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

REPORT

AUGUST 2018

Images obtained from the Web

Jeanetta Selier, Lizanne Nel, Ian Rushworth, Johan Kruger, Brent Coverdale, Craig Mulqueeny and Andrew Blackmore

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

Task team

Dr Jeanetta Selier, South African National Biodiversity Institute

Mr Andy Blackmore, Ezemvelo KZN Wildlife

Mr. Brent Coverdale, Ezemvelo KZN Wildlife

Mr. Johan Kruger, Limpopo, Department of Economic Development, Environment and Tourism

Mr. Craig Mulqueeny, Ezemvelo KZN Wildlife

Mrs. Lizanne Nel, South African Hunters and Game Conservation Association

Mr. Ian Rushworth, Ezemvelo KZN Wildlife

Contributing authors

Prof Paulette Bloomer, Head Department of Genetics, University of Pretoria

Dr Mike Jordan, IUCN Species Survival Commission Conservation Breeding Specialist Group

Dr Kelly Marnewick, Endangered Wildlife Trust

Dr Tim O'Connor, University of the Witwatersrand

Prof Michael Somers, Centre for Wildlife Management, University of Pretoria

Dr Andrew Taylor, Endangered Wildlife Trust

Dr Gerhard Verdoorn, Griffon Poison Information Centre

External reviewers

Prof Brian Reilly, Tshwane University of Technology

Dr David Mallon, IUCN Antelope Specialist Group

Dr Harriet Davies-Mostert, Endangered Wildlife Trust

Michael 't Sas-Rolfes, University of Oxford

Prof Paul Grobler, University of the Free State

Dr Peter Novellie, Research Associate at Nelson Mandela Metropolitan University

Dr Adrian Tordiffe, University of Pretoria

Figure 1: Task team members.

From left to right: Dr Andy Blackmore, Ezemvelo KZN Wildlife; Mr. Johan Kruger, Limpopo, Department of Economic Development, Environment and Tourism; Dr Jeanetta Selier, South African National Biodiversity Institute; Mrs. Lizanne Nel, South African Hunters and Game Conservation Association; Mr. Brent Coverdale, Ezemvelo KZN Wildlife; Mr. Ian Rushworth, Ezemvelo KZN Wildlife and Mr. Craig Mulqueeny, Ezemvelo KZN Wildlife.

Well done to you and the team! You have accomplished an enormous task and this report will be hugely valuable in guiding policy and practice to avoid some potentially irreversible impacts from the industry. I found the report incredibly interesting and have also learned a lot in reading it.

Dr H Davies-Mostert

I am really impressed with this report. It is comprehensive, well thought through and clearly laid out. The authors should be commended for completing such a mammoth task and doing so with such a high level of care and professionalism.

Dr A Tordiffe

This is an exceedingly thorough piece of work. The team and their coordinator should be congratulated on a job well done.

Prof P Grobler

This a complex and frequently controversial topic. A structured approach to assessment evaluation is therefore needed and the IPBES (2014) methodology selected is both fully appropriate, already tested in several global situations and widely appreciated and understood.

This is an admirably thorough, carefully researched, and professionally executed report that explores every detail (that I can think of) of a problematic – and increasingly widespread – set of issues. I greatly enjoyed reading it.

Dr D Mallon

What a wonderful piece of work and congrats to all involved. I particularly liked the rational approach and the excellent referencing. I hope the authors can be encouraged to publish this in a reduced form in a scientific journal and present at conferences. It is a fine example of what can be done with existing info.

Prof B Reilly

South Africa's hitherto world-leading track record in conservation, driven by a unique combination of public management and private enterprise, currently faces new challenges. This important and thorough research highlights some divergence between private and public interests in the biodiversity sector in relation to breeding practices. Hopefully South Africa's vibrant wildlife ranching industry will take these concerns to heart and help to forge an improved regulatory environment that more closely aligns the objectives of ranchers and sustainable landscape conservation, thereby mitigating the risks identified here.

Michael 't Sas-Rolfes

TABLE OF CONTENTS

TABLE OF FIGURES

TABLE OF TABLES

DEFINITIONS AND ACRONYMS USED

DEFINITIONS

The concepts below are defined as follows for the purpose of this report:

Adaptation is a trait that permits an organism to function well in its environment and endows it with capabilities especially appropriate for its particular environment.

Allele is any of several forms of a gene, usually arising through mutation, which are responsible for hereditary variation.

Bred in captivity or captive-bred means that the animal was bred in a controlled environment.

"Canned" hunting refers to the shooting and killing of an animal in a confined space, such as a fenced enclosure, whereby the animal has no chance of escape or when the animal is drugged.

Conservation unit refers to a genetically distinct group that may be a subspecies, subpopulation, ecotype or evolutionary significant unit.

Commercial purposes mean that the primary purpose of the activity is to obtain economic benefit, including profit in cash or in kind, and is directed towards trade, exchange or another form of economic use or benefit

Controlled environment means an enclosure designed to hold game in a way that prevents them from escaping and that facilitates intensive human intervention or manipulation in the form of the provision of food or water; artificial housing; planned parasite control; mate selection; or health care; and may facilitates the intensive breeding of game but excludes fenced land on which self-sustaining wildlife populations of that species are managed in an extensive wildlife system.

