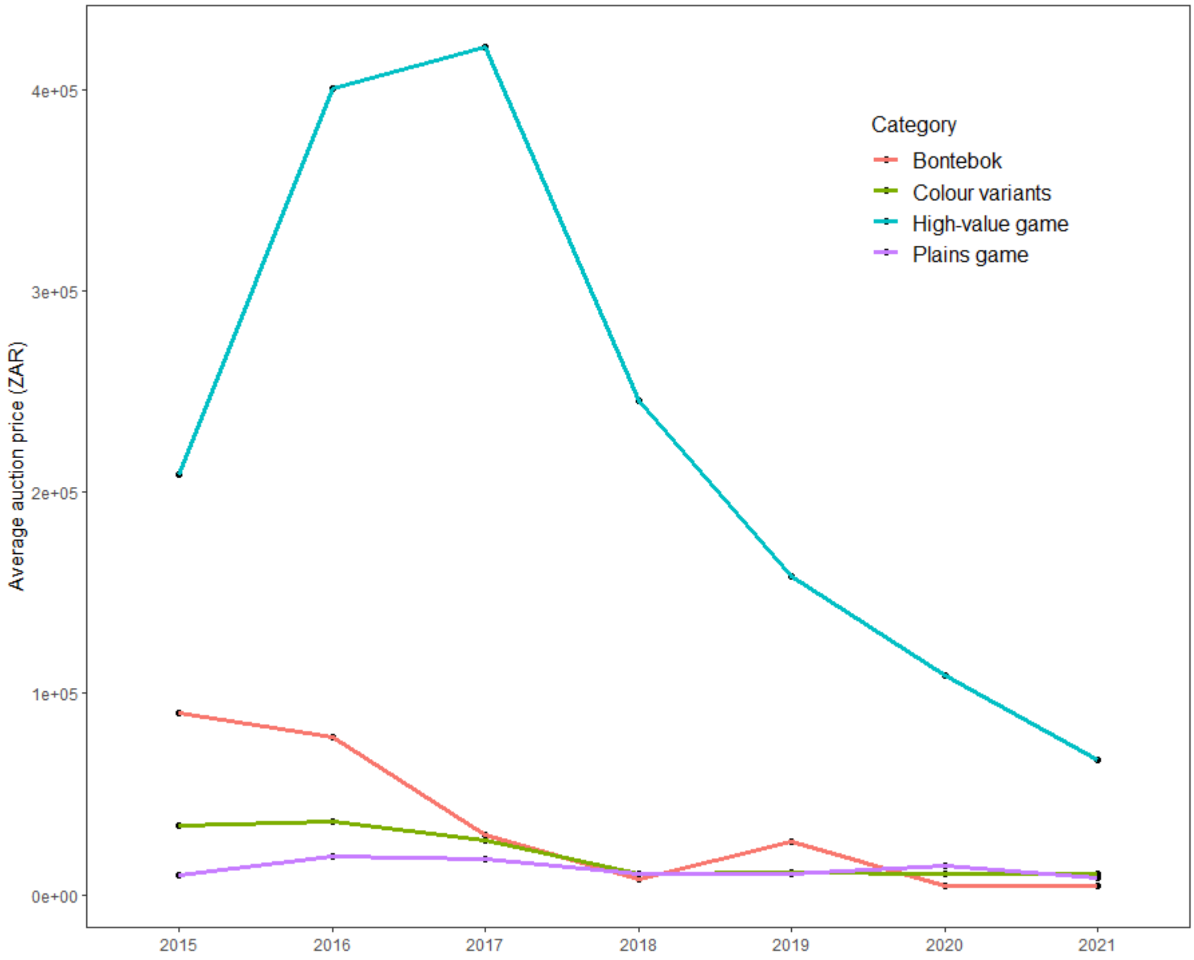

Figure S4.11. Average game price values at Eastern Cape auctions between 2015 and 2021 for different categories of game. While there has been a general decline in high-value game species, the value of bontebok has decreased over time to the point that its value in 2021 was lower than plains game species.

Supporting Information 4.2 Adaptations to wildness framework for Bontebok

Bontebok owner questionnaire

General Questions	
Name, contact number and email address	
Property name (please include cadastral name)	
GPS coordinates (if available)	
Property location (Province and district)	
Property size in ha	
How much of this area is used for wildlife? (provide either area or %)	

Bontebok Questions	
7	What is the total area available to bontebok on your property?

8 Please complete the table below for all areas where bontebok are kept

Bontebok areas	Size of area or camp	Total number of bontebok in area/camp	Number of breeding males	Number of breeding females	Number of young	Frequency of replacing males (Never or No of yrs)	Type of fencing around area (Describe and indicate whether electrified)	Number of artificial waterpoints that provide water all year round accessible to bontebok within the area	Frequency of supplementary feeding (Daily, Monthly, Dry season, Only during droughts, Never)	Type of supplementary food provided (example lucerne, mineral & salt licks; artificial planted areas)	Frequency of veterinary care provided	Briefly provide information on the type of veterinary care provided	Have the herd been tested for genetic purity (Yes / No)	List other species present in the same area
Open area														
Camp 1														
Camp 2														
Camp 3														
Camp 4														
Camp 5														
Camp 6														
Camp 7														
Camp 8														
Camp 9														
Camp 10														

9 What other rare and/or threatened species do you also have on your property?

10 Is your fencing permeable to predators? If yes, what species do you think could get through?

11 Do you practice any measures to control predator numbers in your bontebok areas?

	Yes/No	Which predator/s is this relevant to?
Nothing		
Live capture and translocation		
Guarding dogs		
Lethal, Selective - Shoot problem animals		
Lethal, Selective - Hunting or culling		
Lethal, Non-selective - poison		
Lethal, Non-selective - gin traps		
Lethal, Non-selective - hunting dogs		

12 Contact details of other possible bontebok owners that we can include in survey

Name	Surname	Phone Number	E-Mail Address

Letter to landowners from SANBI

SANBI
Biodiversity for Life
South African National Biodiversity Institute

22 July 2019

Dear Sir/Madam

The Biodiversity Research, Assessment and Monitoring Division within the South African Biodiversity Institute (SANBI) is inviting you to participate in a study to update the current non-detriment finding and a future revision of the regional Red List assessment. In order to do this the wildness of the bontebok populations on private and State land needs to be assessed in line with a framework developed for this specific purpose and used very successfully in the information document provided by South Africa at CITES CoP17 in support of the down-listing of Cape mountain zebra from Appendix I to Appendix II. A scientific publication describing the framework can be found at the following link (https://conbio.onlinelibrary.wiley.com/doi/abs/10.1111/cobi.13299).

The approach followed will be that of a questionnaire survey which will be circulated through the various industry organisations and will include questions addressing specific aspects to wildness such as property variables, species composition, abundance, and management interventions. The length of the questionnaire will take approximately 25 – 35 minutes to complete.

Your participation in this study is voluntarily and all information provided through this process are be considered confidential and will not be made available without the consent of the landowner. We believe that your participation in the questionnaire would assist in meaningful outcomes directed towards better decision-making and policies for private bontebok owners in the country. Ascertaining the wildness of populations will assist government to unlock the economic value of managed populations and design better incentives for lands that are managed both for socio-economic prosperity and biodiversity.

For further inquiries kindly contact the project leader Dr. Jeanetta Selier (J.Selier@sanbi.org.za) and/or research intern Sinozuko Silanda (S.Silanda@sanbi.org.za).

Kind regards

Dr Jeanetta Selier

Senior Scientist, Zoological support to Scientific Authority & Wildlife Economy
Biodiversity Research, Assessment and Monitoring
South African National Biodiversity Institute (SANBI)
Cell: +27 (0)83 45 88063
Tel: +27 (0)12 843 5117

Criteria for adopting the wildness framework to bontebok

The following modified protocol was established to adapt the wildness framework (developed in chapter 3) to the bontebok dataset (and see Table S4.7): If the subpopulation occurred in camps, the size of the camp area was used when evaluating space and artificial water-point densities density thresholds. For artificial water-points: I followed the original framework from Child et al. (2019) where CM = more than 1 AWP for animals in camps and IM = more than 1 AWP in the extensive areas. For reproduction, I assumed that if there was only 1 breeding male, even if they are replaced, breeding competition is still negated and thus scores as Captive Managed.

Table S4.7. Modifications to the scoring protocol for three wildness attributes, based on the bontebok (*Damaliscus pygargus pygargus*) dataset.

Category	**Explanation**	**Wildness score**

Disease and parasite resistance		
Never	No vet care listed	5
Ad hoc - preventative population level	If landowner listed vaccinations against non-native diseases	4
Ad hoc - population level	General annual or periodic population level health checks (such as to de-tick, or cattle dips, or blood tests)	3
Ad hoc - individual	Care provided to sick individuals directly when needed (prevents animals from dying)	2
Continuous (more than once a year)	Direct vet care provided to all animals more than once a year	1
Supplementary feeding		
Never	No supplementary feeding listed	5
During droughts	Lucern and/or mineral licks provided during drought only (WC and EC properties only)	4
Dry season	If lucern is provided in dry season. If just mineral licks are provided in dray season this becomes Near Natural	3
Weekly / Monthly	If feeding of lucern or mineral licks listed as weekly or monthly frequency then Intensive Management assumed	2
Daily	If daily is listed, then assume this is Captive Managed	1
Predator control		
NA	Apex predators assumed not to be resident on any land	5
NA	Apex predators assumed not to be resident on any land	4
No control	If no predator control there will be continuous exposure to meso predators. This includes properties with live capture of predators.	3
Lethal control but permeable fencing no camp	Population not in predator proof camp and /or perimeter fence permeable to mesopredators = occasional exposure to mesopreators.	2
Lethal control and impermeable perimeter or camp fencing	This assumes there is no exposure to mesopredators.	1

Supporting Information 4.3 Perverse policy incentives in multilateral environmental agreements

Most countries in the world are party to both the Convention on Biological Diversity (UNCBD) and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). These multilateral environmental agreements (MEAs) set the targets of what *should* be done but do not produce the theories of change of how things should be done (Burgass et al. 2020); they often lack quantifiable terms, are overly complex or redundant with one another (Butchart et al. 2016); and do not consider synergies or trade-offs between the target (Di Marco et al. 2016). Worryingly, within an MEA, there is no theory of change for how the various targets link together philosophically or operationally, and the drivers behind multiple MEAs have not been thoroughly scrutinized for possible synergies and perverse incentives. Here, I discuss how the interactions between the IUCN Red List of Species (under UNCBD) and complementary indicator, the Living Planet Index, interact negatively with the socioeconomic drivers regulated by CITES and ultimately inhibit African countries from being able to achieve the headline 30x30 target under target 3 of the Global Biodiversity Framework (GBF). Policy incoherence at the global and national scales is fuelled by two perverse and countercurrent feedback loops: public pressure (especially from the Global North citizenry) and the drive towards open-access biodiversity data endorsed by the Global Biodiversity Information Facility (GBIF) (Figure S4.12).

Much of conservation policy and subsequent regulation at global and national scales is determined by the threat status of species as determined by the IUCN Red List of Species assessments (Butchart et al. 2006; Rodrigues et al. 2006, 2014; Bubb et al. 2009; Pfab et al. 2011; Child et al. 2016; Bennun et al. 2018; Betts et al. 2020); although the use of the Red List for many decision-making processes, including setting policies on sustainable use, is inappropriate (Possingham et al. 2002; Jepson 2022). For example, in South Africa, Red List assessments form the backbone of the Threatened or Protected Species List (TOPS), an instrument under the National Biodiversity Act, which then sets in motion regulations that trigger permit applications for those species, which has promulgated backlash from the private wildlife sector who are attempting to move under agricultural regulation and increase their capacity to manage wildlife like domesticated stock (Somers et al. 2020), which is the opposite of what is intended by the Biodiversity Act. The problems with using the global Red List to set policy are threefold: 1) **scale mismatch** - the assessment is at the global scale, so any differences in conservation status regionally or nationally are obfuscated even though the global status will come to set national regulations, which is amplified through other multilateral mechanisms that incorporate global Red List data, such as CITES. To ameliorate this, countries can produce national Red List assessments. For example, the South African national Red List of mammals shows a much more accurate portrayal of wildlife conservation than the global Red List in that key species, such as Lion (*Panthera leo*) and African Elephant (*Loxodonta africana*) are listed as Least Concern (Child et al. 2016) compared to the Vulnerable[8] and Endangered[9] respectively on the global Red List. However, Global North decision-makers and

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[8] https://www.iucnredlist.org/species/15951/115130419
[9] https://www.iucnredlist.org/species/181008073/223031019

funders will preferentially use the more 'trusted' brand of the global IUCN Red List rather than regionally or national produced data (Bennun et al. 2018), which means these assessments still trigger enhancement findings relating to being listed on the Endangered Species Act and other regional or domestic legislative mechanisms that inhibit use and trade (for example Figure S4.12). As such, the efficacy of producing national Red Lists is undermined by the mainstreaming of the global Red List into decision-making processes that impact investment into the wildlife economy.

Secondly, 2) **data availability and conflation** – the Red List assessments are often based on poor or outdated data and then extrapolated across the range to produce an assessment (Cazalis et al. 2022). Population trend data come from a few sites, and these sites are often in poorly managed or underfunded protected areas or in commons where wildlife has no economic value and then becomes subject to illegal harvesting. This is exemplified by the Living Planet Index, which quantifies changes in abundance over time of different taxa in different regions (Collen et al. 2009) and famously asserts there has been a 70% decline in abundance since the 1970s[10]. However, deeper analysis of these trends reveals that only a small proportion of the population data drives the negative trends, and if they were removed, a net positive trend would be observed (Leung et al. 2020); that random fluctuations spuriously exacerbate the negative trend (Buschke et al. 2021); shows different trends when disaggregate regionally – such as the positive trend shown in southern Africa (Craigie et al. 2010); does not really measure abundance at all (Puurtinen et al. 2022); and that managed populations show under sustainable use show positive trends but are lumped with declining populations under a catch-all category of 'utilised' (McRae et al. 2022). Despite these failings, the composite LPI is used extensively as an indicator in the UNCBD targets and for media campaigns for conservation (Ledger et al. 2023), which reinforces the public perception that all wildlife subpopulations are threatened and that hunting, culling or trading individual animals is a morally incorrect thing to do, despite legal international hunting and trade causing negative population impacts in the minority of cases (Di Minin et al. 2021; Challender et al. 2023). Conversely, population data from well-managed areas or from socio-economic systems that employ legal harvesting are often not included in these assessments because the i) the assessor team does not have access to it and

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[10] https://www.livingplanetindex.org/latest_results

ii) the landowners are often disincentivised from sharing data due to perceived risk of further regulation (see below). Additionally, the IUCN Red List criteria do not adequately distinguish between legal and illegal use in the threat categories, leading to conflation between legal (hunting) and illegal use (poaching) and overestimation of wildlife use as a threat to species (Marsh et al. 2022).

Finally, 3) **negative framing** – the Red List is inherently a negative framing of conservation, emphasising the threat of extinction of species, which feeds into an alarmist public purview and introduces a cognitive dissonance that a species cannot at once be listed as threatened and also be sustainably hunted. In other words, the context of the socioeconomic system and legal harvesting of certain subpopulations is overridden by the pervasive public perception that hunting must innately be unsustainable if the species is ‘threatened’. This public pressure, facilitated and amplified by special interest groups like animal welfare organisations, then applies pressure to policymakers and funders to both further uplist or retain precautionary listings on the Red List and CITES as well as suspend or ban the very practices that enabled the conservation and expansion of many ‘threatened’ species, culminating in high-level political decisions such as the proposed ban on trophy hunting imports in the UK (Webster et al. 2022; Clark et al. 2023).

The tendency to over-list species subject to sustainable use on the IUCN Red List combined with the general headline message of the LPI that wildlife is declining ripples into Global North institutions that regulate wildlife trade (CITES) and provide intergovernmental funding for conservation projects (the Global Environment Facility, GEF). Because the Red List is an easy proxy for identifying at-risk species, combined with public pressure to save ‘threatened species’, there has been a tendency to try ‘match’ or ‘cross-walk’ the species listed on the Red List with the species listed on CITES appendices to restrict trade (Gorobets 2020), despite these two bodies having separate assessments and decision-making processes (Challender et al. 2019; Berec & Šetlíková 2021). This has led to conflation of these two policy processes with pseudo-replication of the underlying data and theories of change, undermining the independence of CITES and CBD as well as mischaracterising the threat of trade and misallocating resources to deal with the root causes (Challender et al. 2022). It is also well-known that media and public pressure play a role in CITES decision-making, as animal welfare and wildlife conservation NGOs unduly influence state actors during conference of parties negotiations or portray legal trade as a threat to species (Tyrrell & Clark 2014; Challender &

MacMillan 2019; Cusack et al. 2020), which leads to scientific evidence being eschewed in favour of sentimentality and over-listing of 'charismatic species' (Shiffman et al. 2021). Similarly, public pressure has overridden scientific consensus and led to the likely ban of trophy hunting imports from legally and sustainably harvested systems into the UK, despite the obvious contradictions with domestic hunting policy (Clark et al. 2023). The result is that over-listing on CITES and lack of agreement on regional differences in population trends (to list some species under Appendix II and allow best-performing countries to benefit from trade) triggers listing on national legislation, such as the Endangered Species Act of the USA, which can then negatively affect livelihoods of the people living with wildlife and ultimately be counterproductive to conservation (Weber et al. 2015; Booth et al. 2021; Natusch et al. 2021; Nyamayedenga et al. 2021; Clark et al. 2023; Coe et al. 2023). Listings on the ESA tend to remain or lag behind downlistings on CITES because of the different decision-making processes involved in de-listing (Smith et al. 2018a), and because the USA has pressure to reduce wildlife imports that are not listed on CITES (Watters et al. 2022). Similarly, private corporations looking to comply with international frameworks simply adopt static lists from the IUCN Red List and CITES as a basis to restrict commercial trade in wildlife products (Green et al. 2022), without assessing the provenance of the product and its potential positive impacts.

On a national scale (using South Africa as a case study), the combined effects of blunt threat assessments combined with intense public pressure to uplist species on CITES and restrict trade results in three regulatory processes that block the implementation and potential value of legal and sustainable wildlife hunting. Firstly, the Red List is a key input into the Threatened or Protected Species List (TOPS), which is a gazetted list of species in various threat categories under the National Environmental Management: Biodiversity Act (NEMBA), 2004 (Act No. 10 of 2004). The TOPS listings trigger a slew of administrative barriers to wildlife use and trade, including permit applications – where individuals and enterprises must apply for a permit for hunting, capturing, killing, importing, exporting, re-exporting, transporting, and selling any specimen of a listed species. Such permits also carry penalties for non-compliance. While TOPS is intended to ensure that South Africa meets its international obligations under conventions like CITES and prevent negative consequences for biodiversity during use and trade, it often presents logistical barriers that can impede revenues from international hunters. Such impediments are also facilitated by provincial management agencies who base their provincial ordinances on principles embedded in the UNCBD and Red List, such as the preservation of genetic diversity and the delineation of historical distribution ranges to guide translocations (Blackmore 2017b), which can make it more difficult for permits to be issued. As such, the wildlife ranching industry becomes frustrated with the government and recently sued the Department of Forestry, Fisheries and Environment (DFFE) for amendments made to the TOPS list[11].

Secondly, the CITES listing triggers non-detriment findings (NDFs) conducted by the appointed Scientific Authority in each member state to assess if national populations will be affected by the proposed listing, enabling national quota setting for the species in question, and restricted hunting quotas. While no trade is permitted for species listed on Appendix I, hunting quotas for species listed on Appendix II are primarily determined by the Scientific Authority because Appendix II species are not necessarily threatened with extinction, their trade must be controlled to avoid any negative impacts of utilisation. The country's scientific authority must provide a non-detriment finding, meaning they must determine that the export (which could include hunting as a form of off-take) is not detrimental to the survival of the species. While well-intentioned, these processes tend to take time as the NDFs must be compiled by the SA members in consultation with key stakeholders who contest the data (see below), must undergo gazettement by the DFFE, must undergo public consultation, and then must finally be adopted by the relevant provincial management authorities responsible for issuing permits, who often are not briefed / trained on the conditions underlying permit approval (such as the development of specific monitoring and harvesting plans for local subpopulations). More seriously, the landowners and communities themselves often do not understand the technical details of what is needed to apply for a permit for a species listed on CITES and/or TOPS. Only around half of wildlife ranchers have formal management plans to begin with (Cousins et al. 2010; Pienaar et al. 2017) (see Chapter 5), and most do not have the capacity or time to add biodiversity monitoring plans to these plans without assistance from extension officers. However, extension support is severely under-capacitated and declining in its effectiveness for both the agricultural and environmental sectors (Barendse et al. 2016; Gelderblom et al. 2020). Compounding this are the technical assessments the importing countries need to demonstrate conservation and

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[11] https://www.sanews.gov.za/south-africa/creecy-withdraws-tops-regulations

community enhancement under the ESA and EU regulations. These technical barriers and long processing times of permit application (both domestically and internationally) require significant investment into producing the evidence and documentation required, which can both increase the costs of the hunting operation and delay or even cancel some hunts due to lags between the various permits needing to be processed. For example in South Africa, while the expansion of rhino range and increase in population numbers was facilitated by the sustainable use of the species, most notable trophy hunting ('t Sas-Rolfes et al. 2022), the facilitation of revenue-earning options from these rhinos have not kept pace. While the number white rhinos (*Ceratotherium simum*) have not been subject to hunting quotas, high prices and high administrative barriers have generally ensured that only a limited number are hunted each year ('t Sas-Rolfes et al. 2022). This leaves open the exciting possibility that if these technical barriers were brought down by specialised extension support and streamlined enhancement findings such as through wildness assessments, the number of white rhino hunted could increase and enable viable community-based wildlife economies. However, compounding these technical issues is that the demand for hunting 'threatened' species may be declining (especially when prices are high) due to the extremely negative press and personal attacks on trophy hunting (Nelson et al. 2016; Macdonald et al. 2016). Taken together with the absent or weak stance of the government to defend sustainable use, it means that the battle for the hearts and minds of consumers is being lost.

All of which is reinforced, thirdly, by public pressure. Such pressure is again facilitated through NGOs (similar to the lobbying of CITES member states), which successfully interrupt the implementation of hunting quotas through legal challenges enabled by the false dichotomy of species being listed as threatened on global databases and also subject to legal wildlife trade through CITES. For example, in 2022, Humane Society International successfully blocked the issuing of hunting quotas for leopards, black rhinos (*Diceros* spp.) and elephants on a technical procedural basis but with the substantive underpinning that "there was no scientific proof that such hunts would not be detrimental to the species"[12] (despite NDFs having been conducted and gazetted for all three species as per CITES regulations). As such, the public perception of threatened species being decimated by legal hunting, combined with the national legislation

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[12] https://www.dailymaverick.co.za/article/2022-04-26-animal-protection-ngo-stops-trophy-hunting-quotas-for-now/

and tools used is heavily influenced by Global North principles, has left South Africa's (and Africa's) sustainable use stance open to attack and sensationalism, where even allotted hunting quotas are routinely not filled (for example, Nyamayedenga et al. 2021). Similarly, public pressure has crept into national policy itself, with the latest White Paper on Conservation and Sustainable Use of Biodiversity, recently approved by cabinet[13], now considering the 'mental health' of wildlife under the definition of 'animal well-being'[14], which opens sustainable use up for further legal contestation and conflates animal rights with welfare. Essentially, these philosophical positions pit the morality of killing individual animals, and thus the rights of individual animals, against Leopold's (1968) 'land ethic' that justifies sustainable use through the culling/hunting of individual animals on the basis that it leads to net flourishing of the population involved and of the ecosystem overall (King 1991; Di Minin et al. 2021). This polemic also does not acknowledge that 'wild' subpopulations are inherently maintaining their well-being through expressing their evolutionary and ecological roles and that the animal rights of individual animals of wildlife populations should be reserved for captive breeding situations where the animals are reliant on humans for survival. As such, these two arguments ballast opposite ends of a spectrum and should not be seen as mutually exclusive.

All these factors amount to serious impediments in creating value out of wildlife through sustainable use, especially species listed on the Red List, TOPS and CITES. The combination of high technical requirements in proving enhancement and compliance with multiple pathways of permit application, the contested hunting quota allocation and severe public pressure on hunting causing delays in approval and declining demand, and the increasing cost of protecting wildlife without equivalent revenue-earning options mean that private landowners are increasingly looking to transition to more agricultural models of wildlife production and/or (re)convert to livestock agriculture. For example, the DFFE and the Department of Agricultural, Land Reform and Rural Development (DALRRD) were recently involved in a legal contestation to list key game species under the Animal Improvement Act to enable domestication of these species (Somers et al. 2020). This originated from the frustration of the wildlife ranching sector with DFFE regulations and governance. Additionally, the cost of

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[13] https://www.dffe.gov.za/mediarelease/whitepaper_biodiversitysustainableuse

[14] South Africa's Biodiversity 2023: White Paper on Conservation and Sustainable Use of South Africa's Biodiversity (pg 21) https://www.dffe.gov.za/legislation/whitepapers

keeping high-value species like rhinos is escalating, meaning that many rhino owners are disinvesting and remaining rhinos are being consolidated into fewer properties (Clements et al. 2020), leading to more intensive rhino ranches (Clements et al. 2023), which is entirely contrary to the goals of the High-level Panel that called for more extensive rhino ranching and greater 'wildness' of rhino subpopulation (DFFE 2020b). It also means that, ironically, all listed species (which are threatened and thus species of conservation concern) may be losing value and/or becoming too expensive to keep without being able to generate revenue, which may reflect a trend where such species are increasingly being replaced by common plains game and/or livestock (*Supporting Information 4.1*, Figure S4.13, Figure S4.14). For example, properties conducting trophy hunting had 25% fewer threatened species than properties conducting ecotourism only (Taylor et al. 2021). This trend is corroborated by recent studies that show, firstly, 63% of wildlife ranchers in South Africa would consider transitioning back to traditional livestock and/or crop agriculture if revenue from trophy hunting continues to be restricted or banned (Parker et al. 2020); and 91% of Namibian conservancy residents were not in favour of a trophy hunting ban where only 11% of respondents would continue to support wildlife on communal lands if a ban were enacted (Angula et al. 2018). Finally, between 46% (Taylor et al. 2015, 2020) and 54% (SANBI, unpubl. data) of wildlife ranches already constitute mixed wildlife and livestock/crop farms; with these more agro-ecological enterprises able to better adapt (and in some cases increase revenue) to global economic shocks like the COVID-19 pandemic by scaling up agricultural production (Clements et al. 2022). These results suggest that wildlife ranching in South Africa is poised to transition back to livestock after having 'rewilded' their lands over the past few decades (Carruthers 2008).

If this happens, it will significantly impede South Africa's potential to meet target 3 of the GBF – the 30x30 headline target – of putting 30% of terrestrial areas under formal conservation protection[15]. The primary mechanism to achieve this is the declaration of working lands as Other Effective Area-based Conservation Measures (OECMs), as reflected in the headline indicator for target 3 of 'coverage of protected areas and OECMs'. While a full analysis of the potential of wildlife ranches to qualify as OECMs is presented in Chapter 3, the bottom line is that the OECM framing alienates private landowners as being a 'biodiversity first' assessment

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[15] https://www.cbd.int/gbf/targets/3/

without any economic incentives to go through the effort of being recognised as such. If OECMs were to be used as part of the evidence for enhancement findings to facilitate trophy hunting permits, perhaps landowners would be more willing to consider it. However, this iscurrently, not the framing of the OECM assessment, and it seems like the government is 'wanting its cake and eating it too' when it comes to counting the land under wildlife ranching to its national targets but not supporting the socio-economic drivers that enable these land-uses to exist. Considering that the extent of trophy hunting estates across Africa and in South Africa specifically exceeds the total area of the formal protected area estate (Lindsey et al. 2007b; Taylor et al. 2020), the failure to declare these areas as OECMs, or even actively lose these areas to other land-uses, means that the opportunity for national governments in Africa to meet the 30x30 target would be severely compromised.

Under this scenario, African range states would have to rely on failing protected area models and traditional protected area expansion using state resources. The problem is that there are minimal and declining state resources to manage protected areas effectively, let alone expand them (Geldmann et al. 2018, 2019; Lindsey et al. 2018, 2020, 2021, 2022; Coad et al. 2019; IUCN ESARO 2020). For the Global North to rely on the Global South to meet protected area targets without facilitating relevant economic drivers will result in the same patterns of fortress conservation and biodiversity decline everywhere else (Obura et al. 2021). Relying on protected areas then feeds back into the headline wildlife declines exemplified in the LPI and global Red Lists because there is no budget to manage them properly and no incentive for communities to live with wildlife, exacerbating future declines and feeding back into further hunting and restrictions. No matter what, governments will need to bring private landowners and communities into the 30x30 tent. Even if the sustainable use model is abandoned completely, only one third of wildlife ranchers would consider following the ecotourism / stewardship route, and the ecotourism model requires large capital investment and is only viable in certain landscapes as opposed to trophy hunting that can operate viably in less 'spectacular' and remote areas (Clements & Cumming 2018; Parker et al. 2020; Clements et al. 2022), so it is very doubtful this model alone could provide the necessary coverage – and provide the socioeconomic benefits needed – to meet 30x30 equitably and justly and effectively.

Global North priorities are firmly set on achieving Target 3 of the GBF where its researchers are prolific at devising spatially explicit strategies for expanding protected areas in the Global

South to meet the targets (Saura et al. 2018; Barnes et al. 2018; Donald et al. 2019; Visconti et al. 2019; Sayre et al. 2020; Chauvenet et al. 2020; Maxwell et al. 2020; Gurney et al. 2021, 2023; Jetz et al. 2022). This target is then preferentially funded by the Global Environment Facility[16], focusing on establishing OECMs (Alves-Pinto et al. 2021). As such, there is no global or national emphasis on the links between Target 5 ("Use, harvesting and trade of wild species is sustainable, safe and legal") and Target 3. Compounding this, the headline indicator for Target 5 is a fisheries indicator, and terrestrial sustainable use indicators rely on both the LPI and the Red List Index 'disaggregated by species in use'[17], which then feeds back into the problems described above as no positive use cases are documented, and there is now no global mechanism to collect such data. From the wildlife ranching sector itself, there is active resistance to sharing data because of fear of further government regulation and penalties, poaching and a general perception that these data will be used against them. This then reinforces the idea that wildlife is in decline. The positive population trends on private land sustainable use models cannot be properly monitored.

This conceptual model has shown a scale mismatch between global assessments and national reporting and socio-economic drivers needed at local scales to deliver on the targets. The global assessments feed into public misinformation, which pressures African states to limit sustainable use, thereby disincentivising local landowners and communities from protecting wildlife as the options to earn revenue are increasingly curtailed. National governments, with their focus on meeting area-based targets are heavily influence by international funding that focuses on illegal wildlife trade (and not proactively supporting legal wildlife trade) and is increasingly hostile towards trophy hunting. Currently, CITES does not require or encourage an impacts-based evaluation of its decision-making (Cooney et al. 2021), which should be integrated into the framework alongside broadening the range of criteria used to make decisions (such as wildness assessments of subpopulations) and the input of local communities who live alongside wildlife. Additionally, CITES should consider socio-economic considerations alongside biological criteria in decision-making (Abensperg-Traun 2009). The development of conceptual feedback models like that presented in Figure S4.12 could facilitate such

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[16] https://www.thegef.org/projects-operations/database

[17] https://www.cbd.int/gbf/targets/5/

amendments to CITES and provide much-needed functional links between Targets 3 and 5 in the GBF.

An interesting case study in the interplay between global listings, data availability and economics is the Cape mountain zebra (*Equus zebra zebra*), an endemic subspecies to South Africa (Figure S4.13). Due to historical threats, it was listed as Vulnerable on the IUCN Red List until a survey of private landowners and their subpopulations was conducted in 2015 (Hrabar & Kerley 2015). This enabled the wildness framework (presented in Chapter 3 and this chapter) to be applied to the private population to quantify the number of subpopulations with conservation value. This enabled the downlisting to Least Concern in the 2016 national Red List assessment of mammals (Child et al. 2016), as 81–98% of individuals on private land were considered wild and free-roaming subpopulations and were thus eligible for inclusion in the assessment, which raised the total population size within the natural distribution range to 2,381–3,247 mature individuals with an increasing population trend over three generations (Hrabar et al. 2016)[18]. The downlisting, based on an assessment of wildness of the private subpopulations, then enabled the national government through the Scientific Authority to propose a downlisting from Appendix I to Appendix II at the 2016 CITES conference of parties[19], which was duly granted and the subspecies was moved to Appendix II in 2017[20]. This triggered a petition to the US Fish and Wildlife Service in 2017 by the Professional Hunters Association of South Africa (PHASA) to delist the subspecies from “endangered” to “threatened” to reduce the delays and technical barriers associated with trophy exports by US hunting clients (see Figure S4.12)[21]. However, that petition, although supported by US Fish and Wildlife Service (USFWS) in that “we find that the petition presents substantial scientific or commercial information indicating that the petitioned action may be warranted for the Cape

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[18] https://www.iucnredlist.org/species/7959/45171853

[19] Consideration of proposal for amendment of appendices I and II and related news article: https://www.news24.com/news24/cape-mountain-zebra-downlisted-at-cites-cop17-20160929

[20] https://speciesplus.net/species#/taxon_concepts/12266/legal

[21] Evaluation of a Petition To Delist the Cape Mountain Zebra or Reclassify the Subspecies as a Threatened Species Under the Act and associated news article: https://www.nrahlf.org/articles/2018/4/24/usfws-moves-to-delist-cape-mountain-zebra/

mountain zebra due to the reduction or elimination of threats", the subspecies remains listed as Endangered[22], which indicates the significant lags between CITES and domestic legalisation.

When we look at the potential impacts of these various down-listings and market signals, we see that the average game auction price (live animal sales) was higher than the average plain games price until 2016 (the year the subspecies was listed as Least Concern) before it declined

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[22] https://ecos.fws.gov/ecp/species/10539

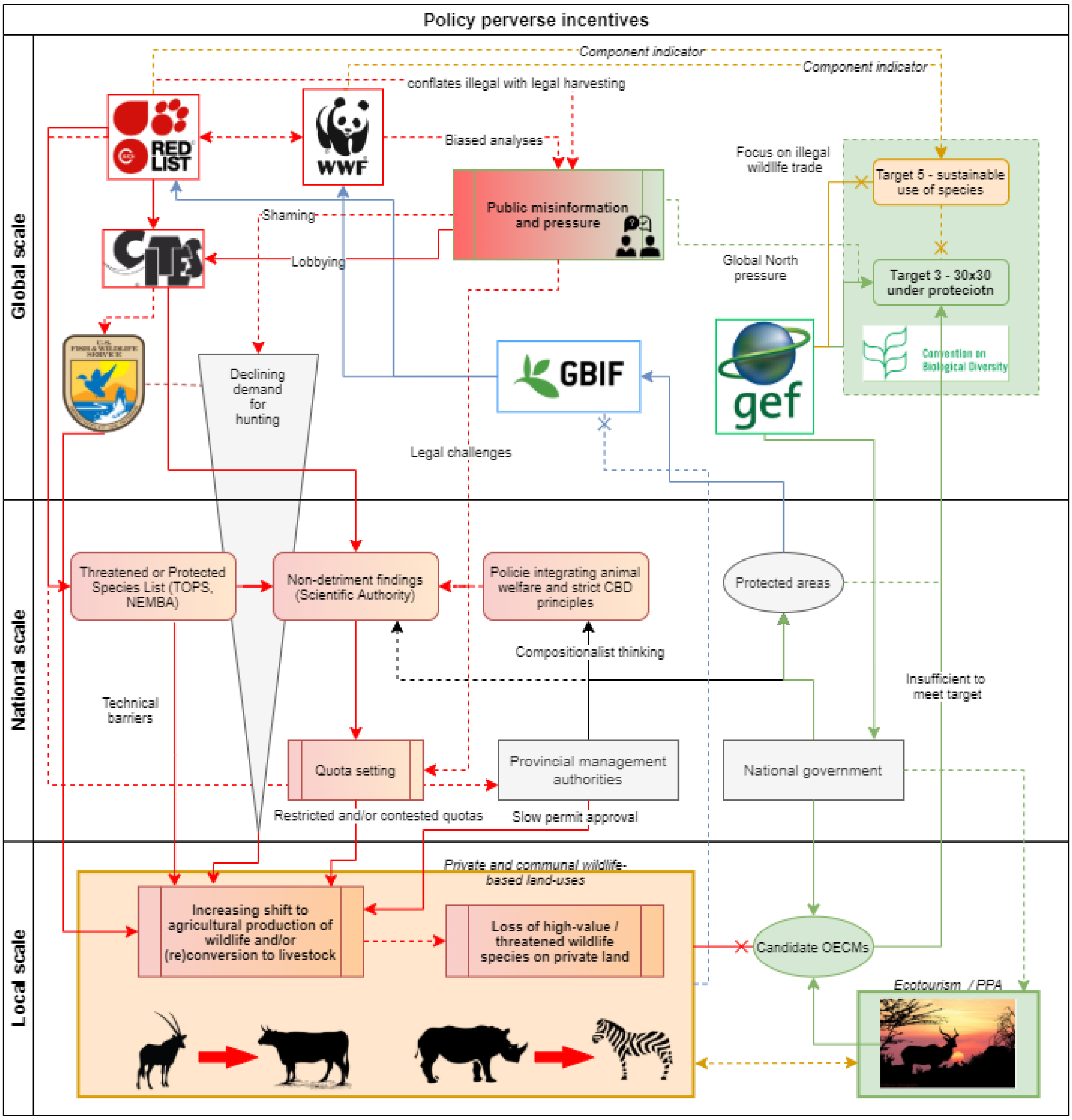

Figure S4.12. Conceptual diagram of the links between the Convention on Biological Diversity Global Biodiversity Framework targets for 2030, the primary assessment mechanisms, the IUCN Red List and the Living Planet Index (depicted by the WWF symbol), and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Solid arrows depict direct influences while dotted arrows are indirect, and arrows with crosses represent potential but inactive links. Red colours indicate links and feedback associated with global assessment data and wildlife trade policy making; orange colours indicate systems to do with sustainable use; green corresponds to area-based conservation; and blue indicates open-access data flows.

significantly to a current value of below the average plains game species. Correspondingly, the average trophy price for Cape Mountain Zebra increased substantially, peaking in 2016-2017, before declining markedly in 2019 and remaining low. These declines in game auctions and trophy value occurred before the COVID19 lockdown in 2020 (although the number of trophies exported dipped in this year for obvious reasons). The number of trophy exports was low between 2002 and 2017, averaging just 2 per annum, until 2019, when reported exports increased to 12[23]. While difficult to infer causal factors from these data, a plausible hypothesis would be that landowners were speculating on the easing of conditions to conduct trophy hunting following the downlisting on the Red List and CITES. However, the continued block at USFWS (with its requirement for enhancement findings despite the petition to de-list the subspecies on the ESA) might have prompted landowners to disinvest from Cape mountain zebra and focus on more reliable species (plains game) that don't require high administrative hurdles. If this is the case, it could indicate a worrying trend in that landowners are generally disinvesting from wildlife species of conservation concern (endemics and threatened species) to de-risk their hunting operations, which would mean the area of occupancy for these taxa will gradually erode after the previous decades of expansion and subpopulation increases. More generally, this demonstrates the limitations of CITES for stimulating legal trade for conservation as opposed to actively suppressing it and the power of national regulations to negate the impacts of downlisting. We must move towards more agile information systems that enable site-specific wildness assessments to be integrated into export/import permit applications to avoid the need for lengthy bureaucratic wheels turning.

This discussion supports findings from elsewhere in the world that hunting bans, when extrapolated from non-existent threats or threats unrelated to sustainable use, have counterproductive impacts on the conservation of the species. For example, polar bears species name? were listed as threatened under the US ESA in 2008 due to climate change. The resulting trade ban reduced US hunter participation and the proportion of quotas taken by sport hunters from specific populations, negatively impacting Arctic indigenous communities' livelihoods and reducing their tolerance for living with wildlife (Weber et al. 2015). If more blanket trade bans and regulations are enacted because of general threats like climate change, without considering the regional and subpopulation contexts, many more wildlife species risk losing habitat due to loss of revenue to the local landowners and communities. The problem with such clampdowns is that they quash nuances related to the legal trade in wildlife, where all taxa are

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[23] CITES trade database: https://trade.cites.org/

treated equally, and both legal and illegal trade systems are conflated. In South Africa, while the illegal trade of some taxa, such as succulent plants and reptiles, is a serious threat and should be addressed through trade bans and punitive measures, trade relating to managed herbivore populations through legal hunting is a major economic driver of the expansion of wildlife habitats and wildlife populations.

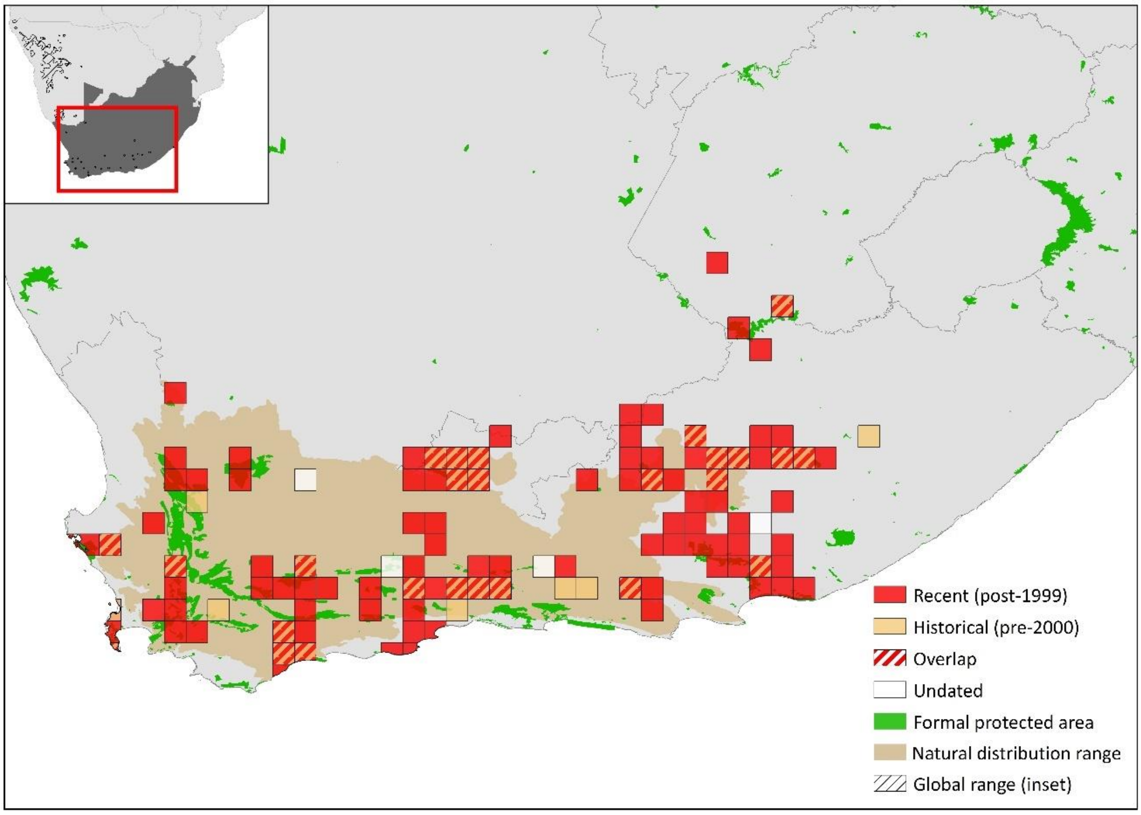

Figure S4.13. Distribution records for Cape mountain zebra (*Equus zebra zebra*) within South Africa showing the global range of the species (including Hartmann's mountain zebra *Equus zebra hartmannae*). The Cape subspecies is endemic to the Western and Eastern Cape provinces of South Africa.

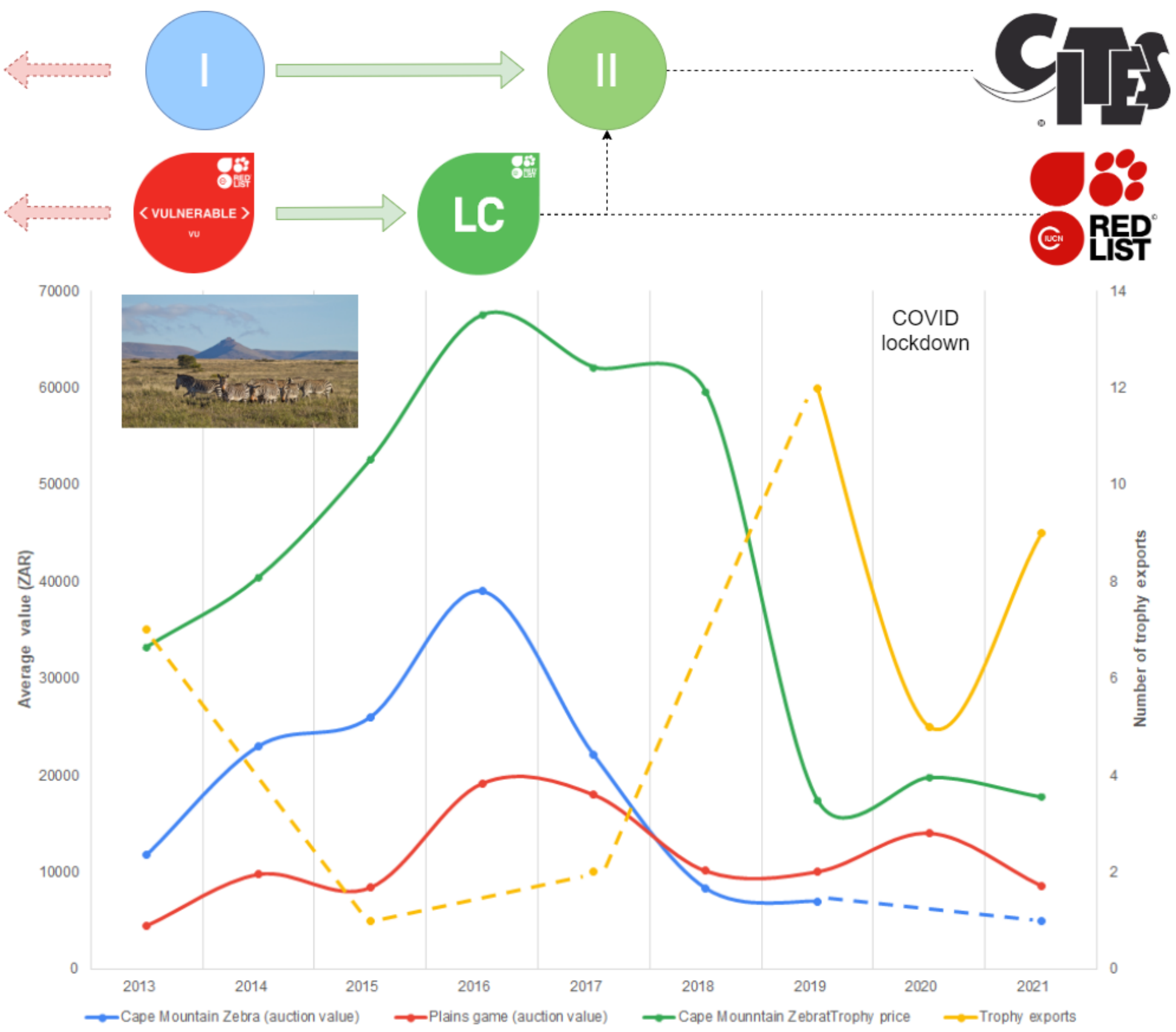

Figure S4.14. Timeline of changes to the Cape mountain zebra (*Equus zebra zebra*) listing under the IUCN Red List and CITES assessments. The inclusion of private sector data and assessments of wildness enabled the downlisting of the subspecies from Vulnerable to Least Concern in the national Red List assessment of 2016. This enabled the national government to lobby for a downlisting on CITES from Appendix I to Appendix II to enable trade. While the number of trophy exports has increased since 2017, the average auction value has dropped drastically to below that of the average plains game species. The average trophy hunting value has also decreased substantially since 2018.

Chapter 5 Rewilding rangelands restores productivity and socioeconomic opportunity

5.1 Abstract

Transitioning to sustainable societies requires integrating production and conservation in multifunctional landscapes, but little evidence exists demonstrating ecologically and socioeconomically viable land-uses. Rangelands across Africa have undergone the replacement of indigenous wild herbivores with livestock, which, in combination with suppression of fire and elevated carbon dioxide levels, has led to the loss of grass productivity through overgrazing and woody plant encroachment and long-term loss of economic viability based on animal production systems. Rewilding these rangelands through indigenous herbivores and natural disturbances, such as fire management, is a potential solution to restore grass productivity. The conversion from livestock farming to wildlife-based land-uses (WBLUs) in South Africa, which now constitutes at least 14% of the country's land area, presents a natural experiment to test this hypothesis. I used residual trend analysis to assess grassy and woody productivity trends. This was done using the Enhanced Vegetation Index (EVI) before and after converting to WBLUs. I compared these trends between wildlife-based land-uses (N = 254), commercial livestock ranches (N = 48), formally protected areas (N = 59), and communal rangelands (N = 10) using a dataset of 30 m x 30 m pixels aggregated into 5 km^2 grids. I then used general linear mixed models (GLMMs) to understand the relationship between vegetation productivity, natural resource management and enterprise profitability. Residual productivity trends were lower than expected before conversion and higher after, while there was no difference for woody productivity (which shows similar rates of increase across periods). Rotationally grazed commercial livestock farms and WBLUs experienced the highest residual gains in grass productivity and communal rangelands the lowest, while livestock farms also showed the highest gains in woody productivity. Revenue generation (and thus job creation) was significantly correlated with grassy productivity, and grassy productivity was correlated with fire management and increasing trophic diversity. These results demonstrate that the sustainable use of wildlife can restore a key rangeland ecosystem service – grassland productivity – and contribute to rural development through revenues associated with the

wildlife economy whilst reducing bush encroachment. It is the first study on a national scale to demonstrate the return on investment for rewilding rangelands through the restoration of trophic diversity and fire regimes. We recommend that policymakers in Africa establish enabling conditions for WBLUs to thrive as ecosystem-based adaptations to climate change.

5.2 Introduction

'Working lands conservation' seeks to identify production-orientated land-uses that reverse biodiversity loss, mitigate climate change and sustain ecosystem services (Kremen & Merenlender 2018). Rangelands are the world's most extensive and productive working land, comprising native grasslands, shrublands and savannahs where grazing herds and agricultural enterprises can thrive (Godde et al. 2020). Rangelands potentially provide multiple ecosystem services, such as food and fibre production, recreational opportunities, ecotourism, carbon sequestration and water security (Havstad et al. 2007; Follett & Reed 2010; Bengtsson et al. 2019; Buisson et al. 2019; Maestre et al. 2022). While rangeland condition has different meanings in different habitats and to different stakeholder groups, we define it as the availability of palatable grass and browse species in an ecosystem, and thus, the potential to support herbivore populations and economic activities. In particular, good condition rangelands are those with a high above ground biomass of perennial tall grass swards that contain palatable forage and which also provide water absorption and carbon sequestration services (Trollope 1990; Peel et al. 1999; Lohmann et al. 2012; Treydte et al. 2013; Vundla et al. 2020). For example, rangelands in good condition produce significantly more phytomass and use soil water more efficiently than rangelands in poor condition (Snyman 1999), thereby reducing the need for forage inputs and irrigation.

Inappropriate land management, such as overgrazing and suppression of fire, together with increasing aridity from climate change, has led to the degradation of large swathes of rangeland in Africa (Akinyemi et al. 2021; Montfort et al. 2021; Reith et al. 2021); and will increasingly undermine their economic value (Scheiter et al. 2019). Of particular concern in Africa is the intensifying woody plant encroachment (O'Connor et al. 2014; Russell & Ward 2016; Skowno et al. 2017; Venter et al. 2018; Godde et al. 2020), which is compounded by inappropriate global 'reforestation' policies such as the Bonn Challenge (Bond et al. 2019; Veldman et al.

2019), and will result in widespread loss of rangeland productivity and thus decrease socioeconomic potential. For example, a global meta-analysis found that invasive plant encroachment results in water loss from the ecosystem through higher evapotranspiration rates of the invading species (Cavaleri & Sack 2010). Bush encroachment also alters the available soil carbon balance (Jackson et al. 2002), which can lead to lowered grass productivity. For example, in South Africa, grass biomass was halved in areas invaded by wattle compared to cleared and uninvaded areas (Vundla et al. 2020).

Rewilding rangelands through reintroducing indigenous ungulate species and where appropriate, predators (hereafter 'wildlife'), could restore rangeland condition and diversify economic options for communities. We define rewilding as the restoration of trophic complexity, natural disturbances, and dispersal to improve ecosystem functioning and sustain biodiversity (Perino et al. 2019). In Africa, replacing wildlife with livestock across the continent has historically contributed to the loss of rangeland productivity through woody plant encroachment, the collapse of lateral nutrient dispersal while doubling methane emissions, and the suppression of key ecosystem processes such as fire (Hempson et al. 2017). Reducing the ratio of browsers to grazers results in the loss of shrub control services and causes overgrazing of the herbaceous cover (O'Connor et al. 2014; Guyton et al. 2020; Irob et al. 2022). Wildlife provides multiple cross-scale ecosystem functions through diverse feeding guilds, nutrient dispersal and ecosystem engineering (Du Toit & Cumming 1999; Waldram et al. 2008; Asner et al. 2016; le Roux et al. 2018; Louw et al. 2019). Reintroducing wildlife on formerly degraded rangelands can increase perennial grass biomass (Kraaij & Milton 2006; Treydte et al. 2013; Masubelele et al. 2014; Odadi et al. 2017; Keesing et al. 2018; McDonald et al. 2018; Achieng et al. 2020), improve forage quality through increased nutrient dispersal (Keesing et al. 2018; McDonald et al. 2020); reverse the impacts of invasive aliens (Parker et al. 2006; Veblen et al. 2015; Guyton et al. 2020; Wells et al. 2021); and improve soil carbon sequestration (Sitters et al. 2020). The foraging patterns of wildlife are also better indicators of rangeland condition than cattle because they are wide-ranging (less reliant on water) and tend to sample all available patches within the landscape (Ranglack & du Toit 2015).

Across most of Africa, however, wildlife remains locked within isolated protected areas. This presents two problems: firstly, because wildlife is state-owned, local communities and landowners cannot manage wildlife as assets and derive economic benefits from them; thus, there is no incentive to reintroduce or even tolerate wildlife on rangelands. Secondly, conservation is overly reliant on ecotourism and philanthropic funding (IUCN ESARO 2020), which makes protected areas and ecotourism models vulnerable to international market shocks (Lindsey et al. 2020). These problems exacerbate each other because protected area expansion creates 'conservation refugees' who can no longer use the natural resources inside, leading to intensified poaching or habitat degradation on the periphery or both. Policy-makers require bottom-up solutions that enable communities to benefit from wildlife assets at local scales, situating ecotourism as just one strand of economically viable social-ecological systems within a sustainable use paradigm.

Legislation in South Africa, Namibia and Zimbabwe enables landowners to own and utilise wildlife directly (Carruthers 2008; Child 2012a; Blackmore 2020), providing national-scale natural experiments to assess the impacts of reintroducing wildlife to rangelands. This has led to unprecedented increases in wildlife abundance (Taylor et al. 2021) to the extent that southern Africa is the only region on the continent where wildlife is increasing (Craigie et al. 2010). Wildlife-based land-uses (WBLUs) provide diverse economic portfolios (see *Supporting Information 5.1* for more detail), including ecotourism, trophy hunting, "biltong" hunting, game meat production and live animal sales, which can generate considerable revenues (Barnes & Jager 1996; Kreuter & Workman 1997; Barnes et al. 2002; Lindsey et al. 2007b; Cloete et al. 2007; Child et al. 2012b; Di Minin et al. 2013; Chiyangwa 2018; Saayman et al. 2018; Taylor et al. 2020); often outperforming livestock farming in profitability (Musengezi 2010; Lindsey et al. 2013b; Clements et al. 2016a; Taylor et al. 2020); and providing more job opportunities than equivalent agricultural land-uses at average higher monthly salaries (Sims-Castley et al. 2005; Langholz & Kerley 2006; Achieng et al. 2020; Taylor et al. 2020). Similarly, adding wildlife to livestock production systems can diversify income streams, engender resilience, and increase net revenue (Lindsey et al. 2013b; Allan et al. 2017; Taylor et al. 2020; Clements et al. 2022). Additionally, WBLUs are seen as potential ecosystem-based adaptations (EbA) to climate change because wildlife species are more adapted to environmental variance and water scarcity (Cromsigt et al. 2018; Malhi et al. 2022), whereas traditional livestock farming may be vulnerable to increasing aridity. However, the effectiveness of WBLUs as EbAs will correlate inversely with increasing intensity of management (see Chapter 3; Holling & Meffe 1996).

While WBLUs are generally considered a form of 'sustainable use' of natural resources, the impacts of this broad-scale shift from livestock to wildlife have not been empirically evaluated against potential long-term trade-offs between economic viability and ecosystem productivity. In South Africa, managers of WBLUs have reported employing actions that might restore rangelands (Pienaar et al. 2017). However, such management systems are often not based on ecological principles or monitored adequately. The imperative to maximise revenue generation has eroded the natural variation of ecosystem processes and thus potentially degraded long-term condition trajectories (Clements & Cumming 2017b). The impacts of these potential trade-offs between revenue and rangeland condition have not been explored. Additionally, there is concern about the cost-effectiveness of restoration. For example, restoration on farmlands in Namaqualand, while successful in improving condition, was less cost-effective than that of importing supplementary forage (Bourne et al. 2017). As such, evidence documenting whether restoration can produce a return on investment or is an opportunity cost in novel ecosystems is needed to inform policies regulating WBLUs.

Here we assess the long-term impacts of WBLUs on ecosystem functioning and enterprise-level viability. We use two independent field survey datasets constituting geo-referenced WBLUs across South Africa to assess how rangeland rewilding and investment in restoration practices have impacted profitability and rangeland condition trends. We use the Enhanced Vegetation Index (EVI) to assess long-term changes in both herbaceous and woody biomass. In South Africa, EVI trends can be considered a proxy of rangeland condition, revealing both degradation (such as woody plant encroachment) and impacts of restoration (such as alien plant clearing) on grass biomass productivity (Venter et al. 2020; Vundla et al. 2020). We distinguished between wildlife-only and mixed livestock-wildlife ranches and contextualised WBLU rangeland trends through comparisons to formally protected areas, commercial livestock rangelands under rotational grazing, and communal livestock rangelands under continuous grazing. We aim to evaluate the impacts of rewilding rangelands on ecosystem functioning compared to other broad land-use classes so that the benefits of the wildlife economy can be unlocked through policy.

5.3 Methods

5.3.1 Study sites

Study sites and WBLU data collection. *Wildlife-based land-uses (WBLUs)*. I used a primary survey dataset of 281 WBLU properties with detailed information on management activities, socioeconomic output and wildlife population information (Taylor et al. 2015). Participants were drawn from a database of 1,540 private landowners / managers from across South Africa, of which 427 were randomly selected to be telephonically requested to participate between September 2014 and January 2017 (Taylor et al. 2020). Only properties that generated revenue directly from wildlife were eligible (where economic activities are defined as per *Supporting Information 5.1*). For further details on questionnaire design (see Taylor et al. 2015, 2020, 2021). In some cases, landowners declined to answer certain questions, in which case the results present subsets of the data.

I combined this dataset with an independently conducted survey of 60 unique WBLUs from the Western and Eastern Cape provinces of South Africa collected between April 2014 and February 2015 (Clements et al. 2016a). Similarly, this survey collected data on management strategies, revenues and wildlife composition and abundance (Clements & Cumming 2017a; Clements et al. 2018). When combining datasets, 347 individual ranches were represented. In both datasets, cadastral information was provided by the landowners, which was then digitised into shapefiles by extracting the relevant farm portions from the Chief Surveyor General cadastral database (Chief Surveyor-General 2017). After excluding farms that did not provide, or provided cadastral information incorrectly, the total WBLU sample size was 254 (Figure 5.1). In South Africa, all properties that utilise wildlife commercially (including formally protected areas) are fenced and thus are not influenced by external management systems unless part of a conservancy (see below). See Hayward et al. (2009), Carruthers (2008), Clements and Cumming (2017b), and Child et al. (2019) (Chapter 3) for critical discussions related to the fencing of WBLUs.

Commercially managed livestock farms (livestock farms). I used a survey dataset of 48 livestock farms distributed across South Africa collected between 2016 and 2017 as the primary reference sites for commercial rangelands under some form of rotational grazing management system (Venter et al. 2019). The primary survey datasets from Taylor et al. (2015) and Venter et al. (2019) are thus directly comparable in that the wildlife ranches and livestock farms were surveyed over the same general time period (2014-2017). The commercial livestock farms were under consistent management for 15 ± 0.8 years (mean ± std. error), and 60% contained a mixture of livestock and small stock (Venter et al. 2019).

Communally managed livestock farms (communal rangelands). I used the former homelands layer from the Department of Rural Development and Land Reform, which reflects broad areas (N = 10) under predominantly communal grazing (Palmer & Bennett 2013). I omitted all land parcels that are privately or state owned within the former homelands layers to focus on areas most likely to be under communal rangeland management. However, because this is a relatively broad layer, it can be considered a null land-use in that it likely contains multiple land-use types within broad swathes of communal grazing lands. Compared to the commercial livestock farms, most communal rangeland systems are subject to continuous grazing regimes and thus subject to potential overgrazing (for example, Abdalla et al. 2022).

Formally protected areas. Protected area reference sites were extracted from the latest South African Protected Area Database (SANBI & SANParks 2018). I filtered for state-run protected areas that manage wildlife populations for conservation objectives where ecotourism is the primary commercial activity. As such, I omitted all private protected areas (which may overlap with certain WBLUs in our dataset that are declared nature reserves but deploy other commercial activities). I also omitted categories such as forest reserves (which do not have significant herbivore populations), mountain catchment areas, World Heritage Sites and Protected Environments (that have complex management systems and encompass multiple land-uses), which correspond to IUCN categories V and VI. As such, I only included non-overlapping protected areas (N = 59) to avoid pseudo-replication.

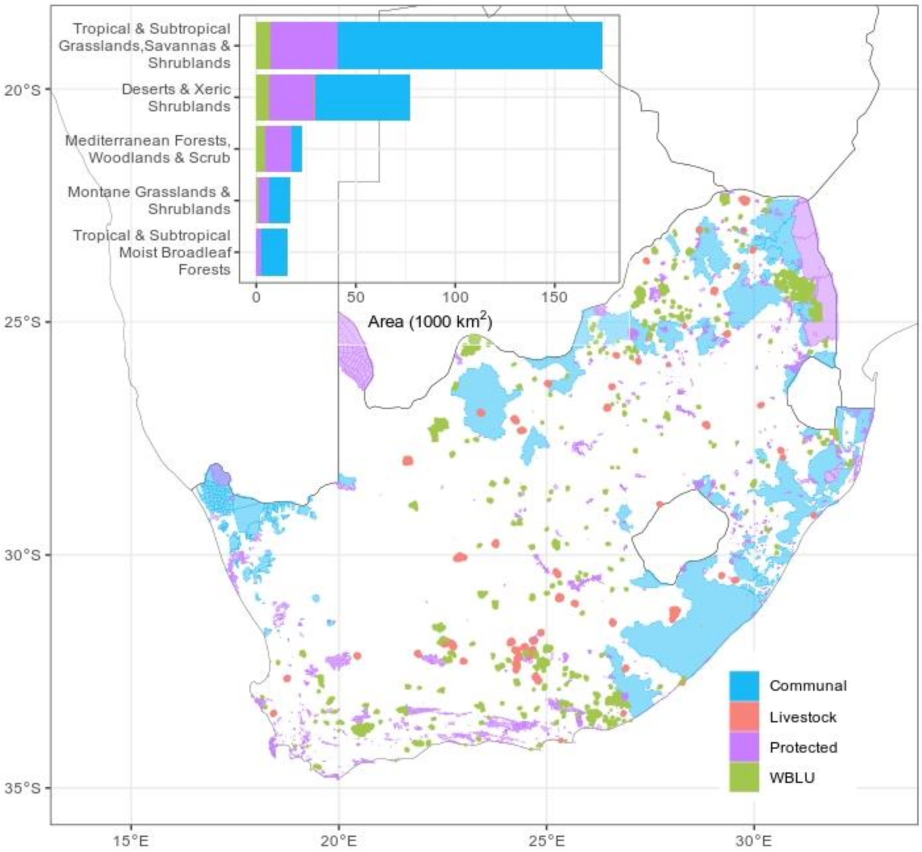

Figure 5.1. The study areas and proportion of different land-uses in different biomes. Most of the sample corresponds to the Grasslands and Savannah ecoregion, but there is also a good representation in the Shrublands ecoregion. Sites across categories are evenly distributed across the various ecoregions. WBLU = wildlife-based land use.

5.3.2 Data sources and analysis

Rangeland condition data. I used the Enhanced Vegetation Index (EVI) as a proxy for rangeland condition to estimate long-term changes. The EVI data have been calibrated against very-high-resolution satellite imagery (c. 3 m resolution), and there is a correlation between patterns of degradation (woody plant encroachment, desertification) and restoration (e.g. increased rangeland productivity, alien species clearing) (Venter et al. 2020). These data use Landsat (30 m resolution) satellite imagery between 1999 (the median conversion date to wildlife ranching in our sample; see Supporting Information 1) and 2014 (the earliest survey start date in our sample) to derive EVI trends within the Google Earth Engine cloud computing environment (Gorelick et al. 2017; Venter et al. 2020). To control for the effect of woody plant encroachment and ecoregion-specific patterns of woody vegetation, EVI trends over grassy and woody pixels were extracted separately. I classified woody and grassy pixels for 2014 using newer satellite data using methods from Venter et al. (2018). Sentinel-2 imagery was used to map woody cover over South Africa for 2018/2019 (including shrub cover below 5 m), and those pixels classified as woody in 2018/19 were used as woody pixels. If a pixel started as 'grassy' in 1984 and became bush encroached, it was classified as a woody pixel in our analysis. This ensures we are capturing the end state of the system transition. Greening trends over woody pixels may be a product of encroachment (previously grassy and now woody) or increase in vegetation cover (less woody to woodier). By considering EVI trends in grassy and woody areas separately, I can be more confident that greening trends are not just a signal of encroachment. For grassy productivity, while remotely-sensed proxies do not always guarantee that rangeland condition has improved, as biomass might increase but not grassland quality (Seymour et al. 2010), EVI proxies have been shown to correlate with field-based condition assessments (Venter et al. 2020). I used a pilot dataset of perennial grass biomass trends in WBLUs around Kruger National Park to validate the relationship between EVI and site-level perennial grass biomass measurements, which showed a significant positive correlation (*Supporting Information 5.2*).

