Full title
A framework to measure the wildness of managed large vertebrate populations

Article Impact Statement
This is a regulatory tool to empirically measure the wildness of managed populations to bridge the science-policy divide.

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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 protected areas and other wildlife-based land-uses are self-sustaining and free to adapt. We present a framework comprising six attributes relating to the evolutionary and ecological dynamics of vertebrates. For each attribute, we 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. We 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.

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. 2016), 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 2017). 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 (Mortiz 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 2011:191) 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 2017). 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, we 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. Our 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.

Methods

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).

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.

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 (F. Cloete unpubl. data): 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.

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 Appendix S1 and S2 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 Appendix S2 and S3 for more detail). All analyses were performed in R 3.4.2 (R Core Team 2014).

Results

The Framework

Six interlinked attributes relating to evolutionary and ecological dynamics were identified as contributing to the wildness of a population (Table 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 2). The quantifiable variables for each attribute from Table 1 were then converted into empirical thresholds (both binary and continuous) to delineate between states (see framework summary in Table 3). The division between non-wild and wild states was drawn between IM and SN (Table 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.

Framework application

In testing the framework, we 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; Fig. 1), 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*) (Fig. 1, Appendix S2). 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 had 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 (Appendix S3).

Discussion

We 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. We have taken these foundational frameworks one step further by testing whether their theoretical underpinnings have efficacy as a regulatory tool. We 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), 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.

In line with species conservation guidelines (IUCN Standards and Petitions Subcommittee 2017), 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.

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 our 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.

We 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. The current evidence demonstrates that captive-bred animals have reduced fitness in unmanaged landscapes (McPhee 2004, Jules 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. We 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 us 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. Our 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 paragraph

Explanation of wildness scoring system (Appendix S1) and ordered logistic regression results (Appendix S2) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Tables

Table 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 state threshold values.

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. (2013) McArthur et al. (2014) Terborgh (2015)	Presence/absence of predators. Functional composition of predator community. Frequency of exposure to predators.

		Owen-Smith (2015)	
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
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 and Hoyle (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.

Table 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 population. .

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-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,	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 can be supported naturally (without intervention). Population likely to become extinct over time.	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.

	harvesting and translocation.		
Near Natural (NN)	Very few interventions that 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-sustainable 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.

Table 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.

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). Mesopredators: 0 species (excluded or removed) Apex predators: 0 species	Small predators: ≥ 1 species continuous exposure Mesopredators: ≥ 1 species occasional exposure (removed) Apex predators: 0 species (excluded or removed or absent)	Small predators: ≥ 3 species continuous exposure Mesopredators: ≥ 2 species continuous exposure (removed *ad hoc*)	Small predators: ≥3 species continuous exposure Mesopredators: ≥ 2 species continuous exposure Apex predators: ≥ 1 species	Small predators: ≥ 3 species continuous exposure Mesopredators: ≥ 2 species continuous exposure Apex predators: ≥ 2 species

	(excluded or removed)		Apex predators: ≥ 1 species occasional exposure (removed *ad hoc*, controlled or absent)	continuous exposure (removed *ad hoc*)	continuous exposure
Exposure to natural food limitations and fluctuations	Continuous food provision to all individuals in enclosures.	> 1 supplementary feeding events per year on average; salt licks	= 1 supplementary feeding event per year on average	< 1 supplementary feeding events per year on average	No supplementary feeding events
	Habitat management: no access to natural habitat	Habitat management: ≥ 1 habitat modifications 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:	Off-take / augmentation: individuals	Off-take / augmentation: individuals	Off-take / augmentation: non-selective	Off-take / augmentation: non-