Domestication (from Latin *domesticus*: "of the home") is the process whereby a population of living organisms is changed at the genetic level, through generations of selective breeding, to accentuate traits that ultimately benefit the interests of humans.

Epistasis is where the action of one gene is influenced by the interaction with other genes

Extensive wildlife system means a system that is large enough, and suitable for the management of self-sustaining wildlife populations with natural mate selection in a natural environment which requires minimal human intervention in the form of the provision of the supplementation of food, except in times of drought; control of parasites; provision of health care.

Fitness is a measure of an animal's ability to produce offspring that can contribute genetically to the next generation.

Founder effect is the loss of genetic variation that occurs when a new (daughter) population is established by a very small number of individuals from a larger population (parent) and the founders of the new (daughter) population were not a representative sample of the larger (parent) population.

Game refers to all species of terrestrial mammals which are utilised through hunting and for the purposes of this document refers to antelope and large predators.

Gene is a unit of heredity which is transferred from a parent to offspring and is held to determine some characteristic of the offspring.

Genotype refers to a particular set of genes possessed by an individual.

Home range of an animal is the area where it spends its time; it is the region that encompasses all the resources the animal requires to survive and reproduce.

Hybridization is the term used for the making of crosses or hybrids. However, a distinction should be made in terms of interspecific and intraspecific crosses.

Inbreeding is used to describe various related phenomena that all refer to situations in which matings occur among close relatives and to an increase in homozygosity associated with such matings.

Intensive and selective breeding refers to the deliberate selection of and breeding for selected animal traits, usually in controlled conditions.

Introgression is the genetic exchange that occurs between species (rather than between populations of a single species). Introgression occurs when hybridization leads to the creation of fertile offspring, which can then backcross with one or both parental species.

Line breeding is a mating system which is designed to maintain a substantial degree of relationship to a highly regarded ancestor or group of ancestors without causing high levels of inbreeding. It is considered a mild form of inbreeding.

Natural selection is the differential reproductive success of different phenotypes, a process which slowly changes gene frequencies in populations.

Outbreeding depression is the phenomenon where the interbreeding between two populations may result in a reduction in number, viability, or fitness of offspring.

Phenotype is the outward appearance and behaviour of the individual, which is determined by interactions between the genotype and the environment.

Piospheres are bare tramped-out areas that develop around water-points and other areas of animal concentration.

Pleiotropy is the phenomenon where different traits may be influenced by the same genes.

Problem animal is a wild vertebrate animal which has the perceived potential to create a degree of conflict with landowners, either directly or indirectly, resulting in economic losses.

Put-and-take hunting is defined as hunting of animals bred (intensive or extensive) in one area and then released into another for the purpose of hunting.

Receiving environment means the area/property into which a problem animal is released.

Reputation can be defined as a stakeholder's overall evaluation of an enterprise over time where this evaluation is made up from the stakeholder's experience of the visible behaviour of the enterprise, as well as the images based on its communication and its symbolism.

Scientific Authority means the Scientific Authority referred to in section 60 of the Biodiversity Act.

Sensitive environments are considered as Critical Biodiversity Areas as designated in bioregional plans or systematic biodiversity conservation plans, Threatened Ecosystems, National Protected Area Expansion Strategy Focus Areas or any other ecologically sensitive areas.

ACRONYMS

African Lion Working Group	ALWG
Associations of Zoos and Aquariums	AZA
Boone & Crockett Club	B&CC
Endangered Wildlife Trust	EWT
Confederation of Hunters Associations of South Africa	CHASA
Department of Environmental Affairs	DEA
International Council for Game and Wildlife Conservation	CIC
Federation of Namibian Tourism Associations in Namibia	FENATA
Game Rangers Association of Africa	GRAA
Hohe Jagd & Fischerei Fair	HJFF
International Union for Conservation of Nature	IUCN
IUCN Antelope Specialist Group	IUCN SSC ASG
Jagd & Hund Exhibition	J&H
Environmental Affairs and tourism lab process to develop strategies for growing the biodiversity economy	LAB
Limits of Acceptable Change	LAC
National Association of Conservancies/Stewardship of SA	NACSSA
Namibia Professional Hunting Association	NAPHA
Namibian Ministry for Environment and Tourism	MET
National Environmental Management Act 107 of 1998	NEMA
Nordic Safari Club	NSC
National Council of Societies for the Prevention of Cruelty to Animals	NSPCA
Outfitters and Professional Hunters Associations of Africa	OPHAA
Professional Hunters Association of South Africa	PHASA
South African Hunters and Game Conservation Association	SAHGCA
South African Movement for the Promotion of Ethical Outfitters	SAMPEO
South African National Biodiversity Institute	SANBI
Safari Club International	SCI
Spiral Horn Antelope Club	SPAC
Threshold of Potential Concern	TPC