Controlling for environmental variance. I used GLOBLAND30 (Arsanjani et al. 2016) to mask out urban, cultivated and waterbody areas to focus the analysis on rangelands. To isolate the effects of management on rangeland condition between the different land-uses from rainfall effects, I extracted the residual trends (RESTREND), which has been widely used to detect management-induced degradation trends in drylands globally (Evans & Geerken 2004; Wessels et al. 2007b; Burrell et al. 2017). Rainfall data from the Climate Hazards group Infrared Precipitation with Stations (CHIRPS) dataset (Funk et al. 2015; Arsanjani et al. 2016) (5 km resolution) going back to 1984 was used for the RESTREND analysis. Rainfall data values were calculated for each Landsat pixel and then averaged over the woody and grassy areas within each site or property. A linear regression model between annual mean EVI and precipitation was established for every pixel over the study area. The differences between the observed EVI values and the EVI values predicted by the linear model were then calculated, hereafter referred to as EVI residuals. After calculating the EVI residual time series in the woody and herbaceous pixels, I aggregated all 30 m x 30 m pixel-based trends to a 5 km^2 grid (same resolution as CHIRPS dataset). All 5 x 5 km pixels within land management units (Figure 5.1) were sampled for further analysis. In this way, statistics were area-weighted because large land management units will have multiple 5 x 5 km pixels within them. The pixel-specific trend in the EVI residuals is considered the management-driven trend. The magnitude and significance of the trend in EVI residuals were calculated using the Sen's slope (Sen 1968) estimator and Mann-Kendall test (Mann 1945), respectively. The Sen's slope estimator is a non-parametric linear regression that is robust against outliers and skewed data.

To control for differences in vegetation productivity between biomes, I relativised EVI and RESTREND values on a scale from 0 to 1 for both grassy and woody series separately using the rescale function in the "dplyr" package (Wickham et al. 2020). To further control for the confounding effects of different climactic regions and soil types, I subsampled the WBLUs and reference sites according to ecoregion, which act as robust proxies for ecosystem-level patterns of biodiversity reflected by environmental and climatic gradients (Smith et al. 2018b). I used the 2017 ecoregion map available from Resolve[24] (Dinerstein et al. 2017).

Trends in productivity since conversion to WBLUs. I tested the hypothesis that rangeland productivity improves after converting to WBLU. The majority (85%) of WBLUs were converted from livestock or mixed livestock and cultivation land-uses (see *Supporting Information 5.1*). Thus, long-term trends can be seen as testing the impact of rewilding livestock or small stock rangelands with indigenous herbivores. I assessed EVI values and trends before and after converting to WBLUs by calculating the time difference in years before and after the 'rewilding' date (see *Supporting Information 5.1* for more information). I used a paired t-test to assess changes in property-level EVI values before and after rewilding, and I used linear models to assess differences in EVI trends between the two time periods. I assessed long-term trends for the period 1999 to 2014, reflecting the median conversion date of wildlife ranches from livestock farms and the beginning of the data collection period, respectively. The conversion date aligns with other independent assessments of peak conversion to WBLUS. For example, WBLU conversion in the Eastern Cape peaked around 2000 (Smith & Wilson 2002), consistent with Child et al. (2013). A minority of properties (11%) were converted outside the 1984 to 2014 EVI analysis window and thus were effectively 'always' WBLUs, and thus were excluded from the calculation of the median conversion date. Trends were measured in the 'before' period from 1984 to the year the property was converted to wildlife, and the after

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[24] Resolve ecoregion application: https://ecoregions2017.appspot.com/

period was measured from the conversion year to 2014. We assessed all trends and then only those that identified as significant through Sen’s slope estimator. To control for the effects of a general greening pattern in South Africa (Venter et al. 2020), I compared the temporal trends in EVI in WLBUs to commercial and communal livestock areas and protected areas in each biome. I tested differences in the EVI slopes between land-uses through analysis of covariance (ANCOVA). All analyses were conducted using R (R Core Team 2023).

Management impacts on profitability and productivity. *Defining the response variables*. Grassy and woody EVI patterns were considered the primary response variables, where long-term residual trends and current biomass were considered separate responses potentially impacted by different management interventions. I used the sum of residual trends in the relationship between EVI and rainfall between 1999 and 2014 to detect the accumulation of residual biomass trends due to management effects. I also analyzed socioeconomic potential as a response variable, where vegetation trends and management interventions were included as predictors. The socioeconomic potential is defined as the profitability and employment density of WBLUs using revenue / ha as a proxy (to control for potential economies of scale). Annual revenue is strongly linearly correlated with indicators of financial viability (annual profit and return on investment) as well as employment figures and well-being (jobs / ha and average monthly salary) (*Supporting Information 5.3*). Thus, I considered the more widely reported revenue / ha (N = 97 WBLUs) to be a robust indicator of socioeconomic impact. I used linear regression to assess whether there are significant associations between WBLU revenue and EVI residual trends.

Defining the fixed and random effect variables. Management responses were used as fixed effects for both the EVI trends and the socioeconomic indicators. These include the use of fire, alien plant / bush encroachment control, stocking density, trophic diversity (including predator guilds), breeding camp extent, mixed farming intensity and artificial water point density (see *Supporting Information 5.4* for more details). The benchmark stocking density for each region was extracted from the national long-term grazing capacity guidelines (DAFF 2018). To make the predictors congruent with relativised EVI, they were rescaled as values from 0 to 1. The predictor variables were used as terms in Generalised Linear Mixed Models (GLMMS), using biome as a random effect to account for differences in broad-scale climatic and environmental variables. To avoid pseudo-replication of subsampling at a property level, I summarised EVI grid values per property to yield an overall value for grassy and woody productivity per year.

All GLMMs were produced using the “lme4” package in R. I performed model selection through backwards selection based on Akaike’s Information Criterion (AIC). Only best-fit model diagnostics are reported.

5.4 Results

5.4.1 Impacts of converting to wildlife-based land-uses

Grass and wood productivity was higher after transitioning to wildlife-based land-uses. Within our sample of WBLUs, most properties reintroduced wildlife into former livestock farms or livestock and crop farms, with ‘rewilding’ mainly occurring between 1990 and 2010, with the median rewilding date being 1999 (*Supporting Information 5.1*). Figure 5.2 shows the long-term trends in EVI residual values between 1984 and 2014 in the periods before and after properties were converted to WBLUs. While grass productivity was initially high (with high variability), it dropped substantially before increasing around the time of conversion to become higher (with less variability) in the post-conversion state. On the other hand, woody productivity began lower in the pre-conversion state but has been steadily increasing over time.

Using residual (rainfall-corrected) and relativised (rescaled to account for biome difference) values, paired t-tests revealed that both grass (paired t-test: $t_{(256)} = 11.0$, $p < 0.001$, Cohen’s $d = 0.69$) and woody (paired t-test: $t_{(120)} = 18.2$, $p < 0.001$, Cohen’s $d = 1.65$) productivity was significantly higher following conversion to WBLUs with moderate and large effect sizes respectively (Figure 5.3). The mean EVI value for grass across properties was -0.048±0.006 (mean±se; N = 261) before transitioning to WBLU and 0.038±0.003 after (across time). Similarly, for woody growth, the before period was lower than the after period (-0.115±0.009 compared to 0.068±0.005; N = 124). The ‘before’ period measured the difference between 1984 and the start of the WBLU enterprise, and the ‘after’ period measured differences from enterprise inception to 2014.

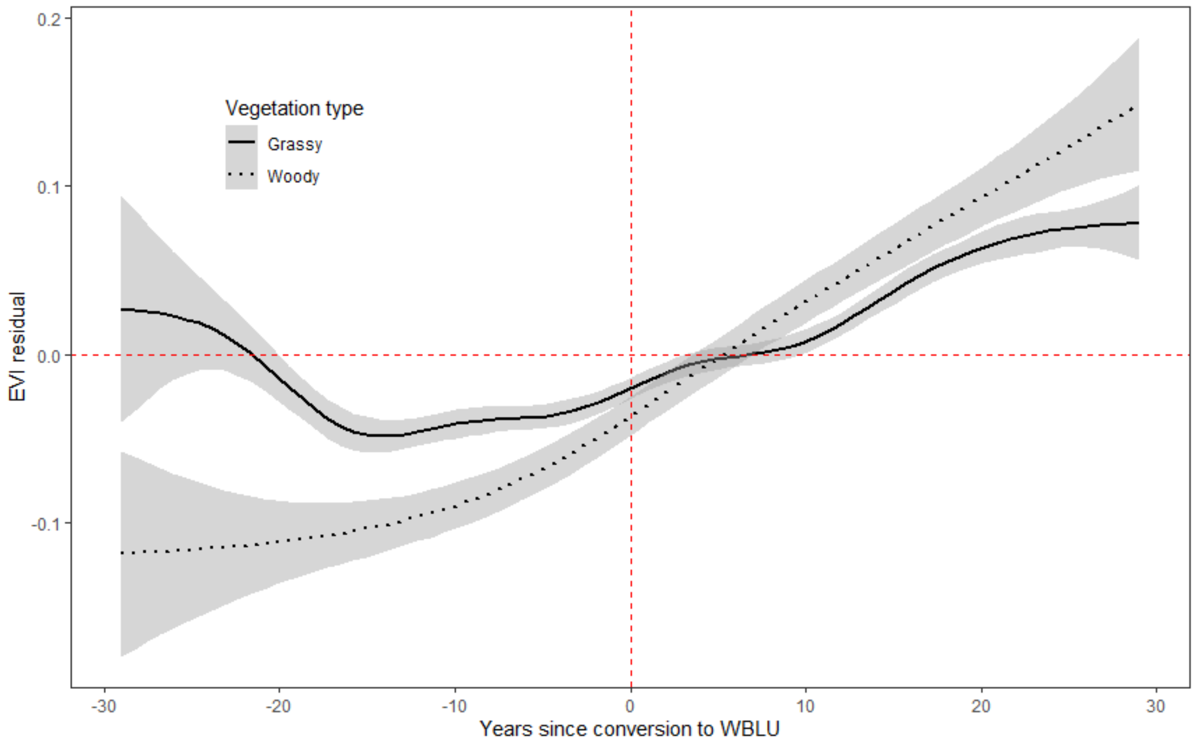

Figure 5.2. Long-term vegetation productivity trends measured in the decades before and after converting to wildlife-based land-uses (and spanning the period 1984-2014). While grassy productivity has experienced an initial dip in productivity pre-conversion, it has increased following conversion. Woody productivity has increased steadily over time and does not appear to be affected by conversion period. Grey error bands represent 9% confidence intervals.

Although the sample size is large, meaning that assumptions of normality will not overly confound the results, I tested whether the differences in values across pairs were normally distributed using a Shapiro-Wilk normality test, which revealed a significant deviation from a normal distribution (Shapiro-Wilk test: $W = 0.97$, $p < 0.01$). As such, I repeated the analysis using the paired samples Wilcoxon test, which also found a significant difference before and after conversion with a large effect size for both grass ($p < 0.01$, effect size $r = 0.61$) and wood ($p < 0.01$, effect size $r = 0.84$) productivity. Additionally, the variance of the EVI values is much lower in the period after converting to wildlife, as measured by both the standard deviation and the confidence interval for both grassy (Std dev: 0.10 before and 0.04 after; 95% CI: 0.012 before and 0.06 after) and woody trends (Std dev: 0.10 before and 0.06 after; 95% CI: 0.018 before and 0.011 after).

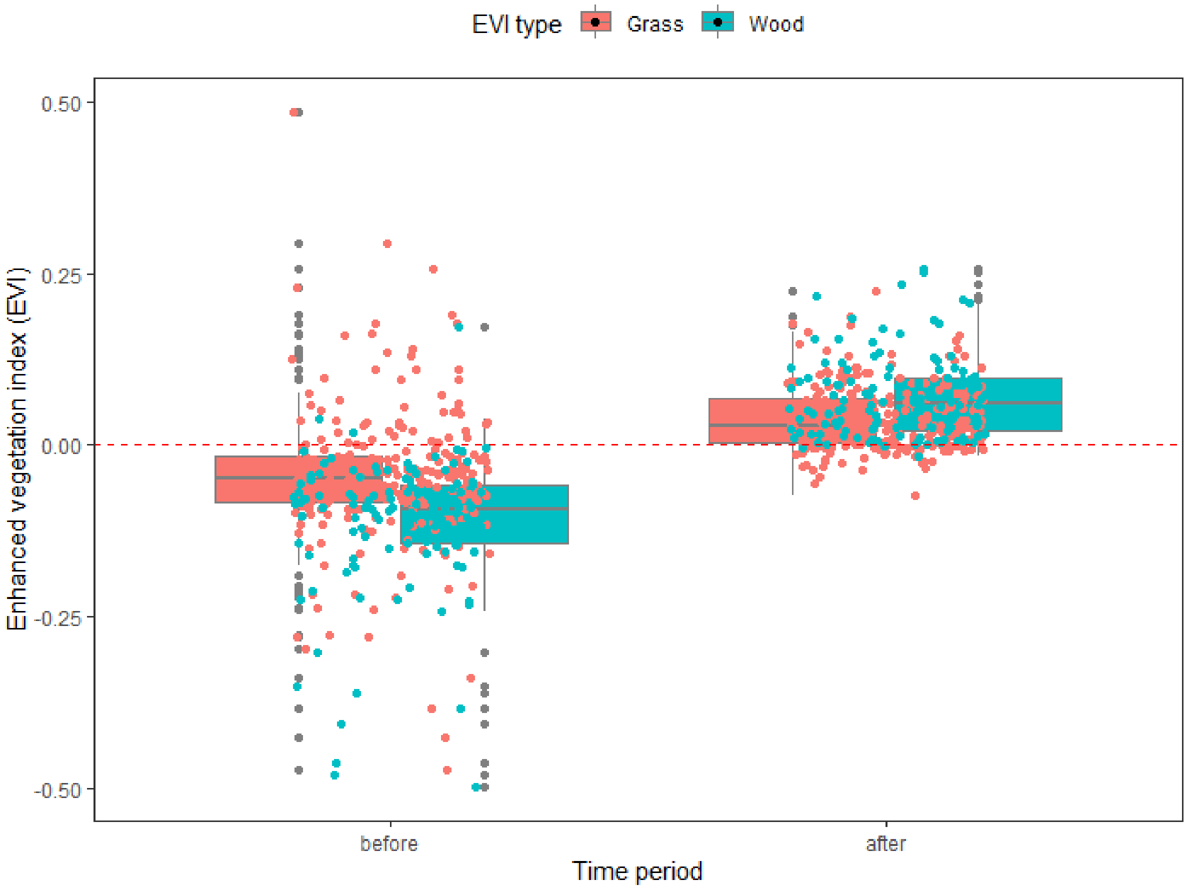

Figure 5.3. Enhanced vegetation index values for grassy and woody patches across 257 wildlife ranches before and after the first rewilding. Following conversion to wildlife-based land-uses, both grass and wood productivity increased significantly, with wood productivity increasing the most. Additionally, the variance in vegetation productivity appears to be lower after rewilding.

When breaking down these patterns by primary business model, ecotourism-only models show lower gains in grass productivity since rewilding compared to mixed-use and use-only properties, as well as relatively higher gains in woody productivity (Table 5.1, Figure 5.4). The same pattern is seen for the density of EVI trends (*Supporting Information 5.5*). However, the difference in means is not significant between the business models in either the period before or after (ANOVA: all $p > 0.16$ for both grassy and woody EVI values).

Table 5.1. Differences in vegetation productivity magnitude before and after converting to wildlife-based land-uses, broken down by primary business model (ecotourism, mixed activities, and use-only / hunting). Before and after values are medians, and mean ± standard errors are in square brackets. While means between business models are not significantly different in either period, the models based on sustainable use (mixed and use-only) show a higher magnitude.

Business model	**Before**	**After**	**Magnitude of change**	**N**
Grass productivity				
Ecotourism	-0.04 [-0.04 ± 0.01]	0.01 [0.03 ± 0.01]	0.05	73
Mixed	-0.06 [-0.07 ± 0.01]	0.03 [0.04 ± 0.004]	0.10	93
Use-only	-0.05 [-0.03 ± 0.01]	0.03 [0.04 ± 0.005]	0.08	91
Wood productivity				
Ecotourism	-1.00 [-0.12 ± 0.01]	0.06 [0.07 ± 0.01]	0.16	73
Mixed	-0.90 [-0.11 ± 0.02]	0.05 [0.06 ± 0.01]	0.14	93
Use-only	-0.08 [-0.11 ± 0.02]	0.07 [0.08± 0.01]	0.15	91

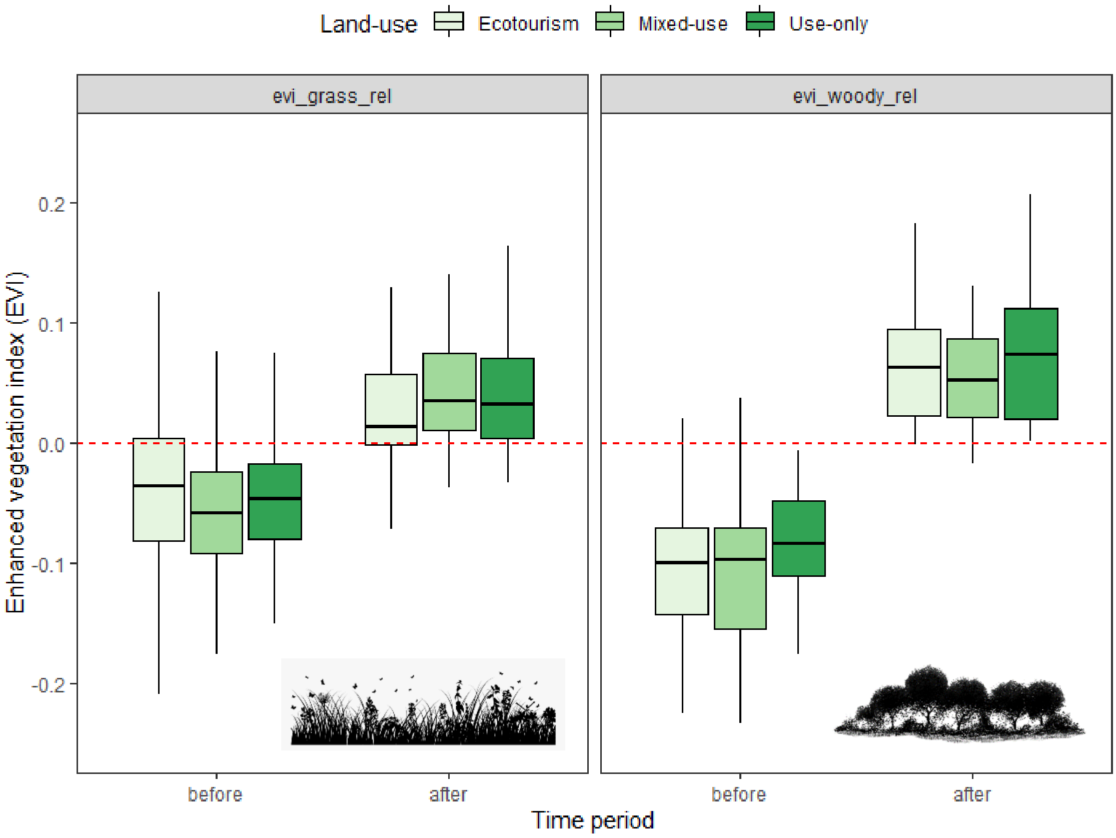

Figure 5.4. Enhanced vegetation index values for grassy and woody patches across 257 wildlife ranches before and after first rewilding broken down per primary economic model (ecotourism, mixed ecotourism and consumptive use, and consumptive use only). Ecotourism-only models show the lowest gains in grass productivity compared to models based on hunting. However, woody productivity gains are similar across land-uses. The graph is mapped without outliers to emphasise differences.

In addition to assessing how the magnitude of residual EVI values change before and after converting to wildlife-based land-uses, I also assessed whether the productivity trends, as measured by the value of the EVI slopes over time in both ‘before’ and ‘after’periods were impacted (Table 5.2). For all analyses, both parametric and non-parametric tests were conducted to control for skewed data (see above). For the full set of trends, using residual (rainfall-corrected) and relativised (rescaled to account for biome difference) values, paired t-tests revealed that grassy trends were significantly more positive since converting to WBLUs (paired t-test: $t_{(209)} = 2.90$, $p < 0.01$, Cohen’s $d = 0.2$). Woody productivity trends, while slightly more positive in the post-conversion period and on the cusp of being significant, were not significantly different between periods (paired t-test: $t_{(103)} = 2.01$, $p = 0.05$, Cohen’s $d = 0.1$). Overall, the ratio of negative to positive trends was 47:53% in the before period and 25:75% in the after period for grassy productivity and 42:58% before and 14:86% after for woody productivity (Figure 5.5). The patterns were similar when filtering for significant trends only (Figure 5.6). Grass productivity trends were highly significantly more positive in the period after converting to WBLU (paired t-test: $t_{(62)} = 7.44$, $p < 0.01$, Cohen’s $d = 1.4$). Woody productivity trends were significantly more positive, but less so than grassy trends (paired t-test: $t_{(24)} = 2.32$, $p = 0.04$, Cohen’s $d = 0.94$). However, the woody trend was not significant when using the non-parametric Wilcoxon test ($W = 175$, $p = 0.07$, effect size $r = 0.31$).

Table 5.2. Differences in vegetation productivity trends before and after converting to wildlife-based land-uses (WBLUs). Before and after values show both medians and mean ± standard errors. While mean grassy productivity trends are generally negative in the period before converting WBLUs and positive after, woody trends are positive in both periods. Asterisks indicate significant differences as determined through parametric paired t-tests.

Vegetation type	**Period**	**Mean (standard error)**	**Median**	**N (number of trends)**
All trends				
Grassy	Before	-0.002 ± 0.002**	-0.001	762
Grassy	After	0.004 ± 0.001**	0.007	768
Woody	Before	0.001 ± 0.002	0.002	124
Woody	After	0.01 ± 0.002	0.01	127
Significant trends				
Grassy	Before	-0.01 ± 0.004**	-0.02	45
Grassy	After	0.02 ± 0.011**	0.02	96
Woody	Before	0.01 ± 0.005ǂ*	0.02	12
Woody	After	0.02 ± 0.002*	0.02	43

**significant at P < 0.05; ** at P < 0.01; ǂ A significant outlier was omitted*

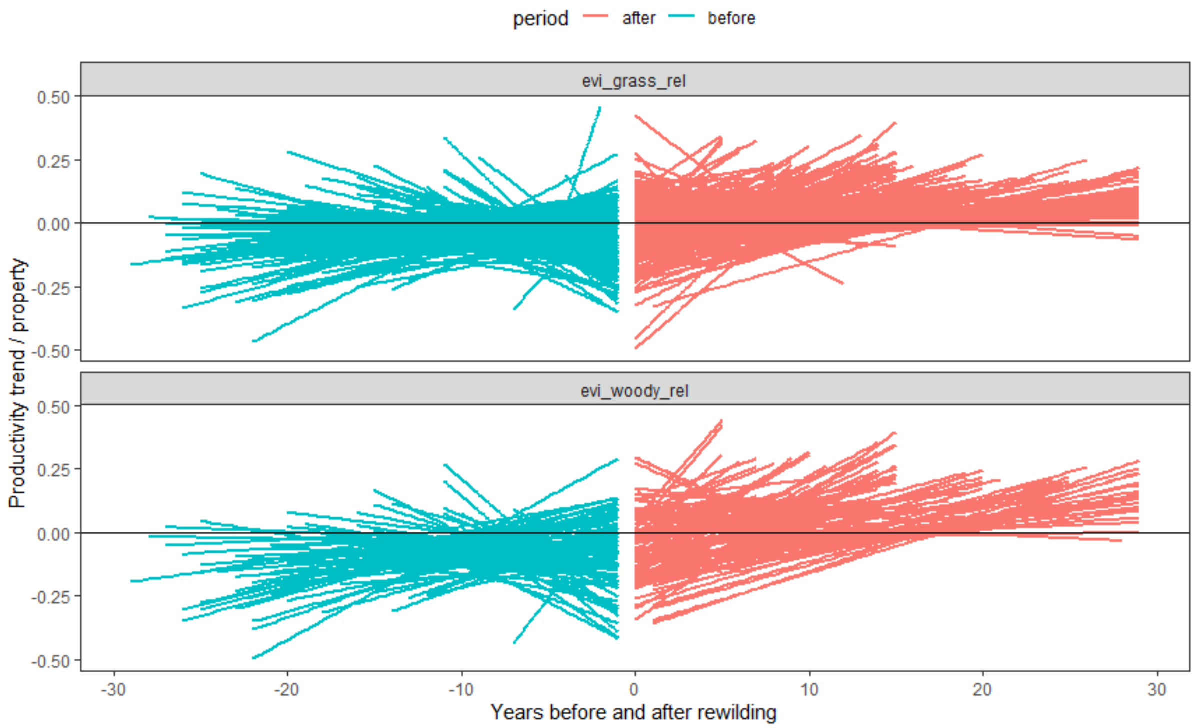

Figure 5.5. Comparison of residual trends in enhanced vegetation index values before and after properties were converted from agricultural land-uses to wildlife-based land-uses. Lines represent smoothed linear trends. The trendlines are drawn as the difference in time between the conversion date and the EVI values in 2014. The slopes of EVI residual trend values are generally lower before rewilding than after.

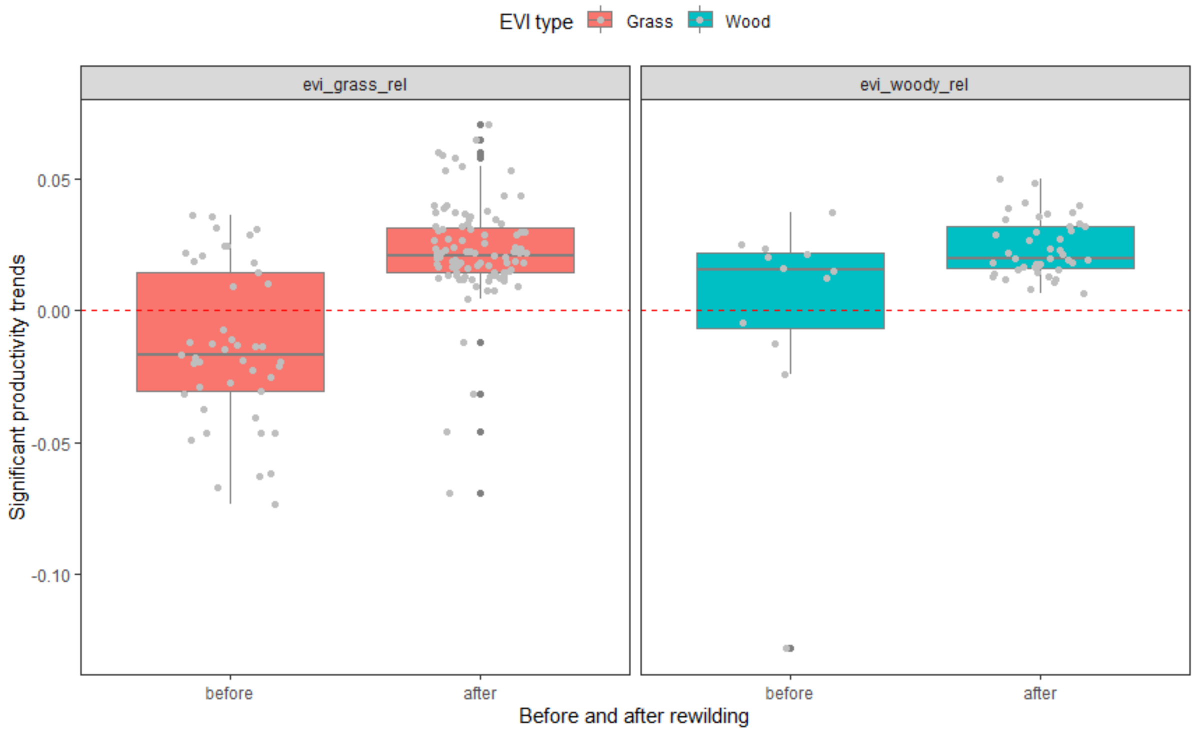

Figure 5.6. Significant grass and wood productivity trends in the periods before and after rewilding. The data were filtered for only significant slopes in both grassy and woody productivity trends split in the time before converting to wildlife and the time after. While there is no significant difference in the productivity trends of woody growth between periods, grassy productivity slopes were significantly higher since converting to wildlife-based land-uses.

5.4.2 *Vegetation productivity trends across different land-uses*

We compared vegetation productivity trends across different land-uses to contextualise the impacts of converting to wildlife-based land-uses. We compared trends between 1999 (the median conversion date across wildlife-based land-uses, *Supporting Information 5.1*) and 2014. Overall, wildlife-based land-uses appear to have higher grass productivity than expected from rainfall alone (*Supporting Information 5.6*). The mean residual trends for relativised grassy productivity were significantly different between land-uses (ANOVA test: $F_{(3)} = 123$, $p < 0.01$), with a post hoc test revealing significant differences between all land-use comparisons besides livestock and WBLUs (Table 5.3, Figure 5.7). Similarly, mean trends for woody productivity were significantly different between land-uses (ANOVA test: $F_{(3)} = 123$, $p < 0.01$), except for livestock and WBLUs (Table 5.3, Figure 5.7). However, when looking at non-relativised woody productivity trends, livestock farms had significantly higher mean trends over time than WBLUs (*Supporting Information 5.6*).

Table 5.3. Comparison of residual and relativised trends in vegetation productivity between different land-uses. Positive differences mean that land-use B has higher mean residual slope values than land-use A. For example, all land-uses have higher productivity trends for grassy and woody habitats than communal rangelands, whereas protected areas have lower grassy and woody productivity than livestock farms.

Land-use A	**Land-use B**	**Mean difference (B-A)**	**95% CI (lower)**	**95% CI (upper)**
Grassy productivity				
Communal	Livestock**	0.008	0.005	0.010
	Protected**	0.003	0.002	0.004
	Wildlife**	0.007	0.005	0.008
Livestock	Protected**	-0.004	-0.007	--0.001
	Wildlife	-0.001	-0.004	0.002
Protected	Wildlife**	0.003	0.002	0.004
Woody productivity				
Communal	Livestock**	0.011	0.006	0.02
	Protected**	0.004	0.002	0.006
	Wildlife**	0.007	0.004	0.01
Livestock	Protected**	-0.007	-0.01	-0.002
	Wildlife	-0.004	-0.009	0.001
Protected	Wildlife*	0.003	0.0001	0.006

**significant at P < 0.05; ** at P < 0.01*

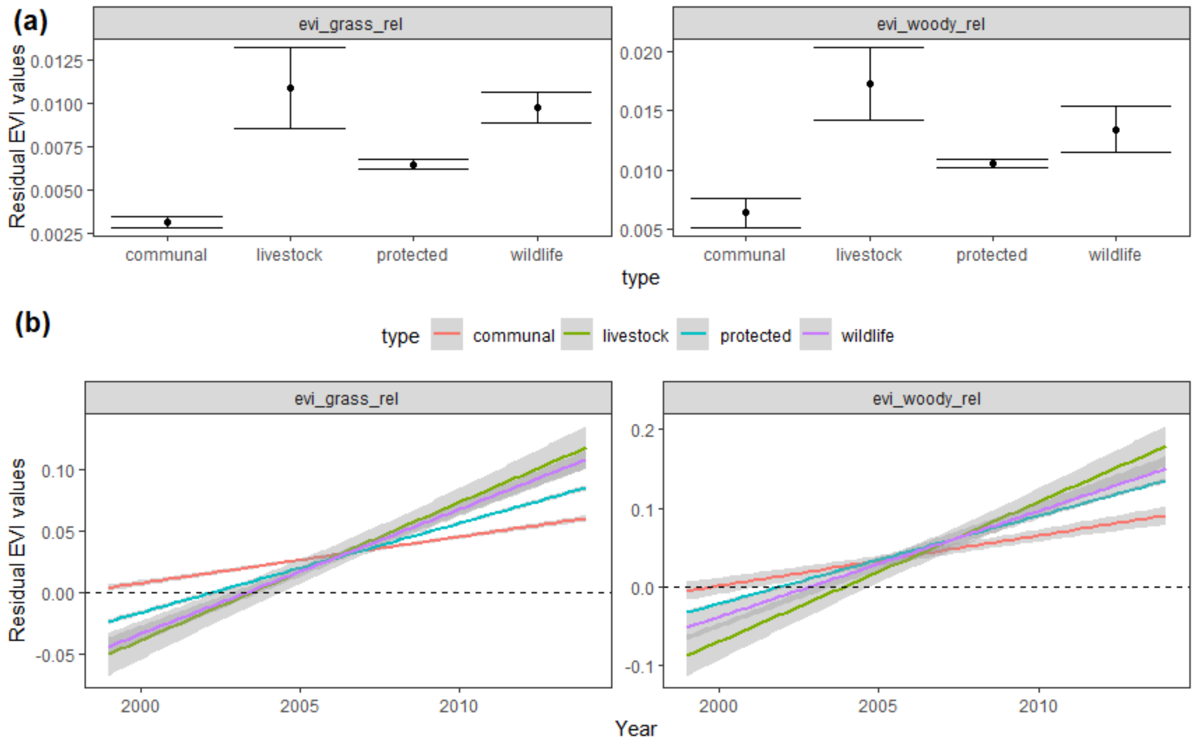

Figure 5.7. (a) The mean residual trend values of woody and grassy EVI values between land-uses. Error bars are 95% confidence intervals. Livestock and wildlife-based systems differ significantly from communal and protected area trends but not from each other. (b) Smoothed linear models in grassy and woody biomass across land-use types between 1999 (the median conversion date of wildlife ranches) and 2014 (the beginning of the data collection period). EVI relativised shows the rescaled EVI values to control for biome-specific productivity patterns. The trends show that both WBLUs and commercial livestock ranches have increased grass biomass at higher rates than communal rangelands and PAs. However, commercial livestock ranches show the highest relative increases in woody biomass.

The density distribution of residual trends between land-uses is shown in Figure 5.8. While all land-uses have more positive than negative trends for the time period, WBLUs and livestock have the highest density of positive trends for grassy productivity, and livestock has the highest density of woody trends. For both grassy and woody productivity trends, protected areas and communal rangelands are more normally distributed. However, for grassy productivity, both WBLUs and livestock farms were skewed right, with livestock farms also showing a bimodal distribution.

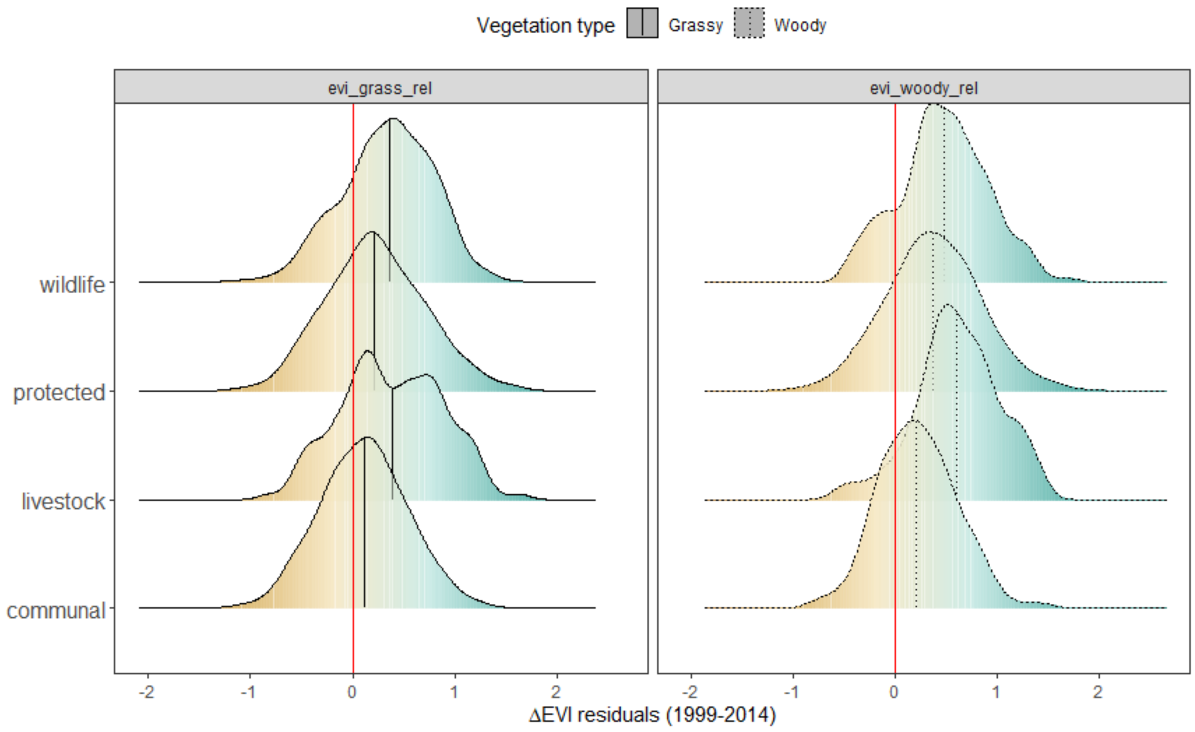

Figure 5.8. Ridge plots showing the distribution of the EVI residual trends between 1999 and 2014 (black vertical lines are median values). Slopes were derived from linear models of residual EVI trends over the time period. While most sites have improved in productivity over time, WBLUs and commercial livestock farms show the highest rates of increase grassy productivity and livestock farms show the highest rates of increase in woody productivity. Additionally, the distribution of grassy residual EVI trends for livestock farms is bimodal.

When filtering for only significant residual trends (Figure 5.9), the pattern remains similar: grassy productivity is higher in WBLUs, protected and extensively managed livestock farms compared to communal rangelands, and woody productivity is highest in livestock farms and lowest in communal rangelands. Grassy productivity means for significant trends were significantly different between communal rangelands and all other land-uses but not between livestock, protected and WBLUs (ANOVA test: $F_{(3, 1843)} = 15.7$, $p < 0.01$); and a similar pattern was found for woody productivity (ANOVA test: $F_{(3, 1063)} = 15.7$, $p < 0.01$). Proportions of trends represent the total land area sampled through the 5 x 5 km pixels.

Table 5.4. Comparison of proportion of residual trends before and after wildlife-based land-uses (WBLUs) converted to wildlife. The 'after conversion' period was defined as being between 1999 and 2014 for land-uses, including both significant and non-significant trends. Significant differences in means between the significant trends were tested through analysis of variance and indicated by asterisks.

	Non-significant trends (%)				**Significant trends (%)**			
Land-use	**Increase**		**Decrease**		**Increase**		**Decrease**	
	Before	*After*	*Before*	*After*	*Before*	*After*	*Before*	*After*
Grass productivity								
Communal**	34	55	61	39	0.2	5	5	1
Livestock	40	60	56	22	0	18	4	0
Protected	35	53	55	31	3	15	7	1
Wildlife	36	65	56	21	1	13.5	7	0.5
Woody productivity								
Communal**	57	65	40	28	2	6	2	1
Livestock	45	61	53	7	0	32	2	0
Protected	50	57	42	20	4	22	3	1
Wildlife	54	60	41	16	2	24	3	0

**significant at $P < 0.05$; ** at $P < 0.01$*

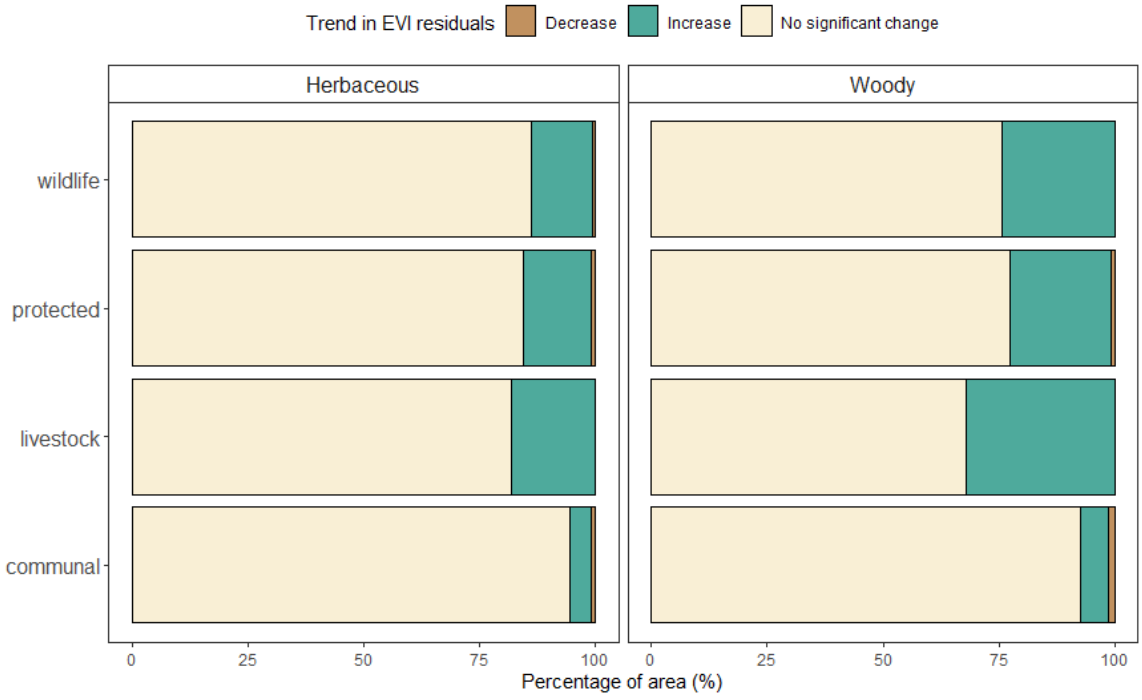

Figure 5.9. Significant EVI residual trends between 1999 and 2014 expressed as a proportion of the sampled land area. Significant positive trends are blue, while significant negative trends are brown. For significant trends in both grassy and woody productivity, communal rangelands were significantly different from other land-uses, but other land-uses were not significantly different from each other.

When looking specifically at the commercial land-uses (livestock farming and WBLUs), it is interesting to note that for some biomes, the grassy productivity is not necessarily synchronised between WBLUs and livestock farms (Figure 5.10), with some productivity lags and asynchronous productivity peaks occurring between the two land-uses over time. This is particularly pronounced in the Fynbos and Savannah biomes and least pronounced in the Karoo biomes. Finally, we used abline plots to assess the relationship between woody and grassy productivity trends between 1999 and 2014. Figure 5.11 shows the ration between woody and grassy productivity trends summarised to property or landscape scale between different land-uses to assess the relative changes in vegetation productivity when either woody or grassy productivity is more dominant. If woody and grassy productivity were independent and unaffected by one another, one would expect the ratio relationship to follow the abline.

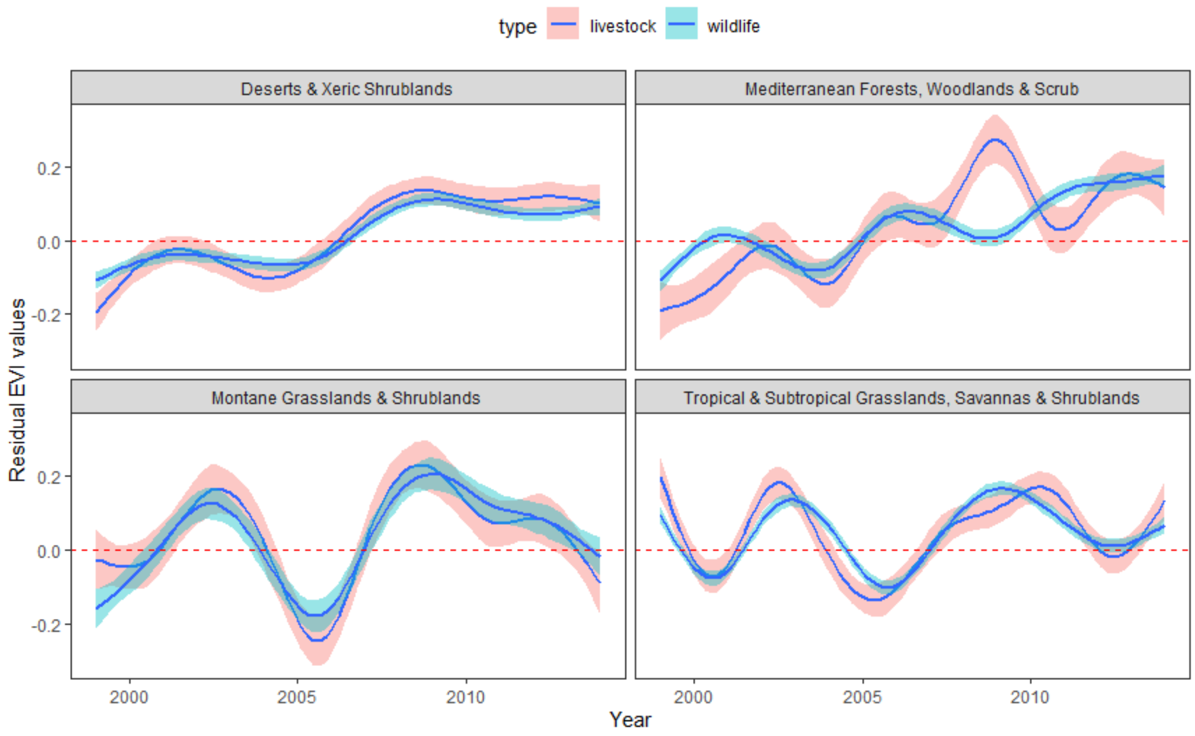

Figure 5.10. Interannual variation in the residual trends of grass biomass production (non-relativised) between WBLUs and livestock. While mostly tracking each other in the savannah, there are some years where one land-use does relatively better than the other.

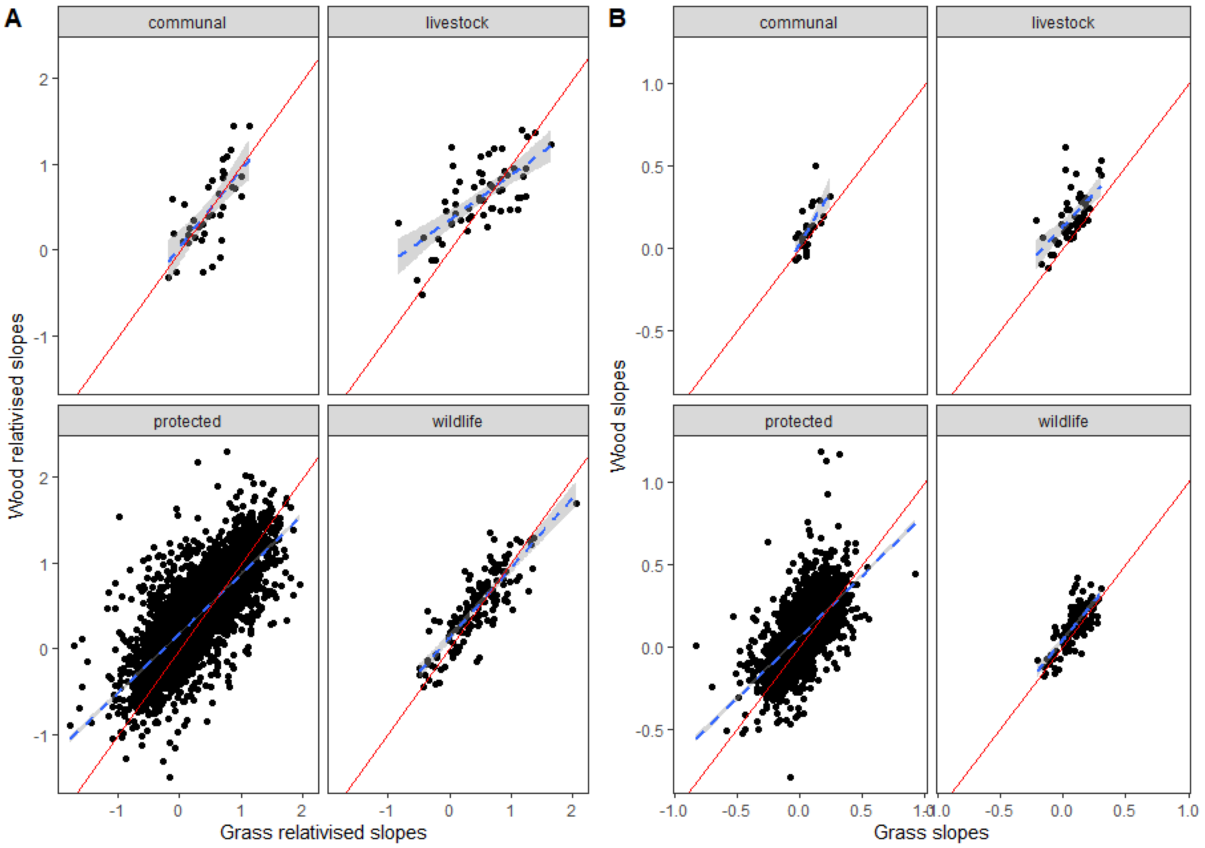

Figure 5.11. Correlation between (A) relativised and (B) un-relativised woody and grassy residual trend slopes (Δ EVI 1999-2014) averaged to property-scale. While communal rangelands show a trendline that mimics the expected change if woody and grassy biomass increased equally, other land-uses show tendencies that suggest higher woody rates of change suppress relative grass biomass on the property and vice versa for grassy biomass. In particular, commercial livestock farms show less grassy productivity than expected when there is relatively high woody productivity.

5.4.3 Effects of management on vegetation productivity and profitability

I assessed the impacts of grassy and woody productivity on revenue generation for WBLUs. Grassy and woody productivity are highly significantly correlated (OLS regression: $t_{(152)} = 9.61$, $p < 0.01$, $R^2 = 0.37$). I used linear regressions to test the relationship between WBLU revenue generation and vegetation productivity. I log-transformed the response variable revenue / ha to meet the assumptions of normality. Grassy productivity was significantly positively correlated with revenue when using the sum of biomass value as the predictor, but only grassy productivity was significant when using the sum of the residuals (rainfall-corrected) (Table 5.5, Figure 5.12). I also investigated the relationship between revenue / ha and the ratio of grassy:woody productivity trends on WBLUs where data were available to test whether properties with relatively higher or lower rates of grass productivity compared to woody productivity had higher revenues (*Supporting Note 5.7*).

I then used a wider set of predictor variables (fire management, alien invasive species and bush encroachment clearing, stocking density, trophic diversity of herbivore populations) to detect any other drivers of revenue generation in WBLUs through generalised linear models (GLMs). The sum of residuals was used as the predictor variables for grassy and woody productivity. Interactions between terms were included in the models. The only significant predictor variable remained grassy productivity, with stocking density being close to significance in its own model (OLS regression: $t_{(95)} = 1.72$, $p = 0.08$, $R^2 = 0.02$).

However, when looking at both fire management and bush clearing through t-tests, the use of fire is significantly correlated with higher revenues (Welch's t-test: $t_{(76)} = 2.12$, $p = 0.04$); while the use of alien plant and bush clearing was almost significant (Welch's t-test: $t_{(53)} = 1.80$, $p = 0.07$) (Figure 5.13).

Table 5.5. Regression model results for the relationship between revenue / ha of wildlife-based land-uses (log-transformed) and the sum of vegetation productivity values (both absolute and residual trends) between 1999 and 2014. Values are estimates, and t statistics are in brackets. Significance values are marked by asterisks.

	Revenue / ha (response variable)			
	Grassy	*Woody*	*Grassy*	*Woody*
Sum EVI values	0.280*** (0.077)	0.238** (0.104)		
Sum EVI residuals			2.440** (1.090)	1.579 (1.230)
Constant	2.759*** (0.486)	2.961*** (0.884)	3.943*** (0.275)	4.419*** (0.444)
Observations	91	48	91	48
R^2	0.13	0.10	0.05	0.03
Residual Std. Error	1.510 (df = 89)	1.490 (df = 46)	1.575 (df = 89)	1.544 (df = 46)
F Statistic	13.31** (df = 1; 89)	5.17** (df = 1; 46)	5.02** (df = 1; 89)	1.65 (df = 1; 46)

**p<0.1; **p<0.05; ***p<0.01*

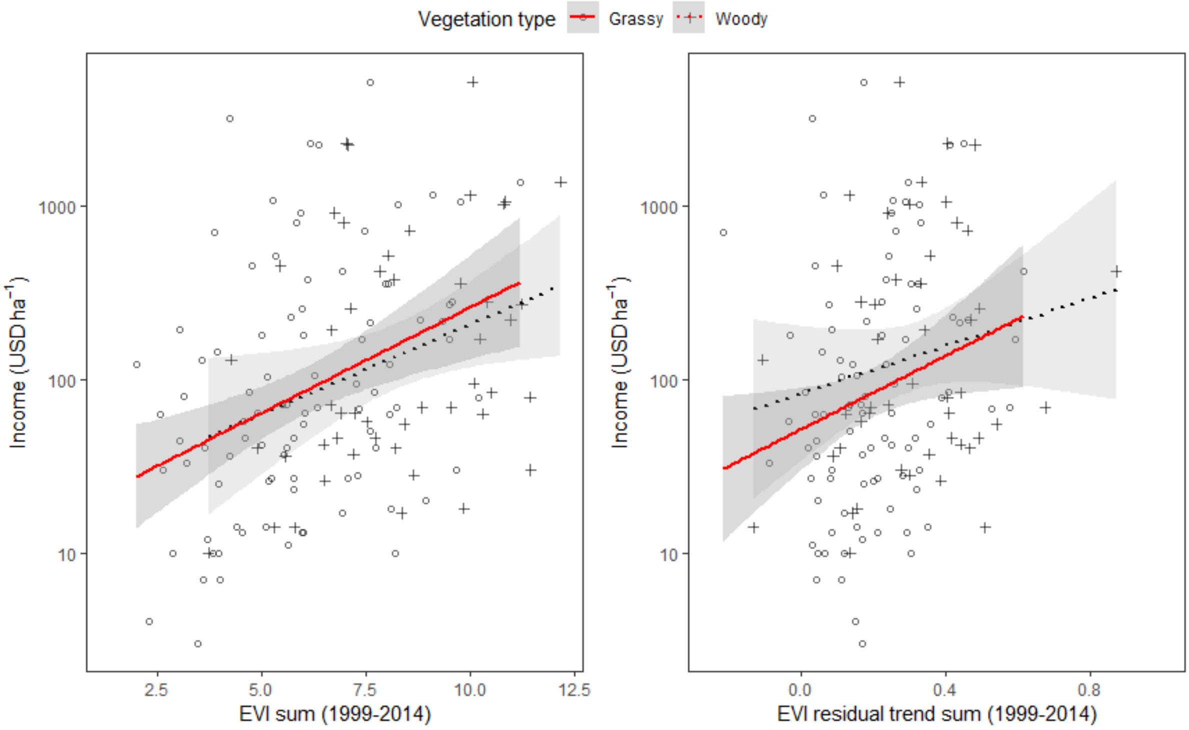

Figure 5.12. Regression between vegetation productivity and enterprise profitability. For the EVI straight biomass, both grassy and woody productivity are significantly correlated with WBLU revenue. For the residual trends in EVI biomass (rainfall corrected), there is only a significant positive relationship between grassy productivity and revenue.

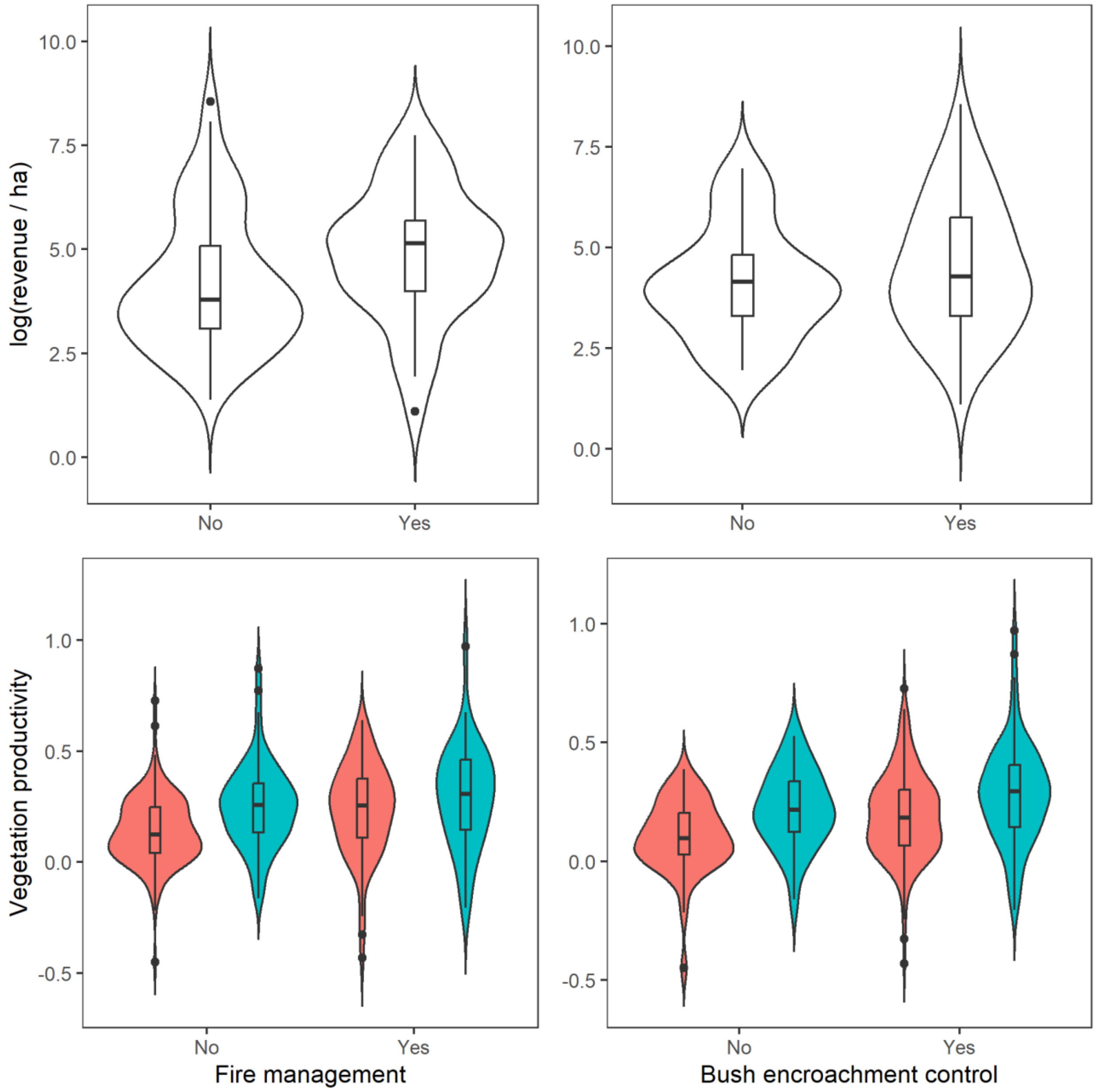

Figure 5.13. The difference in natural resource management techniques on revenue generation (top panel) and vegetation productivity (bottom panel) in wildlife-based land-uses where both the use of fire and the use of alien plant bush clearing are correlated with higher revenues and vegetation productivity. For revenue, only the use of fire is significant. For grassy productivity (red plots), the use of fire similarly is significantly correlated with higher productivity. While control of bush encroachment qualitatively increases grass productivity, it is not significant. For woody productivity, while the use of fire is significantly associated with higher productivity in a single term model, neither fire management nor bush encroachment control remains significant in the best-fit models. Violin plots show density distributions of values, while the boxplots nested within show the minimum, maximum and interquartile range of the values, with solid lines showing the median.

I investigated what management characteristics of WBLUs drive residual changes in vegetation productivity and, thus, revenue and job creation. I used the sum of EVI residual trends as response variables for grassy and woody productivity. I selected explanatory variables based on their potential impacts on vegetation productivity, which included fire management, alien invasive species and bush encroachment clearing, stocking density, and trophic diversity of herbivores. I ran regressions to understand the impacts of farm size and mixed wildlife and livestock farms on vegetation productivity dynamics so that we could separate out potentially confounding variables from the GLMMs, which were then used to understand potential management drivers of vegetation productivity on WBLUs. Farm size was not significantly correlated with either grassy or woody productivity (OLS regression: $t_{(315 / 152)} = -0.78 / 0.86$, $p = 0.46 / 0.39$, $R^2 = 0.002 / 0.004$ for grassy and woody productivity, respectively). Whether a WBLU is a mixed livestock / wildlife farm or a wildlife-only property does not significantly impact EVI trends, although wildlife-only WBLUs have slightly higher grassy productivity than mixed farm, but with low explanatory power (OLS regression: $t_{(314)} = 1.96$, $p = 0.05$, $R^2 = 0.003$); and there is no relationship with woody productivity (OLS regression: $t_{(152)} = -0.46$, $p = 0.64$, $R^2 = 0.001$).

I used AIC model selection to distinguish among a set of possible models describing the relationship between the two vegetation response variables (grassy and woody productivity sum of residuals) and the predictor variables as well as their interactions. For grassy productivity, the best-fit model (model 1) included just the use of fire management (GLM: $F_{(2,315)} = 13.07$, $p < 0.01$, $R^2 = 0.07$) (Table 5.6, Figure 5.13), carrying 98% of cumulative model weight. Combining fire management and trophic diversity with no interaction effects (model 7) was the next best fit model and explained 100% of cumulative model weight (GLM: $F_{(3,257)} = 11.01$, $p < 0.01$, $R^2 = 0.10$) (Table 5.6, Figure 5.14). Including biome as a random effect in the generalised linear mixed modelling produced the same results (Table 5.6).

For woody productivity, the best-fit model (model 2) included just the use of bush encroachment control (GLM: $F_{(1,113)} = 1.91$, $p = 0.17$, $R^2 = 0.02$), carrying 82% of the cumulative model weight. Combining bush encroachment control and fire management with interaction effects (model 3) was the next best fit model and explained 89% of cumulative model weight (GLM: $F_{(4,110)} = 0.85$, $p = 0.49$, $R^2 = 0.03$) (Table 5.6). The next best fit model (model 5) included just trophic diversity as an explanatory factor and explained 91% of cumulative model weight (GLM: $F_{(1,119)} = 6.53$, $p = 0.01$, $R^2 = 0.05$). Including fire management as a single term (model 1) was close to being significant for the use of fire GLM: $F_{(2,151)} = 2.34$, $p = 0.09$, $R^2 = 0.03$) (Figure 5.13). In the two best fit models, no terms were significant, including biome as a random effect through generalised linear mixed modelling produced the same results (Table 5.6).

Table 5.6. Results of generalised linear models and mixed models (including biome as a random effect) for the relationship between grassy and woody productivity (as measured through the sum of residuals between 1999 and 2014) and various natural resource management predictor variables. Best -fit models were determined through Akaike's Information Criterion (AIC), and only these are reported.

	Grassy productivity (GLMs)					
	Grass model 1			Grass model 7		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	0.11	0.07 – 0.15	**<0.001**	0.04	-0.03 – 0.11	0.295
Fire management (No)	0.03	-0.01 – 0.08	0.165	0.01	-0.05 – 0.07	0.733
Fire management (Yes)	0.13	0.08 – 0.19	**<0.001**	0.11	0.05 – 0.18	**0.001**
Trophic diversity				0.17	0.05 – 0.29	**0.004**
Observations	318			261		
R^2	0.08			0.12 (fire) / 0.10 (trophic)		
AIC	-196			-171		
	Grassy productivity (GLMMs)					
(Intercept)	0.12	0.01 – 0.22	**0.035**	0.06	-0.07 – 0.19	0.351
Fire management (No)	0.06	0.01 – 0.11	**0.012**	0.04	-0.01 – 0.10	0.126
Fire management (Yes)	0.12	0.07 – 0.18	**<0.001**	0.11	0.05 – 0.17	**0.001**
Trophic diversity	0.12	0.01 – 0.22	**0.035**	0.14	0.03 – 0.25	**0.014**
				0.06	-0.07 – 0.19	0.351
	Woody productivity (GLMs)					
	Wood model 2			Wood model 3		
(Intercept)	0.22	0.14 – 0.30	**<0.001**	0.27	0.04 – 0.51	**0.022**
Bush control (Yes)	0.06	-0.03 – 0.15	0.17	0.02	-0.13 – 0.17	0.786
Fire management (No)				-0.08	-0.34 – 0.17	0.509
Fire management (Yes)				0.01	-0.19 – 0.20	0.956
Fire management (No) × Bush control (Yes)				0.07	-0.12 – 0.25	0.488
Observations	115			115		
R^2	0.02			0.03		
AIC	-37			-32		
	Woody productivity (GLMs)					
	Model 2			Model 3		
(Intercept)	0.26	0.14 – 0.38	**<0.001**	0.36	0.11 – 0.61	**0.005**

Bush control (Yes)	0.01	-0.08 – 0.09	0.902	-0.06	-0.20 – 0.09	0.444
Fire management (No)				-0.13	-0.37 – 0.11	0.29
Fire management (Yes)				-0.05	-0.24 – 0.13	0.573
Fire management (No) × Bush control (Yes)				0.09	-0.08 – 0.26	0.305

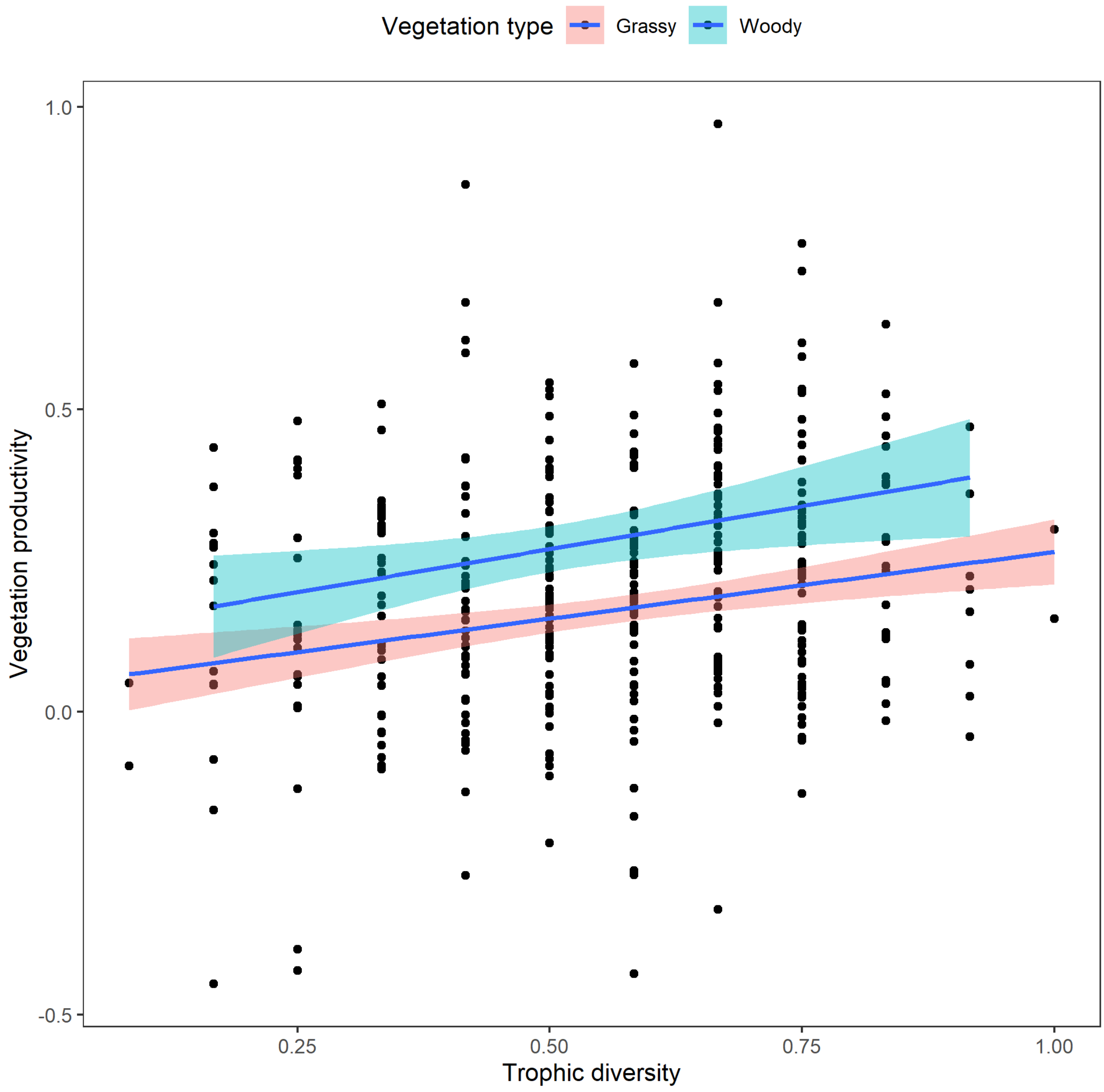

Figure 5.14. The relationship between vegetation productivity, as measured through the sum of residuals over 1999-2014, and trophic diversity of the herbivore guilds (grazers, browsers, mixed feeders and megaherbivores) on wildlife-based land-uses. For both grassy and woody productivity trends, higher levels of trophic diversity are significantly correlated with higher productivity when used as single terms in linear models, but only remains significant as a term in a second best fit model for grassy productivity.