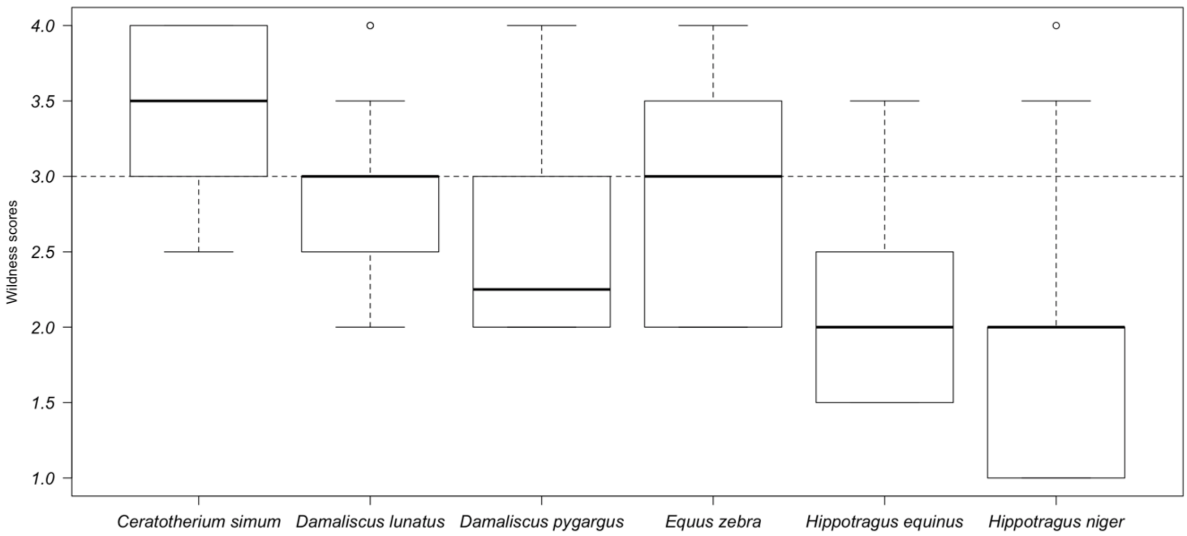
	individuals selected for genotypes (based on stud book)	selected for specific traits	selected to simulate dispersal as part of metapopulation strategy	(based on post-reproductive age where appropriate)	selective (based on post-reproductive age where appropriate); no hybridisation; no augmentation following initial reintroduction

* The division between wild and non-wild populations is drawn between Simulated Natural (SN) and Intensively Managed (IM) respectively.

Figure-legend page

Figure 1. 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).

Figures with legends



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Explanation of wildness scoring system (Appendix S1)

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 S1). 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. (2006) 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 a the size of a typical breeding herd (*sensu* Skinner and 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 and 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 diseases 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 refer 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 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 absolutely 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 2017). 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 2017). 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 taken into account. 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 S1. 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	Occasional exposure to apex predators and continuous exposure to other predator guilds.	All predator functional guilds present. No predator control except occasional live capture	Full complement of predators present. No predator control.

		where mesocarnivores actively managed (through hunting and culling).	Apex predators may be subject to contraception control. No active predator control but damage-causing individuals removed non-lethally (live capture and translocation).	and release of damage-causing apex predators.	
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 to disperse). ≥ 50% water-points are artificial (dams and boreholes), thus limited seasonality in water availability	<1 water-point / home range unit of the species (i.e. some social units do not have access to water and must disperse). <50% water-points are artificial (dams and boreholes), thus water sources are predominantly seasonal	<0.5water-points / home range unit of the species < 25% water-points are artificial (dams and boreholes), thus water sources are predominantly seasonal.	<0.25 water-point / home range unit of the species. 100% natural water-points, thus water sources are all seasonal.
Reproduction	Presence of 'stud' male in camps (1 breeding male / enclosure)	Population size < 1 social unit – no natural breeding competition =. Selective	Population size comprises at least 1 social unit (i.e. natural	Population size ≥ 2 social units. No selective breeding in place. No	Population size ≥ 3 social units . No selective breeding in place. No alien

	(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	breeding for colour variants or specific traits. Presence of alien / extra-limital species. Trophy hunting or live sales activities	competition for mates). No selective breeding in place. No alien / extra-limital species. Subsistence hunting or culling for habitat management	alien / extra-limital species. Subsistence hunting or culling for habitat management	/ extra-limital species. Subsistence hunting or culling for habitat management

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Wildness scores: full results and summary tables (Appendix S2)

The individual attribute score for each population of each species are displayed in Table S2. 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 1).
2. We identified wildness states (or nodes) along the wildness spectrum (adapted from Redford et al. 2011) and defined these (Table 2). Identifying discrete states is necessary so as to develop quantitative thresholds to make the framework measureable.
3. We used the indicator variables of each attribute to set empirical thresholds between wildness states (Table 3).
4. The thresholds were used to assign a score for each attribute for a given population (see S1), 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 S2).

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 S2. 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 (km^2)	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. The framework was applied to all populations of the six pilot species (Table S1). The commercial value of the species is taken from the average game auction prices (114 auctions) in 2014 (F. Cloete unpubl. data). "Populations" refers to the number of properties in the dataset in which the species occurred.

Table S3. 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