5.5 Discussion

5.5.1 Rewilding rangelands enhances grass productivity

Rewilding rangelands significantly increases residual grass productivity compared to formal protected areas and communal rangelands (and shows similar trends to commercial livestock farms). Since conversion to WBLUs, the residual grass productivity trends of both WBLUs and commercial livestock farms have risen significantly above that of both formal protected areas and communal rangelands, despite starting from a lower productivity point (Figure 5.5, Figure 5.7); and WBLUs show even greater increases in residual grass productivity than livestock farms when using un-relativised EVI data (*Supporting Information 5.6*). While the proportion of significantly positive productivity trends per unit area is similar between WBLUs, commercial livestock farms and formal protected areas, WBLUs have the highest proportion of non-significant positive increases in grassy productivity with lower rates of woody productivity compared to commercial livestock farms. These trends occur despite WBLUs existing in areas with the lowest average and aggregate rainfall of all comparative land-uses (*Supporting Information 5.6*). Thus, these results demonstrate the efficacy of WBLUs as ecosystem-based adaptations to climate change whilst still sustaining viable socioeconomic enterprises. These results validate the landowners' own perceptions, where 71% of landowners reported seeing an improvement in rangeland condition since converting to wildlife ranching, and of those, 60% reported it as an increase in grass cover (*Supporting Information 5.1*).

The significant increase in grass productivity in WBLUs is remarkable, considering that maximising animal (and, therefore, grass) productivity is not generally the core business model of WBLUs (Taylor et al. 2020), as it is for commercial livestock farms. Our sample of commercial livestock farms all practice some form of high-density rotational grazing system (Venter et al. 2019), which improves grass productivity and increases soil carbon sequestration (Odadi et al. 2017; Peel & Stalmans 2018; Hillenbrand et al. 2019; Mosier et al. 2021) [but see (O'Connor et al. 2010; Hawkins 2017; Venter et al. 2019; Hawkins et al. 2022)]. The sampled livestock farms had on average 59 ± 12% (mean ± standard error) higher stocking densities than recommended by the long-term grazing capacity guidelines for South Africa (DAFF 2018;

Venter et al. 2019); whereas the sampled WBLUs had an average of 16 ± 17% (mean ± standard error) lower stocking rates on the extensive areas than recommended (*Supporting Information 5.4*). As such, the generally lower stocking rates of WBLUs compared to livestock farms could be one mechanism by which residual grass productivity has improved, which is supported by Venter et al. (2019), who found that farms implementing moderate (rather than high or low) stocking densities undergo the least declines in vegetation cover (see also Masubelele et al. 2014). Interestingly, stocking density did not correlate with grassy productivity in management impacts analysis of WBLUs, perhaps because the generally low stocking densities were insufficient to cross critical grazing thresholds and produce signals in the data or because the impact of fire management has larger absolute effects (see Little et al. 2013). Corroborating evidence for WBLUs having higher grass productivity trends than cattle farms comes from within the WBLU dataset where mixed cattle and wildlife WBLUs are almost significantly ($p = 0.05$) more likely to have lower grass residual trends than wildlife-only WBLUs (but no difference for woody productivity), which further underscores the rangeland restoration potential of wildlife-based systems.

A second mechanism for the relative gains of grass productivity on WBLUs could be the higher trophic diversity of rewilded rangelands, compared to the single guild system of cattle farming (Hempson et al. 2017; Cromsigt et al. 2018). Increased levels of trophic diversity had weak explanatory power for increases in grass productivity in the second best-fit model. While also correlating with woody productivity (Figure 5.14), model fit was better for grassy productivity. Wild herbivores play crucial nutrient dispersal roles through diverse spatial ecologies and seasonal movement patterns, enabling the distribution of nutrients more evenly across the landscape and, thus, improvements in forage quality and landscape heterogeneity (Charles et al. 2017; Keesing et al. 2018; Sitters et al. 2020). The nutrient cycling functions of wild herbivores are mediated by body size, where more diverse herbivore guilds cycle nutrients more evenly across the landscape (le Roux et al. 2018), and a more diverse guild of herbivores may spread grazing pressure amongst a greater diversity of forage species (O'Connor et al. 2010; Kartzinel et al. 2015). Wildlife may also create positive feedback loops for improving rangeland condition. In North America, American bison (*Bison bison*), rather than 'surfing' the green wave of vegetation productivity, actively create it through intensive and aggregate grazing as they move across the landscape, stimulating new green growth and increasing forage quality (Geremia et al. 2019).

Additionally, research has shown that these benefits may not rely on huge landscapes with migratory movements: experimental evidence shows that holistic planned grazing in adaptive multi-paddocks with bison herds led to significant improvements in rangeland condition (increased fine litter cover, improved water infiltration, two to three times the available forage biomass, improved plant composition, decrease in invasive plants, and decrease in bare ground) and increases in soil organic carbon compared to light and heavy continuous grazing by cattle (Hillenbrand et al. 2019). South Africa has several aggregate and migratory grazing species. However, the high-density and intense grazing pressure of migratory herds has ceased due to population declines and fencing (Roche 2008), so it is unclear whether the green wave feedback with grazing is one of the mechanisms reflected in this study. Regardless, restoring large-scale movement of wildlife across the landscape as part of conservation corridors and conservancy formation could lead to even greater gains in grass productivity and forage quality, and presents an opportunity to create 'rewilded commons' in large communal grazing areas as community property associations are formed and begin entering the wildlife economy.

5.5.2 Rewilding rangelands slows rates of woody productivity

Woody productivity trends were significantly higher in proportion than grassy productivity trends in all land-uses besides communal rangelands. While much of this productivity is likely associated with natural tree and shrub growth in healthy rangelands, a large proportion of it will be due to bush encroachment from both indigenous and alien species, which has been extensively documented to be increasing both in South Africa and throughout African rangelands (O'Connor et al. 2014; Skowno et al. 2017; Venter et al. 2018; Turpie et al. 2019). This phenomenon is thought to be due to several factors, including elevated carbon dioxide levels (Skowno et al. 2017; Venter et al. 2018; Piao et al. 2020), replacement of the browsing guild of wildlife by cattle (Hempson et al. 2017), as well as suppression of natural fire regimes and intensive grazing that further reduces fire fuel load (O'Connor et al. 2014; Skowno et al. 2017). These results support the contention that general greening caused by woody plant productivity is occurring at broader spatial and temporal scales than local farms for the following reasons: Firstly, woody productivity has increased continuously and linearly over time and shows no step change pre and post conversion to WBLUs, which suggests that woody growth is being driven by broad-scale environmental drivers, namely elevated CO_2 levels (Skowno et al. 2017; Venter et al. 2018). Conversely, grassy productivity dipped to lows in the pre-conversion period and has dynamically increased to its highest level in the post -conversion period, with an acceleration of productivity post-conversion, suggesting WBLU dynamics and management are important for driving grassy growth.

Secondly, Figure 5.6 shows how the increase in the number of significant grass productivity trends after converting to WBLUs is far more numerous than the increase in woody productivity trends, where grass productivity was generally less than what is expected from rainfall before and more than expected after. Woody productivity, conversely, had high numbers of significant increase trends both before and after rewilding, and there is no difference in the number of significant woody productivity trends before and after conversion. In contrast, there is a significant increase in significant trends after rewilding. Finally, generalised linear models of management interventions failed to sufficiently fit woody productivity data, which implies there are unmeasured variables that more adequately account for woody productivity trends. Overall, these patterns suggest that this vegetation component is generally being driven by external factors such as elevated CO_2.

Heightened woody plant productivity can severely affect rangeland ecosystem functioning by reducing or suppressing grassy productivity (Williams & Albertson 2006; O'Connor et al. 2014; Turpie et al. 2019), and thus undermining the potential for rewilding to positively create viable WBLUs within the wildlife economy. In this study, while woody and grassy growth are highly significantly correlated (*Supporting Information 5.7*), analysis of woody:grassy ratios showed that woody productivity is generally higher in landscapes with low grassy productivity and vice versa for grassy productivity for all land-uses besides communal rangelands. While communal rangelands show a trendline that mimics the expected change if woody and grassy biomass increased equally, other land-uses show tendencies that suggest higher woody rates of change suppress relative grass biomass on the property and vice versa for grassy biomass. However, WBLUs have relatively higher grass productivity at higher levels of woody productivity than commercial livestock farms. Generally, WBLUs show a more balanced vegetation productivity trend across the woody:grassy productivity spectrum, which may indicate that they are relatively more resistant to woody plant encroachment than other land-uses. This may be because WBLUs have both grazing and browsing guilds and so can make better use of available resources on a rangeland and thus keep the woody:grassy productivity trends more balanced. The browse component is most often unaccounted for when developing rangeland management guidelines, as most guidelines refer to grazing capacity only (Bothma et al. 2004; DAFF 2018). Additionally, when looking at the straight (non-relativised) EVI slopes, while commercial livestock ranches and WBLUs have the same trendline, almost no commercial ranches have relatively higher grassy gains than expected. This means that grassy productivity is less than expected, and woody productivity is greater in these farms. This has important implications for ecosystem-based adaptations to climate change because establishing WBLUs may be a more resilient land-use to drivers of global land change, much like many WBLUs are more socio-economically resilient to global disturbances than protected areas (Clements et al. 2022).

Commercial livestock farms have the highest proportion of significant woody productivity trends compared to other land-uses (no significant positive trends in the before period compared to 32% in the period after) and have significantly higher woody productivity trends than WBLUs. This may be because livestock ranches generally lack the browsing guild. Rewilding rangelands with an indigenous herbivore assemblage has been effective in reducing the extent of alien plant coverage and woody plant encroachment because wild herbivore communities encompass browsing and mixed feeder guilds that facilitate density-dependent shifts to eating back shrub invasions and different browsers eat the same plant differently (McGranahan 2008; Pringle et al. 2014; Otfinowski et al. 2017; Calleja et al. 2019; Guyton et al. 2020; Irob et al. 2022). The reintroduction of browsing species tends to reduce bush encroachment, increase grass cover (which leads to improved water uptake by plants) and restore plant functional diversity (Irob et al. 2022). Model simulations reveal even high densities of browsers can lead to increased perennial grass biomass and thus sustain forage production potential (Irob et al. 2022). Considering many WBLU models focus on hunting of browsers or mixed feeders, this might be a crucial business model to restore a key ecosystem service (palatable grass production) to formerly degraded rangelands. The lack of fire management in commercial livestock farms may also play a role, especially since fire use was significantly positively correlated with grass productivity in WBLUs. Parsons et al. (1997) found commercial game farms had the lowest proportion of unpalatable grass species and the highest proportion of perennials, noting that in their study area, only the game farms used fire as a management tool. However, in this study, neither fire management nor bush clearing corresponded to lower woody growth rates, which means that the systems had already crossed a regime shift that is difficult to reverse without broadscale clearing. Similarly, we did not find any effect of megaherbivores on rates of woody increase in contrast to previous studies (O'Connor et al. 2014; Stevens et al. 2016), but a more detailed analysis is required. However, the bimodal curve for grass productivity in commercial livestock farms shown in Figure 5.8 suggests that overgrazing and suppression of fire in some livestock sites may promote tree encroachment, whereas in other sites, more active mitigation measures may reverse this trend.

5.5.3 Counterfactual land-use comparison and landscape planning

These results are supported by comparing livestock and wildlife working lands to two counterfactual land uses – formal protected areas and communal rangelands. Protected areas (PAs) can be thought of as the null model for rewilding as management systems are set up to restore biodiversity and ecosystem functioning, while communal rangelands are the null model for livestock agriculture as most are under continuous grazing systems. In this study, the full rewilding scenario represented by PAs exhibited the highest average grass biomass, and absolute but residual productivity trends were lower than WBLUs. This may be because many PAs have been under wildlife land-use for a long time, grass productivity has decelerated or passed an asymptote, whereas – due to the relatively young nature of many WBLUs (*Supporting Information 5.1*), there is a steeper grass productivity trend from relatively more degraded conditions. These results show that it is possible for WBLU 'working landscapes' to have similar vegetation productivity to set-aside PAs. This has important implications for how policy makers view wildlife areas and how WBLUs might be used in conservation planning and mixed production landscapes. For example, investing in WBLUs on the border of PAs may have mutual benefits, such as reducing poaching by allowing sustainable game meat offtakes in the WBLUs, reducing the need for PA culling by allowing dispersal into the WBLUs (and also reducing the capital investment of buying animals by the WBLUs), and ultimately creating landscape-scale restoration corridors through investment into surrounding rangeland ecological infrastructure (Desmet et al. 2019). This is not a new idea – buffer zones have always been a feature of PA planning. However, these results contribute to a growing body of evidence that suggests the sustainable use of wildlife in buffer zones can improve livelihoods, restore degraded habitats and function similarly to PAs (Chidakel et al. 2020).

Communal rangelands, conversely, can be considered counterfactual to commercial livestock farming as they encompass broad areas (former homelands) under various forms of continuous grazing systems. Correspondingly, communal rangelands had the lowest grass productivity trends, possible because continuous grazing in African savannahs impacts grass productivity relative to other grazing systems (Oomen et al. 2016; Mokgotsi 2018). Communal rangelands also had significantly lower rates of woody productivity than all comparative land-uses, possible because of fuelwood harvesting (Wessels et al. 2011; O'Connor et al. 2014). There are two implications from these results. Firstly, given the general trend of de-ruralisation and abandonment of farm plots (Shackleton et al. 2019), these areas may increasingly be considered for rewilding and the wildlife economy to restore grass production ecosystem services. For example, abandoned communal cropland in the Drakensberg grasslands reverted to indigenous grassland almost devoid of alien species within 20 years (O'Connor 2005). Secondly, communal rangelands can be crucial in connecting ecosystems and creating landscape corridors for wildlife movement (Kiffner et al. 2016). In exchange, policy-makers could establish a 'forage market' through leased grazing lands on WBLUs where grass productivity is higher and more sustained (Fynn et al. 2016). For example, cattle farmers in the Zambezi Region of Namibia (within the Kavango-Zambezi Transfrontier Conservation Area) cannot afford to offset the loss of condition in the dry season with supplementary forage. However, at a landscape scale, there is sufficient forage in the national parks and conservancies to sustain animal condition throughout the year, but these areas with 'surplus' forage exist beyond the grazing range of villages and perennial rivers where cattle congregate (Van Rooyen 2016). Therefore, the opportunity to access these underutilised areas has been suggested as a solution to mitigate the impacts of climate change.

Similarly, the interplay between commercial livestock farms and WBLUs on a landscape scale could produce emergent effects on ecosystem service provision. The results show that cattle farms and WBLUs show similar productivity trends, but these seem to be lagged and sometimes the two land-uses experience troughs and peaks in production alternately. Intriguingly, this may imply that multi-functional landscapes, combining livestock-only, mixed farms, and wildlife-only farms, may enhance the landscape-scale resilience of grass productivity. For example, in poor livestock-farm years, schemes could be established to allow grazing access for cattle in exchange for reciprocal options in the future. This 'forage barter' economy could be quantified, coordinated and audited using remote sensing tools. Within mixed farms, wildlife and cattle can have unique, additive, and interactive effects on productivity in an African savanna (Charles et al. 2017), reducing spatial and temporal variability in ecosystem services such as productivity. Cattle farming also tends to create long-lived nutrient hotspots in kraals that establish plant communities favoured by herbivores and thus provide positive feedback for grazing capacity of rangeland ecological infrastructure (Fynn et al. 2016; Sitters et al. 2020).

Even within WBLU business models, there are different effects of grass productivity. For example, ecotourism-only models show significantly lower gains in grass productivity since rewilding compared to mixed-use and use-only properties, despite woody growth trends being similar across WBLU models (*Supporting Information 5.5*). This may be because ecotourism models do not need to invest as much in fire management and bush clearing to increase rangeland condition, as their models are not based on animal production but on the presence of predators and large mammals (Clements et al. 2016c). This is similar to the findings of Child et al. (2013), who found that commercial ecotourism WBLUs had more intensive management and lower residual grass productivity than WBLUs focussing on extensive hunting. However, the trade-offs between production systems, biodiversity conservation and socioeconomic contribution amounted to functional diversity of systems at a landscape scale. Taken altogether, these results suggest that different land-uses and WBLU business models may be complementary on a landscape scale and enhance the overall resilience of social-ecological systems (Chidakel et al. 2020; Clements et al. 2022). For example, Holecheck and Valdez (2018) suggest that low-input WBLUs for meat and hunting may be an economically viable alternative to livestock production systems in some areas. However, in other areas, communal grazing of livestock and wildlife will most efficiently use rangeland forages and diversify income.

5.5.4 Restoring socio-economic opportunity through grass productivity

Vegetation productivity over time is a significant predictor of profitability, especially grassy productivity, as woody productivity is not significantly linked to revenue when looking at the sum of residuals. Thus, these results demonstrate that rewilding rangelands can make them more profitable and create more employment opportunities by restoring a critical ecosystem service – forage production. The use of fire management is significantly correlated with both higher grass productivity and higher revenue generation, and increased levels of trophic diversity also contribute to grass productivity trends. Woody productivity variation, conversely, is associated with more variables with weaker explanatory power, including bush encroachment control and fire management and their interactions, as well as trophic diversity. This may corroborate other results in that woody productivity is being driven by external factors, such as elevated CO_2 levels, rather than the effects of management alone. Also, no terms were significant in the best fit models, indicating that no management variable had sufficient explanatory power to explain changes in woody productivity. Taken together, these results provide crucial evidence for the long-term return on investment for rewilding rangelands and establishing ecologically managed WBLUs. This study contributes to previous studies that demonstrate the profitability, socioeconomic contributions and resilience of WBLUs (Taylor et al. 2020; Chidakel et al. 2020; Clements et al. 2022).

The use of fire as an ecological management tool to restore ecosystem functioning and biodiversity is well documented and is especially effective when combined with trophic rewilding and grazing management. This ‘pyric-grazing model’ creates a dynamic habitat mosaics on a landscape scale around the burn cycle, which sustains rangeland ecosystem service, improves herbivore condition and creates spatial heterogeneity that leads to higher species richness (Tomor & Owen-Smith 2002, 2002; Gureja & Owen-Smith 2002; Fuhlendorf & Engle 2004; McGranahan 2008; Fuhlendorf et al. 2009; Little et al. 2013; Smit & Archibald 2019).

The interaction between fire and herbivory in time and space produces both positive and negative feedbacks in productivity to create a shifting and dynamic pattern of disturbance across the landscape (Fuhlendorf et al. 2009), which corresponds to the definition of rewilding by Perino et al. (2019) as restoring stochastic disturbances and trophic complexity. In the sample of commercial livestock farms used in this study, only 15% of farmers used fire as a management tool (Z. Venter, unpubl. data; Venter et al. 2019), whereas, in our WBLU sample, 38% of ranchers employed fire management. Thus, converting to WBLUs results in both restoring trophic complexity and disturbance stochasticity and this ‘pyric-herbivory’ interplay has the strongest explanatory power for gains in grassy productivity and ultimately revenue. While WBLU managers employ different burning regimes (e.g., block versus patch mosaic burning) and at different burning intervals, the specific burning regime may not matter as much in the long term as opposed to the complete absence of burning. For example, Everson & Everson (2016) found no difference after 30 years in the standing live grass biomass between annual winter and biennial spring burning in montane grasslands, but did find significantly lower productivity in unburnt areas compared to regularly burnt areas.

Several other variables qualitatively increased grass productivity and revenue but were insignificant. Mechanical clearing of invasive alien plants (IAPs) and woody encroachers was almost significantly associated with higher revenues (p = 0.07) and also correlated with higher grass productivity. This finding requires more analysis and quantification of clearing effort as the signal may have been lost when converting to binary responses. This natural resource management technique is especially important because IAPs drastically reduce rangeland productivity and, for example, are estimated to reduce the value of rangelands in South Africa by ZAR 340 million annually (O'Connor & van Wilgen 2020). Similarly, revenues in WBLUs appear to be higher in properties with relatively higher grass productivity compared to woody as determined by the ratio of grassy:woody productivity per property [see Figure 5.11 and *Supporting Information 5.7*]. However, the relationship is insignificant (p = 0.12), possibly reflecting the small sample size (N = 48) for this analysis. This echoes assessments that show woody encroachment decreases economic opportunity due to the loss of productive grasslands and thus animal production potential, creating poverty traps for poor rural communities as remaining grassy areas become overgrazed, and decreased fuel loads reduce fire frequency and intensity that further facilitates encroachment (Turpie et al. 2019; White et al. 2022).

Rewilding increases revenue through increased grass productivity and creates more stable and consistent production systems, which may also unlock revenue streams from soil carbon credits (*Supporting Information 5.8*). While vegetation productivity has increased following conversion to WBLUs, the variance in productivity has decreased. The standard deviation in productivity between years has decreased by 60% for grass patches, while for wood patches, it has decreased by 40%. This could mean that the ecological functions of wildlife have enhanced ecosystem functioning, as has been documented elsewhere, or it could be that WBLU managers have improved habitat condition through sustainable land management interventions or a combination of both. Additionally, WBLUs generally show a more balanced vegetation productivity trend across the woody:grassy productivity spectrum, as revealed by the abline plots (Figure 5.11), meaning that woody productivity does not tend to dominate overall system productivity. The lower variance in grass productivity may reflect the importance of this resource in agro-ecological systems. Wildlife are able to respond opportunistically to local rainfall events and increase their grazing activity where and when primary productivity is high, and the diverse set of trophic guilds leads to a more evenly distributed spatial impact on the herbaceous layer, which may lead to more spatially and temporally consistent productivity (Charles et al. 2017; Keesing et al. 2018; Sitters et al. 2020). These data reveal the importance of WBLUs as sources of resilient ecosystem service provision in rangelands.

Interestingly, sustainable use models, based on hunting, game meat production and wildlife breeding, or models combining ecotourism with sustainable use, show significantly greater grass productivity trends than ecotourism-only models. This may indicate that sustainable use models will achieve more return on investment for long-term restoration of rangelands and sustained job creation opportunities (as revenues correlate with profit and job numbers) than many ecotourism enterprises (for example, see Clements & Cumming 2018). Further research should investigate these findings in more detail to ensure the various attributes of different wildlife economy business models – landscape health, biodiversity, resilience and job creation – are adequately understood to attract more effective investment into new market entrant enterprises.

5.5.5 Policy implications

Diversify investment into different WBLU business models, especially those based on sustainable use. Rewilding rangelands improves vegetation productivity, especially the residual increase in grass productivity, which is correlated with higher revenues / ha (and thus higher employment rates, *Supporting Information 5.3*). These patterns are contrasted to communal rangelands, which are predominantly under continuous grazing systems with livestock, and where most of the country’s previously disadvantaged individuals live. Our results demonstrate the potential of what rewilding these commons means for simultaneously improving ecosystem functioning, biodiversity and socio-economic development. The National Biodiversity Economy Strategy (NBES, 2016) aims to add 100,000 jobs through new market entrants to the wildlife economy, which these results support could be feasible. However, policy-makers should be careful to promote sustainable use models as well as more traditional ecotourism models, as sustainable use models show significantly greater grass productivity trends than ecotourism. Both ecotourism and sustainable use models (including mixed farms) have differential responses to socioeconomic resilience, whereas more agriculturally focussed models are more resilient to global perturbations because of a more diversified revenue stream and capacity to adapt production (Clements et al. 2022). These results add another resilience dimension – rangeland ecosystem service provision. Rather than pursuing a perceived 'silver bullet' land-use, evidence is emerging that a mosaic of different wildlife economy models, with various trade-offs in conservation, production and job creation, should be pursued on a landscape scale to enhance rural socioeconomic and social-ecological resilience (Child et al. 2013; Chidakel et al. 2020; Clements et al. 2022). This underscores the imperative for policies to incentivise conservancy formation, comprising multiple WBLU enterprise models, as shared working landscapes while also designing policies to attract investment into the infrastructure necessary to unlock the wildlife economy. Given the generally low stocking rates on WBLUs, payment for ecosystem schemes could be used to establish access to grazing areas inside WBLUs by herders and livestock farmers, which in turn would promote co-existence with wildlife, reduce poaching rates and potentially create wildlife corridors on a landscapes scale, as has been done in the conservancies of Kenya (Fynn et al. 2016).

Government institutions must communicate clearly the importance of South Africa's hunting model. There is a strong global north anti-hunting sentiment, where the legal sustainable use of game species is often conflated with the illegal trade in wildlife. This is leading to many global north countries beginning to implement trophy hunting bans, which may have dire consequences for the wildlife habitats and employees of ranches that implement sustainable and ethical hunting. For example, 36% of landowners reported that they would transition back to livestock or crop agriculture if a trophy hunting ban were to be fully implemented (Parker et al. 2020). These results add to a growing evidence base that shows the effectiveness of hunting models for sustainable development in African rangelands. Hunting generates more significant revenues than ecotourism and employs almost as many people (Lindsey et al. 2007; Saayman et al. 2018; Clements et al. 2022) while restoring rangeland condition.

While 'silver bullet' WBLU models should not be pursued above all else, equally perceived 'negative' WBLU models should not become the sole focus of excessive regulation. Generally, WBLUs have substantially lower stocking rates on extensive areas than commercial livestock farms, including within the breeding camps (*Supporting Information 5.4*). However, there is currently substantial concern in public discourse and, resultingly, policy design around the potential impacts of breeding camps and subsequent impacts on rangeland condition. For example, the recent High-Level Panel report is pursuing a policy of dropping fences and de-intensifying wildlife management to create more extensive areas, without appreciating that some business models may rely on erecting small camps to make their enterprises viable. Considering the total area of intensive breeding camps is only 10% of properties conducting intensive breeding and 5% of all WBLUs (Taylor et al. 2020), it seems a relatively small footprint of intensive management may offset a much larger area of ecological stocking rates, thereby alleviating grazing pressure on 90–95% of all land across WBLUs. Our results show that, on the whole farm scale, WBLUs are still generating ecosystem services for rangelands, and grassland productivity is not correlated with WBLU size or area of breeding camp footprint, and it thus may be unnecessary or even counterproductive to regulate land-use and farm management practices.

Employ WBLUs as ecosystem-based adaptations to climate change to enable rural economies to thrive. WBLUs often exist in marginal lands or arid areas, and the results of this study show that increases in vegetation productivity have occurred despite WBLUs having the lowest overall average rainfall (*Supporting Information 5.6*). Indeed, landowners in southern Africa started switching to WBLUs to combat droughts, especially the droughts of 1984 and 1992 (Lindsey et al. 2009; Child et al. 2012b). Wildlife are better adapted to environmental disturbance and stochasticity, such as droughts and disease, and are more adept at finding alternative forage species (e.g., Selebatso et al. 2018). Wild herbivores can also switch between grazing and browsing and, through niche separation, are able to more effectively use the full spectrum of browse and grazing forage available (Taylor & Walker 1978; Du Toit & Cumming 1999). As rangelands become encroached, the role of wild browsers will be key, both to mitigate further encroachment and also to provide revenue from hunting, game meat production and ecotourism. Such adaptions from a diverse herbivore assemblage enable a more consistent production system over time (reviewed in McGranahan 2008), which is supported by the results of this study in that the variance in grass productivity decreased following conversion to WBLUs.

As traditional models of livestock agriculture will become less suitable in many areas of Africa with projected patterns of increasing aridity (Ferner et al. 2018), the potential for WBLUs to improve rangeland condition whilst sustaining commercially viable enterprises make this land-use a key ecosystem-based adaptation (EbA) to climate change. Similarly, rewilding is also a climate change mitigation strategy as increases in vegetation productivity help to sequester soil carbon (Keesing et al. 2018; Sitters et al. 2020). These aspects of WBLUs should be incorporated into spatial planning processes for EbAs. For example, a rangeland ecological infrastructure map has been developed based on land cover classes, and the condition of existing rangelands is estimated from trends in primary productivity (Desmet et al. 2019), which is a similar approach used in this study. This map, combined with the maps of designated Biodiversity Economy Nodes, could be used to prioritise degraded areas of rangeland where WBLUs could help restore rangeland ecological infrastructure in South Africa. Climate financing mechanisms, such as the Green Climate Fund, could then be used to fund infrastructure development and the costs of establishing viable WBLU enterprises. Policy instruments and funding should also incentivise ecological management, such as the use of ecological fire management regimes in ‘pyric-herbivory’ management models, as fire and grazing guild have been shown to significantly influence grass productivity and thus revenue in this study.

Use WBLUs as a land-use bridge between conservation and agriculture mandates. WBLUs are a key interface land-use between conservation and agriculture, existing at the nexus of both the Convention on Biological Diversity (UNCBD) and the Convention to Combat Desertification (UNCCD). Our results show that wildlife ‘working lands’ achieve as much residual grass productivity as commercial, rotationally-grazed livestock farms with less increases in potential bush encroachment; and that the business models employing sustainable use strategies, such as hunting and game meat production, have significantly higher levels of grass productivity than ecotourism models. As such, the wildlife economy should cease being seen as a simple extension of protected area networks (through ecotourism and stewardship models) as there are multiple business models along the wildlife working lands spectrum, each with their own impacts on revenue generation, job creation, resilience (Clements et al. 2022) and, through our results, productivity. WBLUs are key to meeting the goals of the Land Degradation Neutrality (LDN) framework under the UNCCD, where LDN is defined as “a state whereby the amount and quality of land resources necessary to support ecosystem functions and services to enhance food security remain stable, or increase, within specified temporal and spatial scales and ecosystems" (Cowie et al. 2018). Crucially, this conceptualisation views restoration as the result of socioeconomic and social-ecological systems that drive change (Cowie et al. 2018). Shifting to a systems-level understanding of biodiversity conservation and associated policies is a major recommendation in the recent High-Level Panel report (DFFE 2020). Three indicators are used to measure LDN: positive land cover change, land productivity and carbon stocks (UNCCD/Science-Policy Interface 2016; Cowie et al. 2018). The LDN indicators are complementary components of land-based natural capital because gains in one of these measures cannot compensate for losses in another. The "one-out, all-out" principle is also critical to prevent perverse incentives. For example, bush encroachment and inappropriate planting of trees (as through the Bonn Challenge) may increase above-ground carbon stocks but decrease biodiversity and grazing productivity, thus undermining sustainable development in the long run. Linking LDN to the 2030 GBF through an ecosystem type perspective will help to resolve these issues, as land-cover change and restoration strategies can then be contextualised through ecosystem type characteristics (such as the appropriate woody to grassy ratio, or the relevant indicator species). An ecosystem perspective can give land-use change analyses more significance and help identify scenarios where LDN contributes to biodiversity, community livelihoods, and land productivity. Future work on WBLUs should seek to populate these LDN indicators and further establish whether this land-use is contributing to reversing land degradation. To do this, a working group between the Department of Forestry, Fisheries and Environment (DFFE) and the Department of Agriculture, Land Reform and Rural Development (DALRRD) should be established.

5.5.6 Conclusions and future work

African rangelands are under threat. For example, between 2000 and 2015, grassland in Botswana lost approximately 17% of its original extent (Akinyemi et al. 2021); in Mozambique, productivity was reduced by 18% between 2000 and 2016 (Montfort et al. 2021). In South Africa, the grassland biome experienced the highest rate of habitat loss of all inland biomes between 1990 and 2018 (Skowno et al. 2019, 2021). While there has been a general loss of grassiness, there has been a concomitant rise in woody plants (Skowno et al. 2017; Venter et al. 2020), further threatening to reduce grass productivity and undermine the socioeconomic value of rangelands. As such, Africa needs nature-based solutions that can simultaneously improve rangeland ecosystem functioning, reverse degradation, protect biodiversity, and provide rural jobs.

In this study, I assessed changes in vegetation productivity following the conversion of land-use from livestock and/or crop agriculture to wildlife-based land-uses (WBLUs), compared these trends to counterfactual land-uses over the same period and assessed the impacts of rewilding on revenue generation. Grassy productivity trends and biomass production are similar between WBLUs, protected areas and extensively managed livestock farms, which are all significantly higher than communally managed rangelands. However, WBLUs appear to have increased grassy productivity with lower rates of woody encroachment compared to livestock farms and are more resistant to woody productivity than livestock farms. Rewilding former livestock farms has led to increased levels of trophic diversity and the use of fire as a management tool, which both are significantly positive in predicting gains of grassy productivity and revenue generation. This ‘pyric-herbivory’ model is supported by the results of this study in restoring ecosystem functioning through trophic complexity and stochastic disturbance regimes, thus constituting an ecosystem-based approach to rewilding (Perino et al. 2019). WBLUs are more than twice as likely to employ fire management as commercial livestock farmers, and thus WBLUs embody two forms of interconnected rewilding – trophic complexity and stochastic disturbance. These results support previous studies finding positive links between rewilding and perennial grass biomass increases (for example, Parsons et al. 1997; Kraaij & Milton 2006; McDonald et al. 2020), but this is the first on a national scale.

These findings have several implications for policies seeking to employ the wildlife economy for sustainable development. Despite WBLUs existing in areas with the lowest average rainfall of all land-uses, grass productivity trends were the most significant or at least equal to rotationally grazed livestock systems. Given that climate change is predicted to make livestock farming less viable with increasingly arid conditions in Africa (for example, Rahimi et al. 2021), and WBLUs can provide higher employment rates and profitability than conventional agriculture (Taylor et al. 2020), can produce comparable amounts of legally produced meat (Taylor et al. 2020), combined with their biodiversity conservation contributions (De Vos & Cumming 2019; Taylor et al. 2021), mean that WBLUs are a proven model for resilient rural rejuvenation in Africa. The findings of this study build on this evidence base by demonstrating that enhanced grass productivity through greater degrees of rewilding (higher trophic diversity and use of fire) correlates with increased revenues and, thus, employment. Policy-makers should recognise WBLUs as a key land-use and facilitate investment to new market entrants to scale up these impacts on rangeland condition and rural development.

These results also suggest that different land-uses have complementary productivity functions that can be strategically interlinked to enhance the resilience of rangeland ecological infrastructure, such as through the creation of PES schemes for forage between commercial livestock farmers and WBLUs or between WBLUs and pastoralists. The comparative advantage of African rangelands is that both grazers and browsers are important components of the wildlife economy, unlike cattle farming, which relies on the grazing guild, meaning that total forage resource available to wildlife is greater and dynamics between woody and grassy productivity are more balanced than livestock farms. This is especially true of sustainable use models of WBLUs, which have significantly higher grass productivity trends. Thus, these results demonstrate that t the sustainable use of wildlife is sustainable on an ecosystem functioning level. Given the general bush encroachment occurring in rangelands, wildlife (especially browsers) will likely become increasingly viable over livestock.

However, more research remains to quantify the true impact and value of the browsing guild on reducing rates of woody plant encroachment (e.g., Guyton et al. 2020). An especially important line of future research is to assess the rate of change in profitability with woody increases on livestock farms and WBLUs. Additionally, the reintroduction of wild herbivores to degraded rangelands is a conservation success story in South Africa (Clements et al. 2018; Taylor et al. 2021). The relationship between rangeland condition and other facets of biodiversity is unclear (O’Connor et al. 2010). Future research should quantify the biodiversity co-benefits of rewilding rangelands, such as on insect diversity (Pryke et al. 2016) and threatened ecosystems (De Vos & Cumming 2019).

This study provides evidence on a national scale for the positive impacts of rewilding on rangeland productivity and enterprise profitability. Our results add to a growing evidence base showing the sustainability of wildlife-based agro-ecological systems while contributing significantly to socioeconomic development. Africa can rejuvenate rural economies from the bottom-up through indigenous ungulate rewilding to benefit both human well-being and conservation.

Supporting Information 5.1 Description and history of wildlife-based land-use economic activities

Wildlife-based land-uses in South Africa comprise multiple economic activities that are summarised in Taylor et al. (2015) and Table S5.7. Most enterprises (86%) conduct one or of these economic activities (Taylor et al. 2020), which gives rise to a spectrum of management objectives.

The surveys from Taylor et al. (2015) and Clements et al. (2016a) contain information on the date when each property converted to a WBLU and the former land-use. The conversion dates refer to when wildlife were reintroduced into the ecosystem and not the date of commercial lodge establishment. The median conversion date for WBLUs is 1999 (Figure S5.15). The major former land-use is livestock ranching, where 58% (N = 90 out of 155 properties) of WBLUs converted from straight livestock farming; 26% converted from livestock and crop farms; 5% converted from only crop farms; and 11% were effectively ‘always’ WBLUs because their conversion dates fell significantly outside of the analysis window (pre-1984). Thus, most properties (86%) converted from livestock and/or mixed livestock and crop farms.

WBLU managers were asked if they had noticed any improvements in rangeland condition following rewilding with native herbivore species. Most managers (N = 124 respondents) noted they had seen an improvement (71%), while 28% did not notice a discernible change and a small number observed mixed effects (6%) or negative effects (2%). Figure S5.16 shows a breakdown of the respondents’ observations. For respondents who noticed improvements, the most reported impact was increased grass cover (60%), followed by less erosion (20%) – presumably due to increased ground cover. Other positive impacts were related to the quality of the grass sward, including higher frequencies of palatable species and fewer alien invasive species. The respondents ascribed bush clearing, de-stocking, and the selective grazing of wildlife species as major reasons for rangeland improvements.

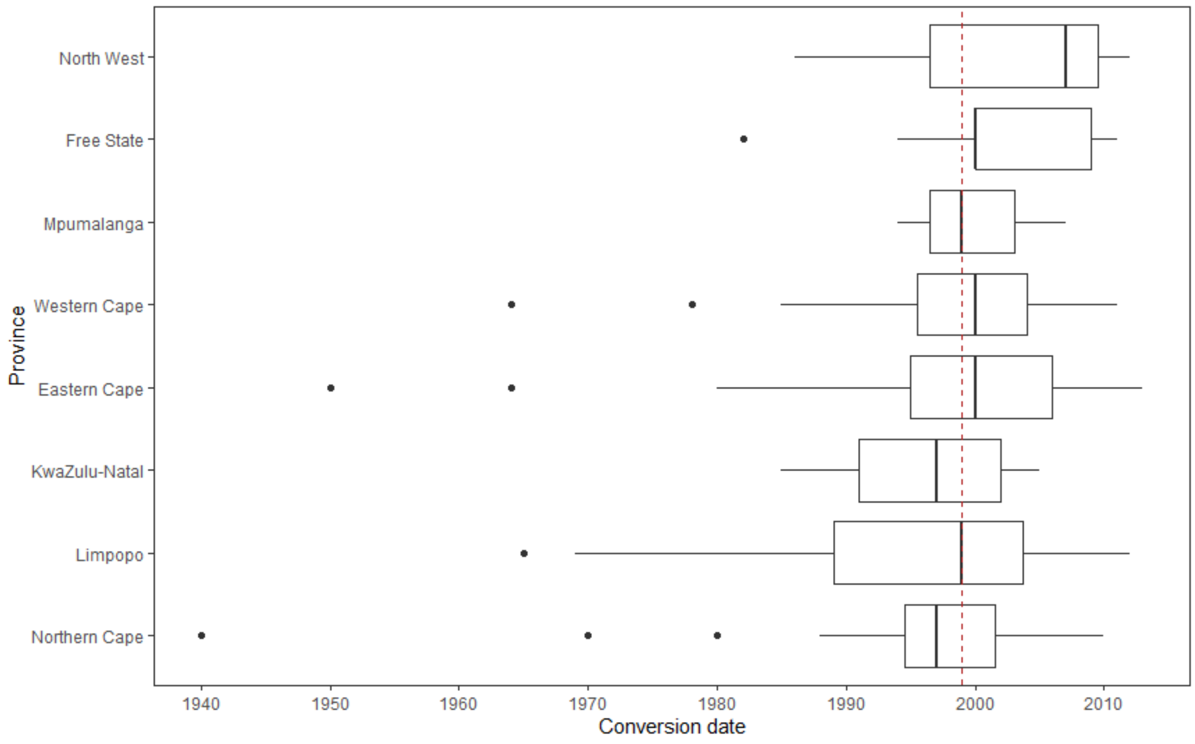

Figure S5.15. The median and interquartile range of conversion dates to wildlife-based land uses in each province. The Northern Cape and Limpopo provinces have the longest history of wildlife ranching based on this sample, whereas the Free State and North West provinces have the shortest. The median conversion date (dotted line) is 1999 across the country.

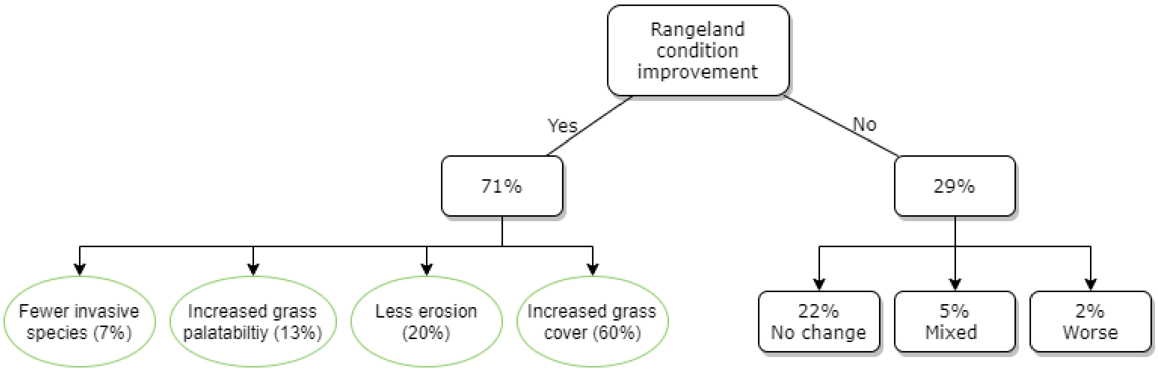

Figure S5.16. Schematic diagram summarising WBLU respondents' perceptions (N = 124) on rangeland condition improvement following conversion. Most respondents observed a positive improvement, where the most observed positive impact was increased grass cover. Of the respondents who observed no improvement, most said there was no change since conversion, while some noticed mixed impacts and a small amount observed worsening rangeland condition.

Table S5.7. Definitions of wildlife-based land uses (WBLUs) on private land in South Africa. Adapted from Taylor et al. (2020).

	Definition
Biltong hunting	The hunting of non-domesticated animals (invariably wild ungulate species), performed as a cultural activity by local hunters, normally using a rifle or bow, with the purpose of obtaining meat (Van der Merwe et al. 2014).
Ecotourism	Non-consumptive activities including photographic tourism, birdwatching, hiking and horseback riding.
Extensive management	Wildlife moves freely on a property within the borders of the perimeter fence and with minimal human interference. For the purposes of this paper, we did not set a minimum property size below which a property would not be counted as extensive, but rather included a property as extensive if management practices provided minimal supplemental food, veterinary care and protection from predation.
Game meat production	The process of culling either with the direct intention of producing meat or as a bi-product of game management. This invariably means wild ungulate species, but we excluded Ostrich and crocodile meat.
Intensive breeding	The confinement of wild species in small- to medium-sized enclosures (hereafter referred to as camps), where they are fenced in, protected from predators and provided with most of, or all their food, water and veterinary requirements. Although camps tend to be small (e.g. 10–100 ha), we did not set a maximum camp size to define intensive breeding. We assigned activities to the intensive breeding category when the landowners self-identified as intensive breeders and if the management activities described by them met the above definition (i.e. we crosschecked survey answers relating to use of camps, supplemental feeding and provision of veterinary care). Some landowners indicated that they preferred the term "semi-extensive breeding", which is an intermediate condition between extensive and intensive management, but which is difficult to define precisely. For the purposes of this paper, we lumped semi-extensive breeding with intensive breeding, but acknowledge that there are different degrees of intensive management.
Live game sales	The selling of live wildlife species through auctions or direct transactions between landowners, communities, conservation authorities and wildlife capture businesses (Bothma et al. 2010). Animals sold may be obtained from extensive or intensive systems.
Mixed farms	Commercial enterprises that comprise of a mix of wildlife, domestic livestock and crops. Mixed farms can run wildlife and livestock at the same time (either

	Definition
	overlapping on the same land or separated but on the same property), wildlife and crops, or all three together.
Selective breeding	The deliberate selection of individual animals of a wild species in an attempt to manipulate the genetic traits of their offspring in order to attain desired phenotypic characteristics, such as large body size or long horns. To achieve this, landowners generally use fenced camps to exclude animals lacking preferred traits and control which animals mate. Although intensive and selective breeding practises frequently occur together, they are separate management approaches. They generally involve high value species (e.g. African buffalo (*Syncerus caffer*), sable (*Hippotragus niger*) and roan (*Hippotragus equinus*)) or colour variants of plains game species (e.g. black impala (*Aepyceros melampus*) and golden wildebeest (*Connochaetes taurinus*)), with the aim of producing "superior" animals for live game sales or trophy hunting.
Trophy hunting	The selective hunting of individual non-domesticated animals (primarily mammals), picked for specific traits such as large horns, tusks or body size, and performed by paying clients using a rifle or bow in the presence of a professional hunter (Lindsey et al. 2007; Van der Merwe et al. 2014).
Wildlife ranching	The management of wildlife on private land for commercial purposes in the agricultural sector (may include any mix of the abovementioned land use types). The term is often used interchangeably with "game farming" or "game ranching".

Supporting Information 5.2 Relationship between perennial grass biomass and EVI

Estimates of vegetation productivity often conflate biomass production with forage quality (Seymour et al. 2010). To mitigate this, I used data from Child et al. (2013) and Venter et al. (2020) to test the relationship between EVI values and grass standing crop data collected by the Agricultural Research Council. Grass biomass data was collected in the Associated Private Nature Reserves (APNR) through disc pasture meters and was focused on the perennial grass component and thus are a good indicator of rangeland quality (Peel et al. 1999). Figure S5.17 shows that average EVI values over the period for which grass biomass data were available for each site are positively correlated and thus can be used as an adequate proxy for rangeland condition. However, there are some instances where EVI will over- or under-estimate relative grass biomass for a particular site, emphasising the need for remote sensing to complement but not replace field surveys. These results corroborate findings from the main section in that wildlife-based land-uses (WBLUs) exhibit rangeland condition between communal rangelands and formally protected areas. Figure S5.18 shows that trend in the residuals between EVI and rainfall for each WBLU in Child et al. (2013). There is a slight positive trend over the study period, perhaps indicating a general increase in rangeland condition in the APNR area.

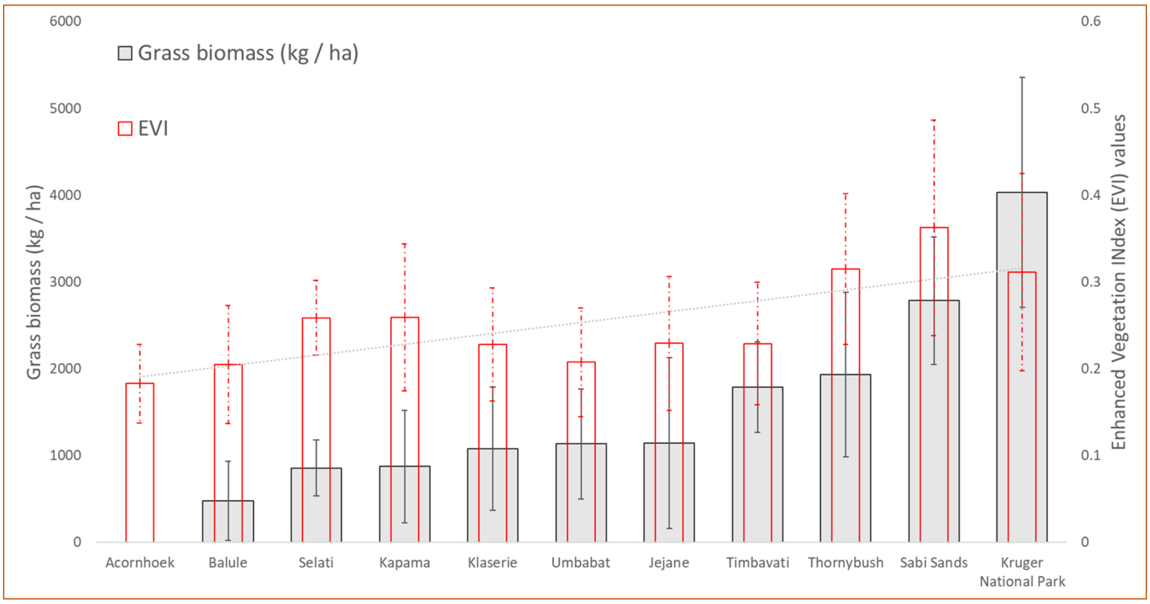

Figure S5.17. Relationship between EVI values and perennial grass biomass in the Greater Kruger area. Values represent averages over site-specific periods for which data were available. EVI generally tracks increases in grass biomass. Acornhoek refers to a communal rangeland in the study area.

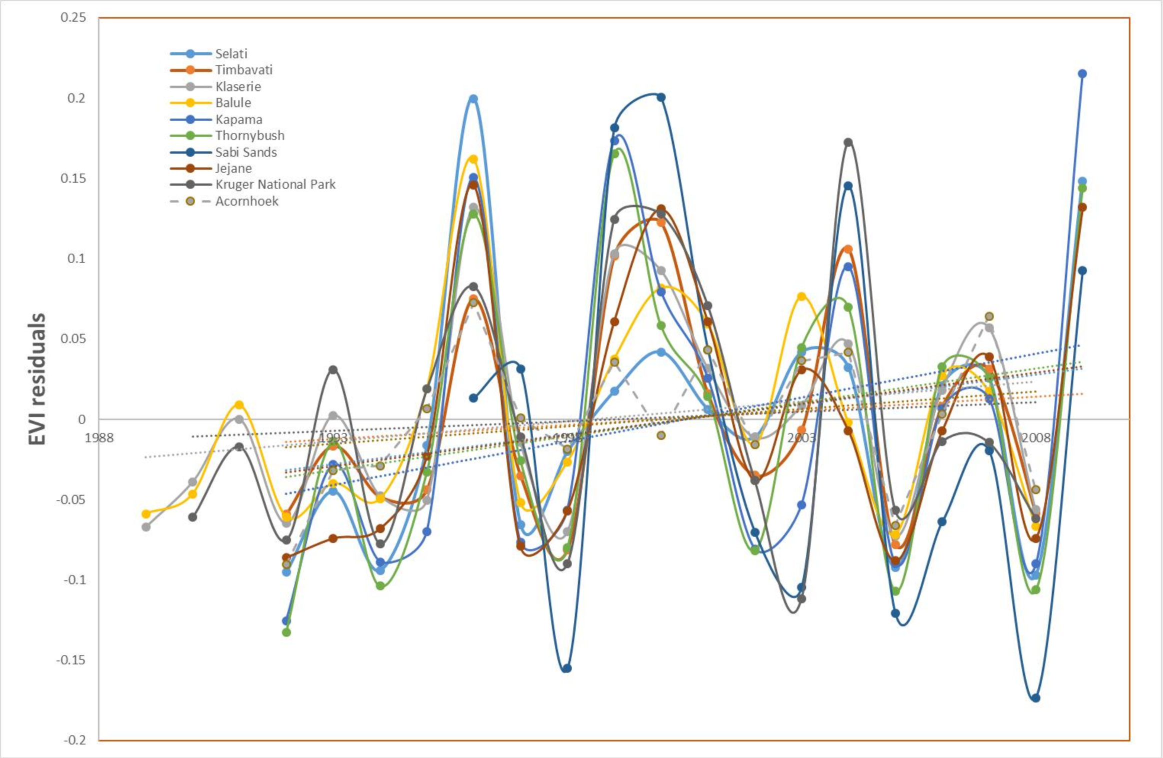

Figure S5.18. Temporal trend in the residuals between EVI and rainfall. There is a slight positive trend in primary productivity over the period recorded in Child et al. (2013) [1988-2008].

Supporting Information 5.3 Relationship between revenue and other socioeconomic factors

I assessed the potential for the most widely reported socioeconomic indicator – annual revenue – to be used as a proxy for broader socioeconomic impact. Direct use values were estimated by Taylor et al. (2020) and were derived from the actual use of an environmental resource, including consumptive and non-consumptive uses, involving the production and/or consumption of marketable products that can be measured through income and spending. These did not include indirect uses, option, bequest or non-use values. Profits were calculated as the difference between total annual revenue and total annual running costs, and return on investment was estimated by dividing operating profit by total property and wildlife asset values as a proxy for capital investment (Clements et al. 2016a; Taylor et al. 2020). Annual revenues included wildlife-related activities and livestock-related activities for mixed farms to assess whole-property economics. Values were converted from South African Rand (ZAR) to United States Dollars (USD) using exchange rates of USD 1: ZAR 13.7 in 2016. Revenue data from 2013 (Clements et al. 2016a) were adjusted to 2016 values using average annual consumer price indices[25]. Employment figures relate to permanent jobs provided by the WBLU and excluded temporary labour needed, for example, to construct game fences (Taylor et al. 2020). As such, it is a more stable indicator of social well-being.

I tested whether revenue could be considered a proxy of broader financial and social viability through linear regressions (Figure S5.19, Table S5.8). Revenue / ha was log-transformed to increase linearity. Model summaries were produced using the “Stargazer” package (Hlavac 2018). Revenue / ha is significantly positively correlated with all socioeconomic variables besides average monthly salary. These results suggest that the more widely recorded value of gross revenue serves as an adequate proxy of WBLU profitability.

Diagnostic plots were used to test the assumptions of linearity of the data, homoscedasticity of the residual variance, normality of the response variable, and presence of high-leverage outliers. The revenue ~ profit model violated assumptions of homoscedasticity and contained significantly influential outliers as identified by Cook’s distance (Bruce & Bruce 2017). The most significant outlier was removed, and the model was re-run using log-transformed response variable values to increase linearity. To avoid negative and zero values, the data were transformed through $\log(Y+a)$ where $\min(Y+a) = 1$. These transformations improved model diagnostics and increased the coefficient of determination ($R^2 = 0.39$) (Figure S5.20).

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[25] https://www.bls.gov/data/inflation_calculator.htm

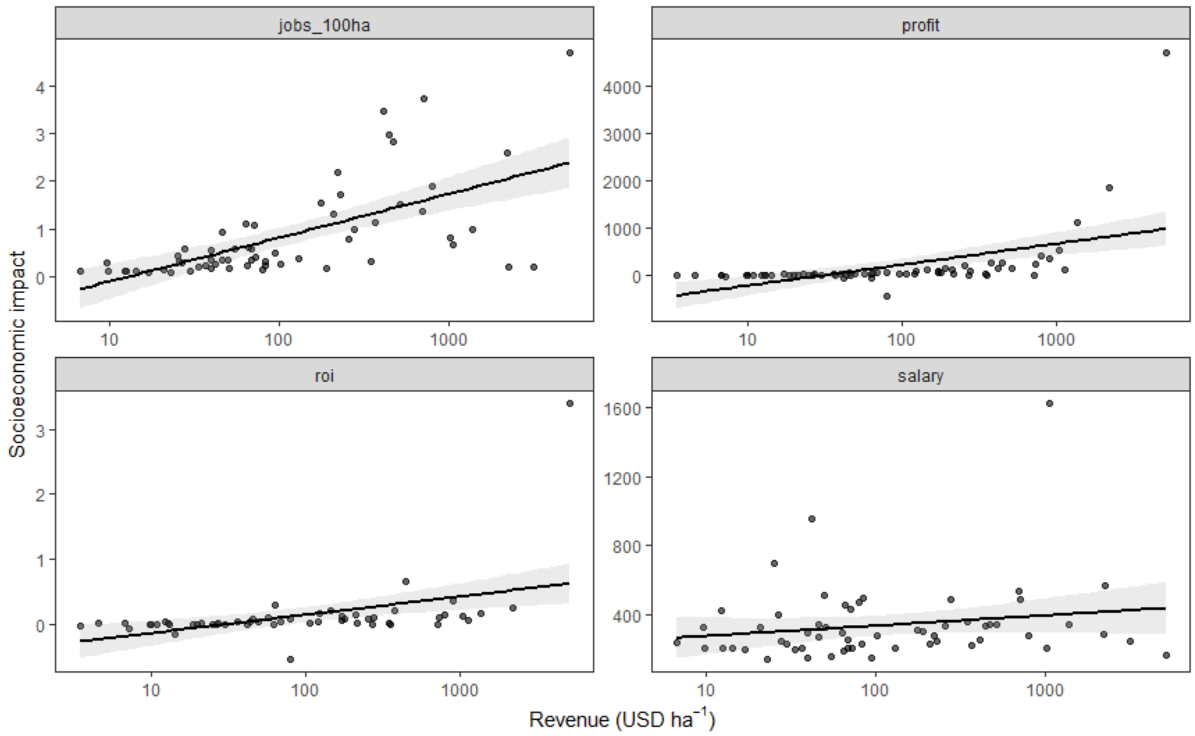

Figure S5.19. The relationship between revenue / ha and profit / ha (USD) (adjusted to 2016 values, “profit”), return on investment (“roi”), jobs / 100 ha (“jobs_100ha”) and average monthly salary (“salary”). Grey bands represent standard error around the regression line. Revenue is significantly positively correlated with all socioeconomic impact indicators besides average monthly salary.

Table S5.8. Results of linear regression models between revenue / ha and other socioeconomic indicators. Model outputs include intercepts (and standard errors), coefficients (and standard errors), F statistics and coefficients of determinations (R2). Significance values are indicated (*P* value; *α* at 0.05).

Model description	**Intercept (std. error)**	**Coefficient (std. error)**	**R-squared**	**Sample size**	**F statistic (df)**
Revenue ~ profit	4.1^{***} (0.18)	0.001^{***} (0.0003)	0.25	68	23.4^{***} (df = 1; 66)
Revenue ~ return on investment (ROI)	4.3^{***} (0.22)	1.6^{***} (0.45)	0.20	54	12.6^{***} (df = 1; 52)
Revenue ~ jobs / 100 ha	3.8^{***} (0.21)	0.9^{***} (0.16)	0.39	61	38.4^{***} (df = 1; 59)
Revenue ~ average monthly salary	4.3^{***} (0.37)	0.001 (0.001)	0.03	59	1.9 (df = 1; 57)

* $P = 0.05$, ** $P < 0.05$, *** $P < 0.01$

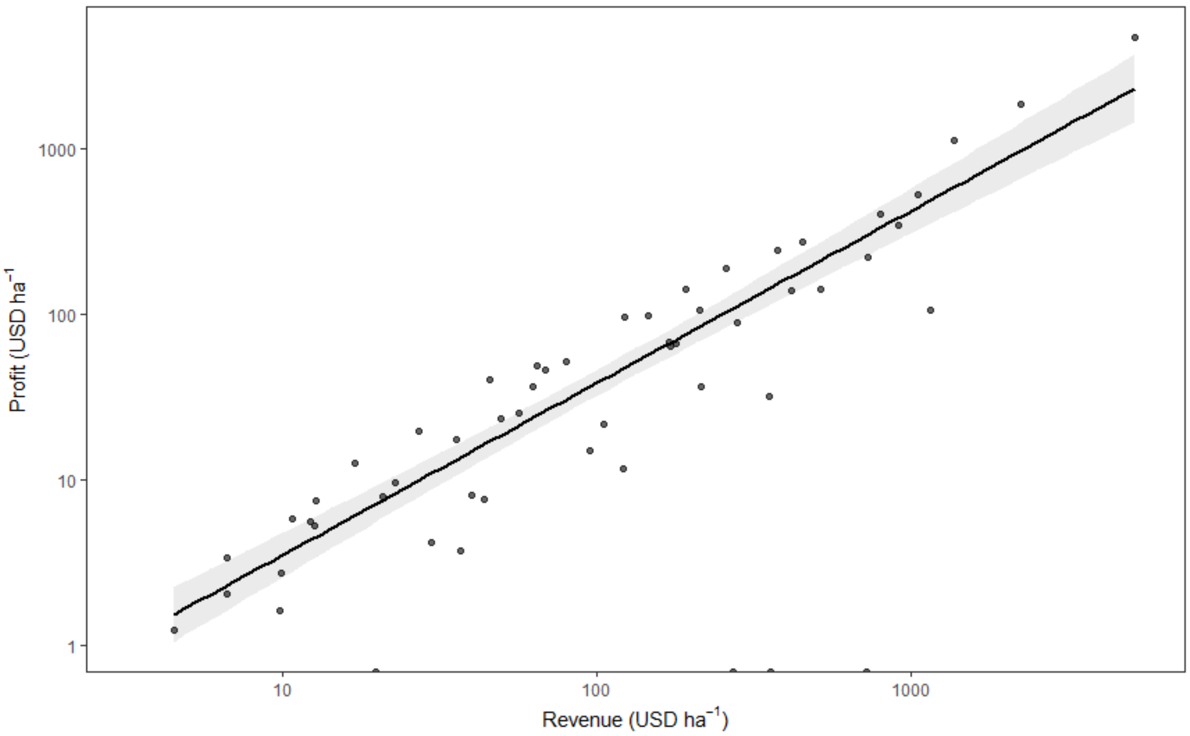

Figure S5.20. Significant positive relationship between log-transformed revenue / ha and profit / ha.

Supporting Information 5.4 Defining the predictor variables

Detailed management information for 281 WBLUs was captured by Taylor et al. (2015) during a survey to determine the economic value of the wildlife industry, as well as its ecological and social impacts. This dataset was supplemented by an additional 61 WBLUs from the Western and Eastern Cape provinces collected by Clements and Cumming (2017b). This survey similarly intended to quantify the economic and ecological viability of WBLUs and captures similar information on management interventions, such as supplementary feeding, management and monitoring plans, breeding camp descriptions, game counts and habitat restoration interventions. The data from the latter survey was cross-walked into the primary survey database.

Predictor variables were selected for their assumed direct (4 indicators) or indirect influence (3 indicators) on rangeland condition and restoration. Direct influence variables comprised ‘habitat restoration intensity’, ‘fire management’, ‘wild herbivore stocking rate management’ and 'mixed farming intensity'. Indirect indicators included 'management plans and monitoring', 'internal breeding camp area intensity', and 'artificial water-point density'. Each variable was converted into a continuous score of 0 to 1 (where 0 is least ecological management and 1 is the most) by calculating continuous variables as proportions and ordinal variables as reference values corresponding to assumed relative impacts on rangeland condition.

For 'Habitat restoration intensity', I considered three management interventions: bush clearing, erosion control and invasive alien plant (IAP) removal. Bush encroachment is defined as woody plant (tree or shrub) proliferation and expansion at the expense of grasses in savannah and grassland ecosystems (Hudak & Wessman 2001). The replacement of indigenous ungulates with domestic ungulates and subsequent overgrazing, together with fire suppression and elevated carbon dioxide levels, has caused bush encroachment across African rangelands (Smit 2004; McGranahan 2008; O'Connor et al. 2014; Hempson et al. 2017; Venter et al. 2018; Luvuno et al. 2018), lowering rangeland productivity and grazing capacity. Managers use a number of techniques to control bush encroachment, including mechanical removal, chemical treatment, the use of fire, sowing grass seed, and stocking higher proportions of browsers to grazers (Pienaar et al. 2017). Of the WBLU managers who provided information on bush encroachment management (N = 210 WBLUs), 51% reported they use various mechanical and sometimes chemical treatments to clear invading bushes. This is very similar to Pienaar et al. (2017), who reported that 57% of respondents used some form of bush encroachment control (Table S5.9).

Invasive alien plant (IAP) species are a serious and widespread threat to rangeland productivity and ecosystem functioning by displacing native vegetation and decreasing forage availability, increasing fire intensity and subsequent erosion, reducing surface water availability and flow, and increasing evaporation rates (Vilà et al. 2011; Chamier et al. 2012; van Wilgen et al. 2012; O'Connor & van Wilgen 2020). Control of IAPs is similar to bush encroachment and most managers use a combination of mechanical and chemical treatment, which are often more intensive due to the recolonization potential of IAPs (Pienaar et al. 2017). Fire and biological control agents are also sometimes used to curb the spread of IAPs. Reintroduction of indigenous wild herbivores has also been shown to mitigate IAP colonisation (Parker et al. 2006; Guyton et al. 2020). Of the WBLU managers who provided information on IAP management (N = 261 WBLUs), 64% reported they attempted to control IAPs, which is very similar to the findings of Pienaar et al. (2017), who found 71% of respondents reported controlling IAPs. This higher proportion of WBLU managers tackling IAPs rather than bush control may reflect the rapid rate of expansion of IAPs in South Africa (van Wilgen et al. 2008, 2012), combined with a shifting baseline of bush encroachment.

Erosion in South African rangeland has been attributed to former overgrazing and the planting of row crops with little ground cover (Pienaar et al. 2017). Efforts to control erosion include maintenance of infrastructure, such as roads and culverts, to rotational grazing management to allow vegetation to recover. Of the WBLU managers who provided information on erosion control and vegetation restoration (N = 265 WBLUs), 70% employed a variety of management erosion control actions, such as road maintenance, installing gabion structures and planting grass species in eroded areas. In the Pienaar et al. (2017) dataset, 46% of respondents employed some form of erosion control.

Fire management is critical in African rangelands, and interacts functionally with herbivory to sustain rangeland condition (Fuhlendorf & Engle 2004; Fuhlendorf et al. 2009; van Wilgen et al. 2011; Little et al. 2013; Everson & Everson 2016; Case & Staver 2017; Venter et al. 2017). Fire management was only used by 25% of respondents in Pienaar et al. (2017). Our dataset showed a similar proportion where some form of fire regime was employed by only 38% of WBLU managers (N = 276 WBLUs; Table S5.9). The predominant method was block burning (84%, total N = 76 WBLUs), but others employed a patch-mosaic approach (16%), defined here as the use of natural asymmetries in the landscape to define burn-lines. While we should not be constrained by historical fire regimes as the normative baseline (Case & Staver 2017; Freeman et al. 2017), misuse of fire can lead to trade-offs between socio-economics, biodiversity and rangeland condition (Archibald 2016; Docherty et al. 2020). The average fire return interval (defined as the average return times of fires to a particular spatial location) for Africa is 2 years (Archibald 2016), and the median interval for grassland and savannah ecosystems in southern Africa is 1.7 – 10 years, depending on rainfall and human impact (Archibald et al. 2010). As such, burning too frequently to stimulate forage production in commercial rangelands may ultimately undermine rangeland condition. Thus, I weighted fire regimes with irregular, ‘patch-mosaic’ or adaptive burn methods (Parr & Brockett 1999; Parr & Andersen 2006) more highly than annual block-burning regimes. However, fire management is a key ecological management method needing cooperation and engagement between WBLU managers and ecologists to determine the appropriate site-scale season, frequency and intensity of burning to sustain rangeland condition and biodiversity (Parr & Andersen 2006; Chown 2010; Little et al. 2013).

Overall, there is strong congruence between these data and the survey from Pienaar et al. (2017) in the proportion of WBLU managers employing various natural resource management techniques (Table S5.9). As these were independently conducted surveys, this suggests that these are robust estimates.

Table S5.9. The list of restoration interventions commonly employed for natural resource management in African rangelands. The proportion of WBLU managers employing the various interventions is similar between the current study and that of Pienaar et al. (2017), despite being independent surveys.

Habitat restoration intervention	**Current study % (N)**	**Pienaar et al. (2017) % (N)**
Bush encroachment control	51 (210)	57 (28)
Alien invasive plant control	64 (261)	71 (28)
Erosion control	70 (265)	46 (28)
Fire management	38 (276)	25 (28)

Stocking rates of herbivores have long been a contested issue in rangeland management. While the evidence is ambiguous around the relative importance of stocking rates in context of rainfall dynamics (equilibrium versus non-equilibrium paradigms), and there is no universally applicable grazing management system (Briske et al. 2008; Hawkins 2017; di Virgilio et al. 2019), most likely due to fine-scale environmental variables influencing the suitability of grazing system as well as the fine-scale adaptive responses of rangeland managers, it is clear that sustained heavy grazing in an area can lead to reduced grass biomass, bush encroachment and long-term loss of ecosystem services (Wessels et al. 2007a; O'Connor et al. 2014; Stevens et al. 2016; Venter et al. 2018). I define 'stocking density' as the density at which wildlife populations are kept through the management practices of landowners (number of animals per unit area) (Taylor et al. 2015), and 'stocking rate' as the number of Large Stock Units (LSUs) per hectare (Venter et al. 2019). An LSU is defined as an animal with a mass of 450 kg that gains 0.5 kg per day on forage with a digestible energy of 55% (Trollope et al. 1990). For the 'wild herbivore stocking rate management' indicator, I converted wildlife populations within each WBLU into LSU equivalents and calculated the relative stocking rate compared to long-term grazing recommendations for the particular ecosystem type (DAFF 2018; Venter et al.

2019). I excluded populations contained within breeding camps in the indicator because these populations do not influence the condition of the broader landscape. Rangeland degradation signals from greater levels of intensive breeding in breeding camps are incorporated by the 'proportion of internal breeding camp area' indicator (see below). However, as part of a sensitivity analysis, I we reran the regression analyses using an index that included encamped populations in the stocking rate indicator.

Herbivore species were converted to LSU equivalents using metabolic mass (coefficient 0.75) (Nagy 1987; Cumming & Cumming 2003; Müller et al. 2013). The average body masses of each species were taken from Bothma et al. (2010b), where body mass was multiplied by 0.75 to account for to account for intra-specific sex and age differences in biomass (Skinner & Chimimba 2005). Whether or not a species was a grazer, was also determined using Bothma et al. (2010b). I compared this method of calculating LSU equivalents to those based on energy requirements and diet digestibility (Meissner 1982; du Toit et al. 2013). The values corresponded well, with an average absolute difference of just 0.06±0.08 in LSU value for a sample of 14 game species.

Long-term grazing capacity values are defined as the area of land required (ha) to sustain a single LSU over time with degrading the vegetation or soil (DAFF 2018). Where WBLUs cut across multiple grazing capacity zones, the average value was taken. I calculated the difference between the actual stocking rate and the recommended government stocking rate (DAFF 2018) and regressed this against residual EVI values. The grazing capacity guidelines do not incorporate browsing capacity and thus may significantly underestimate available forage for wildlife communities. Additionally, tree crown cover was not incorporated and may influence grazing capacity (DAFF 2018). We included only species that are predominantly grazers (including those classified as mixed feeders). Four mixed feeder species were included as grazers, namely eland (*Taurotragus oryx*), African elephant (*Loxodonta africana*), impala (*Aepyceros melampus*) and springbok (*Antidorcas marsupialis*). As I did not have consistent information available on cattle stock numbers in mixed WBLUs, I omitted these from the analysis unless it was clear that the land area referred to the wildlife portion only.

Of 251 WBLU with data to calculate stocking rates, 51% (N = 130) had higher stocking rates than the recommended grazing guidelines. Clements and Cumming (2017b) found that a quarter of WBLUs were overstocked by >50% and that increasing management intensity increased stocking rate, and warned that such overstocking might lead to long-term impacts on the vegetation. However, on average, when looking at the absolute differences between current and recommended stocking rates, our sample of WBLUs had 56 ± 12% (mean ± standard error) lower stocking rates than the long-term grazing guidelines in the extensive areas (Figure S5.21). However, because we did not have data on the cattle stock number for mixed WBLUs, these results could be confounded. Using just the wildlife-only properties (N = 125), the pattern held, however, where WBLUs exhibited 16 ±17% (mean ± standard error) lower stocking rates than the long-term grazing guidelines in the extensive areas (Figure S5.22).

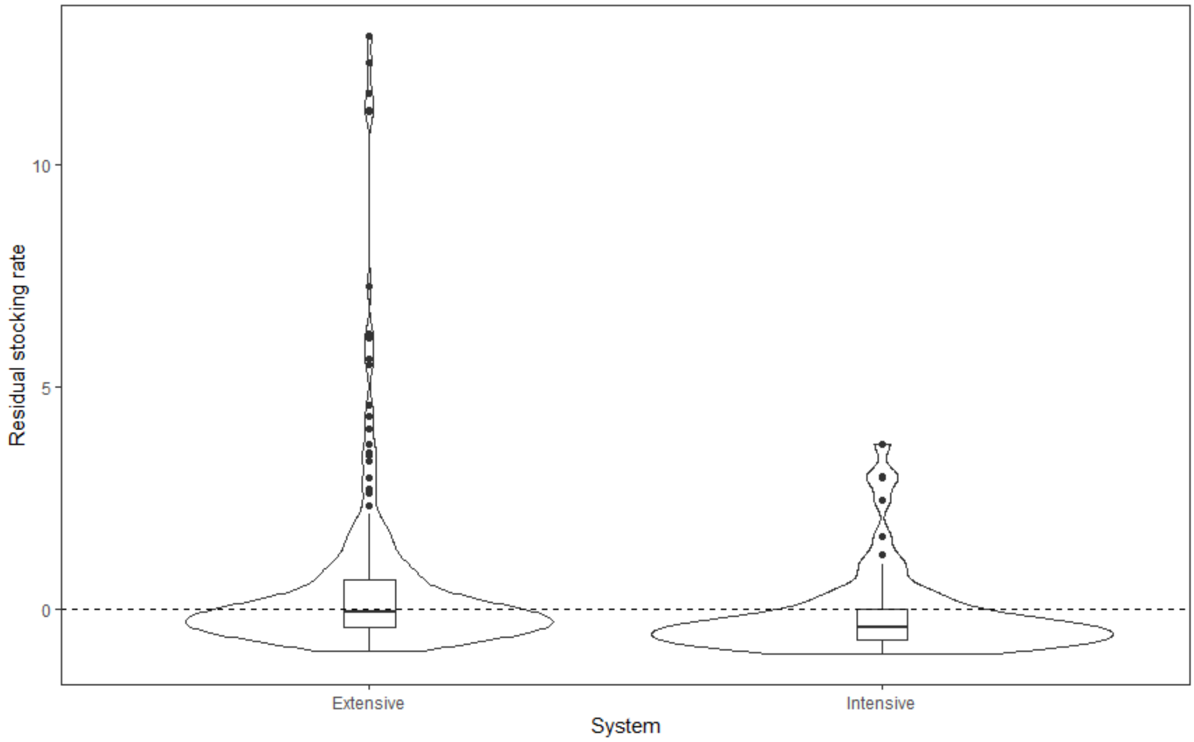

Figure S5.21. Difference in residual stocking rate between extensive and intensive areas of wildlife-based land-uses (for all WBLUs, N = 251). Solid lines represent median values, and boxes are the interquartile ranges. Violin plots show the kernel probability density of the data at different values. Positive values indicate lower than recommended stocking rates (i.e. lower relative grazing densities). The dotted line shows the value at which the actual stocking rate is equal to the recommended stocking rate.

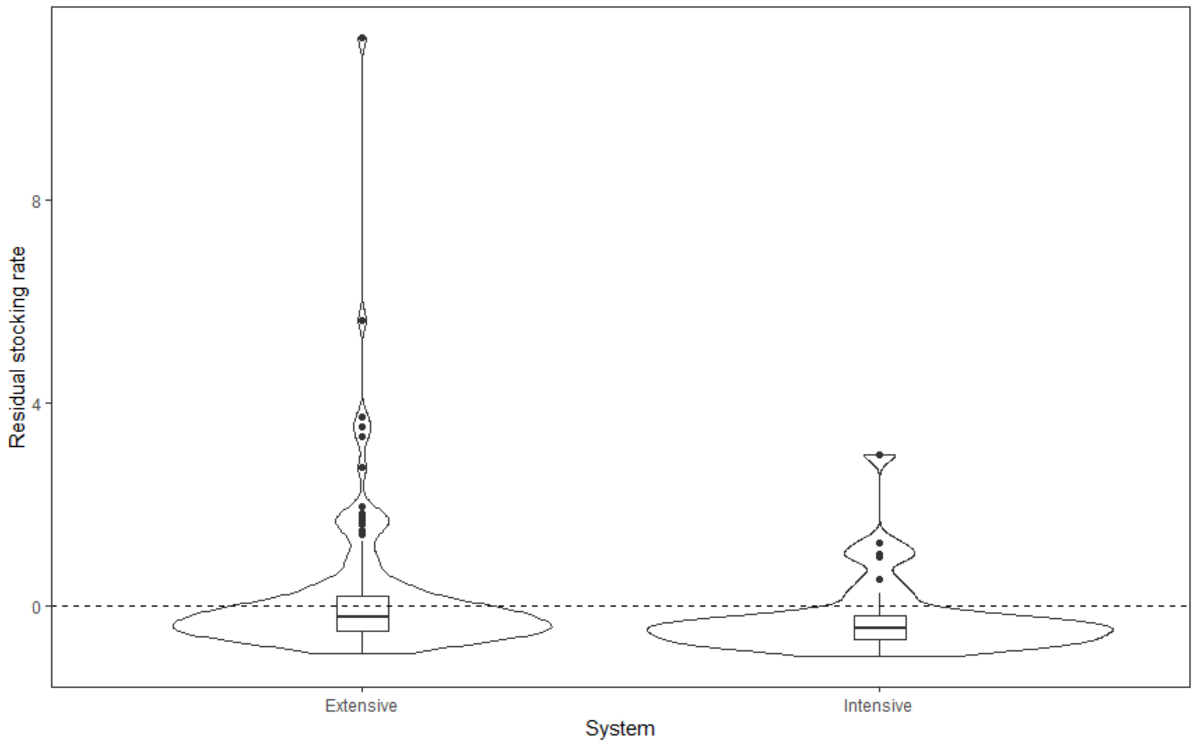

Figure S5.22. Difference in residual stocking rate between extensive and intensive areas of wildlife-based land-uses (for wildlife-only WBLUs, *N* = 164). Solid lines represent median values, and boxes are the interquartile ranges. Violin plots show the kernel probability density of the data at different values. Positive values indicate lower than recommended stocking rates (i.e. lower relative grazing densities). The dotted line shows the value at which the actual stocking rate is equal to the recommended stocking rate. While encamped areas are likely to have higher relative stocking rates than extensive areas, the differences are not significant.

What is clear is that stocking rates on WBLUs are lower than commercial extensive livestock farms. Stocking rates on the commercial livestock farm dataset (N= 48 farms) were, on average, 59 ± 12% (mean ± standard error) higher than those recommended by extension services (Venter et al. 2019). This supports the findings of Pienaar et al. (2017), who found that 87% of managers attempted to control overgrazing by lowering stocking rates. Our results on a national scale suggest a lower overgrazing impact of WBLUs in the extensive areas, as only 18% of WBLUs were estimated to exceed long-term grazing capacity guidelines by >50%. I did not find evidence for stocking rate being correlated with impacts on vegetation productivity or revenue (model results in main section). However, this analysis may mask long-term impacts of stocking rate on vegetation composition and biodiversity, which should be evaluated through field studies. Overall, these results support the supposition that the long-term grazing capacity guidelines may not be suitable for wildlife because wild herbivores comprise grazers, browsers and mixed feeders across different body-mass scales (Cumming & Cumming 2003), and thus may utilise available forage more efficiently even at higher overall stocking rates.

Additionally, I also calculated a trophic diversity score as the proportion of total farm LSUs that constituted grazers, browsers, mixed feeders, megaherbivores (Bothma & du Toit 2010) on each property to assess whether more functionally diverse WBLUs have greater rangeland condition values (*sensu* Venter et al. 2019). I split each trophic guild into sub-guilds, such as selective grazers on short grass (e.g., oribi *Ourebia ourebi*) versus selective grazers on long grass (e.g., sable antelope *Hippotragus niger*). We included predators in this index as top down control of herbivore populations through predation, and 'landscapes of fear' create habitat heterogeneity and are predicted to improve vegetation productivity (reviewed in Child et al. 2019). I excluded megaherbivore populations that are kept in enclosures, breeding camps, or bomas as they do not functionally interact with the broader landscape. We defined megaherbivores as elephant, both rhino species (*Ceratotherium* and *Diceros* sp.) and giraffe (*Giraffe camelopardarlis*) (Sitters et al. 2020). Only populations greater than the minimum population size were included (Hilbers et al. 2017), as only functionally self-sustaining populations are predicted to exert their ecological roles (reviewed in Child et al. 2019) [see Chapter 3]. The score was calculated as the percentage of guilds present on each property where a score of 1 means all guilds present.

For 'internal breeding camp area intensity', I calculated two indicators: firstly, the relative stocking rate of the internal breeding camp area, and secondly, the proportion of the WBLU area occupied by internal breeding camps. I defined breeding camps as the confinement of wild species in small to medium sized internal camps or enclosures, where they are fenced in, protected from predators and provided with most of, or all their food, water and veterinary requirements (Taylor et al. 2015; see also Child et al. 2019). Such populations are 'inert' in that they do not influence the broader landscape but may cumulatively contribute to landscape change in the same way that kraals do. The use of breeding camps might be a way for WBLU managers to 'offset' lower stocking rates in the extensive areas by increasing production in the breeding camps. However, breeding camp stocking rates were only slightly higher on average than the extensive areas, exhibiting 12 ± 25% (mean ± standard error) lower stocking rates than the long-term grazing guidelines. The differences between residual stocking rate between extensive and intensive areas were tested using a paired t-test using wildlife-only WBLUs where there was both an extensive and camp system. The differences were insignificant (t =

1.16, df = 34, $p = 0.25$). The proportion of breeding camp area to total ranch area was used as a proxy of landscape fragmentation as most have electrified game fencing that can cause significant damage to biodiversity (Taylor et al. 2015).

For ‘Mixed farming intensity’, the impacts of livestock and smallstock ranching are highly variable depending on the suitability of the management system and environment (Mysterud 2006; Anderson & Hoffman 2007; Venter et al. 2019; Bailey et al. 2019; di Virgilio et al. 2019). Overall, 40% of the WBLUs were mixed farms, of which 96% contained livestock and 41% contained livestock and crops. Mixed livestock and wildlife farms can improve rangeland condition (e.g., Keesing et al. 2018; Sitters et al. 2020).

There were an additional four indirect variables that might potentially influence rangeland condition. The first of these was the management system of the WBLU. For the ‘management plans and monitoring’ indicator, I incorporated the degree to which WBLU management was based on ecological management plans and regular vegetation monitoring. I did not include wildlife population monitoring, as 93% of 198 WBLUs responded they monitor wildlife numbers, so this was trivial. The quality of population monitoring may vary, but there was no information to discern these differences. However, vegetation monitoring, defined here as fixed transect or repeat photography surveys, is arguably a more important indicator of the capacity to detect changes in rangeland condition. For example, Clements and Cumming (2017b) use vegetation monitoring as a proxy for long-term adaptive management capacity of WBLUs. For the vegetation monitoring component, I only included formal and repeat vegetation monitoring programmes, not those based on informal observation or once-off assessments alone. Overall, only 32% of 263 WBLUs reported positively. This figure falls in between previous estimates of 23% (Du Toit 2000) and 43% (Pienaar et al. 2017) of respondent samples who say they do not conduct formal vegetation monitoring, relying instead on personal observation to assess rangeland condition and quality. In the Western and Eastern Cape provinces, 53% of WBLUs have never assessed habitat condition (Clements & Cumming 2017b).

For the management plan component, I defined management plans as any formal strategy for managing habitats and wildlife numbers to achieve specific management goals. Overall, 60% of 268 WBLUs reported they had developed management plans, which is substantially higher than the 36% reported by Pienaar et al. (2017). The management plans were most commonly called ‘ecological’ or ‘environmental’ plans. Of the WBLUs with monitoring plans (N = 162

WBLUs), 53 (33%) wrote the plans themselves, while the rest utilised a range of expert input from specialist managers to external advisors and consultants. Overall, the high number of WBLUs that either do not have a management plan or do not seek specialist input to produce one (59%) may highlight a gap needing to be filled by extension services and public-private partnerships (Cousins et al. 2008).

Finally, increasing the density of artificial water points (AWPs) can increase habitat degradation, both directly around the water-points ('piospheres') (Gaylard et al. 2003; Smit et al. 2007, 2020; Schmidt et al. 2019), but also indirectly by increasing population abundance unsustainably, leading to population crashes when the AWP fails (Walker et al. 1987; Selebatso et al. 2018). Additionally, increasing the extent of AWP coverage and reducing inter-point distance can homogenise herbivory pressure leading to landscape-level impacts and loss of 'refugia' where grass biomass can recover and rare species can persist (Grant & Van der Walt 2000; Sianga et al. 2017). While guidelines do exist for minimum distances between AWPs (Owen-Smith 1996; Sianga et al. 2017), I did not have information on the configuration of AWPs to measure this factor. Rather, I measured the 'artificial water-point density' indicator as the proportion of water-points or water-sources that are artificial as opposed to naturally occurring in the landscape (rivers, vleis, wetlands) as a proxy for the relative seasonal availability of water, and the density of AWPs in the landscape as a proxy for spatial refuge potential.

The list of predictor variables is summarised in Table S5.10, along with brief justification for their impacts and methods used to construct the indicator score. These variables were used in regression models (described in main text) and also combined into an Ecological Management Index (EMI) as the sum of the standardised variable scores. Properties with missing data for ≥50% of the variables were excluded from the index computation. The EMI differs from the 'rewilding framework' developed by Torres et al. (2018) through the normative values of human participation in the ecosystem and the scale at which the scores are relevant. While the rewilding framework is designed to minimise human input and output in the landscape, the EMI is explicitly based on managing human input and output of the system for sustainable socio-economic activity rather than restricting human activity *de facto*. [See Child et al. (2019) for discussion on ensuring the wildness of these managed systems]. The rewilding framework is more suited to broader landscapes – areas larger than specific properties as it incorporates major land-use activities, such as mining, fishing and hunting (Torres et al. 2018), which are unlikely to occur simultaneously on the sites where specific socioeconomic policies would be applicable. Most importantly, we need to understand whether ecological management results in better revenues in the long-term, which would mean that investing in natural capital and biodiversity makes business sense. The relationship between the rangeland management score and revenue / ha (N = 74 properties) is shown in Figure S5.23, where there is weakly positive relationship ($R^2 = 0.07$).

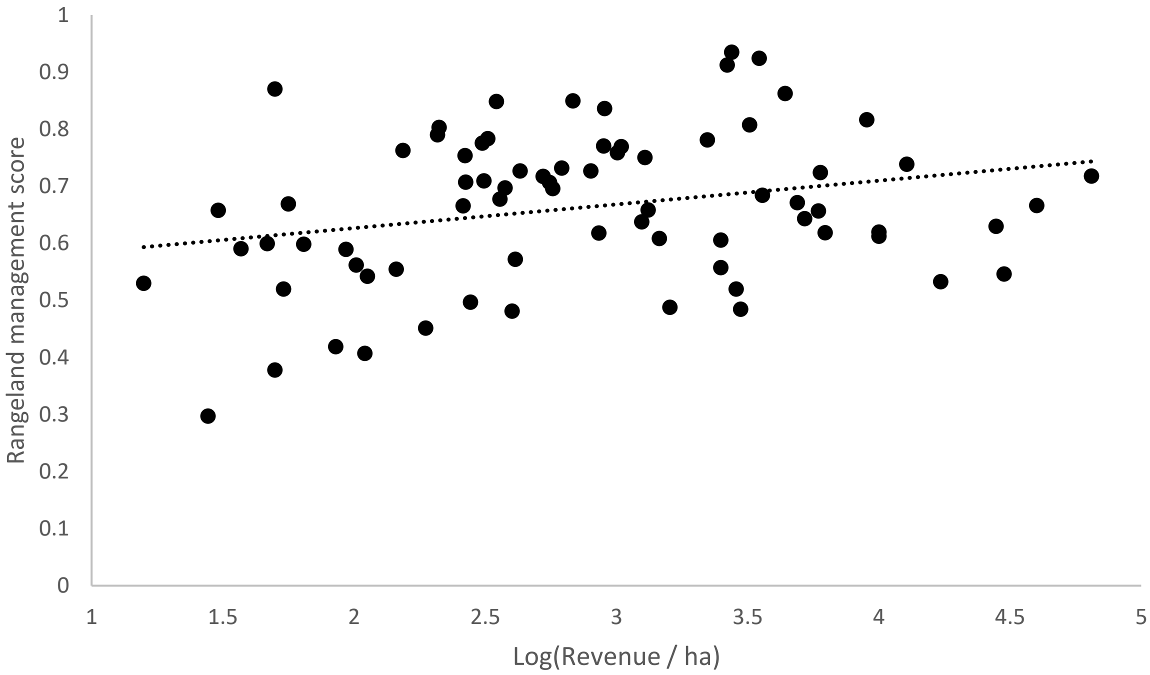

Figure S5.23. The relationship between a rangeland management index, integrating variables that will lead to potentially better ecosystem functioning and restoration of biodiversity, and the revenue generation of wildlife-based land-uses. While there is a positive relationship, it is insignificant. Future research should repeat this analysis on a larger sample.

Table S5.10. The variable list of management actions and attributes used to construct the Ecological Management Index and as independent predictor variables in the Generalised Linear Mixed Models.

Variable name	**Rationale for inclusion**	**Indicator score calculation**
Direct impacts		
Habitat restoration intensity	Management activities intended to reverse the loss of rangeland productivity from woody plant encroachment and former overgrazing should improve indices of rangeland condition over time. The activities included here are bush clearing to curb bush encroachment, erosion control, and removal of alien invasive plants (Taylor et al. 2015). See Pienaar et al. (2017) for an explanation and discussion of how the above restoration activities are conducted on WBLUs in South Africa. These actions are thought to improve rangeland condition, ecosystem functioning and enterprise profitability.	This indicator was calculated as the presence or absence of specific habitat restoration interventions employed by WBLU managers where: $HR = \frac{\sum(RestAction_{1....n})}{RestAction_n}$ *RestAction* interventions were classified as bush clearing to reduce bush encroachment, erosion control and alien invasive clearing*.
Fire management	Fire is a key ecosystem process that restores asymmetric disturbance patterns across the landscape, creating a dynamic matrix of differently aged patches that improves habitat suitability for a wider range of species and acts to increase grass production and reduce bush encroachment and alien invasive colonisation (McGranahan 2008; Fuhlendorf et al. 2009; references in Torres et al. 2018). Controlled burning is necessary to stimulate forage production and prevent woody encroachment. Fire management interacts with herbivory as two driving forces in savannah ecosystems. Fire regime is defined as the frequency, seasonality, intensity, severity, fuel consumption, and spread patterns of fire characteristic of a certain region (reviewed in Archibald et al. 2010). This variable is suspected to have a non-linear relationship with rangeland condition where we assume the absence of fire is worse than minor differences in frequency and burning method. As the median fire return internal for the region is	This indicator was calculated as the average between two ordinal variables pertaining to fire management frequency and method where: $FM = \frac{(FM_{freq} + FM_{meth})}{2}$ Where no fire management = 0 and fire return internal [FM_{freq}] = 0.5 – annual burning; and 1 – fire return intervals > 1 year.* And burning method [FM_{meth}] = 1 – 'patch mosaic'; and 0.75 – block burning *Ultimately, threshold calculations should be spatially-explicit and compared to a reference database on natural dynamics range (*sensu* Archibald et al. 2010).

Variable name	Rationale for inclusion	Indicator score calculation
	a minimum of 1.7 years, annual burning scored lower than longer fire return intervals.	
Wild herbivore stocking rate management	Overstocking properties can lead to habitat degradation through overgrazing and the subsequent loss of palatable plant species and an increase in woody plant encroachment (Wessels et al. 2007a; Teague et al. 2011; Briske et al. 2011; Scott-Shaw & Morris 2015; Hawkins 2017). While wildlife is hypothesised to have lower impacts of habitats due to a diverse feeding guild structure and range of adaptations that commercial livestock to not have (Hempson et al. 2017), overstocking properties with Large Stock Unit (LUS) equivalents may still cause degradation. To calculate farm stocking rates (LSU ha^{-1} yr^{-1}), native herbivore species were converted into LSU equivalents using metabolic mass (coefficient 0.75) (Nagy 1987; Cumming & Cumming 2003; Müller et al. 2013), where body mass was first adjusted for age-structure by multiplying by 0.75 (e.g. Clements & Cumming 2017b). This indicator does not include stocking rates of domestic livestock on mixed farms, as this information was not consistently available. However, the relative land-areas dedicated to livestock farming are incorporated under 'Mixed farming intensity'.	This indicator was calculated on a continuous scale by converting wildlife populations to LSU equivalents. $LSU_{WBLU} = \sum_{n} (BM_i \times 0.75)^{0.75}$ Where BM_i is the average body mass of species *i* Total LSUs on each property were divided by property size to get actual ha / LSU. Relative stocking rates were then calculated as the percentage difference between the actual (*Ra*) and recommended (*Rb*) stocking rates (Venter et al. 2019), $R_{SR} = \frac{Ra-Rb}{Rb} * 100$ where *Rb* was derived from the national Long Term Grazing Capacity norms (DAFF 2018).
Mixed farming intensity	Many wildlife ranches are mixed farms (Du Toit 2000; Taylor et al. 2015), exhibiting a combination of wildlife with either crop or stock farming. The extent and type of agricultural activity may negatively impact rangeland condition through a combination of factors ranging from fire suppression, overgrazing, habitat loss for crop planting and soil erosion from loss of ground cover (Wessels et al. 2007a; O'Connor et al. 2014; Venter et al. 2017, 2018). To compensate for	Mixed farming is calculated as a continuous variable where: $MF = 1 - \left[\sum Agric_{1\ldots n} \times \%\, agric\right]$

Variable name	**Rationale for inclusion**	**Indicator score calculation**
	the potentially positive or neutral effects of domestic herbivore grazing on rangeland condition, we weighted the land-area proportion by 0.5. We assumed the net impacts of livestock and smallstock (sheep and goats) on rangeland condition were potentially similar in magnitude (du Toit et al. 2009, 2018) when practiced separately but additive when practiced simultaneously on the same WBLU (constituting 8% of properties). We assumed crop farming to cause the most severe impacts through outright habitat loss and loss and ground cover and so did not weight the land-area proportion.	Where %agric is the proportion of land area* covered by the agricultural land use; $Agric$ = 0.5 – extensive livestock farming (alone); 0.5– goats and sheep farming (alone); 0.75 (where both livestock and smallstock are present); 1 – crop farming. *As these activities occurred in different parts of the farm, they were spatially zero-sum and thus were not averaged. Where livestock and smallstock co-occurred, they occupied the same proportion of land area.
Indirect impacts		
Management plan and monitoring	A management plan is essential for keeping track of ranch operations, optimising production, and ensuring sustainable ecosystem functioning. However, managers often do not produce management plans or produce plans themselves to save costs (Pienaar et al. 2017). Similarly, ecological monitoring is correlated with lower management intensity and, thus, adaptive responses to natural resource management problems (Clements & Cumming 2017b). This indicator assumes that landowners who have developed an ecological or biodiversity management plan and implement regular habitat monitoring will likely exhibit better rangeland condition through adaptive management responses. We use this indicator as a proxy for adaptive management capacity.	This indicator was calculated on an ordinal scale $AdapMgmt = \frac{(M_{plan} + V_{mon})}{2}$ Where M_{plan} = 0 where no management plan was present; 0.5 for management plans developed by the landowner; 1 for management plans developed by specialists (in-house or consultants). Where V_{plan} = 0 for no vegetation / habitat monitoring; = 1 for regular vegetation / habitat monitoring (once-off surveys counted as 0)
internal breeding camp area intensity’	Increasing landscape connectivity, both within a property and between properties, is critical for sustaining the dispersal capacities of species and enhancing ecosystem functioning. Breeding camps may significantly fragment landscapes and create hotspots of habitat degradation. Breeding camps may also exhibit higher	This indicator was calculated as the relative stocking rate of the total breeding camp area (see ‘Wild herbivore stocking rate management’ above).

Variable name	Rationale for inclusion	Indicator score calculation
	stocking rates than the extensive areas as managers boost production in certain parts of the landscape (Clements & Cumming 2017b).	Additionally, the proportion of the landscape covered by breeding camps / enclosures across species was calculated as: $LF = 1 - (BreedC\%)$
Artificial water-point density	Natural hydrological regimes are important drivers of herbivore movements across the landscape (Derry & Dougill 2008), where reducing inter-point distance can reduce nutrient cycling and increase impact on vegetation, as well as reducing the overall diversity in the ecosystem (Sianga et al. 2017). Reducing artificial surface water availability was the main management intervention employed by Kruger National Park to halt declining populations of rare antelope species (Grant & Van der Walt 2000).	This indicator was measured on a continuous scale: $AWP = 1 - \left[\frac{(\mathrm{AWP}_{seasonality}) + \mathrm{AWP}_{den.})}{2}\right]$ Where $\mathrm{AWP}_{\mathrm{seasonality}}$ = the proportion of permanent water-points that are artificial as a proxy for seasonal availability of water (assuming non-artificial water points will be ephemeral or fluctuate) And $\mathrm{AWP}_{\mathrm{density}}$ = density of artificial water-points in the landscape as a proxy for spatial refuge potential.

Supporting Information 5.5 WBLU sub-model productivity patterns

I assessed the differences in vegetation productivity between three broadly different WBLU models, which earn revenue from ecotourism only (E), consumptive-use only (U) or enterprises that combine both (M). We assessed the difference in relative grassy and woody productivity trends since conversion to WBLU. The EVI productivity trend values were normally distributed (Figure S5.24), so I applied a one-way analysis of variance to test differences in mean productivity trends between broad business models. The model found that the main effect of WBLU model was significant (ANOVA: $F_{2, 848} = 8.8$, $p < 0.01$; Figure S5.25), and a post-hoc Tukey test confirmed this effect to be between the ecotourism (E) and use models (M and U) ($p < 0.05$) but not between the use models ($p = 0.24$). For woody productivity trends, no significant differences were detected between models (ANOVA: $F_{2, 156} = 0.1$, $p = 0.91$; Figure S5.25). Additionally, trophy hunting operations are purported to have lower carbon emissions than ecotourism lodges as more revenue can be generated from fewer tourists and per capita energy use may be lower and infrastructural requirements are fewer (Di Minin et al. 2016) [and see *Supporting Information 5.8*].

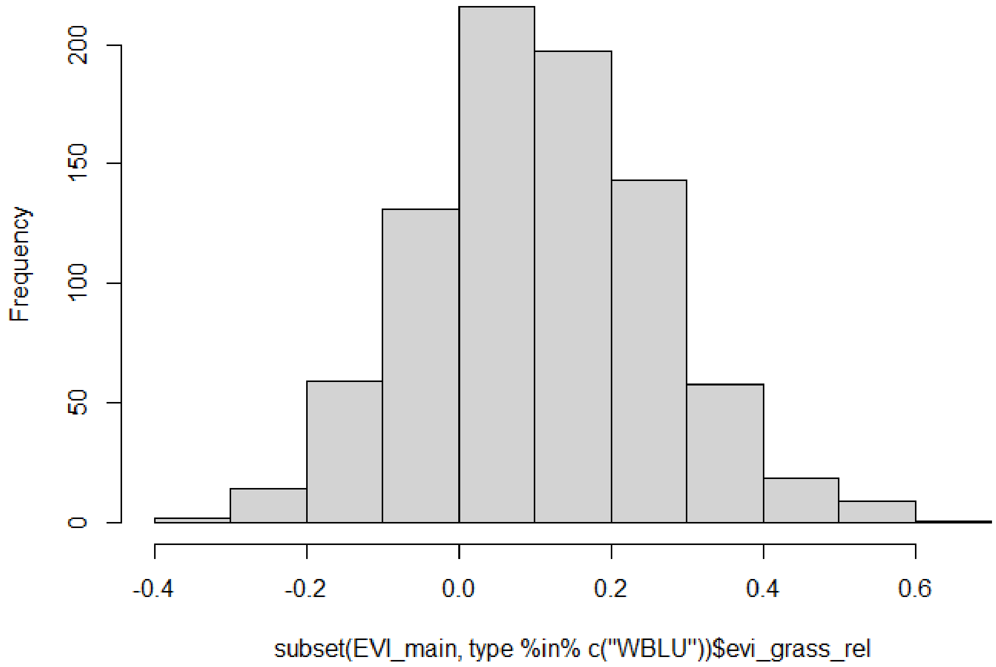

Figure S5.24.Histogram of relativised residual grass productivity trends in the wildlife-based land-use sample, showing the data are normally distributed.

As the data were rescaled to account for inter-biome differences in productivity (essentially comprising relative 'ranks'), this might have confounded the results. Thus, I used the non-parametric Kruskal-Wallis test to assess mean productivity trend differences. This corroborated the ANOVA in that a significant difference was found between models (KW chi-squared = 20.8, df = 2, p < 0.01). A pairwise comparison using a Wilcoxon rank sum test revealed the difference to be between ecotourism-only and the mixed-use and use only models (p < 0.01) but not between the mixed use and use-only models (p = 0.16). Similarly, a KW test found no significant differences in woody vegetation trends between the WBLU models (KW chi-squared = 0.09, df = 2, p = 0.96).

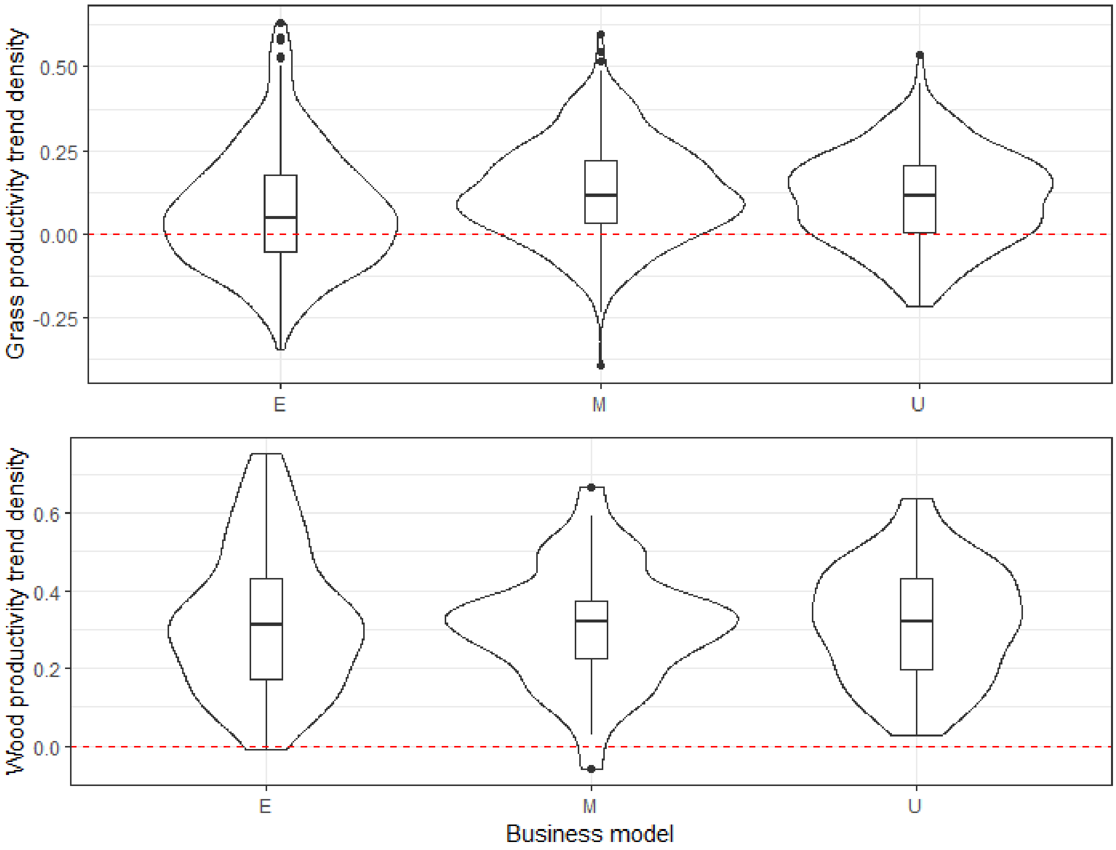

Figure S5.25. Residual trends in grass and wood productivity were assessed between three broadly different business model, corresponding to 'ecotourism-only' (E), mixed ecotourism and consumptive use activities (M) and use-only properties that only conducted hunting and/or game breeding (U). The sustainable-use models (M) and (U) have significantly higher median grass productivity trends than ecotourism-dominated properties. There is no significant difference in wood productivity trends.

Supporting Information 5.6 Vegetation productivity trends across land-uses

Vegetation productivity trends were compared between land-uses. Figure S5.26 shows the non-relativised and smoothed trends (using a linear model). Despite generally lower and declining rainfall trends, wildlife grass productivity was similar to extensively managed livestock farms and more positive than communal rangelands. Woody productivity was initially lowest in livestock farms but has gained most rapidly over time. When accounting for rainfall using residual trend analysis, mean residual trends between 1999 and 2014 were significantly different for grassy productivity (ANOVA test: $F_{(3)} = 45.8$, $p < 0.01$), with a post hoc test revealing significant differences between all land-use comparisons besides livestock and wildlife-based land-uses (WBLUs) and between livestock farms and protected areas (Table

S5.11, Figure S5.27). Mean trends for woody productivity were significantly different between land-uses (ANOVA test: $F_{(3)} = 14.2$, $p < 0.01$), except in protected areas and WBLUs (Table S5.11, Figure S5.27).

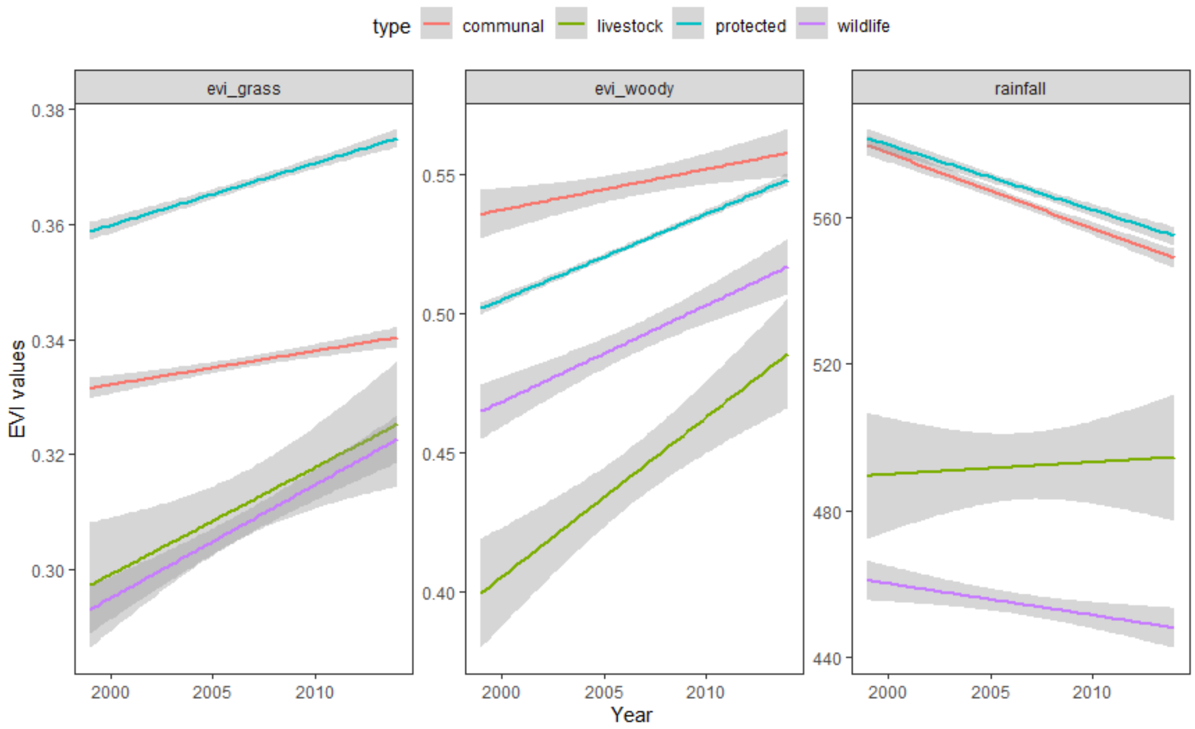

Figure S5.26. Smoothed linear trends (and 95% confidence intervals) of enhanced vegetation index values between 1999 and 2014. Despite a general decrease in rainfall, absolute grass productivity has increased over time in WBLUs – at a greater rate relative to the rainfall received. A similar pattern can be seen for PAs– despite similar rainfall patterns. PAs have much higher and steeper grassy biomass increases than communal rangelands. Over time, WBLUs also have a slower rate of woody increase compared to livestock ranches. The low average rainfall of WBLUs probably reflects the historical precedence of converting to WBLU in arid areas that are marginal for livestock and crop production.

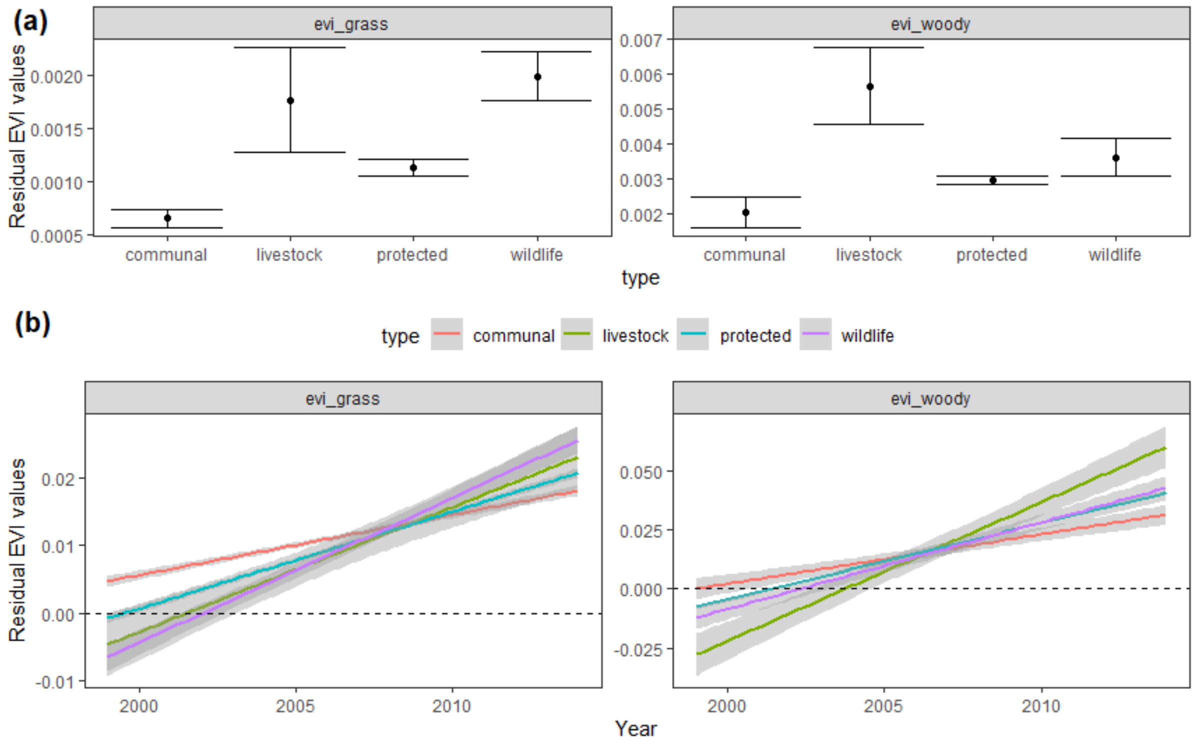

Figure S5.27. (a) Comparison of mean residual trend values between land-uses (non-relativised). Error bars are 95% confidence intervals. Livestock and wildlife-based systems are significantly different from both communal and protected area trends but not from each other for grassy productivity. However, livestock farms have significantly higher woody productivity trends than all other land uses. (b) Smoothed linear models (with 95% confidence intervals) in grassy and woody biomass across land-use types between 1999 (the median conversion date of wildlife ranches) and 2014 (the beginning of the data collection period). The trends show that both WBLUs and commercial livestock ranches have increased grass biomass at higher rates than communal rangelands and PAs. Wildlife ranches show the highest relative increases in grassy biomass, whereas livestock ranches show the highest gains in woody productivity.

Table S5.11. Comparison of residual (non-relativised) trends in vegetation productivity between different land-uses. Positive differences mean that Land use B has higher mean residual slope values than land-use A.

Land-use A	**Land-use B**	**Mean difference (B-A)**	**95% CI (lower)**	**95% CI (upper)**
Grassy productivity				
Communal	Livestock**	0.001	0.0003	0.002
	Protected**	0.0004	0.0003	0.0006
	Wildlife**	0.001	0.001	0.002
Livestock	Protected	-0.001	-0.001	0.0001
	Wildlife	0.0002	-0.0006	0.001
Protected	Wildlife**	0.0009	0.0005	0.001
Woody productivity				
Communal	Livestock**	0.004	0.002	0.005

	Protected**	0.001	0.0002	0.001
	Wildlife**	0.001	0.0005	0.003
Livestock	Protected**	-0.003	-0.004	0.0001
	Wildlife**	-0.002	-0.003	0.008
Protected	Wildlife	0.001	-0.0002	0.001

**significant at P < 0.05; ** at P < 0.01*

Supporting Information 5.7 Tree-grass patterns

The relationship between grassy and woody productivity is highly significantly correlated (OLS regression: $F_{1, 152} = 267.5$, $N = 152$, $p < 0.001$, $R^2 = 0.64$; Figure S5.28a), due to the general greening trend detected in the region (Skowno et al. 2017; Venter et al. 2018; Piao et al. 2020). I also tested the correlation between revenue / ha and the ratio of grassy to woody productivity across properties. While there was a positive relationship, meaning that properties with higher levels of grass productivity compared to woody productivity generally had higher revenues, this relationship was only close to being significant at the 10% level (OLS regression: $F_{1, 46} = 2.46$, $N = 48$, $p = 0.12$, $R^2 = 0.03$; Figure S5.28b). More research is required to unpack this relationship.

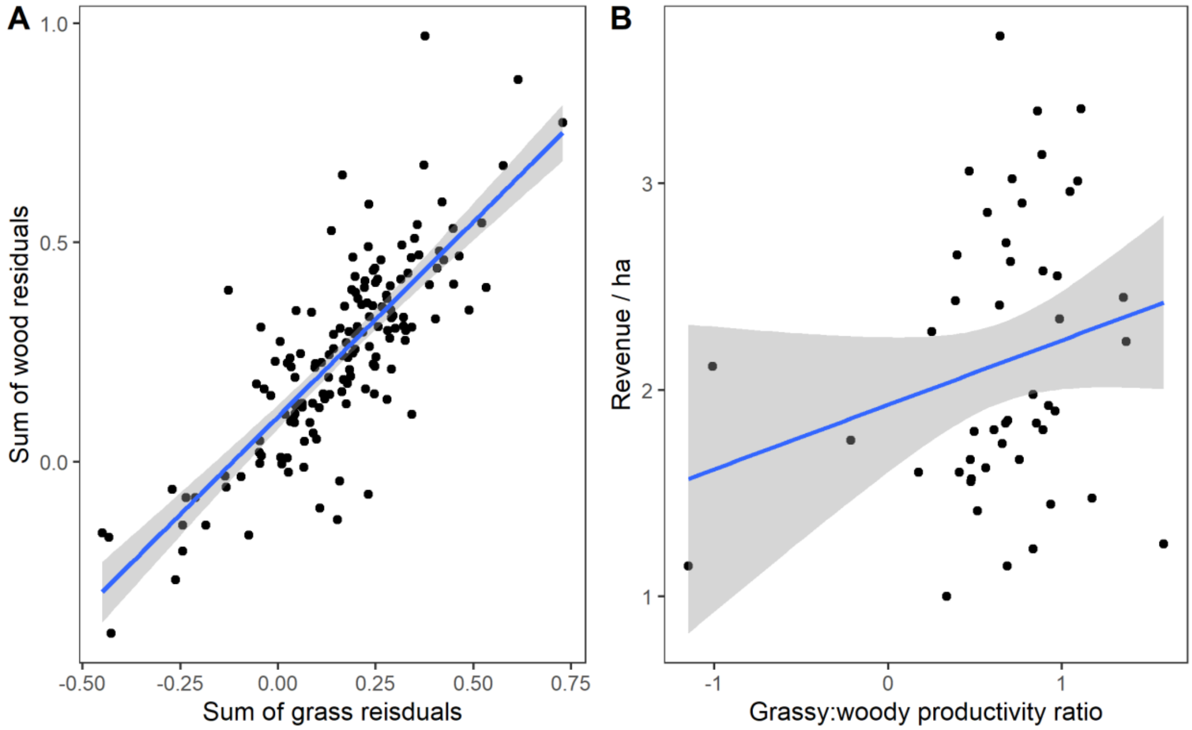

Figure S5.28. Regressions to explore the relationship between grassy and woody productivity (A) The correlation between the sum of woody and grassy residuals across wildlife-based land-uses, which is significantly positively correlated. (B) whichThe relationship between the grassy:woody productivity ratio and revenue / ha on 48 properties where data were available was positive but insignificant (p = 0.12).

Supporting Information 5.8 Rewilding rangelands for soil carbon sequestration

The global nature-based carbon market has predominantly emphasized forest carbon projects, largely driven by the Reduced Emissions through Degradation and Deforestation (REDD+) programme (Bossio et al. 2020), despite significant methodological issues with quantifying ‘avoided emissions’ projects (West et al. 2023). While suitable for temperate and tropical forests, this focus overlooks the unique characteristics of rangelands, which cover approximately 50% of the African continent. These rangelands, primarily grassland or savanna-grassland ecosystems, are unsuitable for afforestation efforts as tree planting can adversely affect biodiversity and hinder communities' ability to derive livelihoods from herbivore production (Vetter 2020). Moreover, the perceived benefits of afforestation and fire suppression in savanna ecosystems have been significantly overestimated, with actual carbon sequestration gains inflated by more than tenfold (Zhou et al. 2022). Increased above-ground carbon, either through planting trees directly (through programmes like the Bonn challenge) or indirectly through fire suppression to increase above-ground carbon, can be termed bush encroachment and has severe socioeconomic and biodiversity impacts in African rangelands (Turpie et al. 2019). Initiatives such as the Bonn Challenge are based on the incorrect assumption that open-canopy vegetation and grassy areas are degraded and in need of afforestation (note not re-forestation) (Bond et al. 2019; Venter et al. 2021). This, together with elevated CO_2 levels, leads to woody encroachment and loss of productivity and biodiversity of rangelands (Skowno et al. 2017; Venter et al. 2018), and is also a problem in temperate rangelands (Ratajczak et al. 2012). As such, interventions focused on forest carbon markets in Africa might often have detrimental effects on biodiversity, socioeconomic development, grazing capacity, and water cycling. Across all of Africa, there is only one registered soil carbon project[26].

South Africa, despite its limited forest cover, is heavily invested in REDD+ initiatives (Rahlao et al. 2012), and will be launching a national REDD+ programme, with a joint task team from DALRRD and DFFE overseeing its implementation (Knowles et al. 2020). This is despite that over 75% of the national terrestrial carbon stock is located in grassland and savanna ecosystems – and within these two ecosystems, over 90% of the total carbon pool is located mainly belowground, mainly in the form of soil organic carbon (Department of Environmental Affairs 2017). Notably, the absence of a formal 'forest' definition under REDD+ allows countries to establish their own criteria. However, this flexibility can inadvertently categorize bush encroachment and alien invasions as positive carbon accounting contributors, especially when using remote sensing (Zomer et al. 2016; Venter et al. 2020). Such classifications might boost above ground carbon stocks but could compromise landscape productivity and water services, especially in regions projected to experience increasing aridity. For example, a review of alternative livelihoods in Africa limits the market potential to REDD+ projects and does not even mention soil carbon (Fabricius et al. 2021). Similarly, countries like Namibia are overlooking the potential of soil carbon, focusing solely on REDD+ (Meyer et al. 2021). REDD+ based carbon offsets run the risk of undermining local communities' autonomy and economic base (Takacs 2020).

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[26] Search for projects on the Verified Carbon Standard project registry

Soil organic carbon (SOC) is the largest terrestrial carbon pool in the world, containing an estimated 2344 gigatons of carbon (Stockmann et al. 2013), which is greater than carbon in the atmosphere and above-ground biomass combined (Georgiou et al. 2022). As such, soil carbon represents 25% of the potential of natural climate solutions (23.8 Gt of CO_2-equivalent per year), of which 40% is protection of existing soil carbon (e.g., Goldstein et al. 2020) and 60% is rebuilding depleted stocks (Bossio et al. 2020). In African savannas, grasses produce more than half of soil organic carbon to a 1 m soil depth even in soils directly under trees, and the largest SOC densities were associated with higher grass productivity, and tree cover does not explain SOC density (Zhou et al. 2023). As such, SOC carbon markets are directly linked to the capacity of rangelands to support herbivore production and, thus, agro-ecological wildlife economies. Griscom et al. (2017) state the greatest threat to natural climate solutions is competition with increasing food production. However, unlike natural climate solutions focused on forests, grassland restoration pathways are congruent with maintaining livestock production and enhancing food sovereignty.

Increasing SOC while maintaining herbivore production and socioeconomic outcomes has traditionally been studied through the lens of converting land-uses from cropland to pastures or ‘natural grasslands’, combined with changes to grazing management. Croplands generally have low SOC, which improves significantly when converted to grasslands (Preger et al. 2010; Loke et al. 2019; Mbanjwa et al. 2022). For example, in KwaZulu-Natal, South Africa, SOC was significantly lower under arable cultivation (range from 1.4 to 2.1% C) compared to matched grasslands and pastures (3.4 to 4.2% C) (Mbanjwa et al. 2022). Preger et al. (2010) found SOC recovery in the secondary pastures (or ‘old fields’) resulted in SOC stocks that were 30–94% greater than those in the arable land in the Free State province of South Africa. This was corroborated by Loke et al. (2019), who found conversion from cultivated soils into perennial pastures restored soil C fractions by 3–129% (i.e. sometimes more soil C than control sites) in the Free State grasslands.

After positive land-use change, grazing management is the next most significant impact on SOC. On a global scale, improved grassland management increases SOC stocks by 0.47 Mg C / ha / year on average (Conant et al. 2017). Whether it is cattle or wildlife, continuous grazing systems at high stocking rates reduce grass and forb cover and thus SOC (Conant & Paustian 2002; Kotzé et al. 2013; Sandhage-Hofmann et al. 2015; Kgosikoma et al. 2015; Wachiye et al. 2022). Reducing plant cover and productivity reduces root- and microbial-mediated SOC formation while also increasing soil bulk density (through overgrazing and trampling) and lowering water infiltration needed to stimulate the soil microbiome (Byrnes et al. 2018; Wachiye et al. 2022). Intensive grazing, by removing the biological soil crust and decreasing soil water content, increases rates of soil CO2 efflux (Thomas 2012; Abdalla et al. 2022; Wachiye et al. 2022), meaning that the capacity of soils to sequester soil carbon is diminished. These impacts also vary according to a precipitation gradient, where SOC loss from arid climates with sandy soils is more severe than in wetter environments, with average SOC loss of 16% compared to 8%, respectively (Dlamini et al. 2016). Wetter environments may have the productivity to sustain higher grazing intensity (McSherry & Ritchie 2013; Hawkins 2017). Impacts also vary by grassland type, where moderate and heavy grazing can increase SOC in grasslands dominated by C4 species (by 7%) but decrease it (by 18%) in grasslands dominated by C3 species (McSherry & Ritchie 2013). This pattern was confirmed by global reviews (Abdalla et al. 2018; Maestre et al. 2022), which confirmed that increasing grazing can increase SOC only in moist warm environments, and lowering grazing pressure in dry or cool environments is better, which presents a bifurcation pathway for using grazing management as a tool to increase SOC.

The question, however, of what the best grazing management systems and stocking rates are, is fraught with contention. This is because overgrazing is a function of both spatial and temporal scales rather than animal numbers alone. Multiple reviews have demonstrated the lack of universality when implementing different grazing systems, attributed to the inherent disequilibrium dynamics of rangeland systems, fine-scale environmental suitability gradients and adaptive management responses that confound the prescription of a silver bullet system (McGranahan & Kirkman 2013; Hawkins 2017; Teague & Barnes 2017; Hawkins et al. 2022). However, the general principle of reducing grazing intensity (either through rotational grazing or lower stocking rate) with adequate resting periods seems to be the most robust heuristic. At the global scale, light to moderate grazing (for example, seasonal or rotational grazing) shows the least negative effects and sometimes promotes soil carbon storage, whereas heavy (continuous) grazing consistently reduces soil carbon stocks (Conant & Paustian 2002; McSherry & Ritchie 2013; Zhou et al. 2017; Byrnes et al. 2018; Chen & Frank 2020; Reinhart et al. 2021; Moore et al. 2023, 2023; Kachler et al. 2023). Africa has been identified as having the most potential to modulate from intensive to moderate grazing (Conant & Paustian 2002).

The most effective way to achieve 'light to moderate grazing' seems to be transitioning to some form of rotational grazing where herds are moved dynamically across the landscape to enable periods of rest for different paddocks, pastures or grassy areas. This contrasts with continuous grazing, where livestock have unrestricted access to a pasture for extended periods. Rotational grazing can increase grass cover and sustain higher proportions of perennial grasses (such as *Themeda triandra)* biomass in African rangelands (Peel & Stalmans 2018), which is an indicator of enhanced rangeland ecosystem functioning (Snyman et al. 2013). Rangelands in better condition have higher levels of ecosystem functioning and make more efficient use of resource inputs, producing more biomass and using water more efficiently than rangelands in poor condition (Snyman 1999), where fodder input in dry seasons and reduced water use efficiency makes them even more vulnerable to the impacts of climate change, thereby reducing the resilience of the overall social-ecological system. Rotational grazing consequently shows consistently higher SOC stocks compared with continuous grazing (Kotzé et al. 2013; Chaplot et al. 2016; Conant et al. 2017; Byrnes et al. 2018; Hillenbrand et al. 2019; Abdalla et al. 2022), with gains observed specifically in the mineral associated fraction (Mosier et al. 2021). In an experimental design in KwaZulu-Natal, South Africa, for example, rotational grazing systems, when compared to livestock exclosure with fertilizer application, annual burning regimes and continuous grazing farms, reduced soil carbon emissions by 17% compared to continuous grazing and increased soil carbon stocks by 50% after just three years of implementation (Abdalla et al. 2022). Globally, improved grazing management generally increases soil C concentration by 10% (Conant et al. 2017); and optimizing grazing intensity through rotational grazing is projected to increase soil carbon sequestration potential by 148 to 699 megatons (Mt) CO2e / year in global rangelands (Griscom et al. 2017), which provides a low-cost / high-carbon gain natural climate solution for grasslands that also improves biodiversity and socioeconomic outcomes (McElwee et al. 2020; Pörtner et al. 2021; Bai & Cotrufo 2022).

However, improved grazing management with cattle is only half the story. Cattle have replaced wildlife over most of the African continent, resulting in a loss of ecosystem functioning, an increase in greenhouse gas (GHG) emissions and a loss of rangeland productivity from bush encroachment (Hempson et al. 2017). Simultaneously, climate change is making large parts of Africa more arid and thus less suitable for livestock ranching, threatening the livelihoods of millions of people and potentially forcing communities into a maladaptive system that perpetuates rangeland degradation through overgrazing. While the available evidence for the impacts of grazing on SOC is limited for tropical grasslands and savannas, it is even less for the impacts of wild versus domestic grazers. Cattle exclusion experiments reveal that heavy cattle grazing uniformly reduces ecosystem carbon uptake and soil storage as well as increases carbon fluxes from ecosystems to the atmosphere (Talore et al. 2016; Xiong et al. 2016; Zhou et al. 2017), which might be different to the effects of wild grazers. Cattle also cannot compensate for the loss of browsing herbivores that impact woody vegetation to produce beta diversity of tree and shrub species (Pringle et al. 2016), and repel bush encroachment to sustain grass productivity (Guyton et al. 2020). Considering bush encroachment reduces grazing capacity of the grassland, removing invading shrubs can triple grazing capacity and double soil carbon content (Yapi et al. 2018), which might be a good additive factor for rewilding rangelands in SOC gains. Similarly, while cattle grazing can be managed in ways that promote SOC sequestrations, these effects are still highly localised due to the intensive herding practices required and cannot replicate the extent of spatial grazing and browsing patterns generated by highly mobile wild herbivores, and thus key nutrient dispersal functions might be lacking.

Rewilding rangelands has thus been conceptualised as a key pathway to improve SOC sequestration by restoring ecological functions (Schmitz et al. 2014, 2017, 2018, 2023; Cromsigt et al. 2018; Sandom et al. 2020; Malhi et al. 2022). Wild grazing has several hypothesised effects on SOC: firstly, the increased nutrient cycling and nutrient dispersal from diverse trophic guilds (Hempson et al. 2017). Wild animals roam widely across landscapes, which creates disturbance via herbivory to create patch dynamics and ensures continuous carbon uptake in the soils (Veldhuis et al. 2018; Schmitz et al. 2018; Schmitz & Leroux 2020). The combination of grazing and browsing has been documented to reduce alien invasion and bush encroachment (Sankaran et al. 2013; Ender et al. 2017; Venter et al. 2018, 2019; Guyton et al. 2020). Greater nutrient cycling and the creation of patches also create higher plant diversity, which increases soil carbon storage (Lange et al. 2015; Chen et al. 2018; Yang et al. 2019). Wild herbivores can also switch between grazing and browsing and, through niche separation, are able to more effectively use the full spectrum of browse and grazing forage available (Taylor & Walker 1978; Du Toit & Cumming 1999). Mixed trophic levels may also contribute to a reduction in forage selectivity through complementary foraging and thus improve SOC (Chang et al. 2018). Wild herbivores are also better adapted to environmental disturbances and stochasticity, such as droughts, diseases, and finding alternative forage species. For example, wildebeest in Botswana were able to disperse and found tsamma melons during a drought when artificial water points dried up (Selebatso et al. 2018). This means they can disperse across the landscape more efficiently.

The grazing patterns and disturbance caused by wild herbivores can also reduce fire intensity and ensure asymmetrical burns to sustain habitat heterogeneity (Kimuyu et al. 2014). For example, wildebeest migrations periodically reduce fuel loads, preventing fires from burning intensively over vast areas and helping to retain carbon in the soil (Holdo et al. 2009). Patchy and heterogenous grazing create patch mosaic fires that help reduce extreme carbon releases into the atmosphere. By removing living and moribund plants, large herbivores reduce material that may fuel wildfires, which can add large amounts of carbon to the atmosphere.

Thirdly, megaherbivores act as key ecosystem engineers and transfer carbon from above-ground to below-ground pools (Cromsigt et al. 2018). Wild herbivores can increase the persistence of ecosystem carbon through redistributing carbon from aboveground vegetation pools vulnerable to disturbances into persistent soil pools (Forbes et al. 2019; Kristensen et al. 2022; Naidu et al. 2022). The flux of matter through the ecosystem increases when large herbivores are present, which increases ecosystem metabolism and fertility. Wild herbivore presence promotes fast-growing, highly productive herbaceous vegetation, which stimulate plant productivity and allocation of carbon to roots. Large animals can expose a larger fraction of the organic matter in soils to organo–mineral interaction through vertical soil mixing, either through their own activity or the activity of associated fossorial mammals or soil fauna. Megaherviores also maintain habitat ‘openness’ by removing shrubs and trees and creating ‘grazing lawns’. For example, in South Africa, elephant reintroduction time was positively correlated with landscape openness, and trophic rewilding with elephants helped promote a semi-open ecosystem structure which increased biodiversity (Gordon et al. 2023). As such, elephants reduce above-ground carbon but increase SOC density through the formation of woody biomass (Sandhage-Hofmann et al. 2021), almost accounting for a net neutral C transfer from aboveground to soil. If these were viewed purely in terms of C gains and losses, elephants are not assets. However, by helping to maintain open savannahs, prevent woody encroachment and increase SOC, elephants will help restore productivity and fertility to African rangelands and thus sustain socio-ecological systems based on mixed production and conservation landscapes.

Restoring apex predators may have interesting impacts on SOC, too. In general, carnivore-herbivore-plant interactions mediate soil and ecosystem carbon and nitrogen turnover rates (Schmitz et al. 2018), thus affecting fundamental properties of the carbon cycle. However, the reintroduction of apex predators may enhance SOC more directly through two mechanisms: firstly, the creation of landscapes of fear creates habitat heterogeneity and ensures continuous grazing is not possible. Secondly, the direct killing of prey controls herbivore population dynamics and directly returns carbon to the soil through carcasses. In South Africa, most managers control the impacts of overgrazing in fenced reserves through annual game capture and removal (‘offtakes’), wherein live animals are sold to other landowners or culled for game meat (Taylor et al. 2020). However, this may lead to critical nutrient losses from the system. For example, in Tswalu Kalahari Reserve (TKR), it was estimated that the removal of wild herbivores from the system to reduce stocking rates between 2009 and 2018 amounted to a net loss of 18.5 kg/km^2 and 40.6 kg/km^2 phosphorous and calcium respectively, which may have severe consequences for rangeland condition and thus productivity in the long-term as well as leading to nutritional stress in the herbivore populations themselves (Abraham et al. 2021). Additionally, the carrion left behind is an important source of nitrogen and, thus, soil carbon (Macdonald et al. 2014; Barton et al. 2019). This presents a conundrum to managers of fenced reserves who wish to use offtakes to improve short-term grass productivity at the potential risk of undermining long-term rangeland fertility and productivity. The reintroduction of apex predators may solve this problem. Remarkably, the biomass of all prey taken by lions in the Lekgaba section of TKR (where lions are present) is 224 kg km^2 / year, which is close to the biomass taken off by management to reduce grazing pressure in the Korannaberg section (where no apex predators are present) of 230 kg km^2 / year (Abraham et al. 2021). This indicates that natural predation can achieve similar herbivore population control as live animal removals and prevents nutrient loss. Correspondingly, the veld condition index of the Lekgaba section is significantly better than that of Korannaberg (van Rooyen & van Rooyen 2017).

These examples demonstrate that wild herbivore grazing may increase SOC. However, a recent synthesis of 174 experiments on the impacts of wild grazing on SOC found the opposite: large herbivore exclusion generally increases SOC storage across different biomes, suggesting a net negative impact of large wild herbivores on soil carbon storage (Forbes et al. 2019). Upon closer examination of this review, however, the confounding effects of wild grazing in different biomes across different soil and precipitation gradients using different experimental designs may have lost any signals for specific regions. Very few studies from Africa were included and, of these, only one provided experimental evidence for SOC sequestration. Additionally, the review of Forbes et al. (2019) did not contextualize the impacts of wild grazing in the context of domestic grazing counterfactual.

To redress, I compiled experimental evidence for the effects of different grazing systems on SOC across African savannas to ensure like-for-like comparisons. I found seven sources containing experimental evidence for SOC impacts from South Africa, Zimbabwe and Kenya (Table S5.12). While there are more experiments on the impacts of grazing systems on SOC across Africa, many of these report only the soil carbon fraction (proportion of soil weight that is made up of organic carbon) and not the carbon mass and bulk density needed to calculate soil carbon stocks. The grazing systems documented in these studies ranged from no grazing to livestock only (both continuous and rotational systems) and various stages of wildlife integration to full wildlife grazing systems. Rainfall was similar across sites at the time of SOC sampling. I selected only SOC values for the 0–30 cm depth (to be congruent with international carbon standard methodologies) and converted all SOC metrics into C kg / m^2 through the following protocol:

- Determine the volume of soil for a specific depth: Volume (m^3) = Soil depth (m) × 1m^2
- Calculate the mass of soil for the specific depth: Mass (kg) = volume (m^3) × bulk density (kg/m^3)
- Calculate the mass of carbon for the soil depth: Carbon mass (kg) = Mass (kg) × carbon concentration (g / kg) × 0.001

These data were then compiled into Figure S5.29 (grazing system codes explained in Table S5.12), where grazing systems were organised left to right from no grazing to cattle grazing and various stages of wildlife integration. While grazing exclusion (code O) has higher SOC density than grazing with cattle (both continuous and rotational), adding wild meso- and megaherbivores systematically increases SOC density to levels above that of total grazing exclusion. There are important implications from these results. Firstly, African rangelands are adapted to disturbance by herbivory and fire, and wild grazing systems can increase the SOC more than the counterfactual. This is in contrast to recent global reviews that combine data across the globe to conclude herbivore exclusion leads to inevitably higher SOC (Tanentzap &

Coomes 2012; Xiong et al. 2016; Forbes et al. 2019). It also supports the theory that shows moderate grazing pressure is necessary to maintain SOC and grass productivity in African rangelands (Fuhlendorf et al. 2009; Hempson et al. 2017; Naidu et al. 2022), which may have a stronger influence on SOC than fire (Ritchie 2014). Consistently, the trials where grazing pressure was light to moderate had higher SOC densities than the counterfactual with the highest SOC of the set being both cattle and wildlife at very low grazing densities (compared to complete exclusion under the O code) (Table S5.12, Figure S5.29).

Table S5.12. List of data sources and characteristics of the different grazing systems used to compare soil organic carbon (SOC) densities in various savanna sites in Africa. Seven studies were used to compare grazing systems, where categorisations were adapted from Sitters et al. (2021). Some grazing systems were replicated between studies providing independent tests of the grazing system on SOC. All SOC densities were converted to kg / m^2. Outliers (two standard deviations from the mean) were removed to increase the readability of the plot.

Grazing system code	**Grazing system**	**Grazing system explanation**	**Fire regime**	**Locality**	**SOC (kg / m^2)**	**Rainfall (mm / annum)**	**Period**	**Source**
O - Kenya	No grazing	Large herbivore exclosure	No fire	Mpala Research Centre, Laikipia County, Kenya	1.9±0.3	612	2003-2015	Sitters et al. 2021
O - South Africa	No grazing	Sheep grazing exclosure (10 years).	No fire	Grootfontein Agricultural Research Institute, Eastern Cape, South Africa	2.1±0.2	373	Long-term average	Talore et al. 2016
C - Ken1	Cattle only	Moderate stocking rate (5-10 ha / LSU); rotational grazing system (plots get max. 16 weeks grazing rest)	No fire	Mpala Research Centre, Laikipia County, Kenya	1.1±0.1	612	2003-2015	Sitters et al. 2021
C - Ken2	Cattle only	Continuous heavy grazing by livestock throughout the year. 7532 cattle and 373 mesoherbivores (5.9 ha / LSU overall)	One fire event in study period	Taita–Taveta County, southern Kenya	1.2±0.3	500	Long-term average	Wachiye et al. 2022
C - SA2	Cattle only	Rotational (1200 LSU / ha for three days per plot) and continuous grazing systems (stocking rate unknown (combined here) in degraded habitat.	Low fire occurrence	Potshini village, KwaZulu-Natal Province, South Africa	0.6±0.2	640	2013-2014	Abdalla et al. 2022
Sh - SA	Sheep only	Merino wethers grazed the treatments at a stocking rate of 0.78 and 1.18 sheep ha1	No fire	Grootfontein Agricultural Research Institute, Eastern Cape, South Africa	1.4±0.1	373	Long-term average	Talore et al. 2016
WC - Ken1	Wild mesoherbivores and cattle	Moderate stocking rate (5-10 ha / LSU) by cattle; rotational grazing system (plots get max. 16 weeks grazing rest). Stocking rate of wildlife not reported	No fire	Mpala Research Centre, Laikipia County, Kenya	1.3±0.3	612	2003-2015	Sitters et al. 2021
WC - Ken2	Wild mesoherbivores and cattle	Moderate continuous grazing (911 cattle and 365 mesoherbivores)	Low fire occurrence	Taita–Taveta County, southern Kenya	1.9±0.7	500	Long-term average	Wachiye et al. 2022

Grazing system code	Grazing system	Grazing system explanation	Fire regime	Locality	SOC (kg / m^2)	Rainfall (mm / annum)	Period	Source
WC - low stocking rate	Wild mesoherbivores and cattle	Fenced reforestation project area located established in 2010. Very low levels of cattle and mesoherbivores.	No fire	Taita–Taveta County, southern Kenya	4.9±1.6	500	Long-term average	Wachiye et al. 2022
W - Ken1	Wild mesoherbivores	Cattle excluded from plots (>10 years). Stocking rate not reported	No fire	Mpala Research Centre, Laikipia County, Kenya	1.9±0.6	612	2003-2015	Sitters et al. 2021
W - Ken2	Wild mesoherbivores	Moderately to lightly continuous grazing of mesoherbivores (1051 mesoherbivores, 125 cattle)	Low fire occurrence	Taita–Taveta County, southern Kenya	2.6±0.9	500	Long-term average	Wachiye et al. 2022
MWC - Ken1	Wild mega- and mesoherbivores and cattle	Moderate stocking rate (5-10 ha / LSU); rotational grazing system (plots get max. 16 weeks grazing rest). Stocking rate of wildlife not reported	No fire	Mpala Research Centre, Laikipia County, Kenya	2.5±0.2	612	2003-2015	Sitters et al. 2021
MWC - Ken2	Wild mega- and mesoherbivores and cattle	Heavily grazed - some elephants (22) and mesoherbivores (1867) but mostly cattle (1458)	Low fire occurrence	Taita–Taveta County, southern Kenya	2.7±1.2	500	Long-term average	Wachiye et al. 2022
MWC - Zim	Wild mega- and mesoherbivores and cattle	Moderate stocking rates of both cattle and wildlife (2.1 LSU / ha). Rotational grazing system.	Low fire occurrence	Shangani Ranch, Zimbabwe	3.2±0.7	607	2012-2021	Rewild Capital (unpubl. Data)
MW - Ken	Wild mega- and mesoherbivores	Cattle excluded from plots (>10 years). Stocking rate of wildlife not reported	No fire	Mpala Research Centre, Laikipia County, Kenya	2.3±0.4	612	2003-2015	Sitters et al. 2021
MW - SA	Wild mega- and mesoherbivores	Natural grazing system with meso- and megaherbivores along a grazing intensity gradient and other environmental gradients (combined here).	Average fire return interval of 2–4 years	Hluhluwe iMfolozi Park, KwaZulu-Natal, South Africa	2.8±1.6	598	2019-2020	Hyvarinen et al. 2023

Secondly, the potential negative impacts of cattle grazing on SOC are turned positive when wildlife are added to the system, especially megaherbivores. Adding mesoherbivores to cattle-only systems increases SOC density (confirmed in two independent studies). Aadding megaherbivores to the system alongside mesoherbivores and cattle improves SOC density even more so than meso-and megaherbivore only systems (confirmed in three independent studies). Thus, mixed cattle and megaherbivore systems might be a sustainable management strategy in African savanna ecosystems with high herbivore diversity (Sitters et al. 2020). This underscores the importance of mixed wildlife and cattle rangelands as key novel ecosystems for Africa, where both herbivore production and ecosystem functioning can be balanced or even designed to generate additional ecosystem services. Cattle export nutrients to small patches where they are kraaled overnight, which in the long-term may lower overall forage quality in the rangeland. However, adding wildlife in the system leads to more even nutrient cycling across the rangelands and net higher habitat quality. Experimental evidence shows that while cattle are the primary drivers of higher mean productivity, mesoherbivores are the primary drivers of more spatially and temporally stable productivity (Charles et al. 2017). These functions also feedback into each other as megaherbivores keep savanna rangelands open ,transfer woody carbon to the soil and maintain highly productive grazing lawns (Gordon et al. 2023; Hyvarinen et al. 2023). The ecological impacts of cattle can, therefore, be more productive rather than leading to rangeland degradation and bush encroachment. Additionally, the use of bomas for overnight kraals creates nutrient hotspots, including high SOC stocks (Valls Fox et al. 2015; Momberg et al. 2023), which increases overall long-term landscape heterogeneity (Fynn et al. 2016; Sitters et al. 2020). This uniquely African novel ecosystem could be further enhanced by rewilding cattle breeds themselves (such as switching to lighter breeds that also browse, like Nguni), which will also mitigate climate change by lowering methane emissions and enhancing nutrient cycling (Jordaan et al. 2021; Casey 2021). For example, increasing cattle productivity and switching to indigenous cattle breeds can lower methane emissions by as much as 44% (Scholtz et al. 2023)

Thirdly, these data, compiled from different sources with different experimental methods, show remarkable consistency for African rangelands. Three different studies, two in Kenya and one in Zimbabwe, showed that mixed cattle and wildlife grazing systems with megaherbivores had similar high levels of SOC density. The two primary study sources came from conservancies in the north (Sitters et al. 2020) and south of Kenya (Wachiye et al. 2022) and again show remarkable consistency between the grazing treatments (Figure S5.29). It is interesting to note in the Wachiye et al. (2022) study that the treatments were essentially replacing one continuous grazing system (cattle) with increasingly more wildlife-integrated continuous grazing systems. Thus, the patterns most likely represent the restoration of ecological functions and lower cattle stocking rates across the treatments. For example, the ‘moderately grazed’ category had a total animal count of 1287 of which 71% was cattle, whereas the ‘moderately to lightly grazed’ and the ‘lightly grazed’ categories had only slightly smaller total animal numbers (918 and 833 respectively) but much lower proportions of cattle (14% and 0% respectively), but the latter sites had significantly higher SOC (1.9 ± 0.7 kg / m^2 compared to 2.8 ± 1.1 and 2.3 ± 0.8 respectively). Collectively, these studies demonstrate robustly the power of rewilding rangelands with wild grazers and show consistently that cattle do not need to be completely removed from the system, but total grazing pressure must be moderate and the ecological functions of wild grazers must be present.

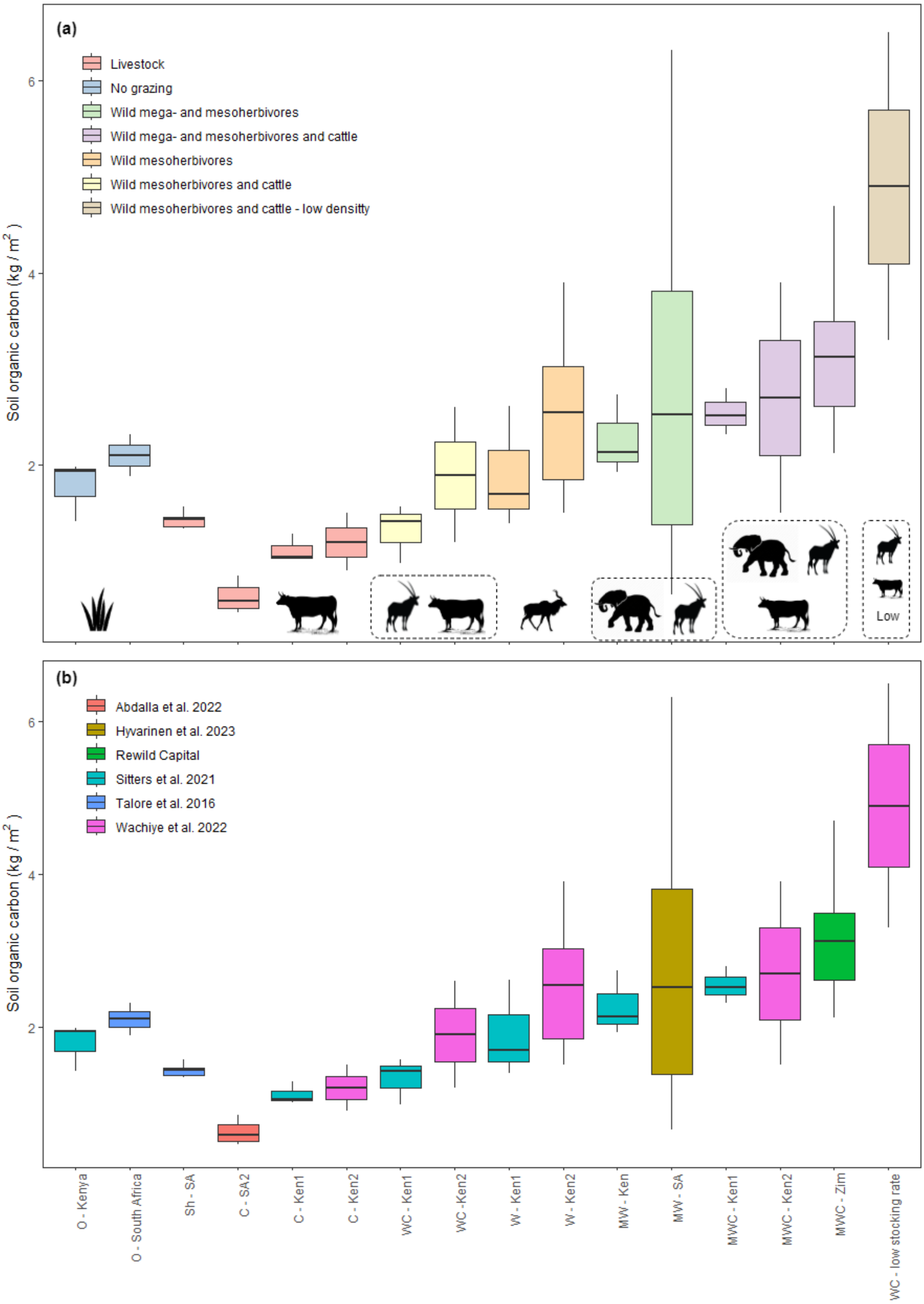

Figure S5.29. Synthesis of soil carbon studies in savanna ecosystems across Africa showing the transition from no grazing to cattle grazing to mixed grazing systems and wild grazing systems. The data, although compiled from different sources, are remarkably consistent in showing that no grazing is mostly better than livestock-only grazing systems and that adding wildlife to cattle grazing systems increases carbon sequestration rates. Most importantly, removing cattle completely is not necessary as wild grazers (especially megaherbivores) are reintroduced. However, the most optimal solution is to have cattle and wildlife grazers at low stocking rates or to facilitate their movement across the landscape to ensure adequate rest periods for grass productivity. (a) shows the grazing systems by colour, while (b) shows the data by study source.

What these studies lack, however, is an analysis of the impacts of fire on SOC and the interactive effects of fire and herbivory. In African rangelands, ecosystem functioning is maintained by the interaction between fire and herbivory. Post-fire regeneration creates green grass flushes that draws herbivores and shifts grazing pressure around the landscape. This dynamic relationship results in a spatial-temporal patchwork across the landscape, with herbivores preferentially grazing on post-fire rejuvenated patches, allowing unburned sections to amass biomass that can serve as future fire fuel. This interplay between pyric effects and herbivory creates a shifting habitat mosaic, fostering enhanced biodiversity by providing varied successional stages favourable to different species. Moreover, these interactions highlight feedback loops, wherein areas frequently rejuvenated by fire but subjected to minimal grazing might exhibit distinct vegetative structures compared to those with pronounced herbivore presence but infrequent fire events. Within the rewilding framework, as delineated by Perino et al. (2019), the pyric-herbivory model underscores the importance of allowing natural processes, like fire and herbivory, to interact freely, facilitating the restoration of resilient ecosystems and enhancing biodiversity. Such a perspective has profound implications for grassland management strategies across Africa, emphasizing the need for holistic approaches that account for these intertwined natural processes.

Initially, fires might lead to the direct loss of SOC as it is combusted and released into the atmosphere (Pellegrini et al. 2018), but fires also contribute to the formation of charcoal and other pyrogenic organic matter, which can be more resistant to decomposition and can enhance SOC sequestration in the long term. For example, fire suppression has little effect on SOC in tropical savannas because C4 grass–derived carbon dominates the SOC, particularly in deeper soil layers, where soil carbon is less affected by changes in fire frequencies (Zhou et al. 2022). The impact of fire on SOC is most strongly driven by frequency. For example, multiple studies have demonstrated that annual burns reduce SOC stocks (Chaplot et al. 2016; Abdalla et al. 2022; Findlay et al. 2022), whereas moderate fire frequency (such as biennial) might create the highest SOC densities compared to no fires and annual fires (H. Hawkins, unpubl. data). Over time, repeated fire events can result in shifts in vegetation composition and structure, further influencing soil carbon storage capacities. For example, restoring fire-herbivory interactions can reduce bush encroachment (Capozzelli et al. 2020; Scholtz et al. 2022), which serve to increase grass-mediated SOC, but a a transition from grass-dominated systems to shrublands or woodlands, in the absence of regular fires, can alter below-ground carbon inputs and potentially increase or decrease soil carbon stocks based on soil types and grazing types (Hyvarinen et al. 2023). Overall, the impact of fire on SOC depends on fire intensity, frequency, and interactions with herbivory. In the context of global climate change and carbon management, understanding the multifaceted interactions between fire and soil carbon in African rangelands becomes paramount, especially as adjusting fire frequencies in Africa is being touted as a way to increase carbon credit generation (Lipsett-Moore et al. 2018; Tear et al. 2021; Awuah et al. 2022; but see Laris 2021), but potentially at the expense of biodiversity that may require different fire regimes to that which would maximize carbon emissions avoidance.

Putting the potential of early dry season burns aside, the results from Figure S5.29 were used to develop a conceptual model of how rewilding can drive the South African National Parks (SANParks) expansion programme (DFFE 2016b) through carbon revenue generated by reducing stocking rate of wildlife in protected areas that are currently not performing optimally for SOC sequestration and donating this wildlife to expansion sites to ‘rewild’ (Figure S5.30). Doing so enables ‘additionality’ for both the target NP(s) and the expansion sites, while also reducing potential leakage associated by translocating wildlife into systems that would add additional grazing pressure (Figure S5.30). This model would help generate revenue through carbon credit sales using existing wildlife assets in NPs, and thus help meet South Africa’s 30x30 target under the Global Biodiversity Framework, whilst also helping to achieve domestic wildlife economy goals under the National Biodiversity Economy Strategy in creating wildlife-based economies for new market entrants. Importantly, this model can be integrated into systematic conservation planning through modelling the costs of management interventions to increase SOC (adjusted grazing and fire management) in the context of the opportunity costs compared to business as usual and the potential revenue from credits earned (Douglass et al. 2011).

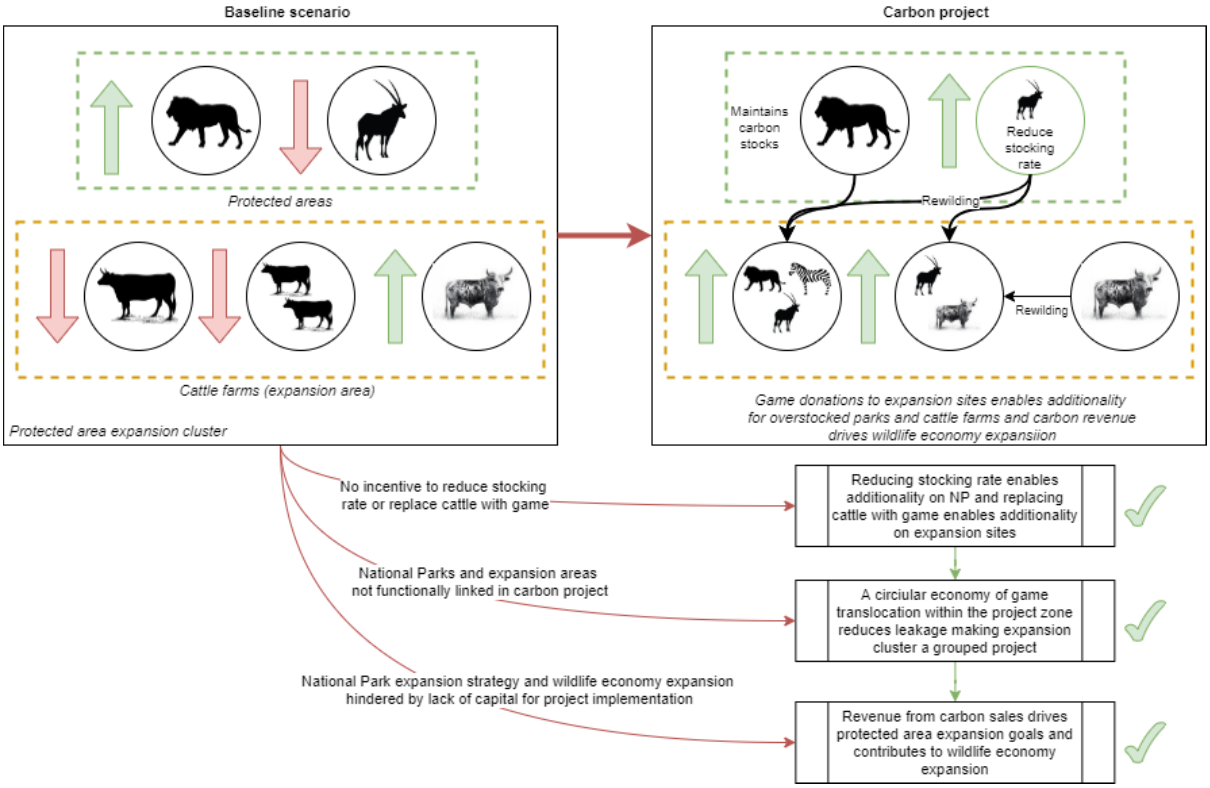

Figure S5.30. Conceptual diagram of driving South Africa’s national park expansion strategy and wildlife economy expansion targets through the use of a grouped soil carbon project. In the baseline scenario, an expansion cluster comprising key national parks and identified expansion areas (here assumed to be mostly privately managed and communally managed livestock farms), will have a combination of sites at soil carbon equilibrium (green arrows) and sites where there is soil carbon sequestration potential (red arrows). Sites with sequestration potential are typically those that are overgrazed due to being overstocked and/or with no grazing management plan to enable resting of veld. As protected areas in South Africa are fenced, there is potential for overstocking and overgrazing, even with wildlife. However, this problem establishes the potential for a virtuous feedback loop where overstocked national parks strategically donate game to expansion areas that are overstocked to bring grazing pressure down and increase nutrient cycling (in the case of cattle farms). Doing so enables a key criterion of a carbon project to be achieved, ‘additionality’, as both management systems undergo a step change through the intervention to draw down additional carbon dioxide. In this example, the donated game could form part of a mixed wildlife and game farm, where exotic cattle are replaced by indigenous cattle types to further alleviate grazing pressure and negative impacts, or become a stewardship site with ecotourism facilities and fully transition into wildlife habitat. This intervention also satisfies a second key criterion of carbon projects – ‘leakage’, in that it ensures grazing pressure is not displaced to neighbouring sites but reduced absolutely and redistributed to net lower levels in the project area (in this case, we assume that cattle on the expansion sites are sold to abattoirs). The revenue from carbon credit sales would enable project proponents to both overcome the opportunity costs of lower stocking rates and also raise capital to implement project activities (such as training in adaptive grazing management practices). Linking project activities between national parks and expansion sites enables the establishment of a grouped project, which would reduce auditing costs associated with individual sites as the project ‘instance’ overall could be audited. A grouped project structure would also allow the entire expansion strategy to be included in one project, where different expansion clusters could be incorporated in a staged manner as they become ready to implement the project interventions.

Chapter 6 Creating inclusive wildlife working lands in South Africa

6.1 Abstract

Rewilding rangelands to create wildlife economies through land reform is seen by many policy-makers as a route to create employment and stimulate rural economies. Land reform is a key social justice movement worldwide and can be integrated into wildlife-based land-uses to promote conservation and socio-economic development. Wildlife-based economies can be profitable, contribute to biodiversity conservation and help restore ecosystem functioning, yet most of the continent’s communities still rely on traditional forms of livestock and crop agriculture that are increasingly untenable under climate change. To effectively design and implement policies to unlock wildlife resources, we need information on the opportunities and barriers confronting the establishment and operation of viable wildlife enterprises. I conducted a survey of communities and landowners awarded land through a government land reform programme in the Eastern Cape province of South Africa to understand the state of wildlife economy development and assess investment needs. Despite all properties listing wildlife-based economic activities in their business plans, and most (84%) having ownership over wildlife assets, only 32% were currently generating income from these activities (range: 5-20% of revenue), accounting for a low total proportion of income across properties (4%). The most mentioned challenges were a lack of infrastructure (fencing, accommodation, irrigation and water storage) and theft of property and animals (32% of respondents). Beyond direct challenges, implementation of the activities listed in the business plans is hindered by the lack of information and decision-support on viable wildlife business models and the subsequent lack of access to skills development and market information. While 90% and 84% of respondents indicated plans to establish ecotourism and trophy hunting operations, respectively, the sizes of new market entrant properties were significantly smaller than the average ecotourism and trophy hunting enterprise from the established sector, indicating a potential mismatch in expectation (and investment) versus viability. I suggest that government programmes initially focus on helping new market entrants establish mixed cattle and wildlife enterprises to address a local hunting and game meat market and develop more impactful infrastructure and skills development investments. If wildlife-based land-uses are seen more as regenerative

agricultural activities and less as protected areas, appropriate support can be mobilized across government departments for communities to develop resilient and viable agro-ecological enterprises.

6.2 Introduction

Rewilding rangelands to create wildlife economies is the focus of current policy formulation and implementation designed to facilitate new market entrants to the sector and create employment as well as conservation benefits. However, the constraints and opportunities for creating new market entrants to the wildlife economy have not been assessed. If rewilding is to become a key policy tool in Africa for sustainable development, existing policies and outcomes must be critically evaluated.

Creating new market entrants for the wildlife economy is most often achieved through various land reform mechanisms. Land reform is a key tool for decolonization in many countries (Boyce et al. 2007; Akinola 2019) and can have far-reaching outcomes for biodiversity conservation and livelihoods alike. These outcomes can be positive, such as increased farm productivity and improved ecological function (Bryan et al. 2018), empowerment of marginalized groups and rural revitalization (Han 2020). However, land reform policy can also have unintended consequences that undermine its original intentions, such as promoting food insecurity (Valente 2009), violence, and the destruction of natural habitats (Alston et al. 2000).

As land reform programmes often focus on rural landscapes that may be untransformed, underdeveloped or protected, they often overlap with areas that are also prioritized for biodiversity conservation. In this context, land reform policies commonly seek win-win outcomes for people and conservation through, for example, co-management agreements of restituted, co-managed conservation areas (Kepe et al. 2005). These “win-win” solutions, however, often undermine both equity and biodiversity goals, such as through ‘elite’ capture of the revenue streams, ‘trickle down’ benefits and lack of decision-making by communities, and loss of productive land-uses from project sites (Wolmer et al. 2004; Kepe et al. 2005; Cundill et al. 2013). Similarly, in areas where the focus is more strongly on agriculture, new owners may choose to transform and cultivate land that may have a host of trade-offs for ecosystem service benefits and beneficiaries (Clements et al. 2021). Key to the success of land reform policy in achieving its stated goals is the recognition that land reform areas are complex adaptive social-ecological landscapes, particularly in agroecological working landscapes (Kremen & Merenlender 2018).

In southern Africa, wildlife-based land uses (‘wildlife ranching’ for short) represent a potential land use option that could reduce trade-offs between objectives related to biodiversity conservation and economic development and empowerment. In recent decades, wildlife ranching, the management of wildlife on private land for commercial purposes, has emerged as a financially viable alternative to farming on private land, particularly in more arid areas (Barnes & de Jager 1996; Child et al. 2012a; Taylor et al. 2020). Wildlife ranches adopt a variety of business models and diverse economic activities that differ in profitability (Clements et al. 2016b, 2022), with the major revenue-generating activities including ecotourism, trophy hunting, meat (‘biltong’) hunting, game meat, skin and hide from culling activities, breeding and selling live animals, selling wildlife meat (‘venison’), as well as mixed livestock / crop agriculture and wildlife farming (Taylor et al. 2020). In South Africa, wildlife ranches cover an estimated 17–20.5 million hectares – 17% of the land area and over double the extent covered by the country’s protected areas (Taylor et al. 2020). The wildlife ranching sector is estimated to provide 68,000 jobs directly on the farm (not including downstream value chains) (Taylor et al. 2020), with trophy hunting alone contributing USD 341 million per year to the South African economy (Saayman et al. 2018).

Wildlife ranching presents a land-use that interfaces between agriculture and conservation. Approximately half of all wildlife ranches are mixed wildlife and livestock farms (Cloete et al. 2015; Taylor et al. 2020; SANBI, unpubl. data). Thus, wildlife ranches are potentially ‘win-win’ social-ecological systems for biodiversity and food production, providing a pathway to enhancing rural livelihoods as well as contributing to the conservation estate. However, the importance of ecosystem services and sustainable land management has not received adequate attention in land reform policies and debates (Shackleton et al. 2001; Clements et al. 2021). Land is a cultural and spiritual resource for people in rural areas; they derive valuable livelihood benefits such as firewood, nutritious wild fruits, medicinal plants and timber and do not merely use it for livestock and crop production. Natural landscapes are vital as water catchment areas for both rural and urban consumers and for maintaining the biodiversity necessary to provide species for food crops (Shackleton & Shackleton, 2004). Rewilding communal rangelands could potentially improve ecosystem service provision, enable ecosystem-based adaptation to climate change and improve livelihood options (Chaminuka 2013; Vetter 2013; Chaminuka et al. 2014; Cromsigt et al. 2018; Sitters et al. 2020; Taylor et al. 2020).

At a policy level, complexity is deepened in that land reform and biodiversity conservation mandates are implemented by different government departments, who often view their mandates as in opposition to each other. In South Africa, the Department of Agriculture, Rural Development and Land Reform (DALRRD) carries the legislative mandate to assist persons (or their descendants) who were excluded from South Africa's formal agricultural economy based on their skin colour, and who have recently begun to engage in farming on a larger scale to sell crops and livestock on the market, with the support and assistance of the state. Since 1994, DALRRD has been executing the mandate of disposing of state land to develop the economic potential of rural communities while at the same time addressing contemporary inequalities in access to commercial agriculture by black farmers. However, the Department of Forestry, Fisheries and Environment (DFFE) is mandated to conserve biodiversity, including regulating the reintroduction, translocation and management of wildlife – even if they are privately managed populations (Carruthers 2008). Attempting to resolve such policy conflicts can lead to unintended and counterproductive consequences, such as attempting to legalize the domestication of wildlife species (Somers et al. 2020), or excluding communities from the land for the sake of 'biodiversity' (Kamuti 2014; Spierenburg & Brooks 2014).

High-level policy design in South Africa recognizes the opportunity to integrate wildlife ranching into land reform to achieve transformation of the sector and produce biodiversity and sustainable agricultural outcomes. The National Biodiversity Economy Strategy (NBES, 2016), which is aligned to the National Development Plan, aims to establish 10 million ha of wildlife ranch estate for communities and previously disadvantaged individuals as well as create 100,000 new jobs in the sector and empower 4000 emerging entrepreneurs and farmers through focused capacity building (DFFE 2016a; Department of Environmental Affairs 2016). To facilitate these goals, DFFE has established spatially-defined biodiversity economy nodes (BENs) across the country to channel infrastructure and asset investment and provide extension support to unlock the wildlife economy for participating communities (DFFE 2020a). Additionally, the Recapitalisation and Development Programme (RECAP) exists to support land reform beneficiaries in establishing viable agricultural enterprises through infrastructural and mentor support (Maka & Aliber 2019; DRDLR 2020). However, the efficacy of these policies in using rewilding as a tool to transform the wildlife industry and create opportunities in rural communities is yet to be fully realized (Kamuti 2014; Spierenburg & Brooks 2014; Pasmans & Hebinck 2017; Mokotjomela & Nombewu 2019). As wildlife economy beneficiation schemes tend to become dominated by wealthy beneficiaries looking to diversify their business, the wildlife economy is not necessarily benefitting poor communities yet (Mtero et al. 2019). Additionally, wildlife economy development often benefits external stakeholders more than the communities themselves (Ngubane & Brooks 2013), which can perpetuate existing inequalities through racialised divisions of labour and displacement of communities from rangelands (Thakholi 2021).

How can the wildlife economy be developed to provide equitable outcomes for people without compromising biodiversity and ecosystem services? Thus far, there has been little work on the nature of wildlife-based activities practiced by land reform beneficiaries, the challenges that they experience, or their desired trajectories and business strategies. Without this basic understanding of the current state of the wildlife economy within the land reform sector, it is impossible for policymakers, government programmes and non-profit organizations to design fit-for-purpose interventions and solutions. In this paper, I aim to address the dearth of knowledge on key barriers and opportunities for realizing the economic and ecological potential of wildlife ranches in land reform programmes through the lens of South Africa’s land reform programme in the Amathole BEN of the Eastern Cape province, South Africa. Through socio-economic and social-ecological surveys of land reform beneficiaries in the Eastern Cape province of South Africa, I identify both barriers preventing rewilding from achieving land reform goals and potential solutions to ensure wildlife-based land-uses become viable for new market entrants. Since property size and age are important determinants of both economic and ecological success in established wildlife ranches and protected areas, we also assess the role that these variables play in the revenue generated by these new entrants, the challenges they experience, and the business models they adopt.

6.3 Methods

6.3.1 *Study sites*

This study was conducted in the Eastern Cape, South Africa. Nineteen land reform sites were purposively selected according to whether the beneficiaries currently have wildlife assets on the property or whether the stated aim of the business plan was to develop wildlife-based enterprises. Sites were also selected to coincide with the Amathole BEN (Figure 6.1), which the government is targeting infrastructure investment, capacity building and other biodiversity economy support activities (DFFE 2020a). Within the sampling region, there are land reform properties acquired through three systems: 1) land restitution, where land was restored to individuals or groups (or their descendants) that were forcefully removed from their land by racially discriminatory policies since 1913; 2) Land tenure system which secures the rights of people living in land parcels with insecure arrangements on land owned by others; and 3) Redistribution systems: which broadens access to land among the country’s black majority correcting historic racial inequalities (Figure 6.1). The nineteen sites, which all fell within the distribution systems, were identified from the transfer documents registered with DALRRD.
Respondents were made aware of the goal of the project, and no financial or in-kind compensation was provided. I compared the results to a more extensive survey (using a similar questionnaire form) conducted on the established wildlife industry between 1 February and 30 June 2021. Established enterprises were simplified according to primary economic activity, which was defined as trophy hunting (“trophy”), biltong hunting and game meat production (“biltong”), game breeding and sales (“breeding”), overnight ecotourism with guest accommodation (“eco-night”), day visitor ecotourism (“eco-day”) and mixed cattle and wildlife ranches (“mixed”) (Clements et al. 2016b; Taylor et al. 2020). Activities were considered significant if they comprised ≥10% of total revenue for the enterprise. Ethical clearance was obtained from the Rhodes University Ethics Committee [no. 2021-2810-5892].

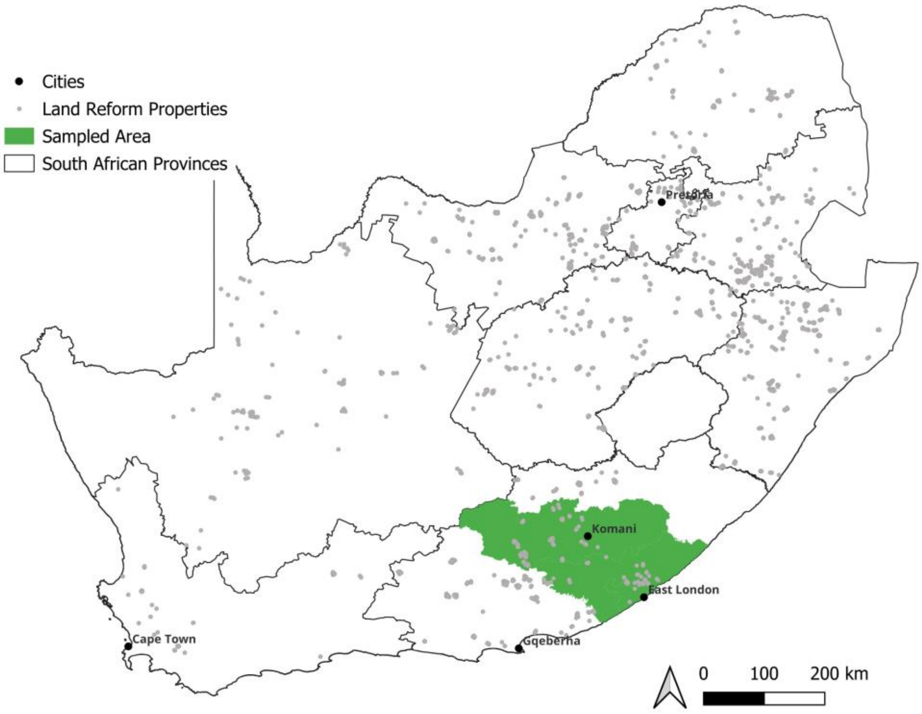

Figure 6.1**.** Distribution of land reform properties across the country. The area sampled in this study is shown in green – it coincided with the Amathole Biodiversity Economy Node earmarked by the Department of Environment, Forestry and Fisheries (DFFE) and the Eastern Cape Parks and Tourism Agency (ECPTA).

6.3.2 Data collection and analysis

I interviewed 19 land reform beneficiaries and community property associations across the Eastern Cape province of South Africa between 1 and 30 June 2021. I developed a semi-structured survey that collected information on aspects of land-use history, governance structures, asset and infrastructure profiles, skills and investment needs, as well as more general problems and opportunities to do with establishing viable wildlife-based enterprises. In December 2020, four experts with knowledge and experience in ecological economics, sustainable land management, wildlife ecology, and biodiversity policy helped compile an initial survey developed for established ranchers. In an additional workshop, this survey was adapted for new beneficiaries on land reform farms by three of the original experts and two additional experts. Three of the experts in this second team work in government departments directly implementing land reform and/or biodiversity economy programmes. The draft questionnaire was then piloted on three representative land reform sites selected by the Chief Environment Scientist of the land reform programme in the Eastern Cape province (S. Shwababa) between 17 and 21 May 2021 and subsequently revised. Of relevance to this study, the questionnaire included questions about current activities, challenges that prohibit land reform beneficiaries from participating in the wildlife economy and critical success factors necessary for land reform beneficiaries to succeed in the wildlife economy.

The South African National Biodiversity Institute then hired four field assistants (from an ecological / natural science background) to conduct the interviews. The expert group trained these assistants to conduct the questionnaire by developing a script to accompany the survey and conducting a series of training sessions before the interviews. This training included covering the scientific background of the project, social process learning techniques, and conflict resolution. Three out of four field assistants spoke isiXhosa (the predominant first language of the interviewees) to facilitate communication during the survey. Additionally, two programme officers from DALRRD accompanied the team during the interviews to facilitate contact with the interviewees and mediate potential administrative issues.

The quantitative results of the survey were analysed using R (R Core Team 2023). We used chi-squared tests to assess differences in frequencies. We used logistic regression (estimated using maximum likelihood) to assess whether financial viability was dependent on time since the establishment of the enterprise, and we fitted a linear regression model to predict whether the time since establishment influenced the proportion of revenue from wildlife-based activities. We use a t-test to compare the size of the land reform beneficiary properties to property sizes of established wildlife ranches implementing different economic activities, and a one-way analysis of variance test and post-hoc Tukey to assess the effect of ‘business model’ category on property size.

6.4 Results

The 19 surveyed land reform properties within the Amathole BEN represented a total area of 47,371 ha. There is a wide range in property size (mean: 2,632 ±SE 645 ha, median: 1667 ha). Only one property reported an increase in size since its establishment (expanding by 1,027 ha). Most (84%) of the properties were leased land from the state, and most were under 30-year leases, where beneficiaries had been managing the land for an average of 7 ± 3 years (2009-2021). Most properties (89%) were already generating revenues (Table 6.1). Of properties that are currently commercially active, the most important revenue-generating activity was cattle farming (mean ± SE: 43 ± 9%), where cattle together with goat (15 ± 5%) and sheep (11 ± 4%) farming, as well as cultivation (14 ± 7%) (Table 6.1), accounted for 95% of total revenue across properties (Figure 6.2). Most (84%) new market entrants had game populations on their properties. However, only 32% of properties were currently generating revenue from wildlife, of which biltong hunting predominated (3 ± 2% average revenue; range: 5–20%), and only two properties were currently practicing trophy hunting (range: 3-5% revenue contribution). No properties currently earned any revenue from ecotourism or game meat sales (Table 6.1, Figure 6.2). Wildlife-based activities only accounted for 4.3% of the total revenue across properties. Overall, 55% of properties were reported as being financially self-sustaining. A logistic regression (estimated using maximum likelihood) was used to assess whether financial viability depended on time since the establishment of the enterprise~~,~~ which was not significant and explanatory power was low (Tjur's $R^2 = 0.03$; Table 6.2). Similarly, I fitted a linear regression model to predict whether time since establishment influences the proportion of revenue from wildlife-based activities, which was not significant ($R^2 = 0.03$, $F_{1, 17} = 0.45$, $p = 0.5$; Table 6.2).

Extension officers developed business plans for the beneficiaries from DALRRD (47%) or through consultants (24%). In contrast, the enterprise owners themselves developed 18% of the business plans, and only 2 sites (12%) were assisted by an industry-specific mentor (Table 6.1). Despite ecotourism currently not contributing to any property’s revenue, 90% of respondents wanted to establish ecotourism enterprises, followed by 84% wanting to establish trophy hunting ventures. In 84% of properties, ecotourism and trophy hunting were both listed as future endeavours. Overall, most properties listed multiple wildlife-based enterprises in their future business development, leading to homogenously distributed future aspiration of wildlife-based economic activities across properties (future economic activities range from 74-89% across all properties (Table 6.1, Figure 6.2).

Table 6.1. Enterprise development patterns and current (2021) revenue contributions from economic activities for nineteen land reform beneficiaries in the Eastern Cape province of South Africa. Property names are excluded to protect personal information.

Enterprise name	**Time since establishment (years)**	**Business plan developer**	**Financially viable**	***Proportion of revenue (%)***										
				Cultivation	**Cattle**	**Goats**	**Sheep**	**Chicken/ pigs**	**Biltong hunting**	**Trophy hunting**	**Game meat**	**Game breeding**	**Ecotourism**	**Events**
P1	1	Enterprise owner(s) and DALRRD	*Did not answer*	0	75	25	0	0	0	0	0	0	0	0
P2	4	Enterprise owner(s) and industry mentor	Yes	80	0	0	0	0	20	0	0	0	0	0
P3	5	Enterprise owner(s) with consultant	No	0	100	0	0	0	0	0	0	0	0	0
P4	5	*Did not answer*	No	0	30	20	50	0	0	0	0	0	0	0
P5	5	Enterprise owner(s) with consultant	No	70	25	0	5	0	0	0	0	0	0	0
P6	5	Enterprise owner(s) and industry mentor	Yes	2	80	6	8	4	0	0	0	0	0	0
P7	6	Enterprise owner(s)	Yes	0	20	20	35	0	20	0	0	0	0	5
P8	7	Enterprise owner(s) and DALRRD	Yes	0	65	0	30	0	0	5	0	0	0	0
P9	8	Enterprise owner(s) with consultant	No	0	100	0	0	0	0	0	0	0	0	0
P10	8	DALRRD	Yes	0	0	60	35	0	5	0	0	0	0	0
P11	8	Enterprise owner(s)	No (not earning revenue)	0	0	0	0	0	0	0	0	0	0	0

P12	9	Enterprise owner(s) with consultant	Yes	0	70	30	0	0	0	0	0	0	0	0
P13	9	*Did not answer*	Yes	0	20	50	20	0	10	0	0	0	0	0
P14	9	DALRRD	No	0	100	0	0	0	0	0	0	0	0	0
P15	10	Enterprise owner(s) and DALRRD	Yes	50	30	10	10	0	0	0	0	0	0	0
P16	12	DALRRD	Yes	0	20	70	0	0	10	0	0	0	0	0
P17	12	Enterprise owner(s)	No	0	75	0	22	0	0	3	0	0	0	0
P18	12	DALRRD	Yes	70	0	0	0	30	0	0	0	0	0	0
P19	32	Enterprise owner(s), DALRRD and SANParks	No (not earning revenue)	0	0	0	0	0	0	0	0	0	0	0
Ranches currently undertaking (%)				*26*	*74*	*47*	*47*	*11*	*26*	*11*	*0*	*0*	*0*	*5*
Average revenue contribution (%)				*14.3*	*46*	*15*	*11*	*1.8*	*3.4*	*0.4*	*0*	*0*	*0*	*0.3*
Standard error (%)				*6.6*	*8.7*	*5.1*	*3.6*	*1.6*	*1.5*	*0.3*	*0*	*0*	*0*	*0.3*
Ranches planning to undertake (%)				*58*	*21*	*26*	*21*	*16*	*74*	*84*	*79*	*84*	*89*	*63*

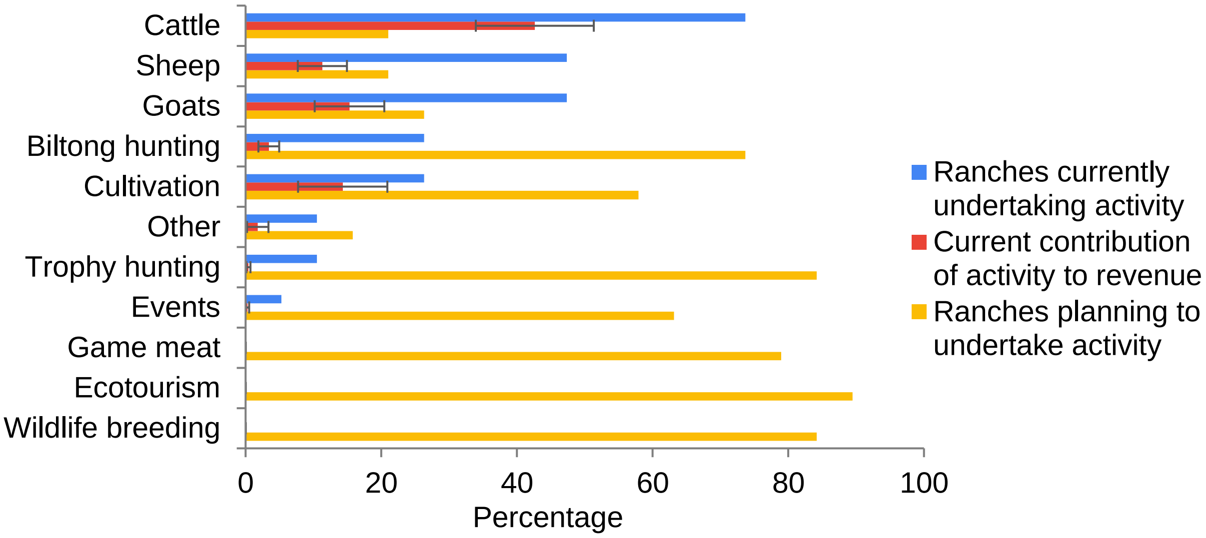

Figure 6.2. Percentage of ranches (n = 19) currently undertaking and planning to undertake different revenue-generating activities, as well as the average (+-SE) percentage that each activity currently contributes to total revenue. “Other” includes pigs and chickens.

Table 6.2. Linear and logistic regression results for modelling the impact of time since establishment on financial viability and proportion of wildlife-derived revenue.

Predictor	**Estimate ± std error**	**95% CI**	**z / t value**	**p value**
Logistic regression of financial viability against time				
(Intercept)	0.8±0.9	[-0.9, 3.2]	0.9	0.37
Time since est.	-0.1±0.1	[-0.3, 0.1]	-0.7	0.46
Linear regression of wildlife revenue against time				
(Intercept)	5.3±2.7	[-0.3, 10.9]	1.9	0.06
Time since est.	-0.2±0.2	-[0.7, 0.4]	-0.7	0.51

I then compared the size of the land reform beneficiary properties to the property sizes of established wildlife ranches implementing different economic activities (Figure 6.3, Table 6.3). The property sizes of the land reform beneficiaries, mixed cattle and wildlife farms and properties offering ecotourism day visits are, on average, around half the size of those enterprises focusing on biltong hunting, trophy hunting, game breeding and overnight ecotourism. A one-way analysis of variance testing the effect of ‘business model’ category on property size revealed a significant effect of business model ($F_{6,\ 178} = 2.32$, $p = 0.04$), where a post-hoc Tukey revealed that "LR Beneficiary", "Eco-day" and "Mixed ranches" categories were significantly different from the "Eco-night" category ($p < 0.05$). Additionally, 54% (N = 64) of the established sector sample (N = 118) conducted overnight ecotourism, 49% conducted trophy hunting, 55% conducted biltong hunting, 50% conducted game breeding, 34% conducted cattle farming and 31% conducted day-visitor ecotourism.

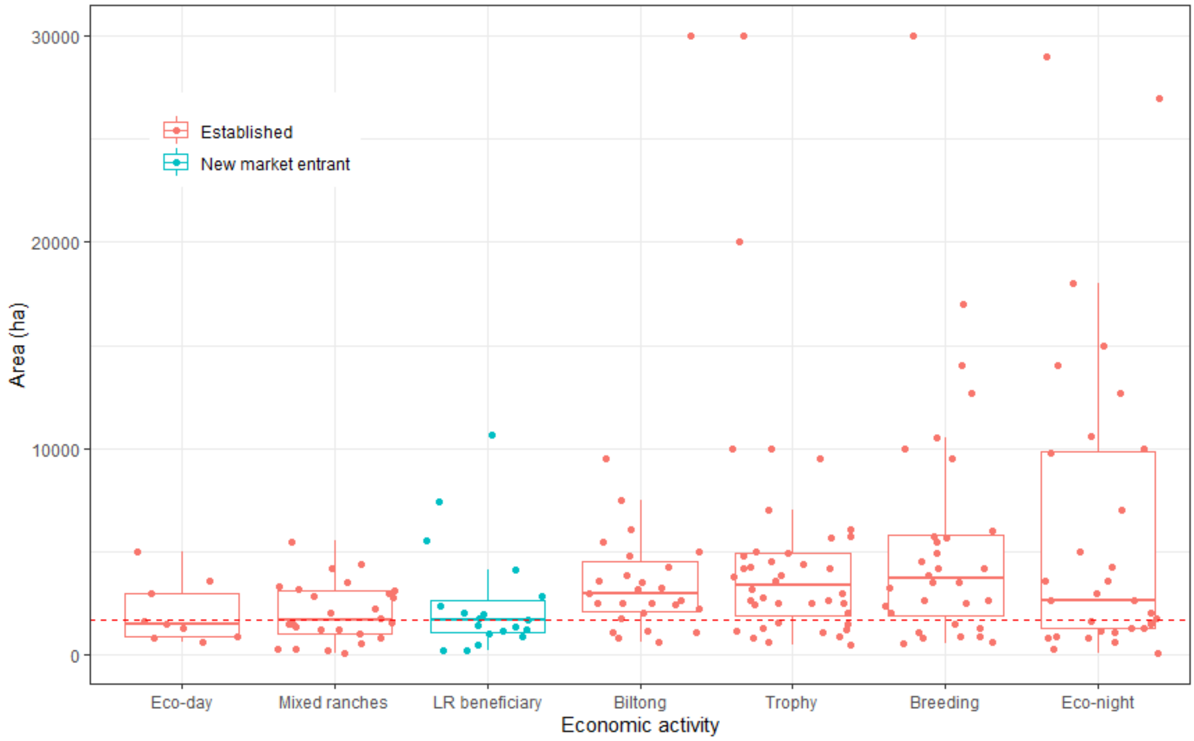

Figure 6.3. A comparison of the property sizes between land reform beneficiaries (new market entrants) and the established wildlife economy business models split by primary economic activity (>10% of total revenue per business model). The property sizes of the land reform beneficiaries are generally lower than the established enterprises operating successful biltong, trophy, breeding and overnight ecotourism models. The red dotted line shows the median property size (1667 ha) for the land reform beneficiaries.

Table 6.3. Property size comparison of the new market entrants and the established wildlife ranches operating different economic activities. Means are reported with standard errors. Significant differences in size refer to a one-way analysis of variance.

Property type	**Sample size (N)**	**Mean ± SE size (ha)**	**Median size (ha)**
LR beneficiary*	19	2541 ± 616	1667
Mixed ranches*	26	2052 ± 281	1660
Ecotourism-day*	9	2031 ± 502	1500
Biltong hunting	27	4303 ± 1066	3000
Trophy hunting	40	4704 ± 857	3380

Property type	Sample size (N)	Mean ± SE size (ha)	Median size (ha)
Game breeding	32	5572 ± 1078	3686
Ecotourism overnight*	32	6079 ± 1334	2628

**significantly different from Eco-night (p < 0.05)*

Total employment density (jobs / ha) of permanent staff was slightly higher in the established sector but not significantly different from the new market entrants (t = 0.5, df = 26.6, p- = 0.63; Table 6.4). The density of seasonal workers was also similar (Table 6.4). The absolute number of people employed per ranch, however, was significantly higher in the established sector (27±7 compared to 5±7 people employed / enterprise for established and new market entrants, respectively (t = 3.3, df = 114, p = 0.001), with a median value also double that of the new market entrants. The proportion of women employed was also significantly higher in the established sector (27 ± 2% compared to 11 ± 5% for established and new market, respectively (t = -3.2, df =23, p < 0.01; Table 6.4). When comparing a subset of the established ranches that are within the property size range of the new market entrants (=< 7400 ha), the patterns remained similar, with the established ranches in this size class employing almost double the number of people (mean in group established enterprise = 10, mean in group new market entrant= 5; 95% CI [1.59, 8.54], t = 2.9, p < 0.05) and almost triple the number of women (t = 1.8, df =16, p = 0.04; Table 6.4, Figure 6.4).

Table 6.4. Employment patterns between new markets entrants and the established wildlife ranch sector. Employment was split into permanent and seasonal workers. The new market entrant data was also compared to a subset of the established industry data where property sizes were similar. All values are mean ± standard errors; and median.

Employment type	Employees / enterprise	Employees / 100 ha	Proportion of women employees (%)
New market entrants (N = 19)			
Permanent	5 ± 7; 4 *	0.44 ± 0.07; 0.21	11 ± 5; 0 *
Seasonal	3 ± 0.9; 1.5	0.28 ± 0.14; 0	42 ± 10; 33
Established sector (N = 110)			
Permanent	27 ± 7; 9 *	0.59 ± 0.14; 0.25	28 ± 2; 28 *
Seasonal	9 ± 2.7; 1 *	0.22 ± 0.05; 0.02	37 ± 5; 30
Established sector <= 7400 ha (N = 84)			

Permanent	10 ± 1.3; 8 *	0.52 ± 0.12; 0.29	27 ± 2; 29*
Seasonal	5 ± 1.2; 0	0.24 ± 0.04; 0	38 ± 5; 23

**significantly different between new market entrants and established sector*

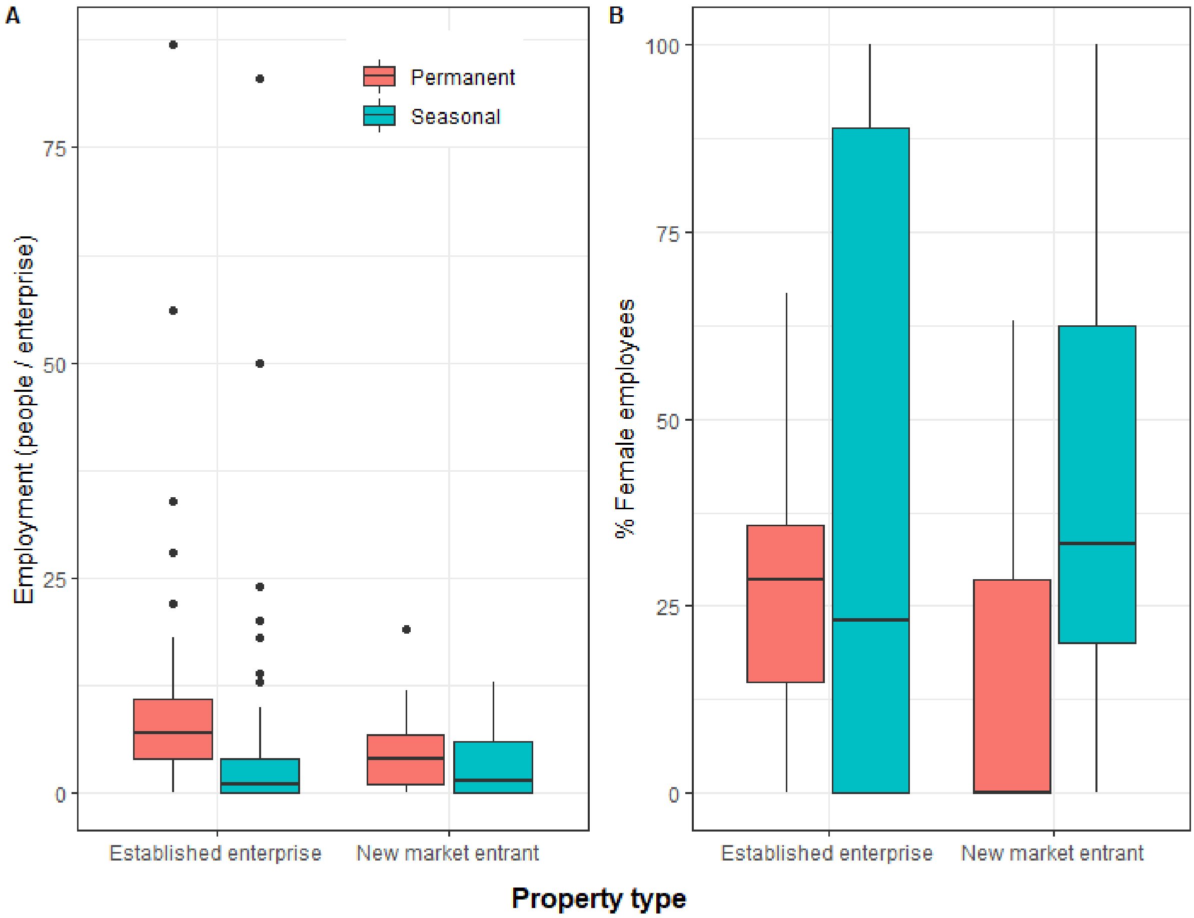

Figure 6.4. Comparison of employment characteristics between established sector and new market entrants. Employment data for established sector are for all properties =< 7400 ha (maximum of new market entrant sample) to control for effects of area size on employment. The established sector employs significantly more permanent employees per enterprise and, on average significantly more women. The differences for seasonal workers are not significant but new market enterprises tend to have more seasonal than permanent employees.

The most abundant animal stocks were sheep (33%), cattle (23%) and goats (23%). Collectively, livestock comprised 80% of the total abundance of animal stocks. Across properties, there were a total of 24 species of wildlife and three extra-limital species (fallow deer *Dama dama*, Barbary sheep *Ammotragus lervia* and lechwe *Kobus leche*), where wildlife comprised 11% of total stocks and extra-limitals comprised 9% (mostly driven by one property with an estimated 800 fallow deer). The average wildlife population across properties was negligible compared to livestock (Figure 6.5, Table 6.5). The relative abundance of wildlife is an underestimate as some properties could not estimate population sizes for various wildlife populations (36 wildlife populations across 10 properties did not have a population estimate, compared to only two missing population estimates for livestock populations). When compared to the established sector, emerging ranches had significantly lower abundances of game than all the established ranch business models (wildlife-agriculture: W = 22, p = 0.003; wildlife mixed: W = 15, p < 0.001; trophy hunting: W = 32.5, p < 0.001; ecotourism: W = 5, p < 0.001), as well as conventional livestock farms (W = 45.5, p = 0.017; Figure 6.5,).

Additionally, of populations where noticeable declines were recorded (N = 12), 67% of these were wildlife populations compared to 25% for livestock populations. The primary reasons given for these declines were poaching of wildlife (80% of properties, n = 10 who answered). Commercial exploitation of wildlife by hunting outfitters (40% of properties, N = 10 who answered) who often do not keep records of the hunts, underpay for hunts (based on market prices in the established industry), or do not pay at all and cause population decline through overharvesting. Conversely, 70% of livestock populations were increasing where trends were compared to only 27% of wildlife populations recorded (N = 41).

Table 6.5. Population estimates for various livestock and wildlife species across sampled properties. Sheep, cattle and goats collectively account for 80% of total abundance, with wildlife populations accounting for at least 11% (no estimates were available for 36 wildlife populations).

Species / type	**Type**	**Total stock size**	**No. of properties present**	**Proportion of abundance (%)**
Sheep	Livestock	3905	11	32.8
Cattle	Livestock	2769	15	23.3
Goats	Livestock	2756	11	23.2
Fallow deer	Extra-limital	808	3	6.8
Springbok	Wildlife	429	6	3.6
Barbary sheep	Extra-limital	300	1	2.5
Blesbok	Wildlife	133	6	1.1
Kudu	Wildlife	101	6	0.8
Mountain reedbuck	Wildlife	100	3	0.8
Impala	Wildlife	77	4	0.6
Eland	Wildlife	50	3	0.4
Black wildebeest	Wildlife	50	2	0.4

Species / type	Type	Total stock size	No. of properties present	Proportion of abundance (%)
Plains zebra	Wildlife	47	3	0.4
Steenbok	Wildlife	40	1	0.3
Red hartebeest	Wildlife	40	1	0.3
Pigs	Livestock	97	2	0.8
Mountain zebra	Wildlife	30	1	0.3
Waterbuck	Wildlife	28	3	0.2
Ostrich	Wildlife	21	2	0.2
Grey rhebok	Wildlife	20	1	0.2
Common duiker	Wildlife	16	4	0.1
Common reedbuck	Wildlife	15	2	0.1
Bushbuck	Wildlife	15	2	0.1
Lechwe	Extra-limital	10	1	0.1
Bushpig	Wildlife	10	7	0.1
Bontebok	Wildlife	10	1	0.1
Horses	Livestock	7	1	0.1
Sable	Wildlife	6	1	0.1
Warthog	Wildlife	No estimate	4	
Nyala	Wildlife	No estimate	1	
Blue duiker	Wildlife	No estimate	1	

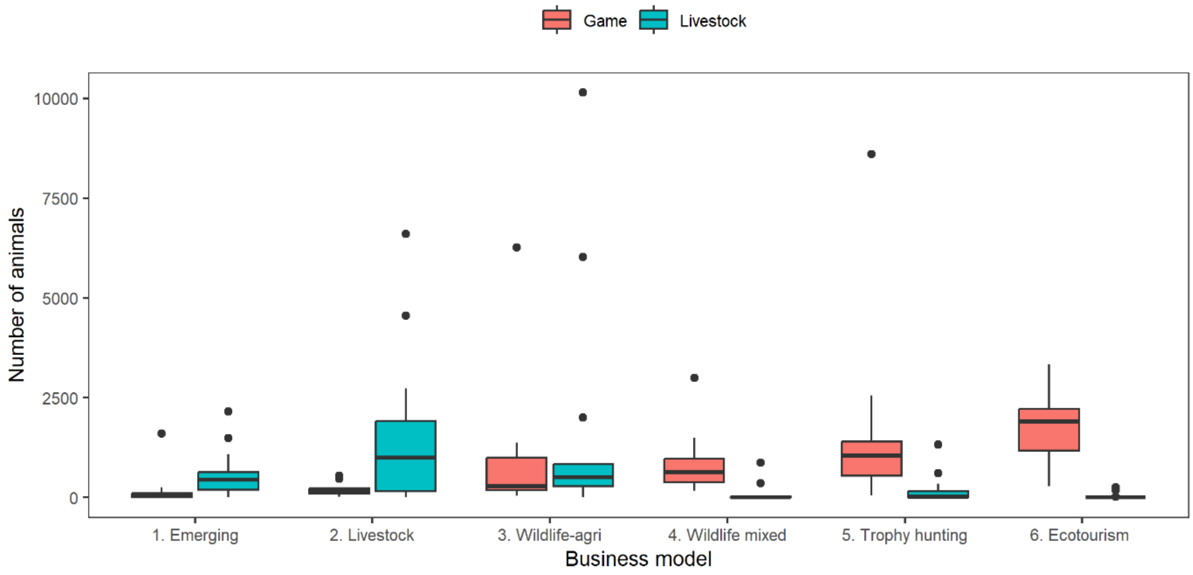

Figure 6.5. Comparison of game and livestock abundance between emerging ranches (1; land reform sites), conventional livestock farms (2), and the established wildlife ranch business models (3–6).

Of the existing infrastructure on the properties, most properties had drinking water for people and animals (88% and 82%, respectively) upon transfer (Figure 6.6). Many also inherited sheds (71%), grazing camps (58%) and game (79%). Only four properties (21%) had an existing perimeter fence, and three properties (16%) inherited vehicles and machinery when they started. As such, vehicles and machinery were the most-invested infrastructural category (53%) post-transfer, followed by drinking water and roads (35% and 29%, respectively). Only two properties (11%) had invested in perimeter fences post transfer. However, plans for infrastructure development focus on perimeter fences (79%), abattoirs (77%), game stocks (71%), and guest accommodation (65%) (Figure 6.6), which is directly linked to planned activities (trophy hunting, biltong hunting and ecotourism). While a third of properties (37%) were focused on future infrastructure investment needs (3 infrastructure types or fewer), the majority (63%) had a future wish list of 7–16 infrastructure assets.

Of the 13 properties (77%) that had invested in infrastructure since transfer, eight indicated that they had received help from the government for dams and irrigation systems (DRDLR), livestock fencing (DRDLR), vehicles and machinery (RECAP, One Household, One Hectare), drinking water (Department of Agriculture), boreholes, and the maintenance of a perimeter fence. Commonly mentioned opportunities included ecotourism (26% of properties) and agriculture (16% of properties). The most commonly mentioned challenges included a lack of infrastructure (fencing, accommodation, irrigation and water storage; 32% of properties), theft of property and animals (32%), electricity and water supply and cost (16%), lack of rain (16%) and problem animals (jackal *Canis mesomelas*, caracal *Caracal caracal*; 11%). Fifteen properties indicated that they needed support to develop further. The most pressing needs were help with fencing, game stocking and water infrastructure (47% or seven properties), whilst four properties (27%) indicated they needed help with alien clearing, bush encroachment and road networks. Two properties (13%) highlighted the need for skills development and training in game ranch management.

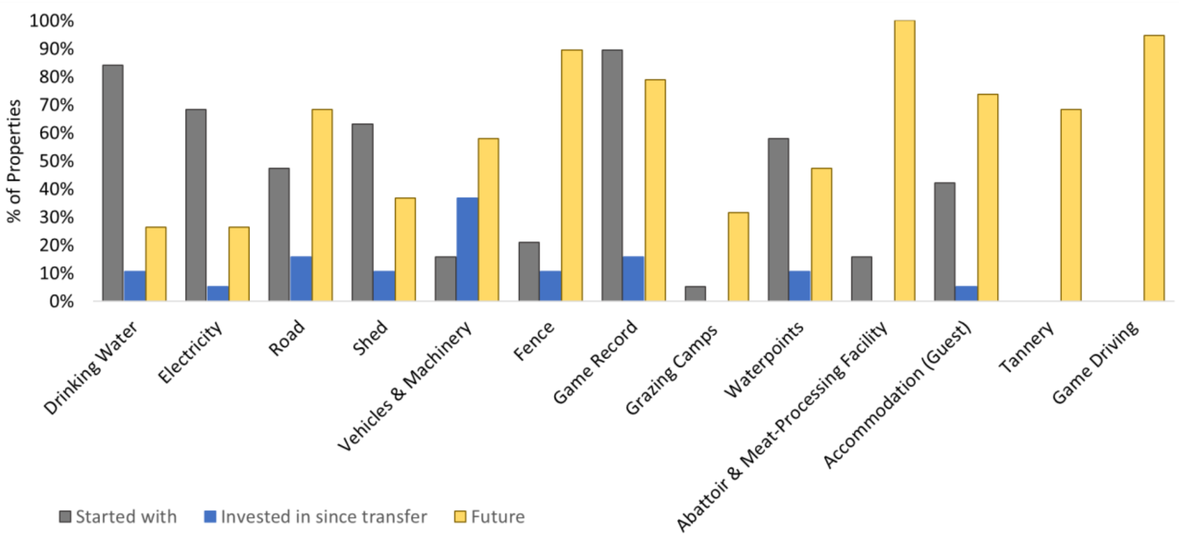

Figure 6.6. Infrastructure and assets inherited by the beneficiaries (grey bars) versus invested in post-transfer (blue bars) and planned investments (yellow bars). Most properties did not start out with perimeter fencing and were not currently able to invest in fencing, but this is the highest future need alongside abattoirs.

Based on the survey response and the institutional mapping conducted at DFFE and DALRRD, a decision-data pathway was designed to enhance the land reform and BEN programmes (Figure 6.7). The key decision-making processes in both programmes are identifying and prioritising land parcels for transfer to beneficiaries and identify and prioritise the investment support provided to that site. Key pathways were to improve geospatial information systems, develop screening tools for assessing the viability of wildlife economy business models, and monitor changes over time. Related to this was the use of empirical data to define the viable business models from the established sector so that they can be converted into open-access knowledge products and decision-support tools to guide beneficiaries' investment and channel impact investment effectively.

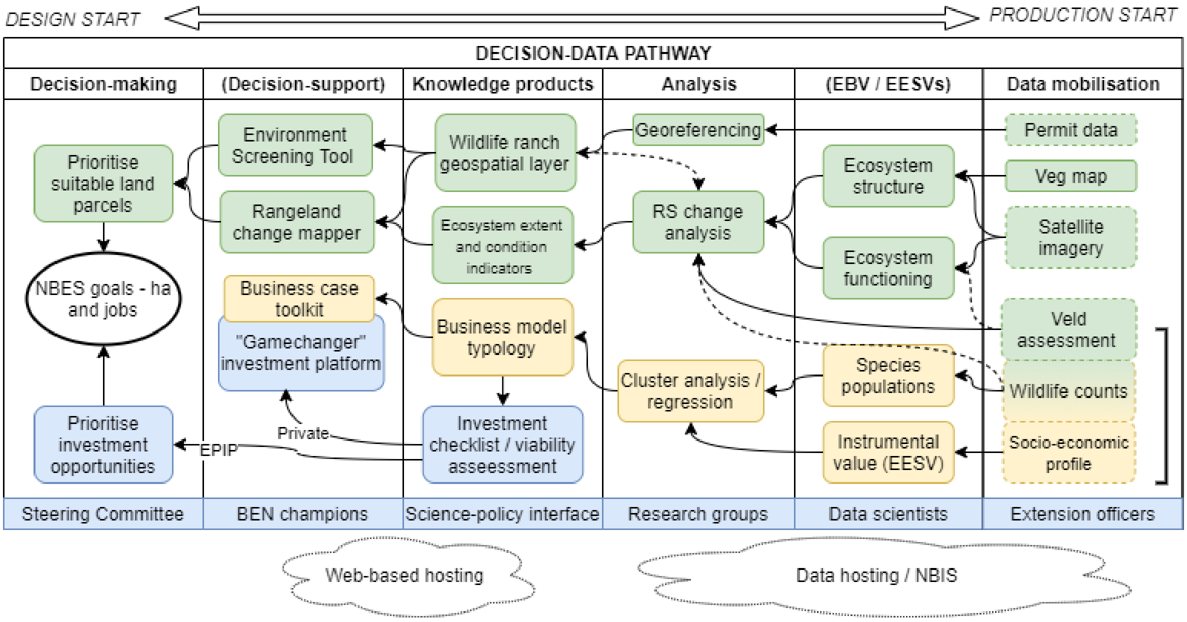

Figure 6.7. A decision-making to data mobilisation schematic pathway for mainstreaming wildlife-based land-uses into the land reform and biodiversity economy node (BEN) programmes. This pathway was developed in collaboration with DFFE and DALRRD and structures SANBI's specific inputs into HLP Goal 2 on transforming the wildlife sector. EBV / EESVs refer to the Essential Biodiversity Variable and Essential Ecosystem Service Variables data cube frameworks for standardising analysis-ready datasets.

6.5 Discussion

These results showed that wildlife-based activities within existing enterprises within the land reform programmes were very underdeveloped, representing only 4% on average in revenue generated on these properties. Only 32% of properties were generating any revenue from wildlife. This lack of investment in wildlife-based land uses was largely related to a lack of funding, infrastructure, knowledge and skills, and government support. Farmers predominantly invested in basic infrastructure such as roads and drinking water and identified the lack of fences as a barrier to the development of wildlife-based activities on their properties. This finding presents a policy incongruity in that the government is pursuing a policy of dropping fences to encourage large conservation areas (DFFE 2020b), but fences are necessary to support enterprise development. Whilst many enterprises indicated ambitions for ecotourism, comparisons with established wildlife ranches suggested they were more similar in size to established ranches that perform mixed-farming and mixed wildlife-based activities, but provided relatively fewer jobs to women and permanent positions than in established enterprises of the same size. These findings have several implications for the development of the wildlife economy in land reform programmes, and the achievement of key biodiversity, economic, equity and sustainable development goals.

The first obvious point is that new entrants to the wildlife economy don't have the capacity to survive and thrive in South Africa's wildlife economy. This trend is not unique to our study – Mokotjomela and Nombewu (2019) found only 21% of new market entrants had game farming capacity, and Mtero et al. (2019) found that 50% of enterprises who had entered the wildlife industry subsequently reverted. Similarly, in the Waterberg region of the Limpopo province, redistribution farms similarly underwent an attribution of wildlife ranching activities (from 5% of farms to none) (Netshipale et al. 2017).

We suggest that the primary barrier is a lack of understanding of appropriate wildlife economy business models and their associated asset and skills requirements, which leads to inappropriate and ineffective investment by the government. For example, despite ecotourism not currently contributing any revenue and trophy hunting contributing very little revenue, 89% and 84% of respondents wanted to establish these enterprises. Additionally, in 84% of properties, ecotourism and trophy hunting were both listed as future endeavours, which seem to be mutually exclusive business models in the established sector (Clements et al. 2016b). In the established sector, only 54% and 49% of enterprises practice overnight ecotourism and trophy hunting, respectively. Additionally, the median size of the land reform properties is significantly below that generally required to support overnight ecotourism and trophy hunting business models (as well as game breeding models) compared to the established industry.

Interestingly, the maximum size of our new market entrant sample (7,400 ha) is similar to the sample in Mokotjomela and Nombewu (2019), which indicates the size range of new market entrants in the Eastern Cape is robust. Mokotjomela and Nombewu (2019) also found that the larger properties that could support more specialised game farms fell outside the Amathole-Great Fish BEN, which casts doubt on the BEN's ability to deliver transformation without investment to expand property size. Similarly, other research has confirmed that smaller farms (typically < 500 ha) had the most frequent incidence of mixed farming (von Solms & Merwe 2020). In our sample, only one property increased in size since establishment, limiting the feasibility of more specialized wildlife economy business models, suggesting that insufficient investment into wildlife-based enterprises has taken place, perhaps resulting from inadequate business plans and lack of revenue generation. Additionally, most of the properties are in marginal areas with poor road infrastructure, so competing with more established ecotourism clusters closer to main urban centres, such as the Indalo Group of properties, would be a challenge.

It is also questionable whether attempting to establish overnight ecotourism and/or trophy hunting operations is desirable, given that one third of high-end overnight ecotourism models are financially unviable (Clements & Cumming 2018), and that these business models may require specialized and focused infrastructure and skills development (Clements et al. 2016b). The emphasis on ecotourism models by the government is understandable as these are conceptually similar to the biodiversity stewardship programme that seeks to expand the protected area estate through non-state actors and thus contribute to the mandate of DFFE (Barendse et al. 2016; Rawat 2017; Wilson et al. 2018; Cockburn et al. 2019). Consequently, the integration of land reform into the biodiversity economy is dominated by biodiversity stewardship that intends to create 'meaningful benefits' for land reform beneficiaries through the development of 'new protected areas' (SANBI 2020). However, such models might undermine community enterprises by restricting access to land and natural resources (Kepe et al. 2005) and pushing communities into models that restrict their ability to adapt. Furthermore, the high capital outlay of establishing ecotourism and/or trophy hunting enterprises (Clements & Cumming 2018), combined with the dependence of these models on foreign patrons (Mbaiwa 2005), make the enterprise more prone to fail (Lindsey et al. 2020). This is similar to one of the reasons land reform based on agriculture transfer often fails – the focus by the government in developing large-scale, capital-intensive agribusiness primarily aimed at export markets (Hall & Kepe 2017; Rusenga 2020).

Returning to a more localised and resilient production system that mixes both cattle and wildlife ranching and responds to domestic markets might create a win-win for community upliftment and smart investment. In our sample, half the enterprises reported being financially self-sustaining, where cattle farming was the most important contributor to revenue generation (mean±SE: 43±9% per enterprise). Cattle are not just production units for communities but are interwoven into social relationships and cultural constructs that connect people to the land and place (Hornby & Cousins 2019). Thus, replacing cattle with wildlife could undermine a key pillar of well-being, and retaining cattle in the landscape would enable cultural continuity, allowing time for people to develop relationships with novel ecosystems rather than feeling excluded from them even if they are slightly wealthier as a result (Achieng et al. 2020). Cattle production, while an important livelihood, may not always be sufficient driver of rural development as the opportunity for skilled employment and enterprise development is low (Chaminuka et al. 2014) and climate change is increasingly making cattle production less viable (Rahimi et al. 2021). To solve these problems, rather than attempt to rewild livestock farms wholesale, which risks alienating the stakeholders and reversing the gains they have made, wildlife could be integrated into existing agricultural models progressively to enhance resilience and revenue generation and enable the development of specialised wildlife economy enterprises in future. The average size of land reform properties is most similar to mixed agriculture and wildlife enterprises as well as properties that offer day-visitor ecotourism. The next size class up that new market entrants are most similar to is biltong hunting operations, which is reflected in our sample by almost all wildlife-based revenue being generated by biltong hunting.

Perhaps mixed livestock and wildlife forms should be the fundamental base model for land reform beneficiaries and communities, where local biltong hunting and day ecotourism are the main additional revenue generating activities to augment livestock farming. Cattle farming would provide a reliable cash flow in wildlife market dips and can also be used as a tool to achieve sustainable land management goals on small properties, such as through rotational grazing, where the combination of well managed cattle and wildlife may help restore ecosystem functioning and thus productivity (Hempson et al. 2017; Keesing et al. 2018; Guyton et al. 2020). As the established industry is more orientated around international hunters, new market entrants could fill a market gap by providing affordable local hunting options that help them to become viable quickly and also improve food sovereignty, which will be especially important in the coming decades as climate change causes increasing food insecurity (Dube et al. 2013). The value of the local South African biltong hunting and game meat market remains to be realised, and a lack of consistent game meat supply is a key barrier to unlocking this industry (DFFE 2021a). Biltong hunting could be easily combined with local ecotourists looking for day excursions for hiking or biking. Focussing on biltong hunting within a mixed farming system could also enable rural entrepreneurs to develop and sell the hides from wildlife, where higher prices could be fetched by processing these hides into tanned leather for further product development (DFFE 2020a). The government could assist these ‘emerging hide merchants’ by constructing additional tanneries, of which only seven currently exist in the Amathole-Great Fish BEN (DFFE 2020a).

Initial business models based on mixed ranches might also help to improve the social inequality currently experienced when converting to high-end ecotourism or trophy hunting wildlife enterprises (Spierenburg & Brooks 2014; Pasmans & Hebinck 2017; Spierenburg 2020; Achieng et al. 2020; Thakholi 2021). This could be done by returning access to land to farm dwellers, reducing the need for heavily electrified game fences, and providing cultural continuation and conservation through livestock ranching. Returning to wildlife enterprise models based on sustainable use, local production and consumption within mixed agricultural and wildlife contexts might thus be more effective in delivering the benefits of the wildlife economy to new market entrants.

The wildlife assets on new market entrant properties need better protection and management. Property sizes are generally smaller than the established sector, but wildlife populations across the new market entrants are far lower compared to livestock. While most (84%) beneficiaries had game populations on the property, it is doubtful that these exist in sufficient numbers to create viable revenue-generating activities. When wildlife is present, the populations often decline due to poaching and unscrupulous hunting outfitters. Poaching is considered a ‘contestation’ by some authors (for example, Pasmans & Hebinck 2017), but in our sample, this issue transcended racial politics and impacted the beneficiaries of transformation. Viable businesses cannot be created if the assets are at risk, and if the enterprises cannot run, they cannot fulfil their employment potential. To equip beneficiaries to combat poaching, perimeter fences and basic game management skills are urgently needed. Only four properties had perimeter fences on transfer, and only two managed to invest in them since transfer, while most properties (79%) listed perimeter fences as a key future infrastructure need. This is a key infrastructure for government investment to reduce poaching losses. To combat the exploitation by professional outfitters, access to market knowledge about average hunting prices and basic game management and hospitality skills, are needed. These skills should be integrated into a training programme or decision support tool accessible to all beneficiaries online and ideally nurtured through mentorship with an established rancher in the areas. Wildlife declines are exacerbated by a lack of adequate game population estimates by many properties, and game census techniques should be learnt so that sustainable hunting quotes can be put in place. Keeping a game record was listed as a key need of the beneficiaries. Underscoring this importance, most beneficiaries listed investing in game as a future investment, but this may be ineffective if perimeter fencing and basic game management skills are not in place.

The list of future revenue plans and asset needs is too sprawling and unstructured. Most properties listed multiple wildlife-based enterprises in their future business development, leading to homogenously distributed future aspirations of wildlife-based economic activities across properties (future economic activities range from 74-89% across properties). This finding demonstrates a lack of knowledge on the feasibility and investment required to establish successful enterprises, by both the beneficiaries themselves and the consultants and extension officers who help develop the business plans. Most of the business plans were developed by DALRRD and/or the enterprise owners themselves (65% in total), where there is likely to be less awareness of and experience in developing wildlife-based enterprises. It might lead to more viable enterprises that integrate wildlife and agricultural enterprises if beneficiaries are paired with industry mentors (currently, only two sites have industry mentors) through market-based incentives such as the mentor owning shares in the new business, or through government providing funds to the mentors to support new enterprises for three to five years (Maka & Aliber 2019). This may also help overcome the lack of inclusivity that currently characterises the industry.

The lack of clear guidance or direction in the business model is also reflected in the scatter-gun approach to infrastructure needs, where most respondents had a wish list of over seven asset types. Similarly to the high evenness of business models being pursued by beneficiaries, the infrastructure and support needed is also even amongst classes, indicating that the beneficiaries themselves may not have any clear goals for business model development and the investment from government and private investors risks being dissipated into unfocussed initiatives. This needs some prioritisation to be effective (beyond basic perimeter fencing). A critical research question will be to assess path dependency in establishing wildlife-based enterprises. Does initial property size limit the enterprises that are viable in the area? Do certain enterprises like mixed farming and biltong hunting form keystone enterprises in unlocking more high-end models such as trophy hunting, game breeding and ecotourism? For example, Clements and Cumming (2018) found that after 13 years, initial land and infrastructure investments limited the ability of the enterprise to develop more complex ecotourism and/or hunting business models.

Currently, several national government programs and projects are intended to expand inclusive wildlife economies. The primary financial instrument of land reform, the Recapitalisation and Development Programme (RECAP), seeks to create employment, enhance food security and farm production and improve market access, but the funding is often not directed at priority needs (Nenngwekhulu 2019). In parallel, DFFE is channelling investment into biodiversity economy nodes across the country to transform an additional 10 million ha into wildlife economy enterprises (DFFE 2016a). These investments include infrastructure (such as perimeter fencing), game donations, skills and mentorship programmes, veterinary services, land restoration and market access. For example, the national government is investing in a Biodiversity Economy Investment Platform to facilitate projects wanting to establish or expand wildlife enterprises (DFFE 2021b). There is little spatial overlap between land reform focus and biodiversity economy node designations (Mokotjomela & Nombewu 2019). The two branches of government have much to gain from cooperating in their investment strategy both spatially and through providing support through time, especially if wildlife ranching for new market entrants is orientated around establishing agroecological ‘working lands’ (Kremen & Merenlender 2018). The key to this will be for both departments to stop viewing private wildlife populations as “objects of conservation” and to start viewing them as “legitimate components of rural livelihoods” (Chaminuka 2013).

Both decision-making processes culminate in identifying suitable land parcels for transfer and prioritising investment into those land parcels based on the beneficiaries’ business plans. Our results show that there is common ground becausethe average new market land parcel is large enough to support mixed ranches where wildlife-based revenue generating activities, such as day-ecotourism, local hunting and game meat production, could bolster the goals of both programmes. Integrating local biltong hunting and game meat is compatible with livestock farming, as well as infrastructure for day-ecotourism models, such as walking trails, which will help to increase the resilience of the business model and employ more people from the community.

Considering the most pressing need is for perimeter fence construction, and the most pressing problems are poaching, and exploitative hunting operators, a prudent first phase of wildlife economy investment would be basic game management and fence maintenance skills. The RECAP programme could invest in game fences for beneficiaries as well as infrastructure related to agro-ecological enterprises, such as mobile abattoirs, meat processing facilities, cold rooms and tanneries. Investment into mobile abattoirs (rather than fixed abattoir infrastructure) could especially help to unlock the potential of game meat production for rural communities (DFFE 2020a). Wildlife enterprises could make RECAP more efficient by enabling the purchase of cheaper, less agriculturally suitable, and marginal land and restoring it through wildlife systems. This could be a two-for-one type of investment for RECAP. Simultaneously, it could allow DFFE to focus on developing game management skills, transferring game assets to beneficiaries, and improving market access through information products and decision-support tools through online resources, manuals and mentors.

Interventions are needed on both the beneficiaries' and the officials' sides. There needs to be more coordinated skills programmes and infrastructure investment for the beneficiaries, enabling new market entrants to start with mixed farms and progress to more niche wildlife business models as capital accumulates. This also includes access to market information on game auction prices, trophy animal prices, and trends. On the official side, better information systems are needed to screen the properties for the type of investment needed – depending on the size, area, and location of the property and the current skillsets of the beneficiaries.

If the government could work effectively to scaffold new wildlife-based enterprises to improve their long-term viability, it would pay dividends for employment and food security in rural areas (see Box 1 for a case study on the challenges and opportunities). Research confirms that traditional agricultural models show a long term trend of job shedding due to mechanisation and a shift to seasonal workers (Cousins et al. 2018), which is also reflected in our data. Yet there is no mention of the wildlife economy to mitigate these job losses through a diversified economic portfolio and a range of skilled labour. Taylor et al. (2020) show that game meat harvesting can be comparable in production to extensive livestock farms, and that wildlife properties employ more people per unit area than livestock farms. Our data support the employment potential of wildlife-based enterprises. Compared to established enterprises, new market entrants employ significantly fewer people on average (approximately half that of established enterprises currently), significantly higher proportions of seasonal (compared to permanent) workers, and only one third the number of women employed by the established sector.

Interestingly, these patterns did not hold when looking at area-based employment figures, where the unit area employment figures were similar between new market entrants and the established sector. This may indicate there is no linear relationship between area and employment potential (or that there are size thresholds on either end), However, that wildlife-based activities unlock more employment opportunities overall, possibly because of the more skilled and varied jobs running a wildlife ranch requires. Mokotjomela and Nombewu (2019), conversely, found that employment density was higher in new market entrants, but these data may be confounded by the fact that a quarter of the sample was practicing game ranching, which could account for the skewed employment distribution in their data. Pasmans and Hebinck (2017) assert that game farms do not generate new employment opportunities compared to crop cultivation and actually reduce the farm workforce because it requires "far less intensive management in the form of external inputs, labour and wages than, for example, pineapple farming". However, they do not provide any data to support their claims. Unsubstantiated claims like this are damaging, and it seems like there is personal bias from the authors.

While capital is needed initially to establish game farms, our results suggest that creating viable wildlife-based enterprises could lead to a local shift from seasonal to permanent employees, thus providing job security and long-term skill development. Employing more women has positive implications for meeting the Sustainable Development Goals and also creates more effective decision-making around natural resource management and equitable beneficiation of community members (Cook et al. 2019). Similarly, Achieng et al. (2020) found that Amakhala Game Reserve in the Eastern Cape has increased employment of women and average wages compared to former land uses. However, erecting fences has also made communities disconnected from the landscape and weakened cultural and social bonds (Mkhize 2014). This might be especially true for high-end ecotourism models that rely on privatising 'pristine wilderness', but perhaps less apparent for wildlife economy models based on mixed livestock and wildlife farms. Such trade-offs should be quantified and considered when investing in wildlife economy ventures and infrastructure.

Double Drift / Likhaya Lethu Wildlife Project case study

The Double Drift community formed a community property association (CPA) called Likhaya Lethu CPA (c. 1500 members) in 2016 to facilitate assistance from the Department of Forestry, Fisheries and Environment (DFFE) to develop wildlife economy enterprises on the 1400 ha farm Naudeshoek. This land was bought for the CPA through a land claim settlement through the Department of Agriculture, Land Reform and Rural Development (DALRRD). The CPA has an agreement with Eastern Cape Parks and Tourism Agency (ECPTA) to provide wildlife management training and 'start-up' game. The economic activities being pursued are breeding of both high-value and plains game for live sales, game meat and hide production, ecotourism and trophy hunting, to which end DFFE has allocated ZAR 6 million to build a perimeter fence (in progress), hunters' accommodation (in progress), animal holding boma (complete), and capacity development for the CPA (ongoing) (DFFE, 2020a).

While DFFE (2020,a) reports that 60 related jobs have been created in the CPA since 2018, survey data from the Sustainable Wildlife Economies Project (SANBI, unpubl. data, 2021), indicate that only 2 permanent employees are present and that no revenue is being reliably generated because there is 'no operational budget or staff offices'. Additionally, there is not enough funding to electrify the perimeter fence to enable permits that would enable high-end ecotourism and/or trophy hunting. With the assistance of DFFE, 4 of the donated buffalo (*Syncerus caffer*) from ECPTA were sold at an auction in 2019. However, before more can be introduced, the fence will need to be improved as one respondent explained: "Buffalo will be the first to challenge your fence so we want to have a double fence before we can be more aggressive with our introductions".

While this was an active livestock farm previously (until 2018), cattle were removed from the farm and game from ECPTA were reintroduced (currently 10 different herbivore species, including buffalo and eland *Taurotragus oryx)*. Continued lack of training and competence of game management in the community is causing the community to 'become restless'. Additionally, the CPA is suffering from poaching and current anti-poaching measures are ineffective. Currently, the CPA is looking for funds to send a youth member to train at the South African Wildlife College (SAWC).

A promising initiative seems to have stalled perhaps because the wrong business model has been pursued initially. It might have been better if the cattle had been retained to enable the CPA to keep a revenue stream active while infrastructure and assets for the wildlife economy were developed. This revenue could have been used to train youth at SAWC and to provide more community members with a basic livelihood. If game meat and hide production had been prioritised ahead of the specialised models of game breeding and international ecotourism and/or trophy hunting (the site is currently half the size needed to run these models based on the data in Table 3), less funding would be needed for the perimeter fencing and infrastructure that could be directed into production activities could also be supported by DALRRD. More CPA members could have been employed by now as a result. Combining both ecotourism and trophy hunting as revenue ambitions shows the confusion over what viable wildlife economy business models look like.

Perhaps given the ineffective start to the project, the CPA, in consultation with ECPTA, have redefined the revenue streams to focus on and have decided that a hunting lodge should be constructed along with an abattoir. This will be enabled through a capital investment from DFFE. This recalibration of business model follows closely the recommendation from this paper in focussing on local hunting and game meat production. Integrating cattle into the system could help to buffer market shocks from wildlife and improve rangeland condition through rotational grazing in a fenced system. Additionally, the plans for the 500 ha Ncaza Game Transformation Project will focus on game meat and hunting, which indicates this is seen as a robust strategy.

Having mixed farms as the basic unit would also help to streamline and focus initial business plan development. Business plans are initiated by the beneficiaries themselves and are used as criteria to determine the seriousness of the prospective beneficiary before they are allocated a farm and RECAP funding. Providing knowledge products and decision support tools on how to structure viable agroecological wildlife-based and mixed farming enterprises would help channel RECAP funding more effectively.

Working with the government would also help consolidate and streamline regulations regarding wildlife ranching. Currently, the sector sits awkwardly between the two departments, which is sometimes leveraged by the wildlife ranching industry to ‘play both sides’ (Kamuti 2014; Somers et al. 2020). Continuing to work in silos also risks creating outcomes that work against the respective mandates of the two departments. The erection of internal fencing to assume ownership of wildlife through ‘certificates of adequate enclosure’ is thought to be counterproductive to long-term conservation goals (Carruthers 2008; Blackmore 2020), and incentivising more extensive wildlife management systems is a prominent policy recommendation by a High-level Panel appointed to investigate the trade in iconic species (DFFE 2020b). How can new market entrants make this capital investment viable, and how can government balance the potential fragmentation caused by further fence erection? One medium could be to assist beneficiaries in establishing conservancies with surrounding lands (with state or private) whereby the partners agree to drop fences and develop cooperative management plans (Lindsey et al. 2009). This would have 3 benefits: 1) infrastructure investment into the conservancy would help to achieve economies of scale with more enterprises using and benefitting from it; 2) reintroducing animals into the conservancy would be more tractable as conservancies can apply for permits as an entity, negating the need for individual enterprises to apply; and 3) creating a larger area overall increases the diversity of business models available (as the data presented here show), helping the beneficiaries to expand their business models. Conservancies more amenable to communal governance structures.

Agricultural decision-makers are also concerned by their environmental colleagues declaring protected areas for biodiversity where no production can take place. Globally, pressure is being applied to governments, particularly in developing countries where much biodiversity remains, to protect at least 30% of the national land area to achieve the biodiversity conservation goals of the 2030 Biodiversity Framework (Target 3 of Goal A) (CBD 2021; Obura et al. 2021). This goal is reflected in policy design aspirations that seek to reduce management intensity of wildlife-based enterprises and move towards more extensive systems (DFFE 2020b). However, it may be counterproductive to push communities into developing extensive, ecotourism-focussed enterprises from inception without considering the viability of such enterprises and the impacts on local cultural continuation, skillsets and food security.

Coincidentally, the South African government is aiming to redistribute 30% of land by 2030 for agricultural and land reform objectives (DRDLR 2020). These two policy programmes could be viewed as being counterproductive, especially if livestock and crop agriculture are the primary land uses pursued by land reform. However, integrating the wildlife-based land uses into this national land redistribution programme enables production landscapes to be maintained whilst also potentially contributing to the conservation estate and sustainable agricultural practices under the Land Degradation Neutrality framework. For example, reintroducing wild herbivores to landscapes has been shown to increase perennial grass biomass (Kraaij & Milton 2006; Keesing et al. 2018), reduce bush encroachment (Guyton et al. 2020), and sequester more soil carbon (Sitters et al. 2020). As many areas targeted for land transfer have become degraded through livestock grazing, rewilding these areas might provide greater socio-economic opportunity and restore rangeland for greater long-term productivity. Similarly, livestock could be used to achieve sustainable land management goals through regenerative grazing practices to enhance the productivity of the overall ecosystem. Approximately 354,000 ha of land within the Amathole-Great Fish BEN is degraded (DFFE 2020a), which represents a key opportunity for the environment and agriculture to collaborate on restoration through the wildlife economy,

There is a gap in thinking through solutions at the nexus of biodiversity and climate change – particularly in how restoring ecosystem service functionality can enhance climate mitigation solutions (Pörtner et al. 2021). Most biodiversity policy is still centred on conserving the ‘feature’ and adding these to the conservation estate (i.e. OECMs). In South Africa, the Land Reform and Biodiversity Stewardship Initiative (LRBSI) considers only those land reform projects in areas of ‘biodiversity importance’ (SANBI 2020). However, this will alienate stakeholders not in biodiversity priority areas and will not achieve the restoration scales needed for nature-based solutions to contribute to carbon drawdown. As such, conservation policy makers should consider biodiversity as ecosystem services more widely (Clements et al. 2021).

Coordinating programmes in a spatially explicit context could also unlock the viability of more specialised business models. For example, high-end ecotourism models are more viable if developed in partnership with existing ecotourism nodes, such as has happened in the Associated Private Nature Reserve area adjoining Kruger National Park (Chidakel et al. 2020). In this case, only one of our sampled sites might have the potential to develop a viable ecotourism venture as it exists on the edge of Addo Elephant National Park, where park management is actively involved in developing business plans and providing game, which includes conducting veld assessments to donate appropriate stocking rates of game and in-sourcing game drives (SANParks, pers. comm.). Interestingly, despite this site being in existence for the longest, it currently does not draw any revenue from any commercial activities, showing how difficult it is to establish successful ecotourism ventures. Geospatial information systems must be developed with shared user access to better coordinate between the two departments through spatial planning. Our current sample of sites are not digitized for use in planning software, and the cadastral information is not captured systematically or available on any web-based portal. This can cause inaccurate estimates of investment needed based on the size of the area and / or environmental suitability and prevents monitoring of restoration impacts over time or integration into spatial development frameworks. Focussing on mixed farms and biltong hunting enterprises as the common denominator between the two departments would help structure investment and skills programmes within the BENs and create a shared monitoring and evaluation framework.

Finally, all stakeholders must open up new spaces, dialogues and models of wildlife ranching that promote transformation. To date, the wildlife industry has become dominated by a discourse that sees wildlife on private land as supporting conservation objectives through various regulations and industry trying to push back against these regulations to promote a free market (Snijders 2014). To transform the industry, government should create institutions and communities of practice that view wildlife as an asset to achieve various socio-economic and land restoration goals, rather than as a reporting requirement. Overall, 72% of projects that have received RECAP funding are still not viable, primarily because there is a lack of mentorship and access to relevant markets and a lack of credit / investment to expand enterprises (Ntlou 2016). Working together across government departments and civil society organisations can provide the necessary scaffolding and smart decisions to make investments worthwhile. For example, one of the key policy interventions listed by Mtero et al. (2019) is to create a decentralised and demand-driven land identification process. Poor people lack the resources and knowledge to independently identify and apply for land, so developing screening tools that assist district-level land reform structures and communities in identifying suitable land parcels for various land uses would enable a more participatory and ultimately successful process. Additionally, making the process spatially explicit would help to prioritise the municipal integrated development plans (IDPs) into land identification, strengthening the win-win between biodiversity and agriculture. Additionally, systematic and updated beneficiary databases are required to ensure structured and transparent allocation processes. Maintaining a geospatial portal linked to beneficiary details would be helpful.

Wildlife ranching has not been adequately integrated into land reform and agricultural programmes, and thus, the potential for rewilding rangelands as a tool to improve biodiversity and socioeconomic benefits is stultified. These beneficiaries stand to gain additional revenue streams and job creation opportunities. For example, only one-third of properties were currently generating revenue from wildlife, and the average revenue contribution was low (range in revenue proportion from biltong hunting: 5–20%; range in revenue from trophy hunting: 3–5%). The number of total jobs and proportion of women could also stand to double and triple, respectively. However, despite not having integrated wildlife enterprises, about half the properties are still financially viable, so such enterprises may require considerable effort to enter a new social-ecological system that has the potential to increase revenue streams and employment but requires a radical change in skillsets and market knowledge. This suggests that integrating wildlife-based activities into existing livestock practices, rather than attempting to change the economic model wholesale, might be more efficient, effective and culturally sensitive.

6.5.1 Mixed rangelands as a key policy intervention

Mixed farming systems may thus be key in unlocking wildlife economies for rural communities. The average size of new market entrant farms is insufficient to support the more specialised and capital-intensive models from the established sector, such as ecotourism and trophy hunting, despite these models being pushed by government and business plan developers in various policies. Rather, a focussed strategy of integrating wildlife into livestock ranching could unlock additional revenue streams from day visitors, local hunters and game meat, while lowering initial capital inputs and providing beneficiaries with a buffer against market shocks. This would enable government infrastructure investment to be scaffolded from low-input (basic perimeter fencing, developing walking trails, abattoirs and basic hutting lodges) to high-input (electrified game fences for dangerous game, luxury lodges, road maintenance, game vehicles) as the enterprises develop and adapt their models through impact investment. It would also enable cultural continuation from traditional practices and ensure that the enterprises are not overcapitalised initially. Once successful in this ‘entry enterprises’, investors could be used to expand the land area and infrastructure needed to access the more specialised wildlife economy models. As such, mixed farms can also be seen as the interface land use between DFFE and DALRRD where combined implementation and investment strategies could focus on the benefits of these systems to local food security, job creation and improved rangeland condition.

Integrating wildlife into cattle farms to form novel ecosystems does not necessarily lead to competition as wildlife and cattle tend to temporarily and spatially partition forage and water resources (Tyrrell et al. 2017; Stears & Shrader 2020; Mwasi & Dheer 2022), especially if cattle are kraaled overnight (Connolly et al. 2021). Mixed systems may be mutually ecologically and economically beneficial. For example, rewilding indigenous herbivores in rangelands can restore nutrient dispersal functions and enhance soil carbon sequestration (Hempson et al. 2017; Sitters et al. 2020), Well-planned grazing management can enhance grass productivity and nutrient hotspots in kraal sites that benefit wildlife and biodiversity (Fynn et al. 2016; Young et al. 2018; Keesing et al. 2018).

Currently, wildlife ranching exists in a space between environment and agriculture, where policy design and implementation might work at cross-purposes from each other. Pursuing a protected area mindset through Other Effective Area-based Conservation Measures (OECMs) to achieve a 30% terrestrial protection target could cause immense food insecurity, malnutrition and death in sub-Saharan Africa (Henry et al. 2022). On a national scale, the High-level Panel report on hunting and the resulting draft White Paper on Sustainable Use similarly focuses only on ecotourism as the basis for the wildlife economy (DFFE 2020b). This might be setting up new market entrants to fail if they become locked into capital-intensive, highly specialised models (Clements & Cumming 2018). Subsequently, decision-makers, programme officers and consultants cannot provide adequate business support because the viable wildlife economy business models have not been empirically defined and translated into new market entrant toolkits. To redress this, the bulk of the wildlife economy models must be seen as working lands, where mixed cattle and wildlife farms form the cornerstone agroecological system. Both environment and agriculture departments must work together to create an evidence-based approach to identifying the viable business models of the wildlife economy that can then be integrated into sustainable use policies. Viewing wildlife as assets for the biodiversity economy rather than entities needing state protection will facilitate conceptual common ground for integrated policy development. Mixed cattle and wildlife farms might then rejuvenate rural economies through multifaceted values within a working lands paradigm. Rewilding, as a tool used in policy, should thus not only seek to restore formerly degraded cattle rangelands to biodiverse protected areas, but also use wildlife as an asset to enhance ecosystem functioning and economic opportunity within existing cattle rangelands by designing policy that emphasises livelihoods, productivity and resilience for local communities (Western et al. 2020).

Chapter 7 Synthesis: wildlife working lands in Africa – from ideals to implementation

"I had come to feel wildness as a quality that flared into futurity, as well as reverberating out of the past. The contemporary threats to the wild were multiple, and severe. But they were also temporary. The wild prefaced us, and it will outlive us."

Robert Macfarlane (Macfarlane 2007 p. 316)

In this thesis, I have explored the concepts of wildness and rewilding as boundary objects that open novel interfaces between conservation, agriculture and human well-being. At the broadest, and ultimately most fundamental level, wildness provides the affordances for self-meaning, especially through its alterity or *unknownness*, which I have argued acts as an inimitable heterotopia for cultural change. While biodiversity is rightly regarded as fulfilling multiple dimensions of human needs, the so-called Nature's Contributions to People (NCP) framework adopted by IPBES (Díaz et al. 2018), my conceptual frameowkr suggests that wildness transcends simple *reflection* of values to actively *catalyse* new cultures through the search for self-meaning. Importantly, I propose the debate over intrinsic versus instrumental ethics misses the functional link: the intrinsic value of species, their capacity to flourish, translates into instrumental value for humans, both physically and psychologically. As such, wildness is a critical societal resource for enabling the transition to ecological mindfulness and a global economic system that moves beyond growth as the sole metric of progress (Otero et al. 2020). The transformative effects of wildness are congruent with South African policy that seeks to enhance the wildness of managed wildlife populations and create larger, more connected and ecologically functional landscapes (DFFE 2020b). However, this goal lacks the nuance that, while wilderness is scale-dependent, wildness is fractal and should be used as a way to assess conservation value within the context of 'working lands' and other key outcomes of socio-economic systems.

To begin distilling wildness as a lens to examine value at different scales, I have used the wildlife ranching industry as the basis for empirical research on wildness and rewilding. I have shown that wildlife management on working lands, where the commercial utilisation of wildlife is the engine of rewilding, introduces caveats to hailing rewilding in South Africa as a success using the framework of Perino et al. (2019) as a baseline definition. The erection of fences to qualify for wildlife ownership, the prevalence of selective breeding, predator persecution, resource provisioning and the lack of fire management in many enterprises impede full trophic restoration, stochastic disturbance regimes and natural dispersal. As such, rewilding in South Africa should be seen as a tool employed in agro-ecological enterprises to achieve socio-economic outcomes, rather than being foremostly concerned with biodiversity conservation based on *prima facie* evidence from increases in wildlife species richness and abundance. The recognition of rewilding as a tool rather than a goal should form the basis of policy and research design to best understand the wildlife economy and how to incentivise triple bottom line solutions.

By developing a tool to measure the wildness of managed populations, I have demonstrated significant variation in wildness between species and properties. These wildness assessments provide a fine-scale and nuanced tool in understanding how the commercial use of wildlife can generate conservation value in different socio-economic systems, and which can be mainstreamed into processes that unlock economic value through sustainable use (such as enhancement findings on USA and European Union regulations on trophy hunting). Wildlife populations, especially outside protected areas, will increasingly rely on management to survive and/or be managed to ‘pay the rent’. Frameworks that consistently discern between ‘command and control’ production systems (Holling & Meffe 1996) and ecosystem-based management associated with the ‘land ethic’ (Leopold 1968) will be crucial in mitigating the shifting baseline of wildness as a core conservation value. I have also demonstrated that there is little scope, as it currently stands, for the rewilding efforts in South Africa to contribute to Target 3 of the Global Biodiversity Framework as the OECM framework does not recognise conservation outcomes but insists on economic-blockers such as legal guarantees for biodiversity conservation.

However, these issues could be resolved with more innovative, market-based approaches that view rewilding as a form of novel ecosystem creation in Africa and not an extension of traditional protected areas. For example, the development of a wildlife ranching certification scheme, currently underway as a collaboration between national government and the wildlife ranching industry[27], could help landowners overcome the opportunity cost of intensive management (through premiums on wildlife products and services) (Turpie & Letley 2018) and provide the vehicle through which to develop management plans that incorporate biodiversity conservation and wildness that are officially recognised and binding, thereby unlocking the potential for these lands to be recognised in conservation targets through OECMs (Figure 7.1). This certification scheme, however, should have an agro-ecological underpinning rather than being focussed on species conservation. It should incorporate standards associated with sustainable land management maintenance of ecological processes to both incorporate the other components of rewilding (stochastic disturbance and natural dispersal) as well as recognise the central goal of restoring land productivity in these working lands (Pienaar et al. 2017). Taking this perspective, rather than a narrow biodiversity focus, enables the wildlife economy to potentially contribute to more agricultural indicators under the Global Biodiversity Framework (GBF) and for the multiplicity of values produced by rewilding to be formally measured, mainstreamed and incentives through positive incentives (Figure 7.1).

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[27] Finance Solution 6: Development and implementation of a voluntary market-based certification scheme in the wildlife sector

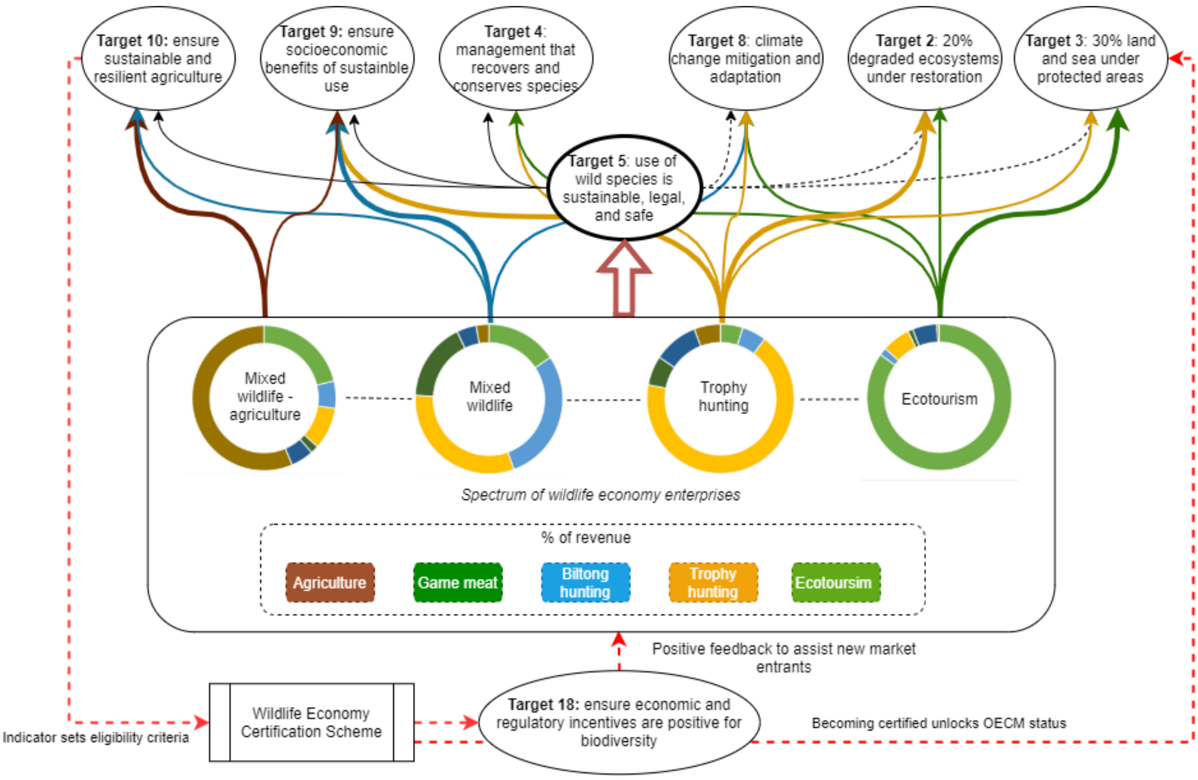

Figure 7.1. The contribution of wildlife-based land-uses (WBLUs) to the 2030 Global Biodiversity Framework targets. WBLUs exist on a spectrum of business models when viewed through primary revenue-generating activities. All WBLU business models rely on the legal and sustainable trade, hunting and use of wild species, which relates directly to Target 5. This target thus directly drives Target 4, 9 and 10 (solid black arrows) and indirectly drives Targets 2,3 and 8 (dotted black arrows). Empirical evidence from the Sustainable Wildlife Economies Project (SWEP) shows that WBLUs exist on a spectrum of business models, variously combining more agricultural production models to models focussed on international hunters and ecotourists. While not mutually exclusive, enterprises focussing on ecotourism are more likely to enter into legal agreements to protect biodiversity features and thus qualify as other effective area-based measures (OECMs), given that the biodiversity features themselves are key parts of the business model. Whereas those focused on more agro-ecological goals are less likely to be interested in such agreements but could qualify for sustainable use and sustainable agriculture targets. Bolded arrows show the hypothetical targets most likely suitable for various WBLU models. The key message is that the wildlife ranching sector should be valued for its multiplicity and measured through multiple indicators in contributing to the GBF and SDGs. Additionally, by recognising the role of sustainable agriculture in WBLUs, a potential positive feedback loop is enabled whereby the indicators from Target 10 help to shape the criteria for a market-based certification scheme for wildlife ranching, which then can be reported under Target 18, and which helps to bring new market entrants into viable models of rewilding that feed into various GBF targets. By signing up to a certification scheme, the legal and lasting agreement will help to resolve criterion 3.2 and potentially unlock more wildlife ranches as OECMs.

Merging this agricultural view with the conservation view is possible as I demonstrate, on a national scale, that restoring trophic diversity and stochastic disturbances (through converting from cattle to wildlife and using fire as a management tool) increases residual grass productivity compared to counterfactual land-uses and increases profitability and thus job density. These data also suggest that rewilding reduces the rate of bush encroachment, where commercial livestock farms show the steepest rate of woody plant productivity. Savannah rangelands in Africa are threatened by increasing woody plant encroachment and alien invasive plant infestation, which undermines grassland productivity and, thus, the potential to support herbivore populations. This is the result of historically poor rangeland management caused by overgrazing and suppression of natural fire-herbivory dynamics and, more recently, the result of elevated CO_2 levels (Venter et al. 2020). Bush encroachment is so widespread and entrenched that even restoring historical fire regimes may be insufficient to prevent or reverse woody plant encroachment (Case & Staver 2017), which may leave managers with a difficult choice of whether to increase fire frequency and intensity to control the spread of shrubs while sacrificing grazer population productivity due to loss of forage (Case & Staver 2017; Smit & Archibald 2019), and potentially foregoing ability to sequester soil carbon and earn carbon revenue (Tear et al. 2021).

Rewilding thus has profound implications for the capacity of African rangelands to maintain productivity and provide inclusive livelihoods based on herbivore production (Eldridge et al. 2011; O'Connor et al. 2014; Stevens et al. 2016; Skowno et al. 2017; Venter et al. 2018). As we are living in the age of the Anthropocene, we no longer have the luxury of rewilding for biodiversity's sake alone. Wildlife must help to solve societal problems. If we can show the relevance of rewilding for agricultural targets and goals, we potentially open limitless opportunities for a wilder world. For example, in South Africa, there is competition between two high-level mandates. For the Department of Forestry, Fisheries and Environment (DFFE), they must respond to Target 3 of the GBF to bring 30% of important biodiversity areas under some form of protection. However, the Department of Agriculture, Land Reform and Rural Development (DALRRD), through the land reform programme, also aims to redistribute 30% of land by 2030 for agricultural transfer. If rewilding can be rightly seen as congruent with this goal and enhancing it, rather than as a competing land-use, then conservationists can work with the agricultural sector too to open new frontiers (Figure 7.2). For example, the concept of Protected Agricultural Areas has been promulgated in policy, which "will ensure that high potential and best available agricultural land are protected against non-agricultural land uses in order to promote long-term agricultural production".[28] However, there is synergy for mixed

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[28] Parliamentary discussion on Preservation and Development of Agricultural Land Bill

farming systems that use forms of rewilding to enhance productivity and resilience to provide win-win solutions for food production and conservation.

Viewing rewilding through an agricultural lens is key to unblocking the adoption of rewilding by previously disadvantaged individuals and communities. I used the agro-ecological framing to assess the barriers to wildlife economy enterprise development by new market entrants and conclude that a primarily conservation focus by key implementing agencies occluded the development of more viable and resilient mixed wildlife and cattle ranching enterprises. Based on the average size of land awarded to beneficiaries, and the source of funding for infrastructure development, wildlife should be seen as enhancing both ecosystem functioning and enterprise viability in communal rangelands, rather than being the goal itself. This necessitates rethinking the National Biodiversity Economy Strategy and the promulgation of more coherent technical working groups between DFFE and DALRRD. However, if this new paradigm could gain traction in working lands of South Africa, both 30% targets of DFFE and DALRRD could be achieved (Figure 7.2).

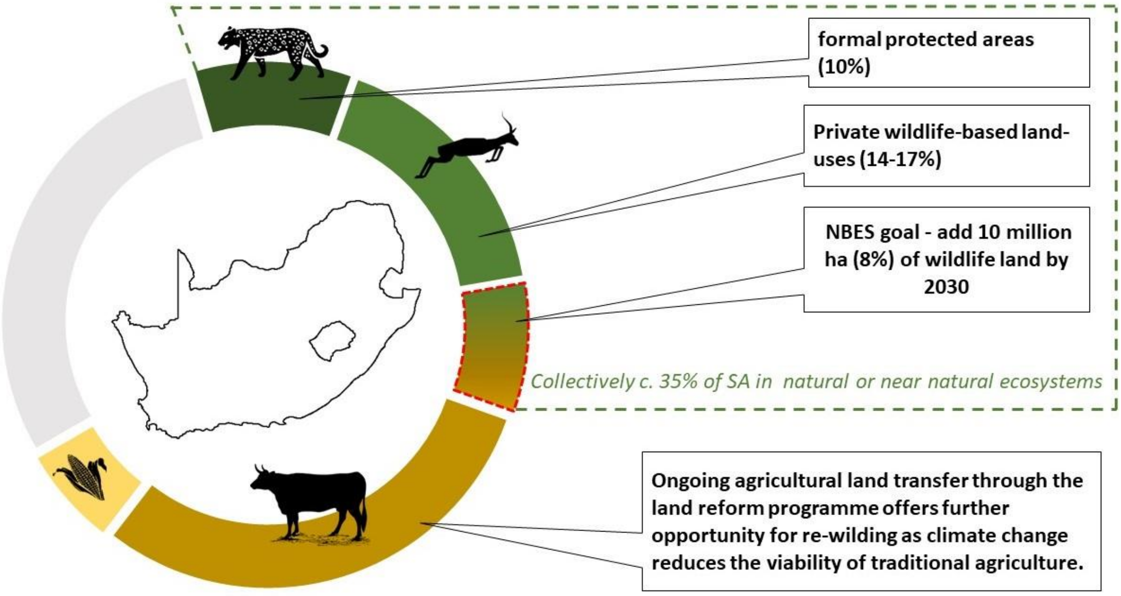

Figure 7.2. Schematic diagram of South Africa’s existing and potential contributions to area-based targets under the Global Biodiversity Framework (Target 3).

Currently, there are key conceptual barriers to creating a fit-for-purpose policy on the wildlife economy. The barriers identified at the Wildlife Economy Lab (DFFE 2016a) provide a good summary of the issues facing African decision-makers. Through situational analysis, I reorganised the identified barriers to be less symptomatic and more reflective of the underlying drivers (Figure 7.3). The main barriers were similar to those identified through a social-ecological systems framework analysis of the wildlife economy in Kenya and Tanzania (Caro & Davenport 2016; Brehony et al. 2020), so they are robust and reflect real issues. There is a strong Global North misperception of the success of the sustainable use model that hinders implementation progress (e.g., Mbaiwa & Hambira 2021), particularly in the global context, (e.g. CITES.) This has meant that, despite the good reputation of South Africa as a global leader in conservation, especially of the iconic elephant (*Loxodonta africana*), lion (*Panthera leo*), leopard (*Panthera pardus*) and rhinoceros (*Ceratotherium* and *Diceros* sp.), there is persistent public concern over policies, legislation and practices relating to the use of these species through their management, breeding, hunting, trade and handling (DFFE 2020b).

One of the fundamental issues is the lack of comprehensive and holistic information on the wildlife economy, especially around the sustainable use of wildlife (Snyman et al. 2021). Lack of information prevents understanding the true value of the wildlife economy at both local and national scales as well as the value chains linked to the wildlife economy (Snyman et al. 2021). This has led to an implicit focus on the ecotourism component of WBLUs and the emphasis on using private wildlife ranches to reach protected area goals. However, while some private WBLUs that focus on ecotourism can be declared as private nature reserves and managed as such, the majority (80%) of WBLUs are production systems with conservation outcomes (Taylor et al. 2020). As such, policies aimed at transitioning wildlife ranches towards protected area models are misplaced and may undermine conservation in the long-run as landowners consider (re)converting to cultivation or livestock (Parker et al. 2020).

Compounding this this, most socioeconomic studies look at economic activities from WBLUs in isolation. For example, the impacts of hunting on a national scale (e.g., Saayman et al. 2018) and not at the combination of economic activities that make individual enterprises viable (e.g., Clements et al. 2022). Consequently, there is almost no holistic information on the socioeconomic, biodiversity and ecosystem restoration value of viable WBLUs (Taylor et al. 2015). This leads to overemphasis of the known negative effects of certain practices, such as electrification of game fences and selective breeding of species, without fully considering how these practices might in effect ‘offset’ greater areas of extensive habitat conservation and biodiversity conservation. The results from this thesis shows that conservation values exist both in the wildness of managed populations and the increases in grass productivity that come from restoring trophic diversity. However, these values are not universal and require fine-scale assessments and evaluation to mainstream into policy.

The lack of holistic information on the impacts of rewilding on a systems level means there is no interface with agriculture and renders wildlife working lands as a conceptual extension of protected areas. This leaves African rangelands prone to misplaced Global North restoration efforts. For example, degraded grasslands or savannas, that are prime sites for the restorative effects of well-designed wildlife economies, might be perceived as needing tree planting under the Bonn challenge, which will ultimately be counterproductive to restoring ecosystem functioning and creating sustainable development opportunities (Bond et al. 2019). Planting trees and economies based around forest products are inherently preservationist, where global financial instruments pay communities not to use resources, whereas with rangeland systems, production and sustainable use can be sustained whilst also improving biodiversity and soil carbon, so these are inherently inclusive working lands. ‘Protected area’ thinking might thus be scuppering attempts for wildlife to create job opportunities and generate wealth for African communities (Mokotjomela & Nombewu 2019; Keane et al. 2020). Generating foundational datasets on ecosystem types to guide restoration will be important, as will generating data on expected employment numbers and returns on investment for designing inclusive wildlife economies.

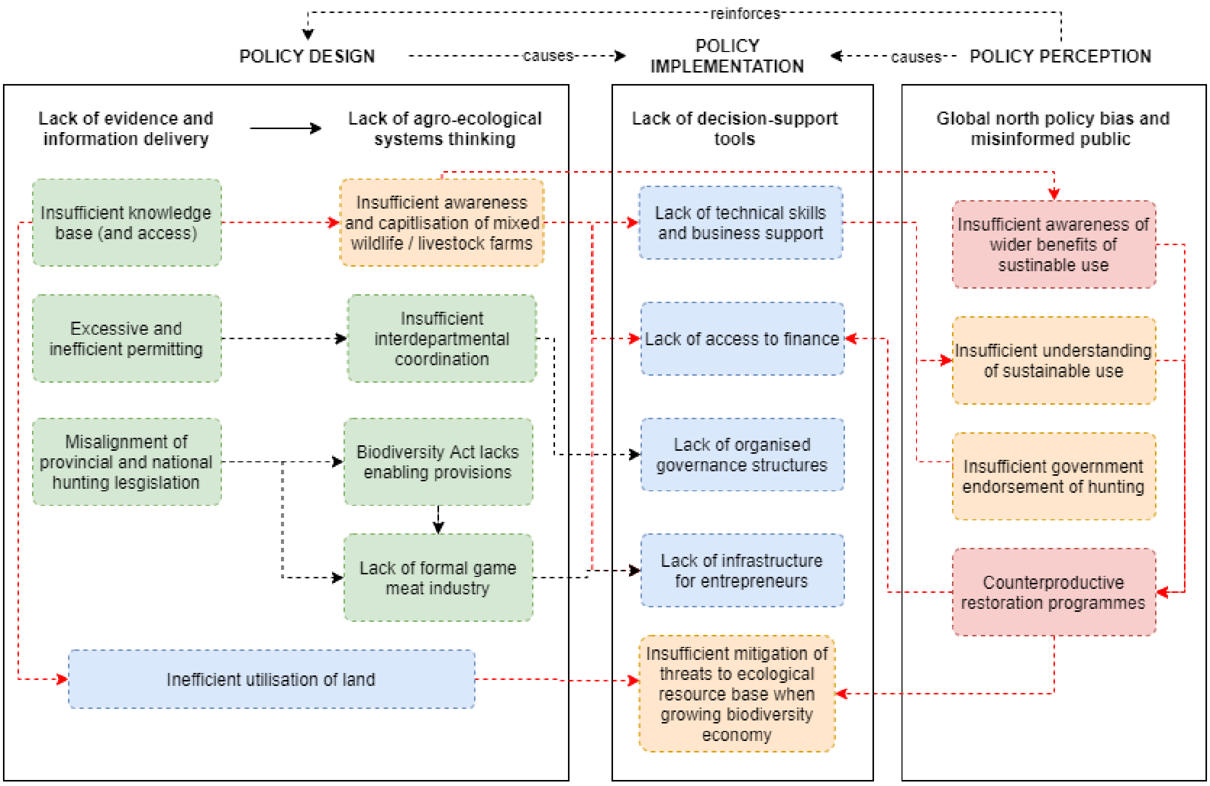

Figure 7.3. Context analysis of barriers to the wildlife economy identified at the 2016 Wildlife Economy Lab convened by the Department of Forestry, Fisheries and Environment and the Department of Tourism (South Africa). Barriers can be categorised according to issues relating to policy design, implementation and 'perception' (e.g. public misinformation). Key problems include an insufficient knowledge base, which prevents policy design of wildlife enterprises as agro-ecological systems and leads to counterproductive investment and infrastructure development (adapted from (DFFE 2016a)). Red arrows, critical pathways and red boxes are additional problems not picked up in the lab but relevant to Africa.

I constructed a theory of change to unlock wildlife economies in Africa (Figure 7.4), which attempts to answer the question, 'what is hindering the wildlife ranching sector from achieving its potential to develop a thriving, inclusive, equitable economy?' The model includes both outputs and outcome layers and key indicators for mainstreaming success that can be used to evaluate its effectiveness and inform subsequent revision (IIED and UNEP-WCMC 2017). This theory of change uses the three key barriers identified in Figure 7.3 to create three interweaving impact pathways.

Firstly, *create better policy design by investing in knowledge products that demonstrate the holistic, systems-level value of rewilded rangelands*. Continentwide, the lack of foundational information on wildlife economy business models prevents effective expansion of the industry and hinders community participation in the wildlife economy (African Leadership University School of Wildlife Conservation 2020; ALU 2020a, 2020b; Snyman et al. 2021). For example, the lack of data on wildlife economy value chains in Kenya prevents opportunities to develop new industries and markets (Ministry of Tourism and Wildlife 2019). In South Africa, there is a general lack of cooperative governance, both between DFFE and DALRRD, and also between national and provincial governments (DFFE 2020b), which is hindered by the lack of common conceptual paradigms. However, several recommendations of the HLP are counterproductive to bridging these divides and seem to alienate the industry further. For example, the HLP recommends phasing out captive rhino breeding and providing clarity that the 'trade in captive rhino horn' will not be approved prior to the Rhino Committee of Inquiry of recommendations. But it does not offer any guidance on what 'captive breeding' constitutes and what rewilding is. There is thus a need to integrate empirical frameworks such as that developed in Chapter 3 to standardise these terms.

One of the keys to developing effective policies is to establish new monitoring and observation networks that can deliver data on the right indicators at the right scale for wildlife-based social ecological systems by integrating natural and social science research (Clements & Cumming 2018; Dressel et al. 2018; Brehony et al. 2020). The wildlife permit system, which interfaces between domestic consumptive use licensing and international trade through CITES, is an excellent foundation for such an information system (Goss & Cumming 2013; DFFE 2020b; van Zyl & Kinghorn 2021), especially as the permitting process can become a geospatial platform linked dynamically to property expansion and contraction; and which enables remotely sensed analyses of changes in natural capital stocks and ecosystem condition. Lessons learned from the tribulations of South Africa's permitting system could be applied to better design similar wildlife trade systems in other countries (van Zyl & Kinghorn 2021).

Additionally, one of the flaws of current policies is focussing on economic activities (such as intensive breeding, trophy hunting, and ecotourism) instead of on the patterns in which these activities are combined in different contexts (business models). This promotes spurious recommendations based on false dichotomies, such as 'extensive' or 'intensive' without understanding that intensive and extensive elements are integrated (DFFE 2020b). Rather, we should be asking how business models drive land-use and land-cover change (outputs) and how different ecosystems and landscapes influence business models (inputs) in an adaptive cycle. Most wildlife ranches (86%) combine at least two wildlife-based economic activities (Taylor et al. 2020), which can be clustered into various business models (Clements et al. 2022). The first step to assisting new market entrants would be to understand the business models so we can develop screening tools for enterprise viability based on multiple characteristics of the proposed project and the business plan goals of the investor. Understanding the attributes of each system in the language of returns on investment will help bring impact investors into the domain of the wildlife economy.

The second impact pathway of Figure 7.4, is to *use these knowledge products to create a functional interface with agricultural institutions and programmes to co-design wildlife economies that operate as multifunctional landscapes*. Currently, wildlife and livestock are viewed competitors (Pozo et al. 2021). However, livestock production alone is insufficient for communities living in communal rangelands and wildlife presents an opportunity to diversify revenue streams (Child et al. 2012a; Chaminuka 2013; Chaminuka et al. 2014). Livestock and wildlife systems can be effectively sustained through age-old practices such as herding and kraaling (Riginos et al. 2012), and the use of livestock guarding dogs that have been effective in deterring predators (Rust et al. 2013; Spencer et al. 2020). Not only do these interventions work, but they also provide skilled labour opportunities and artisanal cultures to be rejuvenated in rural economies. This thesis has used the wildness framework to support an agro-ecologically focussed certification scheme that would ensure the conservation value of wildlife on working lands and reward landowners and communities that practice sustainable land management.

Additionally, I show that rewilding these agricultural lands increases residual grass productivity while lowering woody productivity. This would provide two important revenue streams: firstly, increased forage production from reducing bush encroachment. While bush encroachment leads to woody biomass carbon gains in the order of 4.3–28.5 tonnes per ha in the affected areas. This is worth R23–154 per ha in terms of climate change costs avoided in South Africa. The climate change adaptation benefit of addressing bush encroachment would outweigh the mitigation benefit of allowing it to proceed (Turpie et al. 2019). Secondly, to compensate for opportunity costs of above-ground carbon gains, certified landowners and communities could focus on soil carbon projects. Mixed wildlife and cattle grazing systems show significantly higher soil carbon stocks than cattle or mesoherbivores only grazing systems. Soil carbon sequestration is recognized as playing an increasingly important role in global carbon markets and in enabling viable conservation enterprises in the global south (Bossio et al. 2020, Bai & Cotrufo 2022). However, the carbon markets on the continent are still predominantly focused on above ground carbon storage through forest offsets and planting trees under the Bonn challenge, which can be counterproductive to inclusive economic development and may undermine rangeland productivity. Transitioning from cattle to extensive wildlife and mixed systems will improve soil carbon sequestration whilst sustaining the herbivore populations needed for economies based on sustainable use. The positive effect of rotational grazing with cattle results from the interaction between the carbon inputs associated with the high stocking rate and the long resting periods. The more extended the resting periods, the better for grass and soil recovery; however, farmers incur opportunity costs by resting plots for a long time as that land will be idle. Rewilding and making use of wildlife-based economies might mitigate this opportunity cost, both through revenue from soil carbon credits but also other economic activities, such as hunting, ecotourism and game meat production. Wildlife is an asset to rangelands in that it enhances soil carbon gains.

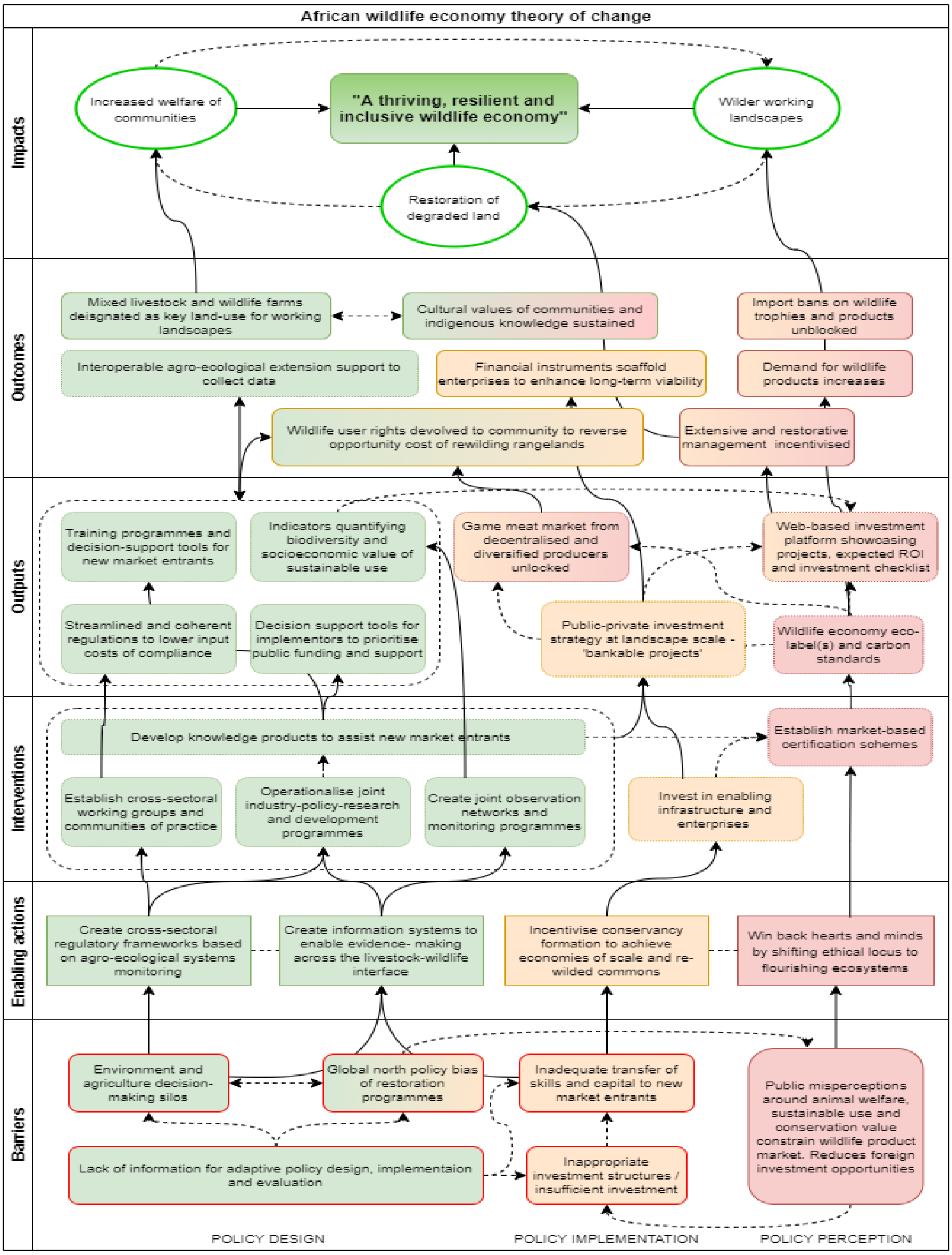

Figure 7.4. A theory of change for unlocking wildlife economies in Africa. The barrier context revolves around ineffective policy design, policy perception (from the public and Global North decision-makers) and, subsequently, policy implementation. This leads to a lack of established workflows for generating and mainstreaming policy-relevant data on rewilding and the wildlife economy, lack of appropriate institutional arrangements, such as environment-agriculture working groups, and a lack of temporally staggered financial instruments to scaffold enterprises for long-term viability. To redress this, I propose three interlinked pathways: firstly, mobilise the correct data and develop appropriate decision-support tools, which can be fed into the second pathway – interfacing with the agriculture sector and viewing mixed rangelands as a key novel ecosystem for Africa. Finally, winning back the trust of stakeholders by using rewilding as a tool to improve socioeconomic conditions for local communities.

The third impact pathway is *to win back 'hearts and minds' of both local stakeholders who live with wildlife as well as consumers to increase demand for wildlife products through setting and communicating sustainability standards around wildlife ranching*. The knowledge products from the first pathway can also be used to channel the investment needed by new market entrants in the third pathway more effectively, thus lower barriers to entry into the wildlife economy and enhancing enterprise viability (Figure 7.5). Cross-cutting actions include mobilising policy-relevant data and implementing more effective agroecological interventions through interdisciplinary teams and conducting foundational research. The key implementing actions are similar to the set of enabling actions identified by the World Bank to unlock nature-smart development, which include integrating nature-based solutions into sectoral investment programmes, enhancing the local benefits to communities of sustainable use of wildlife, mobilising multiple financial instruments, producing decision-support tools and sustainability indicators, and leveraging strategic partnerships (World Bank Group 2021).

There is a proverb from the Kikuyu tribe in Kenya – 'when elephants fight, it is the grass that suffers'. This is true on multiple levels for Africa: decades of colonial rule have morphed into corrupt governance structures, which have facilitated waves of resource-extracting political castes and neo-colonialist multinational corporations (Hughes 2019; Gillies 2020). This deprives local communities of any trickle of trickle-down economics (Holechek & Valdez 2018). On a second front, the counter-rush to protect Africa's wild spaces – to conserve actual elephants – has caused widespread displacement of communities ('conservation refugees') and disenfranchisement of people from the natural resources within which they live (Dowie 2011). The estrangement of communities from their land, the lack of effective infrastructural development, the lack of appropriate laws and policies, and the lack of access to capital have meant widespread degradation in rangelands through overgrazing (e.g., Palmer & Bennett 2013). One of the key pathways to reversing these trends may be to rewild rangelands and devolve sustainable use rights to landowners and communities. Rewilding can be a tool to help communities generate a multiplicity of revenue streams and thus become more resilient. Then, there will be a true incentive to conserve wildlife through direct benefits and autonomy over the socio-economic system.

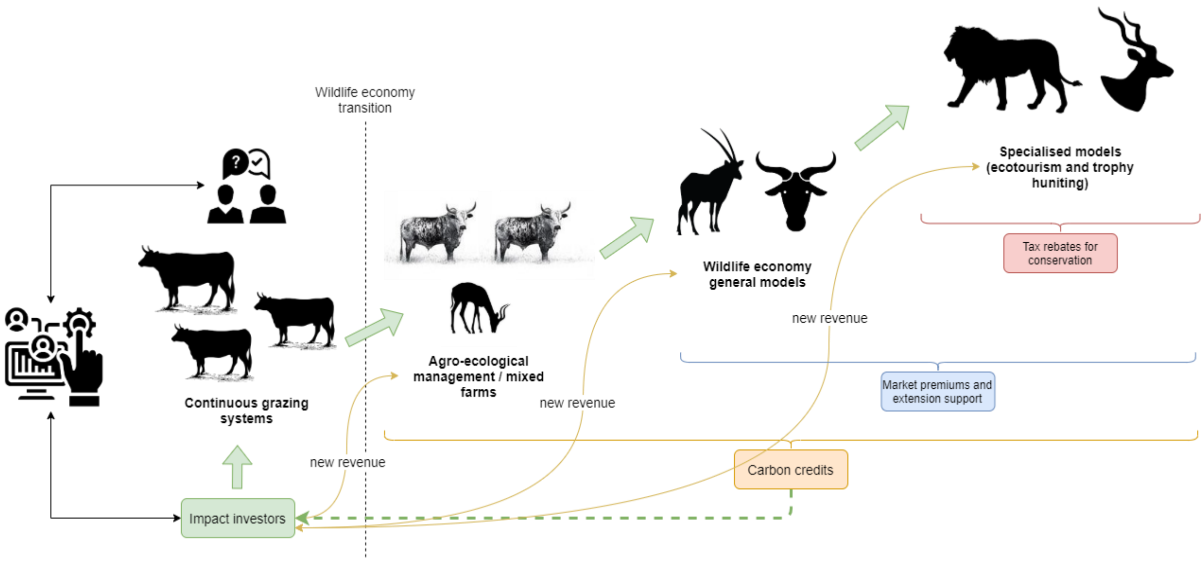

Figure 7.5. Conceptual diagram of potential investment scaffolding for the wildlife economy, where knowledge products on various business models can be used to attract impact investment and overcome the opportunity cost of converting from continuous grazing systems to mixed wildlife and cattle systems and ultimately more specialised wildlife economy revenue streams if desirable. Shifting to better grazing management and improved biodiversity unlocks additional revenue streams, such as carbon and biodiversity credits, that can scaffold the enterprise over long periods and be used to pack back the initial impact investors.

7.1 Conclusions

African wildlife conservation is seemingly stuck in a policy polemic. On the one hand, wildness is a central conservation value embedded within multiple policy frameworks at different scales, but it reflects the intrinsic value of nature and a compositionist paradigm. Conversely, there is the agricultural sector – where wildlife is either viewed as a threat to cattle production, or as an agricultural commodity, such as through the breeding of colour variants or high-quality stock for the hunting industry. This can lead to policy conflicts between sectors (Somers et al. 2020). However, wildness, as defined and explored in this thesis, is a concept that can potentially resolve this conflict: wildness has instrumental value for agro-ecological systems in that it restores ecosystem productivity, enables multiple concurrent revenue streams, and is more feasible on marginal agricultural land. In this way, wildness is an economic asset for socio-economic systems; and wildlife are not static entities to be protected but ecological tools for the management of multifunctional African rangelands.

This new paradigm could form the foundation for common working groups between environment and agriculture, especially through the lens of novel ecosystems, where the landscape is multifunctional and enhanced by rewilding. Under this paradigm, wildlife populations don’t need protection, they need deployment. Biodiversity is often seen as something to be set aside and preserved, but the age of the Anthropocene makes that position untenable. We are still trapped in protected area thinking – which often translates as fortress conservation and is currently manifested as a push to achieve OECMs. Yet strict protected area expansion through OECMs could cause widespread food insecurity, malnutrition, and even death in sub-Saharan Africa (Henry et al. 2022). Rewilding should trigger the shift from a vertical scale (species to be protected) towards a social-ecological perspective based on ecosystem restoration, sustainable land management and ecosystem service provision (Child et al. 2013; Reed et al. 2015; Clements et al. 2016a; Clements & Cumming 2017b; Brehony et al. 2020).

As climate change and biodiversity loss continue to accelerate, large-scale mitigation policies and investment strategies seek to tranistion to low-carbon economies and conserve biodiversity simultaneously (Barbier 2010; Dinerstein et al. 2013; Barbier et al. 2018; Pörtner et al. 2021). Rewilding – the restoration of degraded ecosystems by enabling natural vegetation recovery and the reintroduction of wildlife species (Perino et al. 2019) – is a key intervention to achieve both climate and biodiversity targets because complex ecosystems sequester and store more carbon than simplified land-uses as well as provide habitat for more species (Dinerstein et al. 2013; Pörtner et al. 2021). However, this does not mean abandoning productive agricultural systems. Rather, ‘working lands conservation’ – biodiversity-based production methods (such as silvopastures, diversified farming and holistic grazing management) embedded within multifunctional landscapes (Kremen & Merenlender 2018) – can be actively cultivated by policy and investment to ensure carbon sequestration, biodiversity conservation and socio-economic resilience simultaneously.

Rewilding thus sits at the nexus of biodiversity conservation, climate change mitigation and socio-economic improvement in Africa, potentially providing a boundary object to unify indicators and evaluation schemes under the major multilateral frameworks such as UNCBD, UNCCD and UNFCCC (Pettorelli et al. 2021; Shin et al. 2022), through common implementation pathways such as linking grass productivity and carbon sequestration to biodiversity conservation (Douglass et al. 2011; Soto-Navarro et al. 2020). As such, mixed wildlife and cattle ecosystems are strategic novel ecosystems for Africa – and not just transitional land-uses to protected areas. Africa's rangelands, which cover 43% of the continent (Hoffman & Vogel 2008), are the foundation for wildlife-based working lands. The wildlife economy is a uniquely African solution to sustainable development and is a model of merging rewilding with inclusive economic growth. Integration of cattle and wildlife systems and traditional management methods presents an opportunity to embed indigenous knowledge systems into a 'rewilded commons' approach of conservation, as well as providing a strong voice from the Global South that improves design and implementation of policies for the wildlife sector on an international level. Africa is poised to show the world how valuable wildness is to society, from sustaining the mystery within landscapes that allow human psychological freedom, to instilling the ecological processes necessary to maintain ecosystem services, and enabling the multifaceted enterprises that create skilled employment opportunities and resilient revenue streams. If we get this right, the African model of rewilding could become the benchmark conservation practice for the world.

Literature cited

Abdalla K, Mutema M, Chivenge P, Everson C, Chaplot V. 2022. Grassland rehabilitation significantly increases soil carbon stocks by reducing net soil CO_2 emissions. Soil Use and Management **38**:1250–1265.

Abdalla M, Hastings A, Chadwick DR, Jones DL, Evans CD, Jones MB, Rees RM, Smith P. 2018. Critical review of the impacts of grazing intensity on soil organic carbon storage and other soil quality indicators in extensively managed grasslands. Agriculture, Ecosystems & Environment **253**:62–81.

Abensperg-Traun M. 2009. CITES, sustainable use of wild species and incentive-driven conservation in developing countries, with an emphasis on southern Africa. Biological Conservation **142**:948–963.

Abraham AJ, Webster AB, Prys-Jones TO, Roux E le, Smith D, McFayden D, Jager PC de, Clauss M, Doughty CE. 2021. Large predators can mitigate nutrient losses associated with off-site removal of animals from a wildlife reserve. Journal of Applied Ecology **58**:1360–1369.

Achieng T, Maciejewski K, Dyer M, Biggs R. 2020. Using a social-ecological regime shift approach to understand the transition from livestock to game farming in the Eastern Cape, South Africa. Land **9**:97.

African Leadership University School of Wildlife Conservation. 2020. The State of the Wildlife Economy in Ghana. Country case study for the State of the Wildlife Economy in Africa report. 30 pages. African Leadership University School of Wildlife Conservation., Kigali, Rwanda.

Agrawal A, Redford K. 2009. Conservation and displacement: An overview. Conservation and Society **7**:1–10.

Akçakaya HR, Bennett EL, Brooks TM, Grace MK, Heath A, Hedges S, Hilton-Taylor C, Hoffmann M, Keith DA, Long B. 2018. Quantifying species recovery and conservation success to develop an IUCN Green List of Species. Conservation Biology **32**:1128–1138.

Akinola AO. 2019. Africa and the Land Reform Question. Pages 1–12 in Akinola AO, Wissink H, editors. Trajectory of Land Reform in Post-Colonial African States: The Quest for Sustainable Development and Utilization. Springer International Publishing.

Akinyemi FO, Ghazaryan G, Dubovyk O. 2021. Assessing UN indicators of land degradation neutrality and proportion of degraded land for Botswana using remote sensing based national level metrics. Land Degradation & Development **32**:158–172.

Alkemade R, van Bussel LG, Rodríguez SL, Schipper AM. 2022. Global biodiversity assessments need to consider mixed multifunctional land-use systems. Current Opinion in Environmental Sustainability **56**:101174.

Allan BF et al. 2017. Can integrating wildlife and livestock enhance ecosystem services in central Kenya? Frontiers in Ecology and the Environment **15**:328–335.

Allendorf FW, England PR, Luikart G, Ritchie PA, Ryman N. 2008. Genetic effects of harvest on wild animal populations. Trends in Ecology & Evolution **23**:327–337.

Allendorf FW, Leary RF, Spruell P, Wenburg JK. 2001. The problems with hybrids: setting conservation guidelines. Trends in Ecology & Evolution **16**:613–622.

Alston LJ, Libecap GD, Mueller B. 2000. Land Reform Policies, the Sources of Violent Conflict, and Implications for Deforestation in the Brazilian Amazon. Journal of Environmental Economics and Management **39**:162–188.

Altizer S, Harvell D, Friedle E. 2003. Rapid evolutionary dynamics and disease threats to biodiversity. Trends in Ecology & Evolution **18**:589–596.

ALU. 2020a. The State of the Wildlife Economy in South Africa. Country case study for the State of the Wildlife Economy in Africa report. 44 pages. African Leadership University School of Wildlife Conservation, Kigali, Rwanda.

ALU. 2020b. The State of the Wildlife Economy in Kenya. Country case study for the State of the Wildlife Economy in Africa report. 38 pages. African Leadership University School of Wildlife Conservation., Kigali, Rwanda.

Alves-Pinto H, Geldmann J, Jonas H, Maioli V, Balmford A, Ewa Latawiec A, Crouzeilles R, Strassburg B. 2021. Opportunities and challenges of other effective area-based conservation measures (OECMs) for biodiversity conservation. Perspectives in Ecology and Conservation **19**:115–120.

Anderson PML, Hoffman MT. 2007. The impacts of sustained heavy grazing on plant diversity and composition in lowland and upland habitats across the Kamiesberg mountain range in the Succulent Karoo, South Africa. Journal of Arid Environments **70**:686–700.

Angula HN, Stuart-Hill G, Ward D, Matongo G, Diggle RW, Naidoo R. 2018. Local perceptions of trophy hunting on communal lands in Namibia. Biological Conservation **218**:26–31.

Aplet G, Thomson J, Wilbert M. 2000. Indicators of wildness: Using attributes of the land to assess the context of wilderness. Pages 89–98 in McCool, SF, Cole DN, Borrie WT, O'Loughlin J, editors. Proceedings RMRS-P-15-VOL-2. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT.

Archibald S. 2016. Managing the human component of fire regimes: lessons from Africa. Philosophical Transactions of the Royal Society B: Biological Sciences **371**:20150346. Royal Society.

Archibald S, Scholes RJ, Roy DP, Roberts G, Boschetti L. 2010. Southern African fire regimes as revealed by remote sensing. International Journal of Wildland Fire **19**:861–878.

Arsanjani JJ, Tayyebi A, Vaz E. 2016. GlobeLand30 as an alternative fine-scale global land cover map: Challenges, possibilities, and implications for developing countries. Habitat International **55**:25–31.

Asner GP, Vaughn N, Smit IPJ, Levick S. 2016. Ecosystem-scale effects of megafauna in African savannas. Ecography **39**:240–252.

Awuah J, Smith SW, Speed JDM, Graae BJ. 2022. Can seasonal fire management reduce the risk of carbon loss from wildfires in a protected Guinea savanna? Ecosphere **13**:e4283.

Bai Y, Cotrufo MF. 2022. Grassland soil carbon sequestration: Current understanding, challenges, and solutions. Science **377**:603–608.

Bailey DW, Mosley JC, Estell RE, Cibils AF, Horney M, Hendrickson JR, Walker JW, Launchbaugh KL, Burritt EA. 2019. Synthesis Paper: Targeted Livestock Grazing: Prescription for Healthy Rangelands. Rangeland Ecology & Management **72**:865–877.

Barbier E. 2010. How is the Global Green New Deal going? Nature **464**:832–833.

Barbier EB, Burgess JC, Dean TJ. 2018. How to pay for saving biodiversity. Science **360**:486–488.

Barchett F. 2021. Examining sustainable practices and opportunities for a sustainability certification scheme for trophy hunting in Sub-saharan Africa. 77 pages. Cornell University, USA. Available from https://ecommons.cornell.edu/handle/1813/111064 (accessed March 15, 2022).

Barendse J, Roux D, Currie B, Wilson N, Fabricius C. 2016. A broader view of stewardship to achieve conservation and sustainability goals in South Africa. South African Journal of Science **112**:1–15.

Barnes JI, de Jager JL. 1996. Economic and financial incentives for wildlife use on private lands in Namibia and the implications for policy. South African Journal of Wildlife Research **26**:37–46.

Barnes JI, Jager JLVD. 1996. Economic and financial incentives for wildlife use on private land in Namibia and the implications for policy. South African Journal of Wildlife Research **26**:37–46.

Barnes JI, Macgregor J, Chris Weaver L. 2002. Economic Efficiency and Incentives for Change within Namibia's Community Wildlife Use Initiatives. World Development **30**:667–681.

Barnes MD, Glew L, Wyborn C, Craigie ID. 2018. Prevent perverse outcomes from global protected area policy. Nature Ecology & Evolution **2**:759.

Barton PS, Evans MJ, Foster CN, Pechal JL, Bump JK, Quaggiotto M-M, Benbow ME. 2019. Towards quantifying carrion biomass in ecosystems. Trends in Ecology & Evolution **34**:950–961.

Batavia C, Nelson MP, Darimont CT, Paquet PC, Ripple WJ, Wallach AD. 2019. The elephant (head) in the room: A critical look at trophy hunting. Conservation Letters **12**:e12565.

Bauer S, Hoye BJ. 2014. Migratory animals couple biodiversity and ecosystem functioning worldwide. Science **344**:1242552.

Bengtsson J, Bullock JM, Egoh B, Everson C, Everson T, O'Connor T, O'Farrell PJ, Smith HG, Lindborg R. 2019. Grasslands—more important for ecosystem services than you might think. Ecosphere **10**:e02582.

Bennun L et al. 2018. The value of the IUCN Red List for business decision-making. Conservation Letters **11**:1–8.

Berec M, Šetlíková I. 2021. IUCN-CITES match is not required: A reply to Gorobets. Ecological Indicators **128**:107842.

Betts J, Young RP, Hilton-Taylor C, Hoffmann M, Rodríguez JP, Stuart SN, Milner-Gulland E j. 2020. A framework for evaluating the impact of the IUCN Red List of threatened species. Conservation Biology **34**:632–643.

Birss C, Rushworth I, Collins NB, Peinke D, Buijs D. 2015. Inferred Natural distribution ranges of large mammals in South Africa, Version 1. Unpublished GIS coverage. Department of Environmental Affairs.

Bishop CJ, White GC, Freddy DJ, Watkins BE, Stephenson TR. 2009. Effect of enhanced nutrition on mule deer population rate of change. Wildlife Monographs **172**:1–28.

Blackmore A. 2017a. Selective breeding and the intensive management of wildlife: a legal challenge for Namibian conservation. African Journal of Wildlife Research **47**:65–78.

Blackmore A. 2020. Climate change and the ownership of game: A concern for fenced wildlife areas. Koedoe **62**:1–5.

Blackmore AC. 2017b. Public trust doctrine, research and responsible wildlife management in South Africa. Bothalia - African Biodiversity & Conservation **47**:1–9.

Blanchong JA, Scribner KT, Epperson BK, Winterstein SR. 2006. Changes in artificial feeding regulations impact white-tailed deer fine-scale spatial genetic structure. Journal of Wildlife Management **70**:1037–1043.

Bond WJ, Loffell D. 2001. Introduction of giraffe changes acacia distribution in a South African savanna. African Journal of Ecology **39**:286–294.

Bond WJ, Stevens N, Midgley GF, Lehmann CER. 2019. The trouble with trees: afforestation plans for Africa. Trends in Ecology & Evolution **34**:963–965.

Booth H et al. 2021. "Saving lives, protecting livelihoods, and safeguarding nature": risk-based wildlife trade policy for sustainable development outcomes post-COVID-19. Frontiers in Ecology and Evolution **9**:639216.

Boshoff AF, Landman M, Kerley G. 2015. Filling the gaps on the maps: historical distribution patterns of some larger mammals in part of southern Africa. Transactions Royal Society of South Africa **70**:1–65.

Bossio DA et al. 2020. The role of soil carbon in natural climate solutions. Nature Sustainability **3**:391–398.

Bothma J du P, du Toit JG, van Rooyen J. 2010a. Buying and selling wild animals. Pages 619–639 Game Ranch Management, 5th edition. Van Schaik.

Bothma J du P, Van Rooyen N, Du Toit JG. 2010b. Antelope and other smaller herbivores. Pages 210–245 in Bothma J du P, Du Toit JG, editors. Game Ranch Management 5th edition. Van Schaik Publishers, Pretoria, South Africa.

Bothma J du P, van Rooyen N, van Rooyen MW. 2004. Using diet and plant resources to set wildlife stocking densities in African savannas. Wildlife Society Bulletin **32**:840–851.

Botts EA et al. 2019. Practical actions for applied systematic conservation planning. Conservation Biology **33**:1235–1246.

Bourne A, Muller H, de Villiers A, Alam M, Hole D. 2017. Assessing the efficiency and effectiveness of rangeland restoration in Namaqualand, South Africa. Plant Ecology **218**:7–22.

Boyce JK, Narain S, Stanton EA. 2007. Land reform and sustainable development. Pages 127–150 Reclaiming Nature: Environmental Justice and Ecological Restoration. Anthem Press.

Brehony P, Tyrrell P, Kamanga J, Waruingi L, Kaelo D. 2020. Incorporating social-ecological complexities into conservation policy. Biological Conservation **248**:108697.

Briske DD, Derner JD, Brown JR, Fuhlendorf SD, Teague WR, Havstad KM, Gillen RL, Ash AJ, Willms WD. 2008. Rotational grazing on rangelands: reconciliation of perception and experimental evidence. Rangeland Ecology & Management **61**:3–17.

Briske DD, Sayre NF, Huntsinger L, Fernandez-Gimenez M, Budd B, Derner JD. 2011. Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. Rangeland Ecology & Management **64**:325–334.

Bruce P, Bruce A. 2017. Practical Statistics for Data Scientists: 50 Essential Concepts. O'Reilly Media.

Bryan BA et al. 2018. China's response to a national land-system sustainability emergency. Nature **559**:193–204.

Bubb PJ, Butchart SHM, Collen B, Dublin H, Kapos V, Pollock C, Stuart SN, Vié J-C. 2009. IUCN Red List Index – Guidance for National and Regional Use. Gland, Switzerland: IUCN.

Buisson E et al. 2019. Resilience and restoration of tropical and subtropical grasslands, savannas, and grassy woodlands. Biological Reviews **94**:590–609.

Buk KG, van der Merwe VC, Marnewick K, Funston PJ. 2018. Conservation of severely fragmented populations: lessons from the transformation of uncoordinated reintroductions of cheetahs (*Acinonyx jubatus*) into a managed metapopulation with self-sustained growth. Biodiversity and Conservation **27**:3393–3423.

Bunn D, Büscher B, McHale MR, Cadenasso ML, Childers DL, Pickett STA, Rivers L, Swemmer L. 2023. Golden Wildebeest Days: Fragmentation and value in South Africa's wildlife economy after apartheid. Journal of Southern African Studies **48**: 1013-1035.

Burgass MJ et al. 2020. Three Key considerations for biodiversity conservation in multilateral agreements. Conservation Letters **14**:e12764.

Burrell AL, Evans JP, Liu Y. 2017. Detecting dryland degradation using Time Series Segmentation and Residual Trend analysis (TSS-RESTREND). Remote Sensing of Environment **197**:43–57.

Büscher B, Fletcher R. 2019. Towards Convivial Conservation. Conservation & Society **17**:283–296.

Buschke FT, Hagan JG, Santini L, Coetzee BWT. 2021. Random population fluctuations bias the Living Planet Index. Nature Ecology & Evolution **5**:1145–1152.

Butchart SH, Akcakaya HR, Kennedy E, Hilton-Taylor C. 2006. Biodiversity indicators based on trends in conservation status: strengths of the IUCN Red List Index. Conservation Biology **20**:579–581.

Butchart SHM, Marco MD, Watson JEM. 2016. Formulating smart commitments on biodiversity: lessons from the aichi targets. Conservation Letters **9**:457–468.

Butler MJ, Teaschner AP, Ballard WB, McGee BK. 2005. Wildlife ranching in North America—arguments, issues, and perspectives. Wildlife Society Bulletin **33**:381–389.

Byrnes RC, Eastburn DJ, Tate KW, Roche LM. 2018. A global meta-analysis of grazing impacts on soil health indicators. Journal of Environmental Quality **47**:758–765.

Cain JW, Owen-Smith N, Macandza VA. 2012. The costs of drinking: comparative water dependency of sable antelope and zebra. Journal of Zoology **286**:58–67.

Calleja JA, Escolà M, Carvalho J, Forcadell JM, Serrano E, Bartolomé J. 2019. Cattle grazing fails to control shrub encroachment in Mediterranean landscapes. Rangeland Ecology & Management **72**:803–811.

Capozzelli JF, Miller JR, Debinski DM, Schacht WH. 2020. Restoring the fire–grazing interaction promotes tree–grass coexistence by controlling woody encroachment. Ecosphere **11**:e02993.

Caro T, Davenport TRB. 2016. Wildlife and wildlife management in Tanzania. Conservation Biology **30**:716–723.

Carruthers J. 2008. "Wilding the farm or farming the wild"? The evolution of scientific game ranching in South Africa from the 1960s to the present. Transactions of the Royal Society of South Africa **63**:160–181.

Carver S et al. 2021. Guiding principles for rewilding. Conservation Biology **35**:1882–1893.

Carver S, Tricker J, Landres P. 2013. Keeping it wild: Mapping wilderness character in the United States. Journal of Environmental Management **131**:239–255.

Case MF, Staver AC. 2017. Fire prevents woody encroachment only at higher-than-historical frequencies in a South African savanna. Journal of Applied Ecology **54**:955–962.

Casey NH. 2021. A profile of South African sustainable animal production and greenhouse gas emissions. Animal Frontiers **11**:7–16.

Cash DW, Clark WC, Alcock F, Dickson NM, Eckley N, Guston DH, Jäger J, Mitchell RB. 2003. Knowledge systems for sustainable development. Proceedings of the National Academy of Sciences **100**:8086–8091.

Cavaleri MA, Sack L. 2010. Comparative water use of native and invasive plants at multiple scales: a global meta-analysis. Ecology **91**:2705–2715.

Cazalis V et al. 2022. Bridging the research-implementation gap in IUCN Red List assessments. Trends in Ecology & Evolution **37**:359–370.

CBD. 2021. First draft of the post-2020 global biodiversity framework. Convention on Biological Diversity. Available from https://www.cbd.int/conferences/post2020/wg2020-03/documents.

Challender DWS et al. 2019. Criteria for CITES species protection. Science **364**:247–248.

Challender DWS et al. 2022. Mischaracterizing wildlife trade and its impacts may mislead policy processes. Conservation Letters **15**:e12832.

Challender DWS et al. 2023. Identifying species likely threatened by international trade on the IUCN Red List can inform CITES trade measures. Nature Ecology & Evolution 7: 1211–1220.

Challender DWS, MacMillan DC. 2019. Investigating the influence of non-state actors on amendments to the CITES appendices. Journal of International Wildlife Law & Policy **22**:90–114.

Chamier J, Schachtschneider K, Maitre D le, Ashton PJ, Wilgen B van. 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. Water SA **38**:345–356.

Chaminuka P. 2013. Wildlife or livestock? New directions for developing communal rangelands in South Africa. African Journal of Range & Forage Science **30**:51–55.

Chaminuka P, Udo HMJ, Eilers KCHAM, van der Zijpp A. 2014. Livelihood roles of cattle and prospects for alternative land uses at the wildlife/livestock interface in South Africa. Land Use Policy **38**:80–90.

Champagnon J, Elmberg J, Guillemain M, Gauthier-Clerc M, Lebreton J-D. 2012. Conspecifics can be aliens too: A review of effects of restocking practices in vertebrates. Journal for Nature Conservation **20**:231–241.

Chan KM, Balvanera P, Benessaiah K, Chapman M, Díaz S, Gómez-Baggethun E, Gould R, Hannahs N, Jax K, Klain S. 2016. Why protect nature? Rethinking values and the environment. Proceedings of the National Academy of Sciences **113**:1462–1465.

Chang Q, Wang L, Ding S, Xu T, Li Z, Song X, Zhao X, Wang D, Pan D. 2018. Grazer effects on soil carbon storage vary by herbivore assemblage in a semi-arid grassland. Journal of Applied Ecology **55**:2517–2526.

Chaplot V, Dlamini P, Chivenge P. 2016. Potential of grassland rehabilitation through high density-short duration grazing to sequester atmospheric carbon. Geoderma **271**:10–17.

Chapman R. 2004. Crowded Solitude: Thoreau on Wildness. Environmental Philosophy **1**:58–72.

Chapman RL. 2006. Ecological restoration restored. Environmental Values **15**:463–478.

Charles GK, Porensky LM, Riginos C, Veblen KE, Young TP. 2017. Herbivore effects on productivity vary by guild: cattle increase mean productivity while wildlife reduce variability. Ecological Applications **27**:143–155.

Chauvenet ALM, Watson JEM, Adams VM, Marco MD, Venter O, Davis KJ, Mappin B, Klein CJ, Kuempel CD, Possingham HP. 2020. To achieve big wins for terrestrial conservation, prioritize protection of ecoregions closest to meeting targets. One Earth **2**:479–486.

Chen J, Frank DA. 2020. Herbivores stimulate respiration from labile and recalcitrant soil carbon pools in grasslands of Yellowstone National Park. Land Degradation & Development **31**:2620–2634.

Chen S et al. 2018. Plant diversity enhances productivity and soil carbon storage. Proceedings of the National Academy of Sciences **115**:4027–4032.

Chesson P, Gebauer RL, Schwinning S, Huntly N, Wiegand K, Ernest MS, Sher A, Novoplansky A, Weltzin JF. 2004. Resource pulses, species interactions, and diversity maintenance in arid and semi-arid environments. Oecologia **141**:236–253.

Chidakel A, Eb C, Child B. 2020. The comparative financial and economic performance of protected areas in the Greater Kruger National Park, South Africa: functional diversity and resilience in the socio-economics of a landscape-scale reserve network. Journal of Sustainable Tourism **28**:1100–1119.

Chief Surveyor-General. 2017. Cadastral Spatial Information. Available from http://csg.dla.gov.za/.

Child B. 2012a. Private conservation in southern Africa: practice and emerging principles. Pages 103–111 Evolution and innovation in wildlife conservation.

Child B. 2012b. The sustainable use approach could save South Africa's rhinos. South African Journal of Science **108**:21–25.

Child BA, Musengezi J, Parent GD, Child GFT. 2012a. The economics and institutional economics of wildlife on private land in Africa. Pastoralism: Research, Policy and Practice **2**:18.

Child BA, Musengezi J, Parent GD, Child GFT. 2012b. The economics and institutional economics of wildlife on private land in Africa. Pastoralism **2**:1–32.

Child MF. 2011. Conservation of adaptive self-construction: a flux-centred solution to the paradox of nature preservation. Environmental Values **20**:527–548.

Child MF, Peel MJ, Smit IP, Sutherland WJ. 2013. Quantifying the effects of diverse private protected area management systems on ecosystem properties in a savannah biome, South Africa. Oryx **47**:29–40.

Child MF, Roxburgh L, Do Linh San E, Raimondo D, Davies-Mostert HT, editors. 2016. The 2016 Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. Available from https://www.ewt.org.za/resources/resources-mammal-red-list/.

Child MF, Selier SJ, Radloff FG, Taylor WA, Hoffmann M, Nel L, Power RJ, Birss C, Okes NC, Peel MJ. 2019. A framework to measure the wildness of managed large vertebrate populations. Conservation Biology **33**:1106–1119.

Chiyangwa T. 2018. Financial Implications of Converting from Livestock to Game Farming in the Karoo Region, South Africa. Thesis, 111 pages. Stellenbosch: Stellenbosch University. Available from https://scholar.sun.ac.za:443/handle/10019.1/103526 (accessed March 27, 2023).

Chown SL. 2010. Temporal biodiversity change in transformed landscapes: a southern African perspective. Philosophical Transactions of the Royal Society B: Biological Sciences **365**:3729–3742.

Clark DA, Brehony P, Dickman A, Foote L, Hart AG, Jonga C, Mbiza MM, Roe D, Sandbrook C. 2023. Hunting trophy import bans proposed by the UK may be ineffective and inequitable as conservation policies in multiple social-ecological contexts. Conservation Letters **16**:e12935.

Clayton S, Colléony A, Conversy P, Maclouf E, Martin L, Torres A-C, Truong M-X, Prévot A-C. 2017. Transformation of experience: toward a new relationship with nature. Conservation Letters **10**:645–651.

Clements HS, Balfour D, Di Minin E. 2023. Importance of private and communal lands to sustainable conservation of Africa's rhinoceroses. Frontiers in Ecology and the Environment **21**:140–147.

Clements HS, Baum J, Cumming GS. 2016a. Money and motives: an organizational ecology perspective on private land conservation. Biological Conservation **197**:108–115.

Clements HS, Baum J, Cumming GS. 2016b. Money and motives: an organizational ecology perspective on private land conservation. Biological Conservation **197**:108–115.

Clements HS, Child MF, Lindeque L, Lunderstedt K, De Vos A. 2022. Lessons from COVID-19 for wildlife ranching in a changing world. Nature Sustainability **5**:1040–1048.

Clements HS, Cumming GS. 2017a. Manager strategies and user demands: Determinants of cultural ecosystem service bundles on private protected areas. Ecosystem Services **28**:228–237.

Clements HS, Cumming GS. 2017b. Positives and pathologies of natural resource management on private land-conservation areas. Conservation Biology **31**:707–717.

Clements HS, Cumming GS. 2018. Traps and transformations influencing the financial viability of tourism on private-land conservation areas. Conservation Biology **32**:424–436.

Clements HS, Cumming GS, Kerley GIH. 2016c. Predators on private land: broad-scale socioeconomic interactions influence large predator management. Ecology and Society **21**:45.

Clements HS, De Vos A, Bezerra JC, Coetzer K, Maciejewski K, Mograbi PJ, Shackleton C. 2021. The relevance of ecosystem services to land reform policies: Insights from South Africa. Land Use Policy **100**:104939.

Clements HS, Kerley GI, Cumming GS, De Vos A, Cook CN. 2018. Privately protected areas provide key opportunities for the regional persistence of large-and medium-sized mammals. Journal of Applied Ecology **56**: 537-546.

Clements HS, Knight M, Jones P, Balfour D. 2020. Private rhino conservation: Diverse strategies adopted in response to the poaching crisis. Conservation Letters **13**:e12741.

Cloete PC, Taljaard PR, Grové B. 2007. A comparative economic case study of switching from cattle farming to game ranching in the Northern Cape Province. African Journal of Wildlife Research **37**:71–78.

Cloete PC, van der Merwe P, Saayman M. 2015. Game Ranch Profitability in South Africa. Caxton, Pretoria, South Africa.

Coad L et al. 2015. Measuring impact of protected area management interventions: current and future use of the Global Database of Protected Area Management Effectiveness. Philosophical Transactions of the Royal Society B: Biological Sciences **370**:20140281.

Coad L, Watson JE, Geldmann J, Burgess ND, Leverington F, Hockings M, Knights K, Di Marco M. 2019. Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. Frontiers in Ecology and the Environment **17**:259–264.

Cockburn J, Cundill G, Shackleton S, Rouget M. 2019. The meaning and practice of stewardship in South Africa. South African Journal of Science **115**:1–13.

Coe K, Thomsen JM, Mbaiwa J. 2023. Botswana communities' adaptive capacity to the lifting of a trophy hunting ban. African Journal of Wildlife Research **53**. (6 July 2023)

Collen BEN, Loh J, Whitmee S, McRAE L, Amin R, Baillie JE. 2009. Monitoring change in vertebrate abundance: the Living Planet Index. Conservation Biology **23**:317–327.

Conant RT, Cerri CEP, Osborne BB, Paustian K. 2017. Grassland management impacts on soil carbon stocks: a new synthesis. Ecological Applications **27**:662–668.

Conant RT, Paustian K. 2002. Potential soil carbon sequestration in overgrazed grassland ecosystems. Global Biogeochemical Cycles **16**:90-1-90–9.

Connolly E, Allan J, Brehony P, Aduda A, Western G, Russell S, Dickman A, Tyrrell P. 2021. Coexistence in an African pastoral landscape: Evidence that livestock and wildlife temporally partition water resources. African Journal of Ecology **59**:696–711.

Cook NJ, Grillos T, Andersson KP. 2019. Gender quotas increase the equality and effectiveness of climate policy interventions. Nature Climate Change **9**:330.

Cookson LJ. 2011. A definition for wildness. Ecopsychology **3**:187–193.

Cooney R, Challender DWS, Broad S, Roe D, Natusch DJD. 2021. Think before you act: Improving the conservation outcomes of CITES listing decisions. Frontiers in Ecology and Evolution **9:**631556.

Corson MS, Mondière A, Morel L, van der Werf HMG. 2022. Beyond agroecology: Agricultural rewilding, a prospect for livestock systems. Agricultural Systems **199**:103410.

Cousins B, Genis A, Clarke J. 2018. The potential of agriculture and land reform to generate jobs. Policy Brief 51. Institute for Poverty, Land and Agrarian Studies, University of Western Cape, Cape Town, South Africa.

Cousins J, Sadler J, Evans J. 2010. The challenge of regulating private wildlife ranches for conservation in South Africa. Ecology and Society **15**:28.

Cousins JA, Sadler JP, Evans J. 2008. Exploring the role of private wildlife ranching as a conservation tool in South Africa: stakeholder perspectives. Ecology and Society **13**:43

Coverdale TC, Kartzinel TR, Grabowski KL, Shriver RK, Hassan AA, Goheen JR, Palmer TM, Pringle RM. 2016. Elephants in the understory: opposing direct and indirect effects of consumption and ecosystem engineering by megaherbivores. Ecology **97**:3219–3230.

Cowell C, Birss C. 2017. Biodiversity Management Plan for The Bontebok (*Damaliscus pygargus pygargus*) in South Africa. 71 pages. South African National Parks, CapeNature and the National Department of Environmental Affairs.

Cowie AL et al. 2018. Land in balance: The scientific conceptual framework for Land Degradation Neutrality. Environmental Science & Policy **79**:25–35.

Craigie ID, Baillie JEM, Balmford A, Carbone C, Collen B, Green RE, Hutton JM. 2010. Large mammal population declines in Africa's protected areas. Biological Conservation **143**:2221–2228.

Creel S. 2001. Four factors modifying the effect of competition on carnivore population dynamics as illustrated by African wild dogs. Conservation Biology **15**:271–274.

Creel S, Christianson D, Liley S, Winnie JA. 2007. Predation risk affects reproductive physiology and demography of elk. Science **315**:960–960.

Cromsigt JP, te Beest M, Kerley GI, Landman M, le Roux E, Smith FA. 2018. Trophic rewilding as a climate change mitigation strategy? Philosophical Transactions of the Royal Society B: Biological Sciences **373**:20170440.

Cronon W. 1996. The Trouble with Wilderness: Or, Getting Back to the Wrong Nature. Environmental History **1**:7–28.

Cumming DHM, Cumming GS. 2003. Ungulate community structure and ecological processes: body size, hoof area and trampling in African savannas. Oecologia **134**:560–568.

Cundill G, Thondhlana G, Sisitka L, Shackleton S, Blore M. 2013. Land claims and the pursuit of co-management on four protected areas in South Africa. Land Use Policy **35**:171–178.

Cusack JJ, Duthie AB, Minderman J, Jones IL, Pozo RA, Rakotonarivo OS, Redpath S, Bunnefeld N. 2020. Integrating conflict, lobbying, and compliance to predict the sustainability of natural resource use. Ecology and Society **25**:13

DAFF. 2018. Long-term grazing capacity for South-Africa. Department of Agriculture, Forestry & Fisheries, Directorate Land Use & Soil Management, Pretoria.

Dalerum F, Miranda M. 2016. Game auction prices are not related to biodiversity contributions of southern African ungulates and large carnivores. Scientific Reports **6**: 21922.

DALRRD. 2022. Abstract of Agricultural Statistics 2022. 112 pages. Directorate: Statistics and Economic Analysis, Department of Agriculture, Land Reform and Rural Development, Pretoria, South Africa.

Davies-Mostert HT, Macdonald DW, Mills MG. 2015. The demography and dynamics of an expanding, managed African wild dog metapopulation. African Journal of Wildlife Research **45**:258–273.

De Vos A, Clements HS, Biggs D, Cumming GS. 2019. The dynamics of proclaimed privately protected areas in South Africa over 83 years. Conservation Letters **12**:e12644.

De Vos A, Cumming GS. 2019. The contribution of land tenure diversity to the spatial resilience of protected area networks. People and Nature **1**:331–346.

Deckers J. 2021. On (Un)naturalness. Environmental Values **30**:297–318.

Department of Environmental Affairs. 2016. National Biodiversity Economy Strategy (NBES). Department of Environmental Affairs, Pretoria, South Africa.

Department of Environmental Affairs. 2017. The South African Carbon Sinks Atlas. Department of Environmental Affairs, Pretoria, South Africa.

Dereniowska M, Meinard Y. 2021. The unknownness of biodiversity: Its value and ethical significance for conservation action. Biological Conservation **260**:109199.

Derry JF, Dougill AJ. 2008. Water location, piospheres and a review of evolution in African ruminants. African Journal of Range & Forage Science **25**:79–92.

Desjardins E, Donhauser J, Barker G. 2019. Ecological Historicity, Novelty and Functionality in the Anthropocene. Environmental Values **28**:275–303.

Desmet PG, Knowles T, Blignaut J, Venter Z, King TG, Moyo L, Thompson M. 2019. Ecosystem-based Adaptation: Identifying spatial priorities for EbA implementation in South Africa. Department of Environmental Affairs (DEA), Pretoria, South Africa.

DFFE. 2016a. Biodiversity and Tourism Lab: Wildlife Economy Lab Executive Summary Report. Department of Environmental Affairs and Department of Tourism, Pretoria, South Africa.

DFFE. 2016b. National Protected Areas Expansion Strategy for South Africa 2016. Department of Forestry, Fisheries and Environment, Pretoria, South Africa.

DFFE. 2018. Biodiversity Finance Initiative (BIOFIN) – South Africa: Biodiversity Finance Plan. Department of Forestry, Fisheries and Environment and United Nations Development Programme, Pretoria, South Africa.

DFFE. 2020a. The Amathole – Great Fish Biodiversity Economy Node (BEN) Feasibility Study to Determine Optimal Economic Benefit. Department of Forestry, Fisheries and Environment, Pretoria, South Africa.

DFFE. 2020b. The High-level Panel of Experts for the Review of Policies, Legislation and Practices on Matters of Elephant, Lion, Leopard and Rhinoceros Management, Breeding, Hunting, Trade and Handling. Page 582. DFFE, Pretoria, South Africa.

DFFE. 2021a. Game Meat Strategy for South Africa. Department of Environment, Forestry and Fisheries (DFFE), Pretoria, South Africa.

DFFE. 2021b. Biodiversity Economy Investment Platform - Concept Overview v3. Department of Forestry, Fisheries and Environment, Pretoria, South Africa.

Di Marco M, Butchart SHM, Visconti P, Buchanan GM, Ficetola GF, Rondinini C. 2016. Synergies and trade-offs in achieving global biodiversity targets. Conservation Biology **30**:189–195.

Di Marco M, Ferrier S, Harwood TD, Hoskins AJ, Watson JEM. 2019. Wilderness areas halve the extinction risk of terrestrial biodiversity. Nature **573**:582–585.

Di Minin E, Clements HS, Correia RA, Cortés-Capano G, Fink C, Haukka A, Hausmann A, Kulkarni R, Bradshaw CJA. 2021. Consequences of recreational hunting for biodiversity conservation and livelihoods. One Earth **4**:238–253.

Di Minin E, Leader-Williams N, Bradshaw CJA. 2016. Banning trophy hunting will exacerbate biodiversity loss. Trends in Ecology & Evolution **31**:99–102.

Di Minin E, Macmillan DC, Goodman PS, Escott B, Slotow R, Moilanen A. 2013. Conservation businesses and conservation planning in a biological diversity hotspot. Conservation Biology **27**:808–820.

di Virgilio A, Lambertucci SA, Morales JM. 2019. Sustainable grazing management in rangelands: Over a century searching for a silver bullet. Agriculture, Ecosystems & Environment **283**:106561.

Díaz S et al. 2018. Assessing nature's contributions to people. Science **359**:270–272.

Dicks LV, Baude M, Roberts SP, Phillips J, Green M, Carvell C. 2015. How much flower-rich habitat is enough for wild pollinators? Answering a key policy question with incomplete knowledge. Ecological Entomology **40**:22–35.

Dinerstein E et al. 2013. Enhancing conservation, ecosystem services, and local livelihoods through a wildlife premium mechanism. Conservation Biology **27**:14–23.

Dinerstein E et al. 2017. An ecoregion-based approach to protecting half the terrestrial realm. BioScience **67**:534–545.

Dlamini P, Chivenge P, Chaplot V. 2016. Overgrazing decreases soil organic carbon stocks the most under dry climates and low soil pH: A meta-analysis shows. Agriculture, Ecosystems & Environment **221**:258–269.

Doak DF, Bakker VJ, Goldstein BE, Hale B. 2014. What is the future of conservation? Trends in Ecology & Evolution **29**:77–81.

Docherty TDS, Hethcoat MG, MacTavish LM, MacTavish D, Dell S, Stephens PA, Willis SG. 2020. Burning savanna for avian species richness and functional diversity. Ecological Applications **30**:e02091.

Donald PF, Buchanan GM, Balmford A, Bingham H, Couturier AR, de la Rosa Jr GE, Gacheru P, Herzog SK, Jathar G, Kingston N. 2019. The prevalence, characteristics and effectiveness of Aichi Target 11′ s "other effective area-based conservation measures"(OECMs) in Key Biodiversity Areas. Conservation Letters **12**:e12659.

Donaldson JE, Archibald S, Govender N, Pollard D, Luhdo Z, Parr CL. 2018. Ecological engineering through fire-herbivory feedbacks drives the formation of savanna grazing lawns. Journal of Applied Ecology **55**:225–235.

Douglass LL, Possingham HP, Carwardine J, Klein CJ, Roxburgh SH, Russell-Smith J, Wilson KA. 2011. The effect of carbon credits on savanna land management and priorities for biodiversity conservation. PLOS ONE **6**:e23843.

Dowie M. 2011. Conservation Refugees: The Hundred-Year Conflict between Global Conservation and Native Peoples. MIT Press.

DRDLR. 2020. Strategic Plan 2020-2024. Department of Rural Development and Land Reform, Pretoria, South Africa.

Dressel S, Ericsson G, Sandström C. 2018. Mapping social-ecological systems to understand the challenges underlying wildlife management. Environmental Science & Policy **84**:105–112.

du Toit C, Meissner HH, van Niekerk W. 2013. Direct greenhouse gas emissions of the game industry in South Africa. South African Journal of Animal Science **43**:376–393.

du Toit EA. 2000, March. Certification standards for sustainable game ranching in the Northern Province, South Africa. Thesis. Stellenbosch: Stellenbosch University.

du Toit G van N, Snyman HA, Malan PJ. 2009. Physical impact of grazing by sheep on soil parameters in the Nama Karoo subshrub/grass rangeland of South Africa. Journal of Arid Environments **73**:804–810.

du Toit JC, Ramaswiela T, Pauw MJ, O'Connor TG. 2018. Interactions of grazing and rainfall on vegetation at Grootfontein in the eastern Karoo. African Journal of Range & Forage Science **35**:267–276.

du Toit JG. 2007. Role of the private sector in the wildlife industry. 87 pages. Wildlife Ranching SA/Du Toit Wilddienste, Tshwane.

du Toit JT, Cumming DHM. 1999. Functional significance of ungulate diversity in African savannas and the ecological implications of the spread of pastoralism. Biodiversity & Conservation **8**:1643–1661.

du Toit JT, Pettorelli N. 2019. The differences between rewilding and restoring an ecologically degraded landscape. Journal of Applied Ecology **56**:2467–2471.

Dube S, Scholes RJ, Nelson GC, Mason-D'Croz D, Palazzo A. 2013. South African food security and climate change: agriculture futures. Economics **7(**1): 20130035.

Dudley N, Jonas H, Nelson F, Parrish J, Pyhälä A, Stolton S, Watson JEM. 2018. The essential role of other effective area-based conservation measures in achieving big bold conservation targets. Global Ecology and Conservation **15**:e00424.

Eldridge DJ, Bowker MA, Maestre FT, Roger E, Reynolds JF, Whitford WG. 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. Ecology Letters **14**:709–722.

Elhacham E, Ben-Uri L, Grozovski J, Bar-On YM, Milo R. 2020. Global human-made mass exceeds all living biomass. Nature **588**:442–444.

Elsbeth McPhee M. 2004. Generations in captivity increases behavioral variance: considerations for captive breeding and reintroduction programs. Biological Conservation **115**:71–77.

Ender CL, Christian CE, Cushman JH. 2017. Native herbivores and environmental heterogeneity as mediators of an exotic grass invasion. Ecology and Evolution **7**:1561–1571.

Erwin DH. 2008. Macroevolution of ecosystem engineering, niche construction and diversity. Trends in Ecology & Evolution **23**:304–310.

Evanoff RJ. 2005. Reconciling realism and constructivism in environmental ethics. Environmental Values **14**:61–81.

Evans J, Geerken R. 2004. Discrimination between climate and human-induced dryland degradation. Journal of Arid Environments **57**:535–554.

Everson CS, Everson T. 2016. The long-term effects of fire regime on primary production of montane grasslands in South Africa. African Journal of Range & Forage Science **33**:33–41.

Fabricius C, Currie B, Mbiba M, Ntuli H. 2021. Beyon d Tourism: Assessing the Viability of Community-based Natural Resource Enterprises in Africa. 170 pages. WWF's Nature Pays program by CARMa-Afrika. Nelson Mandela University, South Africa.

Farnsworth KD, Lyashevska O, Fung T. 2012. Functional complexity: The source of value in biodiversity. Ecological Complexity **11**:46–52.

Faulkner W. 2007. As I Lay Dying. Vintage Classics (first published 1930), USA.

Ferner J, Schmidtlein S, Guuroh RT, Lopatin J, Linstädter A. 2018. Disentangling effects of climate and land-use change on West African drylands' forage supply. Global Environmental Change **53**:24–38.

Fincher CL, Thornhill R. 2008. A parasite-driven wedge: infectious diseases may explain language and other biodiversity. Oikos **117**:1289–1297.

Findlay N, Manson A, Cromsigt JP, Gordijn P, Nixon C, Rietkerk M, Thibaud G, Wassen MJ, Beest M te. 2022. Long-term frequent fires do not decrease topsoil carbon and nitrogen in an Afromontane grassland. African Journal of Range & Forage Science **39**:44–55.

Fischer A, Weldesemaet YT, Czajkowski M, Tadie D, Hanley N. 2015. Trophy hunters' willingness to pay for wildlife conservation and community benefits. Conservation Biology **29**:1111–1121.

Follett RF, Reed DA. 2010. Soil carbon sequestration in grazing lands: societal benefits and policy implications. Rangeland Ecology & Management **63**:4–15.

Forbes ES, Cushman JH, Burkepile DE, Young TP, Klope M, Young HS. 2019. Synthesizing the effects of large, wild herbivore exclusion on ecosystem function. Functional Ecology **33**:1597–1610.

Freeman J, Kobziar L, Rose EW, Cropper W. 2017. A critique of the historical-fire-regime concept in conservation. Conservation Biology **31**:976–985.

Fronhofer EA, Altermatt F. 2015. Eco-evolutionary feedbacks during experimental range expansions. Nature Communications **6**:6844.

Fuhlendorf SD, Engle DM. 2004. Application of the fire–grazing interaction to restore a shifting mosaic on tallgrass prairie. Journal of Applied Ecology **41**:604–614.

Fuhlendorf SD, Engle DM, Kerby J, Hamilton R. 2009. Pyric herbivory: rewilding landscapes through the recoupling of fire and grazing. Conservation Biology **23**:588–598.

Funk C et al. 2015. The climate hazards infrared precipitation with stations—a new environmental record for monitoring extremes. Scientific Data **2**:150066.

Fynn RWS, Augustine DJ, Peel MJS, Garine-Wichatitsky M de. 2016. Strategic management of livestock to improve biodiversity conservation in African savannahs: a conceptual basis for wildlife–livestock coexistence. Journal of Applied Ecology **53**:388–397.

Gagic V et al. 2015. Functional identity and diversity of animals predict ecosystem functioning better than species-based indices. Proceedings of the Royal Society of London B: Biological Sciences **282**:20142620.

Gaylard A, Owen-Smith N, Redfern J. 2003. Surface water availability: implications for heterogeneity and ecosystem processes. Pages 171–188 in du Toit JT, Rogers KH, Biggs HC, editors. The Kruger Experience: Ecology and Management of Savanna Heterogeneity. Island Press, Washington, USA.

Gelderblom C, Oettlé N, Clifford-Holmes J, Malgas R, Wilson N, Polonsky S. 2020. Transformative Cross-Sectoral Extension Services Dialogue. SA-EU Dialogue Facility. Available from https://www.wwf.org.za/our_research/publications/?34725/Transformative-Cross-Sectoral-Extension-Services-Dialogue.

Geldmann J et al. 2018. A global analysis of management capacity and ecological outcomes in terrestrial protected areas. Conservation Letters **11**:e12434.

Geldmann J, Manica A, Burgess ND, Coad L, Balmford A. 2019. A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. Proceedings of the National Academy of Sciences **116**:23209–23215.

Genes L, Svenning J-C, Pires AS, Fernandez FAS. 2019. Why we should let rewilding be wild and biodiverse. Biodiversity and Conservation **28**:1285–1289.

Georgiou K et al. 2022. Global stocks and capacity of mineral-associated soil organic carbon. Nature Communications **13**:3797.

Geremia C, Merkle JA, Eacker DR, Wallen RL, White PJ, Hebblewhite M, Kauffman MJ. 2019. Migrating bison engineer the green wave. Proceedings of the National Academy of Sciences **116**:25707–25713.

Gillies A. 2020. Corruption trends during Africa's oil boom, 2005 to 2014. The Extractive Industries and Society **7**:1171-1181.

Godde CM, Boone RB, Ash AJ, Waha K, Sloat LL, Thornton PK, Herrero M. 2020. Global rangeland production systems and livelihoods at threat under climate change and variability. Environmental Research Letters **15**:044021.

Goldstein A et al. 2020. Protecting irrecoverable carbon in Earth's ecosystems. Nature Climate Change **10**:287–295.

Gordon CE, Greve M, Henley M, Bedetti A, Allin P, Svenning J-C. 2023. Elephant rewilding affects landscape openness and fauna habitat across a 92-year period. Ecological Applications **33**:e2810.

Gorelick N, Hancher M, Dixon M, Ilyushchenko S, Thau D, Moore R. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sensing of Environment **202**:18–27.

Gorobets A. 2020. Wild fauna conservation: IUCN-CITES match is required. Ecological Indicators **112**:106091.

Goss JR, Cumming GS. 2013. Networks of wildlife translocations in developing countries: an emerging conservation issue? Frontiers in Ecology and the Environment **11**:243–250.

Grant CC, Van der Walt JL. 2000. Towards an adaptive management approach for the conservation of rare antelope in the Kruger National Park-outcome of a workshop held in May 2000. Koedoe **43**:103–112.

Green J, Schmidt-Burbach J, Elwin A. 2022. Commercial trade of wild animals: examining the use of the IUCN Red List and CITES Appendices as the basis for corporate trade policies. Frontiers in Conservation Science **3**: 902074.

Griscom BW et al. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences **114**:11645–11650.

Gureja N, Owen-Smith N. 2002. Comparative use of burnt grassland by rare antelope species in a lowveld game ranch, South Africa. South African Journal of Wildlife Research **32**:31–38.

Gurney GG et al. 2021. Biodiversity needs every tool in the box: use OECMs. Nature **595**:646–649.

Gurney GG, Adams VM, Álvarez-Romero JG, Claudet J. 2023. Area-based conservation: Taking stock and looking ahead. One Earth **6**:98–104.

Guyton JA et al. 2020. Trophic rewilding revives biotic resistance to shrub invasion. Nature Ecology & Evolution **4**: 712–724.

Haila Y. 1997. “Wilderness” and the multiple layers of environmental thought. Environment and History **3**:129–147.

Hall R, Kepe T. 2017. Elite capture and state neglect: new evidence on South Africa’s land reform. Review of African Political Economy **44**:122–130.

Hall SJG. 2018. A novel agroecosystem: Beef production in abandoned farmland as a multifunctional alternative to rewilding. Agricultural Systems **167**:10–16.

Han J. 2020. How to promote rural revitalization via introducing skilled labor, deepening land reform and facilitating investment? China Agricultural Economic Review **12**:577–582.

Hart AG et al. 2020. Threats posed to conservation by media misinformation. Conservation Biology **34**:1333–1334.

Havstad KM, Peters DPC, Skaggs R, Brown J, Bestelmeyer B, Fredrickson E, Herrick J, Wright J. 2007. Ecological services to and from rangelands of the United States. Ecological Economics **64**:261–268.

Hawkins H-J. 2017. A global assessment of Holistic Planned Grazing$^{\text{TM}}$ compared with season-long, continuous grazing: meta-analysis findings. African Journal of Range & Forage Science **34**:65–75.

Hawkins H-J, Venter Z-S, Cramer MD. 2022. A holistic view of Holistic Management: What do farm-scale, carbon, and social studies tell us? Agriculture, Ecosystems & Environment **323**:107702.

Hayward MW, Child MF, Kerley GIH, Lindsey PA, Somers MJ, Burns B. 2015. Ambiguity in guideline definitions introduces assessor bias and influences consistency in IUCN Red List status assessments. Frontiers in Ecology and Evolution **3**:87.

Hayward MW, Kerley GI. 2009. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? Biological Conservation **142**:1–13.

Hempson GP, Archibald S, Bond WJ. 2017. The consequences of replacing wildlife with livestock in Africa. Scientific Reports **7**:17196.

Henry RC, Arneth A, Jung M, Rabin SS, Rounsevell MD, Warren F, Alexander P. 2022. Global and regional health and food security under strict conservation scenarios. Nature Sustainability **5**:303–310.

Hetem RS, de Witt BA, Fick LG, Fuller A, Kerley GI, Meyer LC, Mitchell D, Maloney SK. 2009. Body temperature, thermoregulatory behaviour and pelt characteristics of three colour morphs of springbok (*Antidorcas marsupialis*). Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology **152**:379–388.

Hilbers JP, Santini L, Visconti P, Schipper AM, Pinto C, Rondinini C, Huijbregts MA. 2017. Setting population targets for mammals using body mass as a predictor of population persistence. Conservation Biology **31**:385–393.

Hillenbrand M, Thompson R, Wang F, Apfelbaum S, Teague R. 2019. Impacts of holistic planned grazing with bison compared to continuous grazing with cattle in South Dakota shortgrass prairie. Agriculture, Ecosystems & Environment **279**:156–168.

Hirons M, Comberti C, Dunford R. 2016. Valuing cultural ecosystem services. Annual Review of Environment and Resources **41**:545–574.

Hlavac M. 2018. Stargazer: Well-Formatted Regression and Summary Statistics Tables. R package version 5.2.2. Available from https://CRAN.R-project.org/package=stargazer.

Hobbs RJ, Higgs E, Harris JA. 2009. Novel ecosystems: implications for conservation and restoration. Trends in Ecology & Evolution **24**:599–605.

Hobbs RJ, Higgs ES, Hall C, editors. 2013. Novel ecosystems: intervening in the new ecological world order. John Wiley & Sons, London, UK.

Hoffman T, Vogel C. 2008. Climate change impacts on African rangelands. Rangelands **30**:12–17.

Holdo RM, Sinclair ARE, Dobson AP, Metzger KL, Bolker BM, Ritchie ME, Holt RD. 2009. A disease-mediated trophic cascade in the Serengeti and its implications for ecosystem C. PLOS Biology **7**:e1000210.

Holechek J, Valdez R. 2018. Wildlife conservation on the rangelands of eastern and southern Africa: past, present, and future. Rangeland Ecology & Management **71**:245–258.

Holling CS. 2001. Understanding the Complexity of Economic, Ecological, and Social Systems. Ecosystems **4**:390–405.

Holling CS, Meffe GK. 1996. Command and control and the pathology of natural resource management. Conservation biology **10**:328–337.

Hoogendoorn G, Meintjes D, Kelso C, Fitchett J. 2019. Tourism as an incentive for rewilding: the conversion from cattle to game farms in Limpopo province, South Africa. Journal of Ecotourism **18**:309–315.

Hornby D, Cousins B. 2019. “Reproducing the social”: contradictory interconnections between land, cattle production and household relations in the Besters Land Reform Project, South Africa. Anthropology Southern Africa **42**:202–216.

Horowitz A. 2019. Trophy hunting: A moral imperative for bans. Science **366**:435–435.

Hrabar H, Birss C, Peinke D, King S, Novellie P, Kerley GIH, Child MF. 2016. A conservation assessment of *Equus zebra zebra*. 14 pages. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Hrabar H, Kerley GIH. 2015. Cape Mountain Zebra 2014/15 Status Report. Report 63. Centre for African Conservation Ecology, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.

Hudak AT, Wessman CA. 2001. Textural analysis of high resolution imagery to quantify bush encroachment in Madikwe Game Reserve, South Africa, 1955-1996. International Journal of Remote Sensing **22**:2731–2740.

Hughes AC. 2019. Understanding and minimizing environmental impacts of the Belt and Road Initiative. Conservation Biology **33**:883–894.

Hulme D, Murphree M. 1999. Communities, wildlife and the ‘new conservation’ in Africa. Journal of International Development **11**:277–285.

Hyvarinen O, te Beest M, le Roux E, Kerley GIH, Findlay N, Schenkeveld WDC, Trouw V, Cromsigt JPGM. 2023. Grazing in a megagrazer-dominated savanna does not reduce soil carbon stocks, even at high intensities. Oikos **9**:e09809.

IIED and UNEP-WCMC. 2017. Mainstreaming biodiversity and development guidance from African experience 2012-17. 60 pages. International Institute for Environment and Development, London, UK.

IPBES. 2022. Thematic Assessment Report on the Sustainable Use of Wild Species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Fromentin, J.M., Emery, MR, Donaldson, J, Danner, M., Hallosserie, A, and Kieling, D. (eds.). IPBES secretariat, Bonn, Germany

Ingold T. 2011. Being Alive. Routledge, London ; New York.

Irob K, Blaum N, Baldauf S, Kerger L, Strohbach B, Kanduvarisa A, Lohmann D, Tietjen B. 2022. Browsing herbivores improve the state and functioning of savannas: A model assessment of alternative land-use strategies. Ecology and Evolution **12**:e8715.

IUCN ESARO. 2020. Closing the gap. The financing and resourcing of protected and conserved areas in Eastern and Southern Africa. 93 pages. IUCN ESARO; BIOPAMA, Nairobi, Kenya.

IUCN Standards and Petitions Subcommittee. 2022. Guidelines for Using the IUCN Red List Categories and Criteria. Version 15.1. 114 pages. Prepared by the Standards and Petitions Subcommittee.

IUCN/SSC. 2013. Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0. Page viiii + 57 pp. IUCN Species Survival Commission, Gland, Switzerland.

IUCN-WCPA Task Force on OECMs. 2019. Recognising and reporting other effective area-based conservation measures. 36 pages. IUCN, Gland, Switzerland.

Jackson CR, Power RJ, Groom RJ, Masenga EH, Mjingo EE, Fyumagwa RD, Røskaft E, Davies-Mostert H. 2014. Heading for the hills: risk avoidance drives den site selection in African wild dogs. PLoS ONE **9**:e99686.

Jackson RB, Banner JL, Jobbágy EG, Pockman WT, Wall DH. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature **418**:623–626.

Jackson ST, Hobbs RJ. 2009. Ecological restoration in the light of ecological history. Science **325**:567–569.

Jager D de, Harper CK, Bloomer P. 2020. Genetic diversity, relatedness and inbreeding of ranched and fragmented Cape buffalo populations in southern Africa. PLOS ONE **15**:e0236717.

Jarman PJ. 1974. The social organisation of antelope in relation to their ecology. Behaviour **48**:215–267.

Jepson PR. 2022. To capitalise on the Decade of Ecosystem Restoration, we need institutional redesign to empower advances in restoration ecology and rewilding. People and Nature **4**:1404–1413.

Jetz W, McGowan J, Rinnan DS, Possingham HP, Visconti P, O'Donnell B, Londoño-Murcia MC. 2022. Include biodiversity representation indicators in area-based conservation targets. Nature Ecology & Evolution **6**:123–126.

Johanisová L, Mauerhofer V. 2023. Assessing trophy hunting in South Africa by comparing hunting and exporting databases. Journal for Nature Conservation **72**:126363.

Jones KE, Bielby J, Cardillo M, Fritz SA, O'Dell J, Orme CDL, Safi K, Sechrest W, Boakes EH, Carbone C. 2009. PanTHERIA: a species-level database of life history, ecology, and geography of extant and recently extinct mammals. Ecology **90**:2648–2648.

Jones KR, Venter O, Fuller RA, Allan JR, Maxwell SL, Negret PJ, Watson JEM. 2018. One-third of global protected land is under intense human pressure. Science **360**:788–791.

Jordaan FJ, Neser FWC, Maiwashe A, King Z, Scholtz MM. 2021. The environmental impact of changes in cow productivity and its component traits in South Africa's landrace beef breeds. Frontiers in Animal Science **2**:743229

Jule KR, Leaver LA, Lea SEG. 2008. The effects of captive experience on reintroduction survival in carnivores: A review and analysis. Biological Conservation **141**:355–363.

Kachler J, Benra F, Bolliger R, Isaac R, Bonn A, Felipe-Lucia MR. 2023. Can we have it all? The role of grassland conservation in supporting forage production and plant diversity. Landscape Ecology, *in press*.

Kamuti T. 2014. The fractured state in the governance of private game farming: the case of KwaZulu-Natal Province, South Africa. Journal of Contemporary African Studies **32**:190–206.

Kareiva P, Marvier M. 2012. What is conservation science? BioScience **62**:962–969.

Kartzinel TR, Chen PA, Coverdale TC, Erickson DL, Kress WJ, Kuzmina ML, Rubenstein DI, Wang W, Pringle RM. 2015. DNA metabarcoding illuminates dietary niche partitioning by African large herbivores. Proceedings of the National Academy of Sciences **112**:8019–8024.

Keane A, Lund JF, Bluwstein J, Burgess ND, Nielsen MR, Homewood K. 2020. Impact of Tanzania's Wildlife Management Areas on household wealth. Nature Sustainability **3**:226–233.

Keesing F et al. 2018. Consequences of integrating livestock and wildlife in an African savanna. Nature Sustainability **1**:566–573.

Kepe T, Wynberg R, Ellis W. 2005. Land reform and biodiversity conservation in South Africa: complementary or in conflict? International Journal of Biodiversity Science & Management **1**:3–16.

Kerley GI, Pressey RL, Cowling RM, Boshoff AF, Sims-Castley R. 2003. Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. Biological Conservation **112**:169–190.

Kgosikoma OE, Mojeremane W, Harvie B. 2015. The impact of livestock grazing management systems on soil and vegetation characteristics across savanna ecosystems in Botswana. African Journal of Range & Forage Science **32**:271–278.

Kiffner C, Nagar S, Kollmar C, Kioko J. 2016. Wildlife species richness and densities in wildlife corridors of Northern Tanzania. Journal for Nature Conservation **31**:29–37.

Kimuyu DM, Sensenig RL, Riginos C, Veblen KE, Young TP. 2014. Native and domestic browsers and grazers reduce fuels, fire temperatures, and acacia ant mortality in an African savanna. Ecological Applications **24**:741–749.

King RJH. 1991. Environmental ethics and the case for hunting. Environmental Ethics **13**:59–85.

Klein CAM de, Nagels JJD, Scarsbrook M, Collins R, Mcdowell RW, Muirhead R. 2002. Environmental impacts of intensive deer farming in New Zealand - a review. NZGA: Research and Practice Series **9**:79–87.

Knowles T, Christie S, Berliner D, Geldenhuys C, Thompson M. 2020. Addressing Specific Elements of REDD+ in South Africa: Comprehensive Assessment of the Definition and Scope of Implementation of REDD+ in South Africa. 36 pages. Department of Environment, Forestry and Fisheries (DFFE), Pretoria, South Africa.

Kosoy N, Corbera E. 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics **69**:1228–1236.

Kotzé E, Sandhage-Hofmann A, Meinel J-A, du Preez CC, Amelung W. 2013. Rangeland management impacts on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa. Journal of Arid Environments **97**:220–229.

Kraaij T, Milton SJ. 2006. Vegetation changes (1995–2004) in semi-arid Karoo shrubland, South Africa: Effects of rainfall, wild herbivores and change in land use. Journal of Arid Environments **64**:174–192.

Kremen C, Merenlender AM. 2018. Landscapes that work for biodiversity and people. Science **362**:eaau6020.

Kreuter UP, Workman JP. 1997. Comparative profitability of cattle and wildlife ranches in semi-arid Zimbabwe. Journal of Arid Environments **35**:171–187.

Kristensen JA, Svenning J-C, Georgiou K, Malhi Y. 2022. Can large herbivores enhance ecosystem carbon persistence? Trends in Ecology & Evolution **37**:117–128.

Laland KN, Boogert NJ. 2010. Niche construction, co-evolution and biodiversity. Ecological Economics **69**:731–736.

Lange M et al. 2015. Plant diversity increases soil microbial activity and soil carbon storage. Nature Communications **6**:6707.

Langholz JA, Kerley GI. 2006. Combining conservation and development on private lands: an assessment of ecotourism-based private game reserves in the Eastern Cape. Report **56**. Centre for African Conservation Ecology (formerly TERU)

Laris P. 2021. On the problems and promises of savanna fire regime change. Nature Communications **12**:4891.

LaRocco AA. 2020. Botswana's hunting ban and the transformation of game-meat cultures, economies and ecologies. Journal of Southern African Studies **46**:723–741.

Laubscher L, Hoffman L. 2012. An overview of disease-free buffalo breeding projects with reference to the different systems used in South Africa. Sustainability **4**:3124–3140.

Laurance WF et al. 2014. A global strategy for road building. Nature **513**:229–232.

le Roux E, Kerley GIH, Cromsigt JPGM. 2018. Megaherbivores modify trophic cascades triggered by fear of predation in an African savanna ecosystem. Current Biology **28**:2493-2499.e3.

Leader-Williams N, Brett RA, Brooks M, Craig I, du Toit RF, Emslie RH, Knight MH, Stanley-Price MR, Stockil O. 1997. A scheme for differentiating and defining the different situations under which live rhinos are conserved. Pachyderm **23**:24–28.

Ledger SEH et al. 2023. Past, present, and future of the Living Planet Index. npj Biodiversity **2**:12.

Lehmann S. 2021. Growing biodiverse urban futures: renaturalization and rewilding as strategies to strengthen urban resilience. Sustainability **13**:2932.

Lele S. 2021. From wildlife-ism to ecosystem-service-ism to a broader environmentalism. Environmental Conservation **48**:5–7.

Leopold A. 1968. A Sand County Almanac and Sketches Here and There. Oxford University Press, USA.

Leung B, Hargreaves AL, Greenberg DA, McGill B, Dornelas M, Freeman R. 2020. Clustered versus catastrophic global vertebrate declines. Nature **588**:267–271.

Lev E, Kahn PHJ, Chen H, Esperum G. 2020. Relatively wild urban parks can promote human resilience and flourishing: a case study of Discovery Park, Seattle, Washington. Frontiers in Sustainable Cities **2:**2.

Lindsey P et al. 2020. Conserving Africa's wildlife and wildlands through the COVID-19 crisis and beyond. Nature Ecology & Evolution **4**:1300–1310.

Lindsey P et al. 2021. Attracting investment for Africa's protected areas by creating enabling environments for collaborative management partnerships. Biological Conservation **255**:108979.

Lindsey PA et al. 2013a. The bushmeat trade in African savannas: Impacts, drivers, and possible solutions. Biological Conservation **160**:80–96.

Lindsey PA et al. 2018. More than $1 billion needed annually to secure Africa's protected areas with lions. Proceedings of the National Academy of Sciences **115**:E10788–E10796.

Lindsey PA et al. 2022. Shepherding sub-saharan africa's wildlife through peak anthropogenic pressure toward a green Anthropocene. Annual Review of Environment and Resources **47**:91–121.

Lindsey PA, Alexander R, Frank LG, Mathieson A, Romañach SS. 2006. Potential of trophy hunting to create incentives for wildlife conservation in Africa where alternative wildlife-based land uses may not be viable. Animal Conservation **9**:283–291.

Lindsey PA, Barnes J, Nyirenda V, Pumfrett B, Tambling CJ, Taylor WA, Rolfes M t'Sas. 2013b. The Zambian wildlife ranching industry: scale, associated benefits, and limitations affecting its development. PLoS ONE **8**: e81761

Lindsey PA, Frank LG, Alexander R, Mathieson A, Romañach SS. 2007a. Trophy Hunting and Conservation in Africa: Problems and One Potential Solution. Conservation Biology **21**:880–883.

Lindsey PA, Romanach SS, Davies-Mostert HT. 2009. The importance of conservancies for enhancing the value of game ranch land for large mammal conservation in southern Africa. Journal of Zoology **277**:99–105.

Lindsey PA, Roulet PA, Romañach SS. 2007b. Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. Biological Conservation **134**:455–469.

Linnell JDC, Strand O. 2000. Interference interactions, co-existence and conservation of mammalian carnivores. Diversity and Distributions **6**:169–176.

Lipsett-Moore GJ, Wolff NH, Game ET. 2018. Emissions mitigation opportunities for savanna countries from early dry season fire management. Nature Communications **9**:2247.

Little IT, Hockey PA, Jansen R. 2013. A burning issue: fire overrides grazing as a disturbance driver for South African grassland bird and arthropod assemblage structure and diversity. Biological Conservation **158**:258–270.

Lohmann D, Tietjen B, Blaum N, Joubert DF, Jeltsch F. 2012. Shifting thresholds and changing degradation patterns: climate change effects on the simulated long-term response of a semi-arid savanna to grazing. Journal of Applied Ecology **49**:814–823.

Loke PF, Kotzé E, Preez CC du, Twigge L. 2019. Dynamics of soil carbon concentrations and quality induced by agricultural land use in central South Africa. Soil Science Society of America Journal **83**:366–379.

Lorimer J, Sandom C, Jepson P, Doughty C, Barua M, Kirby KJ. 2015. Rewilding: science, practice, and politics. Annual Review of Environment and Resources **40**:39–62.

Louw MA, Haussmann NS, le Roux PC. 2019. Testing for consistency in the impacts of a burrowing ecosystem engineer on soil and vegetation characteristics across biomes. Scientific Reports **9**:19355.

Luvuno L, Biggs R, Stevens N, Esler K. 2018. Woody encroachment as a social-ecological regime shift. Sustainability **10**:2221..

Luyt ED. 2005. Models of Bontebok (*Damaliscus pygargus pygargus*, Pallas 1766) habitat preferences in the Bontebok National Park and sustainable stocking rates. MSc. thesis. University of Stellenbosch, Stellenbosch, South Africa.

Macdonald BCT, Farrell M, Tuomi S, Barton PS, Cunningham SA, Manning AD. 2014. Carrion decomposition causes large and lasting effects on soil amino acid and peptide flux. Soil Biology and Biochemistry **69**:132–140.

Macdonald DW, Jacobsen KS, Burnham D, Johnson PJ, Loveridge AJ. 2016. Cecil: a moment or a movement? Analysis of media coverage of the death of a lion, *Panthera leo*. Animals **6**:26.

Macfarlane R. 2007. The Wild Places. Granta Books, United Kingdom.

Madzwamuse M, Rihoy E, Louis M. 2020. Contested conservation: implications for rights, democratization, and citizenship in southern Africa. Development **63**:67–73.

Maestre FT et al. 2022. Grazing and ecosystem service delivery in global drylands. Science **378**:915–920.

Maka L, Aliber MA. 2019. The role of mentors in land reform projects supported through the recapitalisation and development programme: findings from Buffalo City Metropolitan Municipality, South Africa. South African Journal of Agricultural Extension **47**:37–45.

Malhi Y et al. 2022. The role of large wild animals in climate change mitigation and adaptation. Current Biology **32**:R181–R196.

Mallon DP, Stanley Price MR. 2013. The fall of the wild. Oryx **47**:467–468.

Mann HB. 1945. Nonparametric Tests Against Trend. Econometrica **13**:245–259.

Marsh SME et al. 2022. Prevalence of sustainable and unsustainable use of wild species inferred from the IUCN Red List of Threatened Species. Conservation Biology **36**:e13844.

Marvier M, Kareiva P. 2014. The evidence and values underlying 'new conservation.' Trends in Ecology & Evolution **29**:131–132.

Masubelele ML, Hoffman MT, Bond WJ, Gambiza J. 2014. A 50 year study shows grass cover has increased in shrublands of semi-arid South Africa. Journal of Arid Environments **104**:43–51.

Maund PR, Irvine KN, Dallimer M, Fish R, Austen GE, Davies ZG. 2020. Do ecosystem service frameworks represent people's values? Ecosystem Services **46**:101221.

Maxwell SL et al. 2020. Area-based conservation in the twenty-first century. Nature **586**:217–227.

Mbaiwa JE. 2005. Enclave tourism and its socio-economic impacts in the Okavango Delta, Botswana. Tourism Management **26**:157–172.

Mbaiwa JE. 2018. Effects of the safari hunting tourism ban on rural livelihoods and wildlife conservation in Northern Botswana. South African Geographical Journal **100**:41–61.

Mbaiwa JE, Hambira WL. 2021. Can the subaltern speak? Contradictions in trophy hunting and wildlife conservation trajectory in Botswana. Journal of Sustainable Tourism **31**: 1107–1125.

Mbanjwa VE, Hughes JC, Muchaonyerwa P. 2022. Organic carbon and aggregate stability of three contrasting soils as affected by arable agriculture and improved pasture in northern Kwazulu-Natal, South Africa. Journal of Soil Science and Plant Nutrition **22**:2378–2391.

McArthur C, Banks PB, Boonstra R, Forbey JS. 2014. The dilemma of foraging herbivores: dealing with food and fear. Oecologia **176**:677–689.

McDonald SE, Reid N, Smith R, Waters CM, Hunter J, Rader R. 2020. Comparison of biodiversity and ground cover between a commercial rotationally grazed property and an adjacent nature reserve in semi-arid rangeland. Austral Ecology **45**:60–69.

McDonald SE, Reid N, Waters CM, Smith R, Hunter J. 2018. Improving ground cover and landscape function in a semi-arid rangeland through alternative grazing management. Agriculture, Ecosystems & Environment **268**:8–14.

McElwee P et al. 2020. Ensuring a Post-COVID Economic Agenda Tackles Global Biodiversity Loss. One Earth **3**:448–461.

McGranahan DA. 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodiversity and Conservation **17**:1965–1977.

McGranahan DA, Kirkman KP. 2013. Multifunctional rangeland in southern Africa: managing for production, conservation, and resilience with fire and grazing. Land **2**:176–193.

McRae L, Freeman R, Geldmann J, Moss GB, Kjær-Hansen L, Burgess ND. 2022. A global indicator of utilized wildlife populations: Regional trends and the impact of management. One Earth **5**:422–433.

McSherry ME, Ritchie ME. 2013. Effects of grazing on grassland soil carbon: a global review. Global Change Biology **19**:1347–1357.

Meissner HH. 1982. Theory and application of a method to calculate forage intake of wild southern African ungulates for purposes of estimating carrying capacity. South African Journal of Wildlife Research - 24-month delayed open access **12**:42–47.

Meissner HH, Scholtz MM, Engelbrecht FA. 2013a. Sustainability of the South African Livestock Sector towards 2050 Part 2: Challenges, changes and required implementations. South African Journal of Animal Science **43**:289–319.

Meissner HH, Scholtz MM, Palmer AR. 2013b. Sustainability of the South African Livestock Sector towards 2050 Part 1: Worth and impact of the sector. South African Journal of Animal Science **43**:282–297.

Melville HI, Hetem RS, Strauss WM. 2021. Is climate change a concern for the ownership of game within fenced wildlife areas? Koedoe **63**:1–4.

Methorst J, Rehdanz K, Mueller T, Hansjürgens B, Bonn A, Böhning-Gaese K. 2021. The importance of species diversity for human well-being in Europe. Ecological Economics **181**:106917.

Meyer M, Klingelhoeffer E, Naidoo R, Wingate V, Börner J. 2021. Tourism opportunities drive woodland and wildlife conservation outcomes of community-based conservation in Namibia’s Zambezi region. Ecological Economics **180**:106863.

Meyer SM. 2006. The End of the Wild. MIT Press.

Miller SM et al. 2013. Management of reintroduced lions in small, fenced reserves in South Africa: an assessment and guidelines. South African Journal of Wildlife Research **43**:138–154.

Miller SM, Kotze A, Harper CK, Bishop JM, Williams VL, Rossouw C, Schoeman J, de Bruyn M, Dalton DL, Selier SAJ. 2023. Genetic diversity and origin of captive lion (*Panthera leo*) in South Africa: an assessment and comparison to wild populations. Conservation Genetics **24**: 721–737.

Ministry of Tourism and Wildlife. 2019. Report of the task force on consumptive wildlife utilization in Kenya. 101 pages. Ministry of Tourism and Wildlife, Nairobi, Kenya.

Minnie L, Gaylard A, Kerley GI. 2016. Compensatory life-history responses of a mesopredator may undermine carnivore management efforts. Journal of Applied Ecology **53**:379–387.

Mkhize N. 2014. Game farm conversions and the land question: unpacking present contradictions and historical continuities in farm dwellers' tenure insecurity in Cradock. Journal of Contemporary African Studies **32**:207–219.

Mkhize NR, Heitkönig IMA, Scogings PF, Hattas D, Dziba LE, Prins HHT, de Boer WF. 2016. Supplemental nutrients increase the consumption of chemically defended shrubs by free-ranging herbivores. Agriculture, Ecosystems & Environment **235**:119–126.

Mkono M. 2019. Neo-colonialism and greed: Africans' views on trophy hunting in social media. Journal of Sustainable Tourism **27**:689–704.

Mokgotsi RO. 2018. Effects of bush encroachment control in a communal managed area in the Taung region, North West province, South Africa. Thesis. North-West University South Africa.

Mokotjomela TM, Nombewu N. 2019. Potential benefits associated with implementation of the national biodiversity economy strategy in the Eastern Cape Province, South Africa. South African Geographical Journal **102**:190–208.

Momberg M, Haw AJ, Rajah P, van Rooyen J, Hawkins H-J. 2023. Kraals or bomas increase soil carbon and fertility across several biomes. African Journal of Range & Forage Science **40**:32–46.

Montfort F, Bégué A, Leroux L, Blanc L, Gond V, Cambule AH, Remane IAD, Grinand C. 2021. From land productivity trends to land degradation assessment in Mozambique: Effects of climate, human activities and stakeholder definitions. Land Degradation & Development **32**:49–65.

Moore JM, Manter DK, Maczko KA. 2023. Rotational grazing strategies minimally impact soil microbial communities and carbon dynamics—a Texas case study. Land **12**:1517.

Moranta J, Torres C, Murray I, Hidalgo M, Hinz H, Gouraguine A. 2022. Transcending capitalism growth strategies for biodiversity conservation. Conservation Biology **36**:e13821.

Moritz C, Funk V, Sakai AK. 2002. Strategies to protect biological diversity and the evolutionary processes that sustain it. Systematic Biology **51**:238–254.

Mosier S, Apfelbaum S, Byck P, Calderon F, Teague R, Thompson R, Cotrufo MF. 2021. Adaptive multi-paddock grazing enhances soil carbon and nitrogen stocks and stabilization through mineral association in southeastern U.S. grazing lands. Journal of Environmental Management **288**:112409.

Mtero F, Gumede N, Ramantsima K. 2019. Elite capture in land redistribution in South Africa. 90 pages. PLAAS Research Report 55. Institute for Poverty, Land and Agrarian Studies, University of Western Cape, Cape Town, South Africa.

Mukherjee N, Hugé J, Sutherland WJ, McNeill J, Van Opstal M, Dahdouh-Guebas F, Koedam N. 2015. The Delphi technique in ecology and biological conservation: applications and guidelines. Methods in Ecology and Evolution **6**:1097–1109.

Müller DWH, Codron D, Meloro C, Munn A, Schwarm A, Hummel J, Clauss M. 2013. Assessing the Jarman–Bell Principle: Scaling of intake, digestibility, retention time and gut fill with body mass in mammalian herbivores. Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology **164**:129–140.

Musengezi J. 2010. Wildlife utilization on private land: Understanding the economics of game ranching in South Africa. Thesis. University of Florida, USA.

Mwamidi DM, Renom JG, Fernández-Llamazares Á, Burgas D, Domínguez P, Cabeza M. 2018. Contemporary pastoral commons in East Africa as OECMs: a case study from the Daasanach community. Parks **24**:49–58. by: IUCN, Gland, Switzerland.

Mwasi S, Dheer A. 2022. Habitat degradation, vegetation damage, and wildlife-livestock interactions in Amboseli ecosystem wildlife sanctuaries, Kenya. African Journal of Ecology **60**:1201–1209.

Mysterud A. 2006. The concept of overgrazing and its role in management of large herbivores. Wildlife Biology **12**:129–141.

Mysterud A. 2010. Still walking on the wild side? Management actions as steps towards 'semi-domestication' of hunted ungulates. Journal of Applied Ecology **47**:920–925.

Mysterud A, Bonenfant C, Loe LE, Langvatn R, Yoccoz NG, Stenseth NC. 2008. The timing of male reproductive effort relative to female ovulation in a capital breeder. Journal of Animal Ecology **77**:469–477.

Nagy KA. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecological Monographs **57**:111–128.

Naidoo R, Weaver LC, Diggle RW, Matongo G, Stuart-Hill G, Thouless C. 2016. Complementary benefits of tourism and hunting to communal conservancies in Namibia. Conservation Biology **30**:628–638.

Naidu DGT, Roy S, Bagchi S. 2022. Loss of grazing by large mammalian herbivores can destabilize the soil carbon pool. Proceedings of the National Academy of Sciences **119**:e2211317119.

Naor L, Mayseless O. 2017. How personal transformation occurs following a single peak experience in nature: a phenomenological account. Journal of Humanistic Psychology **60**: 865-888.

Natusch DJD, Aust PW, Shine R. 2021. The perils of flawed science in wildlife trade literature. Conservation Biology **35**:1396–1404.

Nelson MP, Bruskotter JT, Vucetich JA, Chapron G. 2016. Emotions and the Ethics of Consequence in Conservation Decisions: Lessons from Cecil the Lion. Conservation Letters **9**:302–306.

Nenngwekhulu N. 2019. Financial analysis of the Recapitalisation and Development Programme in South Africa. Thesis. University of Pretoria, South Africa.

Netshipale AJ, Oosting SJ, Raidimi EN, Mashiloane ML, de Boer IJM. 2017. Land reform in South Africa: Beneficiary participation and impact on land use in the Waterberg District. NJAS - Wageningen Journal of Life Sciences **83**:57–66.

Ngubane M, Brooks S. 2013. Land beneficiaries as game farmers: conservation, land reform and the invention of the 'community game farm' in KwaZulu-Natal. Journal of Contemporary African Studies **31**:399–420.

Norgaard RB. 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. Ecological Economics **69**:1219–1227.

Noss RF et al. 2021. Improving species status assessments under the U.S. Endangered Species Act and implications for multispecies conservation challenges worldwide. Conservation Biology **35**:1715–1724.

Ntlou TF. 2016. Potential for economic sustainability of land reform projects benefitting from the Recapitalisation and Development Programme in South Africa. Thesis. University of Pretoria, South Africa.

Nyamayedenga S, Mashapa C, Chateya RJ, Gandiwa E. 2021. An assessment of the impact of the 2014 US elephant trophy importation ban on the hunting patterns in Matetsi Hunting Complex, north-west Zimbabwe. Global Ecology and Conservation **30**:e01758.

Obura DO et al. 2021. Integrate biodiversity targets from local to global levels. Science **373**:746–748.

O'Connor TG. 2005. Influence of land use on plant community composition and diversity in Highland Sourveld grassland in the southern Drakensberg, South Africa. Journal of Applied Ecology **42**:975–988.

O'Connor TG, Kuyler P, Kirkman KP, Corcoran B. 2010. Which grazing management practices are most appropriate for maintaining biodiversity in South African grassland? African Journal of Range & Forage Science **27**:67–76.

O'Connor TG, Puttick JR, Hoffman MT. 2014. Bush encroachment in southern Africa: changes and causes. African Journal of Range & Forage Science **31**:67–88.

O'Connor TG, van Wilgen BW. 2020. The Impact of Invasive Alien Plants on Rangelands in South Africa. Pages 459–487 in van Wilgen BW, Measey J, Richardson DM, Wilson JR, Zengeya TA, editors. Biological Invasions in South Africa. Springer International Publishing.

Odadi WO, Fargione J, Rubenstein DI. 2017. Vegetation, wildlife, and livestock responses to planned grazing management in an African pastoral landscape. Land Degradation & Development **28**:2030–2038.

Odling-Smee J, Erwin DH, Palkovacs EP, Feldman MW, Laland KN. 2013. Niche construction theory: a practical guide for ecologists. The Quarterly Review of Biology **88**:3–28.

Olden JD, LeRoy Poff N, Douglas MR, Douglas ME, Fausch KD. 2004. Ecological and evolutionary consequences of biotic homogenization. Trends in Ecology & Evolution **19**:18–24.

Oomen RJ, Linstädter A, Ruppert JC, Brüser K, Schellberg J, Ewert F. 2016. Effect of management on rangeland phytomass, cover and condition in two biomes in South Africa. African Journal of Range & Forage Science **33**:185–198.

Oro D, Genovart M, Tavecchia G, Fowler MS, Martínez-Abraín A. 2013. Ecological and evolutionary implications of food subsidies from humans. Ecology Letters **16**:1501–1514.

Otero I et al. 2020. Biodiversity policy beyond economic growth. Conservation Letters **13**:e12713.

Otfinowski R, Pinchbeck HG, Sinkins PA. 2017. Using cattle grazing to restore a rough fescue prairie invaded by Kentucky bluegrass. Rangeland Ecology & Management **70**:301–306.

Owen-Smith N. 1996. Ecological guidelines for waterpoints in extensive protected areas. South African Journal of Wildlife Research **26**:107–112.

Owen-Smith N. 2015. Mechanisms of coexistence in diverse herbivore–carnivore assemblages: demographic, temporal and spatial heterogeneities affecting prey vulnerability. Oikos **124**:1417–1426.

Packer C et al. 2013. Conserving large carnivores: dollars and fence. Ecology Letters **16**:635–641.

Palmer AR, Bennett JE. 2013. Degradation of communal rangelands in South Africa: towards an improved understanding to inform policy. African Journal of Range & Forage Science **30**:57–63.

Parker JD, Burkepile DE, Hay ME. 2006. Opposing effects of native and exotic herbivores on plant invasions. Science **311**:1459–1461.

Parker K, Vos AD, Clements HS, Biggs D, Biggs R. 2020. Impacts of a trophy hunting ban on private land conservation in South African biodiversity hotspots. Conservation Science and Practice **2**:e214.

Parr CL, Andersen AN. 2006. Patch Mosaic Burning for Biodiversity Conservation: a Critique of the Pyrodiversity Paradigm. Conservation Biology **20**:1610–1619.

Parr CL, Brockett BH. 1999. Patch-mosaic burning: a new paradigm for savanna fire management in protected areas? Koedoe **42**:117–130.

Parsons DAB, Shackleton CM, Scholes RJ. 1997. Changes in herbaceous layer condition under contrasting land use systems in the semi-arid lowveld, South Africa. Journal of Arid Environments **37**:319–329.

Pascual U et al. 2023. Diverse values of nature for sustainability. Nature **620**:813–823.

Pascual U, Adams WM, Díaz S, Lele S, Mace GM, Turnhout E. 2021. Biodiversity and the challenge of pluralism. Nature Sustainability **4**:567–572.

Pascual U, Balvanera P, Díaz S, Pataki G, Roth E, Stenseke M, Watson RT, Dessane EB, Islar M, Kelemen E. 2017. Valuing nature's contributions to people: the IPBES approach. Current Opinion in Environmental Sustainability **26**:7–16.

Pasmans T, Hebinck P. 2017. Rural development and the role of game farming in the Eastern Cape, South Africa. Land Use Policy **64**:440–450.

Peel M, Stalmans M. 2018. The effect of Holistic Planned Grazing^{TM} on African rangelands: a case study from Zimbabwe. African Journal of Range & Forage Science **35**:23–31.

Peel MJS, Biggs H, Zacharias PJK. 1999. The evolving use of stocking rate indices currently based on animal number and type in semi-arid heterogeneous landscapes and complex land-use systems. African Journal of Range & Forage Science **15**:117–127.

Pellegrini AFA et al. 2018. Fire frequency drives decadal changes in soil carbon and nitrogen and ecosystem productivity. Nature **553**:194–198.

Pérez-Hämmerle K-V, Moon K, Venegas-Li R, Maxwell S, Simmonds JS, Venter O, Garnett ST, Possingham HP, Watson JEM. 2022. Wilderness forms and their implications for global environmental policy and conservation. Conservation Biology **36**:e13875.

Perino A et al. 2019. Rewilding complex ecosystems. Science **364**: eaav5570

Peterson M, Lopez RR, Laurent EJ, Frank PA, Silvy NJ, Liu J. 2005. Wildlife loss through domestication: the case of endangered Key deer. Conservation Biology **19**:939–944.

Pettorelli N, Barlow J, Stephens PA, Durant SM, Connor B, Bühne HS to, Sandom CJ, Wentworth J, Toit JT du. 2018. Making rewilding fit for policy. Journal of Applied Ecology **55**:1114–1125.

Pettorelli N, Graham NAJ, Seddon N, Maria da Cunha Bustamante M, Lowton MJ, Sutherland WJ, Koldewey HJ, Prentice HC, Barlow J. 2021. Time to integrate global climate change and biodiversity science-policy agendas. Journal of Applied Ecology **58**:2384–2393.

Pfab MF, Victor JE, Armstrong AJ. 2011. Application of the IUCN Red Listing system to setting species targets for conservation planning purposes. Biodiversity and Conservation **20**:1001–1012.

Phukuntsi MA, Dalton DL, Mwale M, Selier J, Cebekhulu T, Sethusa MT. 2022. Genetic patterns in three South African specialist antelope species: Threats, conservation management and their implications. Conservation Science and Practice **4**:e12767.

Piao S et al. 2020. Characteristics, drivers and feedbacks of global greening. Nature Reviews Earth & Environment **1**:14–27.

Pickett STA. 2013. The flux of nature: changing worldviews and inclusive concepts. Pages 265–279 in Rozzi R, Pickett STA, Palmer C, Armesto JJ, Callicott JB, editors. Linking Ecology and Ethics for a Changing World. Springer, Dordrecht.

Pienaar EF, Rubino EC, Saayman M, van der Merwe P. 2017. Attaining sustainable use on private game ranching lands in South Africa. Land Use Policy **65**:176–185.

Pirie TJ, Thomas RL, Fellowes MD. 2017a. Game fence presence and permeability influences the local movement and distribution of South African mammals. African Zoology **52**:217–227.

Pirie TJ, Thomas RL, Fellowes MDE. 2017b. Increasing game prices may alter farmers' behaviours towards leopards (*Panthera pardus*) and other carnivores in South Africa. PeerJ **5**:e3369.

Pitman RT, Fattebert J, Williams ST, Williams KS, Hill RA, Hunter LTB, Slotow R, Balme GA. 2017. The conservation costs of game ranching. Conservation Letters **10**:403–413.

Pongsiri MJ, Roman J, Ezenwa VO, Goldberg TL, Koren HS, Newbold SC, Ostfeld RS, Pattanayak SK, Salkeld DJ. 2009. Biodiversity loss affects global disease ecology. BioScience **59**:945–954.

Pörtner H-O et al. 2021. Scientific outcome of the IPBES-IPCC co-sponsored workshop on biodiversity and climate change; 256 pages. IPBES Secretariat, Bonn, Germany.

Possingham HP, Andelman SJ, Burgman MA, Medellín RA, Master LL, Keith DA. 2002. Limits to the use of threatened species lists. Trends in Ecology & Evolution **17**:503–507.

Pozo RA et al. 2021. Reconciling livestock production and wild herbivore conservation: challenges and opportunities. Trends in Ecology & Evolution **36**:750–761.

Preger AC, Kösters R, Du Preez CC, Brodowski S, Amelung W. 2010. Carbon sequestration in secondary pasture soils: a chronosequence study in the South African Highveld. European Journal of Soil Science **61**:551–562.

Price EO. 1984. Behavioral aspects of animal domestication. Quarterly Review of Biology **59**:1–32.

Pringle RM, Goheen JR, Palmer TM, Charles GK, DeFranco E, Hohbein R, Ford AT, Tarnita CE. 2014. Low functional redundancy among mammalian browsers in regulating an encroaching shrub (*Solanum campylacanthum*) in African savannah. Proceedings of the Royal Society B: Biological Sciences **281**:20140390.

Pringle RM, Prior KM, Palmer TM, Young TP, Goheen JR. 2016. Large herbivores promote habitat specialization and beta diversity of African savanna trees. Ecology **97**:2640–2657.

Pryke JS, Roets F, Samways MJ. 2016. Wild herbivore grazing enhances insect diversity over livestock grazing in an African grassland system. PLOS ONE **11**:e0164198.

Puurtinen M, Elo M, Kotiaho JS. 2022. The Living Planet Index does not measure abundance. Nature **601**:E14–E15.

R Core Team. 2023. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available from https://www.R-project.org/.

Radloff FGT, BIrss C, Cowell C, Peinke D, Dalton D, Kotze A, Kerley GIH, Child MF. 2016. A conservation assessment of *Damaliscus pygargus pygargus*. In Child MF, Roxburgh L, Do Linh San E, Raimondo D, Davies-Mostert HT, editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Rahimi J, Mutua JY, Notenbaert AMO, Marshall K, Butterbach-Bahl K. 2021. Heat stress will detrimentally impact future livestock production in East Africa. Nature Food **2**:88–96.

Rahlao S, Mantlana B, Winkler H, Knowles T. 2012. South Africa's national REDD+ initiative: assessing the potential of the forestry sector on climate change mitigation. Environmental Science & Policy **17**:24–32.

Ranglack DH, du Toit JT. 2015. Wild bison as ecological indicators of the effectiveness of management practices to increase forage quality on open rangeland. Ecological Indicators **56**:145–151.

Ratajczak Z, Nippert JB, Collins SL. 2012. Woody encroachment decreases diversity across North American grasslands and savannas. Ecology **93**:697–703.

Rawat YS. 2017. Sustainable biodiversity stewardship and inclusive development in South Africa: A novel package for a sustainable future. Current Opinion in Environmental Sustainability **24**:89–95.

Raymond CM, Giusti M, Barthel S. 2018. An embodied perspective on the co-production of cultural ecosystem services: toward embodied ecosystems. Journal of Environmental Planning and Management **61**:778–799.

Redford KH et al. 2011. What does it mean to successfully conserve a (vertebrate) species? BioScience **61**:39–48.

Redford KH, Huntley BJ, Roe D, Hammond T, Zimsky M, Lovejoy TE, da Fonseca GAB, Rodriguez CM, Cowling RM. 2015. Mainstreaming biodiversity: conservation for the twenty-first century. Frontiers in Ecology and Evolution **3**:137.

Reed MS, Stringer LC, Dougill AJ, Perkins JS, Atlhopheng JR, Mulale K, Favretto N. 2015. Reorienting land degradation towards sustainable land management: Linking sustainable livelihoods with ecosystem services in rangeland systems. Journal of Environmental Management **151**:472–485.

Reinhart KO, Sanni Worogo HS, Rinella MJ, Vermeire LT. 2021. Livestock increase soil organic carbon in the Northern Great Plains. Rangeland Ecology & Management **79**:22–27.

Reith J, Ghazaryan G, Muthoni F, Dubovyk O. 2021. Assessment of land degradation in semiarid Tanzania - using multiscale remote sensing datasets to support Sustainable Development Goal 15.3. Remote Sensing **13**:1754.

Ridder B. 2007. The naturalness versus wildness debate: ambiguity, inconsistency, and unattainable objectivity. Restoration Ecology **15**:8–12.

Riginos C, Porensky LM, Veblen KE, Odadi WO, Sensenig RL, Kimuyu D, Keesing F, Wilkerson ML, Young TP. 2012. Lessons on the relationship between livestock husbandry and biodiversity from the Kenya Long-term Exclosure Experiment (KLEE). Pastoralism: Research, Policy and Practice **2**:10.

Ringma JL, Wintle B, Fuller RA, Fisher D, Bode M. 2017. Minimizing species extinctions through strategic planning for conservation fencing. Conservation Biology **31**:1029–1038.

Ripple WJ, Larsen EJ, Renkin RA, Smith DW. 2001. Trophic cascades among wolves, elk and aspen on Yellowstone National Park’s northern range. Biological Conservation **102**:227–234.

Ritchie ME. 2014. Plant compensation to grazing and soil carbon dynamics in a tropical grassland. PeerJ **2**:e233.

Roche C. 2008. “The Fertile Brain and Inventive Power of Man”: Anthropogenic factors in the cessation of springbok treks and the disruption, of the Karoo ecosystem, 1865-1908. Journal of the International African Institute **78**:157–188.

Rodrigues AS, Brooks TM, Butchart SH, Chanson J, Cox N, Hoffmann M, Stuart SN. 2014. Spatially explicit trends in the global conservation status of vertebrates. PLoS One **9**:e113934.

Rodrigues ASL, Pilgrim JD, Lamoreux JF, Hoffmann M, Brooks TM. 2006. The value of the IUCN Red List for conservation. Trends in Ecology & Evolution **21**:71–76.

Romañach SS, Lindsey PA, Woodroffe R. 2007. Determinants of attitudes towards predators in central Kenya and suggestions for increasing tolerance in livestock dominated landscapes. Oryx **41**:185–195.

Root-Bernstein M, Guerrero-Gatica M, Piña L, Bonacic C, Svenning J-C, Jaksic FM. 2017. Rewilding-inspired transhumance for the restoration of semiarid silvopastoral systems in Chile. Regional Environmental Change **17**:1381–1396.

Roques K g., O'Connor T g., Watkinson A r. 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. Journal of Applied Ecology **38**:268–280.

Rusenga C. 2020. Setting them up to fail: enforcement of the agribusiness model on land reform projects in South Africa. Review of African Political Economy **47**:382–398.

Russell JM, Ward D. 2016. Historical land-use and vegetation change in northern Kwazulu-Natal, South Africa. Land Degradation & Development **27**:1691–1699.

Russo I-RM, Hoban S, Bloomer P, Kotzé A, Segelbacher G, Rushworth I, Birss C, Bruford MW. 2019. 'Intentional Genetic Manipulation' as a conservation threat. Conservation Genetics Resources **11**:237–247.

Rust NA, Whitehouse-Tedd KM, MacMillan DC. 2013. Perceived efficacy of livestock-guarding dogs in South Africa: Implications for cheetah conservation. Wildlife Society Bulletin **37**:690–697.

Saayman M, van der Merwe P, Saayman A. 2018. The economic impact of trophy hunting in the South African wildlife industry. Global Ecology and Conservation **16**:e00510.

SANBI. 2020. Strategic review of the land reform and biodiversity stewardship initiative. South African National Biodiversity Institute.

SANBI, SANParks. 2018. National Biodiversity Assessment: 2018 Terrestrial Formal Protected Areas. South African National Biodiversity Institute.

Sandhage-Hofmann A, Kotzé E, van Delden L, Dominiak M, Fouché HJ, van der Westhuizen HC, Oomen RJ, du Preez CC, Amelung W. 2015. Rangeland management effects on soil properties in the savanna biome, South Africa: A case study along grazing gradients in communal and commercial farms. Journal of Arid Environments **120**:14–25.

Sandhage-Hofmann A, Linstädter A, Kindermann L, Angombe S, Amelung W. 2021. Conservation with elevated elephant densities sequesters carbon in soils despite losses of woody biomass. Global Change Biology **27**:4601–4614.

Sandom C, Dalby L, Fløjgaard C, Kissling WD, Lenoir J, Sandel B, Trøjelsgaard K, Ejrn\a es R, Svenning J-C. 2013a. Mammal predator and prey species richness are strongly linked at macroscales. Ecology **94**:1112–1122.

Sandom C, Donlan CJ, Svenning J-C, Hansen D. 2013b. Rewilding. Pages 430–451 Key Topics in Conservation Biology 2. John Wiley & Sons, Ltd.

Sandom CJ, Middleton O, Lundgren E, Rowan J, Schowanek SD, Svenning J-C, Faurby S. 2020. Trophic rewilding presents regionally specific opportunities for mitigating climate change. Philosophical Transactions of the Royal Society B: Biological Sciences **375**:20190125.

Sankaran M, Augustine DJ, Ratnam J. 2013. Native ungulates of diverse body sizes collectively regulate long-term woody plant demography and structure of a semi-arid savanna. Journal of Ecology **101**:1389–1399.

Saura S, Bertzky B, Bastin L, Battistella L, Mandrici A, Dubois G. 2018. Protected area connectivity: Shortfalls in global targets and country-level priorities. Biological Conservation **219**:53–67.

Sayre R et al. 2020. An assessment of the representation of ecosystems in global protected areas using new maps of World Climate Regions and World Ecosystems. Global Ecology and Conservation **21**:e00860.

Scheiter S, Schulte J, Pfeiffer M, Martens C, Erasmus BFN, Twine WC. 2019. How does climate change influence the economic value of ecosystem services in savanna rangelands? Ecological Economics **157**:342–356.

Schmidt A, Kerley G, Watson L. 2019. Threshold changes in the structure and composition of bush clumps in piospheres in an arid thicket mosaic, South Africa. Journal of Arid Environments **169**:19–28.

Schmidt KT, Hoi H. 2002. Supplemental feeding reduces natural selection in juvenile red deer. Ecography **25**:265–272.

Schmitz OJ et al. 2014. Animating the Carbon Cycle. Ecosystems **17**:344–359.

Schmitz OJ et al. 2023. Trophic rewilding can expand natural climate solutions. Nature Climate Change **13**:324–333.

Schmitz OJ, Buchkowski RW, Smith JR, Telthorst M, Rosenblatt AE. 2017. Predator community composition is linked to soil carbon retention across a human land use gradient. Ecology **98**:1256–1265.

Schmitz OJ, Leroux SJ. 2020. Food webs and ecosystems: Linking species interactions to the carbon cycle. Annual Review of Ecology, Evolution, and Systematics **51**:271–295.

Schmitz OJ, Wilmers CC, Leroux SJ, Doughty CE, Atwood TB, Galetti M, Davies AB, Goetz SJ. 2018. Animals and the zoogeochemistry of the carbon cycle. Science **362**: eaar3213

Scholte P, Pays O, Adam S, Chardonnet B, Fritz H, Mamang J-B, Prins HHT, Renaud P-C, Tadjo P, Moritz M. 2022. Conservation overstretch and long-term decline of wildlife and tourism in the Central African savannas. Conservation Biology **36**:e13860.

Scholtz MM, Jordaan FJ, Thuli Chabalala N, Pyoos GM, Joel Mamabolo M, Neser FW. 2023. A balanced perspective on the contribution of extensive ruminant production to greenhouse gas emissions in southern Africa. African Journal of Range & Forage Science **40**:107–113.

Scholtz R et al. 2022. High-intensity fire experiments to manage shrub encroachment: lessons learned in South Africa and the United States. African Journal of Range & Forage Science **39**:148–159.

Schou JS, Bladt J, Ejrnæs R, Thomsen MN, Vedel SE, Fløjgaard C. 2021. Economic assessment of rewilding versus agri-environmental nature management. Ambio **50**:1047–1057.

Schulte to Bühne H, Pettorelli N, Hoffmann M. 2022a. The policy consequences of defining rewilding. Ambio **51**:93–102.

Schulte to Bühne H, Ross B, Sandom CJ, Pettorelli N. 2022b. Monitoring rewilding from space: The Knepp estate as a case study. Journal of Environmental Management **312**:114867.

Scott-Shaw R, Morris CD. 2015. Grazing depletes forb species diversity in the mesic grasslands of KwaZulu-Natal, South Africa. African Journal of Range & Forage Science **32**:21–31.

Selebatso M, Bennitt E, Maude G, Fynn RW. 2018. Water provision alters wildebeest adaptive habitat selection and resilience in the Central Kalahari. African Journal of Ecology **56**:225–234.

Selier J, Nel L, Rushworth I, Kruger J, Coverdale B, Mulqueeny C, Blackmore A. 2018. An assessment of the potential risks of the practice of intensive and selective breeding of game to biodiversity and the biodiversity economy in South Africa. 188 pages. Scientific Authority, South African National Biodiversity Institute, Pretoria, South Africa.

Selinske MJ, Cooke B, Torabi N, Hardy MJ, Knight AT, Bekessy SA. 2017. Locating financial incentives among diverse motivations for long-term private land conservation. Ecology and Society **22**: 26270074

Sen A. 2013. The ends and means of sustainability. Journal of Human Development and Capabilities **14**:6–20.

Sen PK. 1968. Estimates of the regression coefficient based on Kendall’s tau. Journal of the American Statistical Association **63**:1379–1389.

Seymour CL, Milton SJ, Joseph GS, Dean WRJ, Ditlhobolo T, Cumming GS. 2010. Twenty years of rest returns grazing potential, but not palatable plant diversity, to Karoo rangeland, South Africa. Journal of Applied Ecology **47**:859–867.

Shackleton CM, Mograbi PJ, Drimie S, Fay D, Hebinck P, Hoffman MT, Maciejewski K, Twine W. 2019. Deactivation of field cultivation in communal areas of South Africa: Patterns, drivers and socio-economic and ecological consequences. Land Use Policy **82**:686–699.

Shackleton CM, Shackleton SE, Cousins B. 2001. The role of land-based strategies in rural livelihoods: The contribution of arable production, animal husbandry and natural resource harvesting in communal areas in South Africa. Development Southern Africa **18**:581–604.

Shiffman DS, Macdonald CC, Wester JN, Walsh MB, Chevalier A, Kachelriess D, Friedman KJ. 2021. Marine species conservation at CITES: How does media coverage inform or misinform? Marine Policy **134**:104813.

Shin Y-J et al. 2022. Actions to halt biodiversity loss generally benefit the climate. Global Change Biology **28**:2846–2874.

Shumba T, De Vos A, Biggs R, Esler KJ, Ament JM, Clements HS. 2020. Effectiveness of private land conservation areas in maintaining natural land cover and biodiversity intactness. Global Ecology and Conservation **22**:e00935.

Sianga K, van Telgen M, Vrooman J, Fynn RWS, van Langevelde F. 2017. Spatial refuges buffer landscapes against homogenisation and degradation by large herbivore populations and facilitate vegetation heterogeneity. Koedoe **59**:1–13.

Sims-Castley R, Kerley GI, Geach B, Langholz J. 2005. Socio-economic significance of ecotourism-based private game reserves in South Africa’s Eastern Cape Province. Parks **15**:6–18.

Sitters J, Kimuyu DM, Young TP, Claeys P, Olde Venterink H. 2020. Negative effects of cattle on soil carbon and nutrient pools reversed by megaherbivores. Nature Sustainability **3**:360–366.

Skinner JD, Chimimba CT. 2005. The Mammals of the Southern African Subregion. Cambridge University Press, Cambridge, England.

Skowno AL, Jewitt D, Slingsby JA. 2021. Rates and patterns of habitat loss across South Africa’s vegetation biomes. South African Journal of Science **117**:1–5.

Skowno AL, Raimondo DC, Poole CJ, Fizzotti B, Slingsby JA. 2019. South African National Biodiversity Assessment 2018 Technical Report Volume 1: Terrestrial Realm. South African National Biodiversity Institute, Pretoria.

Skowno AL, Thompson MW, Hiestermann J, Ripley B, West AG, Bond WJ. 2017. Woodland expansion in South African grassy biomes based on satellite observations (1990–2013): general patterns and potential drivers. Global Change Biology **23**:2358–2369.

Smit C, Ruifrok JL, van Klink R, Olff H. 2015. Rewilding with large herbivores: The importance of grazing refuges for sapling establishment and wood-pasture formation. Biological Conservation **182**:134–142.

Smit GN. 2004. An approach to tree thinning to structure southern African savannas for long-term restoration from bush encroachment. Journal of Environmental Management **71**:179–191.

Smit IP, Grant CC, Devereux BJ. 2007. Do artificial waterholes influence the way herbivores use the landscape? Herbivore distribution patterns around rivers and artificial surface water sources in a large African savanna park. Biological Conservation **136**:85–99.

Smit IP, Peel MJ, Ferreira SM, Greaver C, Pienaar DJ. 2020. Megaherbivore response to droughts under different management regimes: lessons from a large African savanna. African Journal of Range & Forage Science **37**:65–80.

Smit IPJ, Archibald S. 2019. Herbivore culling influences spatio-temporal patterns of fire in a semiarid savanna. Journal of Applied Ecology **56**:711–721.

Smith DR, Allan NL, McGowan CP, Szymanski JA, Oetker SR, Bell HM. 2018a. Development of a species status assessment process for decisions under the U.S. Endangered Species Act. Journal of Fish and Wildlife Management **9**:302–320.

Smith JR et al. 2018b. A global test of ecoregions. Nature Ecology & Evolution **2**:1889–1896.

Snijders D. 2014. Wildlife policy matters: inclusion and exclusion by means of organisational and discursive boundaries. Journal of Contemporary African Studies **32**:173–189.

Snyman HA. 1999. Short-term effects of soil water, defoliation and rangeland condition on productivity of a semi-arid rangeland in South Africa. Journal of Arid Environments **43**:47–62.

Snyman HA, Ingram LJ, Kirkman KP. 2013. *Themeda triandra*: a keystone grass species. African Journal of Range & Forage Science **30**:99–125.

Snyman S, Sumba D, Vorhies F, GItari E, Enders C, Ahenken A, Pambo AFK, Bengone N. 2021. State of the Wildlife Economy in Africa. 219 pages. African Leadership University, School of Wildlife Conservation, Kigali, Rwanda.

Somers MJ, Hayward M, editors. 2012. Fencing for Conservation. Springer New York, New York, NY.

Somers MJ, Walters M, Measey J, Strauss WM, Turner AA, Venter JA, Nel L, Kerley GIH, Taylor WA, Moodley Y. 2020. The implications of the reclassification of South African wildlife species as farm animals. South African Journal of Science **116**:1–2.

Soto-Navarro C et al. 2020. Mapping co-benefits for carbon storage and biodiversity to inform conservation policy and action. Philosophical Transactions of the Royal Society B: Biological Sciences **375**:20190128.

South African National Biodiversity Institute (2006-2018). 2018. The Vegetation Map of South Africa, Lesotho and Swaziland. South African National Biodiversity Institute.

Spash CL. 2022. Conservation in conflict: Corporations, capitalism and sustainable development. Biological Conservation **269**:109528.

Spencer K, Sambrook M, Bremner-Harrison S, Cilliers D, Yarnell RW, Brummer R, Whitehouse-Tedd K. 2020. Livestock guarding dogs enable human-carnivore coexistence: First evidence of equivalent carnivore occupancy on guarded and unguarded farms. Biological Conservation **241**:108256.

Spierenburg M. 2020. Living on other people's land; impacts of farm conversions to game farming on farm dwellers' abilities to access land in the Eastern Cape, South Africa. Society & Natural Resources **33**:280–299.

Spierenburg M, Brooks S. 2014. Private game farming and its social consequences in post-apartheid South Africa: contestations over wildlife, property and agrarian futures. Journal of Contemporary African Studies **32**:151–172.

Stafford W. 1999. The Way It Is: New and Selected Poems. Graywolf Press, Saint Paul, USA.

Star SL, Griesemer JR. 1989. Institutional ecology, "translations" and boundary objects: Amateurs and professionals in Berkeley's Museum of Vertebrate Zoology, 1907-39. Social Studies of Science **19**:387–420.

Statistics South Africa. 2020. Census of commercial agriculture 2017. 104 pages. 11. Statistics South Africa, Pretoria.

Stears K, Shrader AM. 2020. Coexistence between wildlife and livestock is contingent on cattle density and season but not differences in body size. PLOS ONE **15**:e0236895.

Stevens N, Erasmus BFN, Archibald S, Bond WJ. 2016. Woody encroachment over 70 years in South African savannahs: overgrazing, global change or extinction aftershock? Philosophical Transactions of the Royal Society B: Biological Sciences **371**:20150437.

Stockmann U et al. 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystems & Environment **164**:80–99.

Strong M, Silva JA. 2020. Impacts of hunting prohibitions on multidimensional well-being. Biological Conservation **243**:108451.

't Sas-Rolfes M, Emslie R, Adcock K, Knight M. 2022. Legal hunting for conservation of highly threatened species: The case of African rhinos. Conservation Letters **15**:e12877.

Takacs D. 2020. Whose voices count in biodiversity conservation? Ecological democracy in biodiversity offsetting, REDD+, and rewilding. Journal of Environmental Policy & Planning **22**:43–58.

Talore DG, Tesfamariam EH, Hassen A, Toit JD, Klampp K, Jean-Francois S. 2016. Long-term impacts of grazing intensity on soil carbon sequestration and selected soil properties in the arid Eastern Cape, South Africa. Journal of the Science of Food and Agriculture **96**:1945–1952.

Tanentzap AJ, Coomes DA. 2012. Carbon storage in terrestrial ecosystems: do browsing and grazing herbivores matter? Biological Reviews **87**:72–94.

Taylor RD, Walker BH. 1978. Comparisons of vegetation use and herbivore biomass on a Rhodesian game and cattle ranch. Journal of Applied Ecology **15**:565–581.

Taylor WA, Child MF, Lindsey PA, Nicholson SK, Relton C, Davies-Mostert HT. 2021. South Africa's private wildlife ranches protect globally significant populations of wild ungulates. Biodiversity and Conservation **30**:4111–4135

Taylor WA, Lindsey PA, Davies-Mostert HT. 2015. An assessment of the economic, social and conservation value of the wildlife ranching industry and its potential to support the green economy in South Africa. 160 pages. Endangered Wildlife Trust, Johannesburg, South Africa.

Taylor WA, Lindsey PA, Nicholson SK, Relton C, Davies-Mostert HT. 2020. Jobs, game meat and profits: The benefits of wildlife ranching on marginal lands in South Africa. Biological Conservation **245**:108561.

Teague R, Barnes M. 2017. Grazing management that regenerates ecosystem function and grazingland livelihoods. African Journal of Range & Forage Science **34**:77–86.

Teague WR, Dowhower SL, Baker SA, Haile N, DeLaune PB, Conover DM. 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. Agriculture, Ecosystems & Environment **141**:310–322.

Tear TH, Wolff NH, Lipsett-Moore GJ, Ritchie ME, Ribeiro NS, Petracca LS, Lindsey PA, Hunter L, Loveridge AJ, Steinbruch F. 2021. Savanna fire management can generate enough carbon revenue to help restore Africa's rangelands and fill protected area funding gaps. One Earth **4**: 1776–1791

Terborgh JW. 2015. Toward a trophic theory of species diversity. Proceedings of the National Academy of Sciences **112**:11415–11422.

Thakholi L. 2021. Conservation labour geographies: Subsuming regional labour into private conservation spaces in South Africa. Geoforum **123**:1–11.

Thomas AD. 2012. Impact of grazing intensity on seasonal variations in soil organic carbon and soil CO2 efflux in two semiarid grasslands in southern Botswana. Philosophical Transactions of the Royal Society B: Biological Sciences **367**:3076–3086.

Thomson RL, Forsman JT, Sardà-Palomera F, Mönkkönen M. 2006. Fear factor: prey habitat selection and its consequences in a predation risk landscape. Ecography **29**:507–514.

Thoreau HD. 1862. Walking. The Atlantic.

Tomor BM, Owen-Smith N. 2002. Comparative use of grass regrowth following burns by four ungulate species in the Nylsvley Nature Reserve, South Africa. African Journal of Ecology **40**:201–204.

Torres A et al. 2018. Measuring rewilding progress. Philosophical Transactions of the Royal Society B: Biological Sciences **373**:20170433.

Treydte AC, Baumgartner S, Heitkönig IMA, Grant CC, Getz WM. 2013. Herbaceous forage and selection patterns by ungulates across varying herbivore assemblages in a South African savanna. PLoS ONE **8**:e82831

Trollope WSW. 1990. Development of a technique for assessing veld condition in the Kruger National Park using key grass species. Journal of the Grassland Society of Southern Africa **7**:46–51.

Trollope WSW, Trollope LA, Bosch OJH. 1990. Veld and pasture management terminology in southern Africa. Journal of the Grassland Society of Southern Africa **7**:52–61.

Turpie J, Botha P, Coldrey K, Forsythe K, Knowles T, Letley G, Allen J, de Wet R. 2019. Towards a policy on indigenous bush encroachment in South Africa. 16 pages. Department of Environmental Affairs, Pretoria, South Africa.

Turpie J, Letley G. 2018. An initial investigation into the potential feasibility and design of a market-based certification scheme for the wildlife sector of South Africa. 102 pages. Anchor Environmental Consultants Report No: AEC/1810. United Nations Development Programme and Department of Environmental Affairs, Pretoria, South Africa.

Tyrrell M, Clark DA. 2014. What happened to climate change? CITES and the reconfiguration of polar bear conservation discourse. Global Environmental Change **24**:363–372.

Tyrrell P, Russell S, Western D. 2017. Seasonal movements of wildlife and livestock in a heterogenous pastoral landscape: Implications for coexistence and community-based conservation. Global Ecology and Conservation **12**:59–72.

UNCCD/Science-Policy Interface. 2016. Land in balance. The scientific conceptual framework for land degradation neutrality (LDN). Science-Policy Brief 02. United Nations Convention to Combat Desertification (UNCCD), Bonn, Germany.

Valente C. 2009. The food (in)security impact of land redistribution in South Africa: microeconometric evidence from national data. World Development **37**:1540–1553.

Valls Fox H, Bonnet O, Cromsigt JPGM, Fritz H, Shrader AM. 2015. Legacy effects of different land-use histories interact with current grazing patterns to determine grazing lawn soil properties. Ecosystems **18**:720–733.

van der Merwe P, Saayman M. 2003. Determining the economic value of game farm tourism. Koedoe **46**:103–112.

van der Merwe P, Saayman M, Rossouw R. 2014. The economic impact of hunting: a regional approach. South African Journal of Economic and Management Sciences **17**:379–395.

van Horn G, Hausdoerffer J, editors. 2017. Wildness: Relations of People and Place. University of Chicago Press.

van Houdt S, Brown RP, Wanger TC, Twine W, Fynn R, Uiseb K, Cooney R, Traill LW. 2021. Divergent views on trophy hunting in Africa, and what this may mean for research and policy. Conservation Letters **14**:e12840.

van Rooyen J. 2016. Livestock production and animal health management systems in communal farming areas at the wildlife-livestock interface in southern Africa. Thesis. University of Pretoria, South Africa.

van Rooyen N, van Rooyen G. 2017. Ecological Evaluation of Tswalu Kalahari Reserve. 143 pages. Independent Report.

van Wilgen BW, Forsyth GG, Le Maitre DC, Wannenburgh A, Kotzé JDF, van den Berg E, Henderson L. 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biological Conservation **148**:28–38.

van Wilgen BW, Govender N, Forsyth GG, Kraaij T. 2011. Towards adaptive fire management for biodiversity conservation: Experience in South African National Parks. Koedoe **53**:96–104.

van Wilgen BW, Reyers B, Le Maitre DC, Richardson DM, Schonegevel L. 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. Journal of Environmental Management **89**:336–349.

van Zyl H, Kinghorn J. 2021. Effectiveness of Fees for Permits and Licenses under NEMBA: Baseline Report. 69 pages. UNDP and DFFE, Pretoria, South Africa.

Vannini P, Vannini AS. 2019. Wildness as vitality: A relational approach. Environment and Planning E: Nature and Space **2**:252–273.

Veblen KE, Nehring KC, McGlone CM, Ritchie ME. 2015. Contrasting effects of different mammalian herbivores on sagebrush plant communities. PLOS ONE **10**:e0118016.

Veldhuis MP, Gommers MI, Olff H, Berg MP. 2018. Spatial redistribution of nutrients by large herbivores and dung beetles in a savanna ecosystem. Journal of Ecology **106**:422–433.

Veldman JW et al. 2019. Comment on "The global tree restoration potential." Science **366**:eaay7976

Venables WN, Ripley BD. 2002. Modern Applied Statistics with S, 4th edition. Springer-Verlag, New York.

Venter ZS, Cramer MD, Hawkins H-J. 2018. Drivers of woody plant encroachment over Africa. Nature Communications **9**:1–7.

Venter ZS, Cramer MD, Hawkins H-J. 2019. Rotational grazing management has little effect on remotely-sensed vegetation characteristics across farm fence-line contrasts. Agriculture, Ecosystems & Environment **282**:40–48.

Venter ZS, Hawkins H-J, Cramer MD. 2017. Implications of historical interactions between herbivory and fire for rangeland management in African savannas. Ecosphere **8**:e01946.

Venter ZS, Hawkins H-J, Cramer MD, Mills AJ. 2021. Mapping soil organic carbon stocks and trends with satellite-driven high resolution maps over South Africa. Science of The Total Environment **771**:145384.

Venter ZS, Scott SL, Desmet PG, Hoffman MT. 2020. Application of Landsat-derived vegetation trends over South Africa: Potential for monitoring land degradation and restoration. Ecological Indicators **113**:106206.

Vetter S. 2013. Development and sustainable management of rangeland commons – aligning policy with the realities of South Africa's rural landscape. African Journal of Range & Forage Science **30**:1–9.

Vetter S. 2020. With power comes responsibility – a rangelands perspective on forest landscape restoration. Frontiers in Sustainable Food Systems **4**:549483

Vial F, Cleaveland S, Rasmussen G, Haydon DT. 2006. Development of vaccination strategies for the management of rabies in African wild dogs. Biological Conservation **131**:180–192.

Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, Pergl J, Schaffner U, Sun Y, Pyšek P. 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. Ecology Letters **14**:702–708.

Visconti P, Butchart SH, Brooks TM, Langhammer PF, Marnewick D, Vergara S, Yanosky A, Watson JE. 2019. Protected area targets post-2020. Science **364**:239–241

von Brandis RG, Reilly BK. 2007. A temporal analysis of trophy quality in South Africa: has trophy quality changed over time. South African Journal of Wildlife Research **37**:153–158.

von Solms W, Merwe P van der. 2020. Farm size and its impact on land use: The case of the South African private wildlife industry. Open Agriculture **5**:844–856.

Vundla T, Mutanga O, Sibanda M. 2020. Quantifying grass productivity using remotely sensed data: an assessment of grassland restoration benefits. African Journal of Range & Forage Science **37**:247–256.

Wachiye S, Pellikka P, Rinne J, Heiskanen J, Abwanda S, Merbold L. 2022. Effects of livestock and wildlife grazing intensity on soil carbon dioxide flux in the savanna grassland of Kenya. Agriculture, Ecosystems & Environment **325**:107713.

Waldram MS, Bond WJ, Stock WD. 2008. Ecological engineering by a mega-grazer: white rhino impacts on a South African savanna. Ecosystems **11**:101–112.

Walker BH, Emslie RH, Owen-Smith RN, Scholes RJ. 1987. To cull or not to cull: lessons from a southern African drought. Journal of Applied Ecology **24**:381–401.

Wang L, Pedersen PBM, Svenning J-C. 2023. Rewilding abandoned farmland has greater sustainability benefits than afforestation. npj Biodiversity **2**:1–4.

Wanger TC, Traill LW, Cooney R, Rhodes JR, Tscharntke T. 2017. Trophy hunting certification. Nature Ecology & Evolution **1**:1791–1793.

Watson JEM, Dudley N, Segan DB, Hockings M. 2014. The performance and potential of protected areas. Nature **515**:67–73.

Watson JEM, Venter O, Lee J, Jones KR, Robinson JG, Possingham HP, Allan JR. 2018. Protect the last of the wild. Nature **563**:27–30.

Watson LH, Kraaij T, Novellie P. 2011. Management of rare ungulates in a small park: habitat use of Bontebok and Cape Mountain Zebra in Bontebok National Park assessed by counts of dung groups. South African Journal of Wildlife Research **41**:158–166.

Watters F, Stringham O, Shepherd CR, Cassey P. 2022. The U.S. market for imported wildlife not listed in the CITES multilateral treaty. Conservation Biology **36**:e13978.

Weber DS, Mandler T, Dyck M, Van Coeverden De Groot PJ, Lee DS, Clark DA. 2015. Unexpected and undesired conservation outcomes of wildlife trade bans—An emerging problem for stakeholders? Global Ecology and Conservation **3**:389–400.

Webster H, Dickman A, Hart A, Roe D. 2022. Keeping hunting bans on target. Conservation Biology **36**:e13932.

Weeber J, Hempson GP, February EC. 2020. Large herbivore conservation in a changing world: Surface water provision and adaptability allow wildebeest to persist after collapse of long-range movements. Global Change Biology **26**:2841–2853.

Weinzettel J, Hertwich EG, Peters GP, Steen-Olsen K, Galli A. 2013. Affluence drives the global displacement of land use. Global Environmental Change **23**:433–438.

Weise FJ, Wessels Q, Munro S, Solberg M. 2014. Using artificial passageways to facilitate the movement of wildlife on Namibian farmland. South African Journal of Wildlife Research **44**:161–166.

Wells HBM, Kimuyu DM, Odadi WO, Dougill AJ, Stringer LC, Young TP. 2021. Wild and domestic savanna herbivores increase smaller vertebrate diversity, but less than additively. Journal of Applied Ecology **58:**953–963.

Wessels KJ et al. 2011. Impact of communal land use and conservation on woody vegetation structure in the Lowveld savannas of South Africa. Forest Ecology and Management **261**:19–29.

Wessels KJ, Prince SD, Carroll M, Malherbe J. 2007a. Relevance of rangeland degradation in semiarid northeastern South Africa to the nonequilibrium theory. Ecological Applications **17**:815–827.

Wessels KJ, Prince SD, Malherbe J, Small J, Frost PE, VanZyl D. 2007b. Can human-induced land degradation be distinguished from the effects of rainfall variability? A case study in South Africa. Journal of Arid Environments **68**:271–297.

West TAP, Wunder S, Sills EO, Börner J, Rifai SW, Neidermeier AN, Frey GP, Kontoleon A. 2023. Action needed to make carbon offsets from forest conservation work for climate change mitigation. Science **381**:873–877.

Western D, Tyrrell P, Brehony P, Russell S, Western G, Kamanga J. 2020. Conservation from the inside-out: Winning space and a place for wildlife in working landscapes. People and Nature **2**:279–291.

White JDM, Stevens N, Fisher JT, Archibald S, Reynolds C. 2022. Nature-reliant, low-income households face the highest rates of woody-plant encroachment in South Africa. People and Nature **4**:1020–1031.

Wickham H, François R, Henry L, Müller K. 2020. dplyr: A Grammar of Data Manipulation. R package version 1.0.2. Available from https://CRAN.R-project.org/package=dplyr.

Williams CA, Albertson JD. 2006. Dynamical effects of the statistical structure of annual rainfall on dryland vegetation. Global Change Biology **12**:777–792.

Willoughby JR, Ivy JA, Lacy RC, Doyle JM, DeWoody JA. 2017. Inbreeding and selection shape genomic diversity in captive populations: Implications for the conservation of endangered species. PloS one **12**:e0175996.

Wilson N, Kershaw P, Marnewick D, Purnell A. 2018. Biodiversity Stewardship Guideline. A guideline produced for the Department of Environmental Affairs. South African National Biodiversity Institute.

Winterbach CW, Whitesell C, Somers MJ. 2015. Wildlife Abundance and Diversity as Indicators of Tourism Potential in Northern Botswana. PLOS ONE **10**:e0135595.

Wolmer W, Chaumba J, Scoones I. 2004. Wildlife management and land reform in southeastern Zimbabwe: a compatible pairing or a contradiction in terms? Geoforum **35**:87–98.

Woodroffe R, Hedges S, Durant SM. 2014. To fence or not to fence. Science **344**:46–48.

World Bank Group. 2021. Unlocking Nature-Smart Development: An Approach Paper on Biodiversity and Ecosystem Services. 67 pages. World Bank, Washington, DC.

Wright OT, Cundill G, Biggs D. 2018. Stakeholder perceptions of legal trade in rhinoceros horn and implications for private reserve management in the Eastern Cape, South Africa. Oryx **52**:175–185.

Xiong D, Shi P, Zhang X, Zou CB. 2016. Effects of grazing exclusion on carbon sequestration and plant diversity in grasslands of China—A meta-analysis. Ecological Engineering **94**:647–655.

Yang LH, Bastow JL, Spence KO, Wright AN. 2008. What can we learn from resource pulses. Ecology **89**:621–634.

Yang Y, Tilman D, Furey G, Lehman C. 2019. Soil carbon sequestration accelerated by restoration of grassland biodiversity. Nature Communications **10**:718.

Yapi TS, O'Farrell PJ, Dziba LE, Esler KJ. 2018. Alien tree invasion into a South African montane grassland ecosystem: impact of Acacia species on rangeland condition and livestock carrying capacity. International Journal of Biodiversity Science, Ecosystem Services & Management **14**:105–116.

Yoshida T, Jones LE, Ellner SP, Fussmann GF, Hairston NG. 2003. Rapid evolution drives ecological dynamics in a predator–prey system. Nature **424**:303–306.

Young TP, Porensky LM, Riginos C, Veblen KE, Odadi WO, Kimuyu DM, Charles GK, Young HS. 2018. Relationships between cattle and biodiversity in multiuse landscape revealed by Kenya long-term exclosure experiment. Rangeland Ecology & Management **71**:281–291.

Zhou G, Zhou X, He Y, Shao J, Hu Z, Liu R, Zhou H, Hosseinibai S. 2017. Grazing intensity significantly affects belowground carbon and nitrogen cycling in grassland ecosystems: a meta-analysis. Global Change Biology **23**:1167–1179.

Zhou Y et al. 2023. Soil carbon in tropical savannas mostly derived from grasses. Nature Geoscience **16**:710–716.

Zhou Y, Singh J, Butnor JR, Coetsee C, Boucher PB, Case MF, Hockridge EG, Davies AB, Staver AC. 2022. Limited increases in savanna carbon stocks over decades of fire suppression. Nature **603**:445–449.

Zomer RJ, Neufeldt H, Xu J, Ahrends A, Bossio D, Trabucco A, van Noordwijk M, Wang M. 2016. Global Tree Cover and Biomass Carbon on Agricultural Land: The contribution of agroforestry to global and national carbon budgets. Scientific Reports **6**:29987.

zu Ermgassen SOSE, McKenna T, Gordon J, Willcock S. 2018. Ecosystem service responses to rewilding: first-order estimates from 27 years of rewilding in the Scottish Highlands. International Journal of Biodiversity Science, Ecosystem Services & Management **14**:165–178.