United Nations	UN
Wildlife Ranching South Africa	WRSA
Wild Sheep Foundation	WSF

AKNOWLEDGEMENTS

We are grateful to everyone who contributed valuable information, data, photographs or useful opinions including Riel Coetzer and Pieter Nel. We would like to thank Prof Paulette Bloomer of the University of Pretoria, Dr Mike Jordan of Chester Zoo, Dr Kelly Marnewick of the Endangered Wildlife Trust, Dr Tim O'Connor of the University of the Witwatersrand, Prof. Michael Somers of the Centre for Wildlife Management, UP, Dr Andrew Taylor of the Endangered Wildlife Trust and Dr Gerhard Verdoorn of the Griffon Poison Information Centre for reviewing sections of this manuscript and their valuable inputs into the manuscript. We would also like to thank our external reviewers, Prof Brian Reilly (Tshwane University of Technology), Dr David Mallon (IUCN Antelope Specialist Group), Dr Harriet Davies-Mostert (Endangered Wildlife Trust), Mr. Michael 't Sas-Rolfes (University of Oxford), Prof Paul Grobler (University of the Free State), Dr Peter Novellie (Research Associate at Nelson Mandela Metropolitan University) and Dr Adrian Tordiffe (University of Pretoria) for peer reviewing the final manuscript. Our thanks to the South African National Biodiversity Institute, South African Hunters and Game Conservation Association and Ezemvelo KZN Wildlife for providing funding for generating the report.

EXECUTIVE SUMMARY

Over the last three decades the South African wildlife industry has been largely compatible with conserving biodiversity and as such has made a significant contribution thereto (Child et al. 2012). However, in recent years, selective breeding and the intensive management of game has emerged as a new and growing sector within the broader private wildlife industry (Cloete et al. 2015, Taylor et al. 2015). Concerns have been raised about the long-term and potential consequences of the practice on other sub-sectors of the wildlife sector, as well as the country's biodiversity and biodiversity economy (Cousins et al. 2010, Dalerum and Miranda 2016, Pienaar et al. 2017). Following concerns raised within the Scientific Authority of South Africa in 2009 and the subsequent request from the Minister of the Department of Environmental Affairs an expert task team, consisting of scientists with a diverse range of skills and expertise was established by the Scientific Authority on in February 2013. The purpose of the task team was to both identify and assess the full range of potential risks to biodiversity and the biodiversity economy, and to compile a report for submission to the Scientific Authority. The Scientific Authority, in accordance with section 61 of NEMBA, would in turn advise the Minister on appropriate, if required, policy and regulatory responses.

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services' conceptual framework was used to assess the potential risks of selective breeding and intensive management of game to South Africa's biodiversity. Seen potential biodiversity risks/issues relating to the practice of Intensive and Selective Breeding of game were identified using the best available published scientific literature, information obtained from members of the wildlife sector, experts and the national dialogue process (Njobe and King 2016). From these seven potential Issues (risks)**,** 17 potential **Impacts** (harms/stressors) are described with specific **Concerns** highlighted under each Impact Statement. The Impact Statement thus describes the current understanding associated with each anticipated Impact. Each impact was then assessed and scored on the quality of scientific evidence available, the probability of occurrence within the industry, and the likely impact on an ecosystem and species level respectively. The quality of the evidence was evaluated for scientific rigour using the 'uncertainty approach' as used by the UK National Ecosystem Assessment. This approach consists of a set of uncertainty terms derived from a 4-box model and complemented, where possible, and placed on a 'likelihood of manifestation' scale. A hierarchical ranking method was used to rank the impacts on a gradient from highest to the lowest impact at an ecosystem and threatened species level respectively. All Least Concern species were omitted as the assumption was made *a priori* that present the impact on these species is unlikely. However, it is acknowledged that over time, depending on the scale, these risks and concerns may affect even these species. To determine the impacts with the highest potential risk at an ecosystems level, impacts with a score of 1 (Virtually certain) or 2 (Likely) were used. This was followed by ranking the selected impacts according to quality of evidence, only selecting impacts with a score of 1 or 2, and then lastly on the probability of occurrence within the industry. A similar process was followed for assessing the risk to Threatened or Near Threatened species.

Although the focus of this report is primarily on direct risks to biodiversity, the biodiversity economy and its potential in addressing socio-economic challenges of the country, it is an important focus area of government. Social and economic risks related to the biodiversity economy, may pose indirect risks to biodiversity and its contribution to the broader economy. One issue and two impacts relating to the practice of intensive and selective breeding of game on the biodiversity economy were identified based on current events within the biodiversity economy. A similar process to IPBES was used to gather evidence and consider level of agreement. Due to the integrated nature of Biodiversity Economy Risks, it could however not be weighted using similar criteria as for the Biodiversity Risks.

The White Paper on the Conservation and Sustainable Use of South Africa's Biological Diversity, the main policy document pertaining to the use and conservation of biodiversity in South Africa, is modelled on the Convention on Biological Diversity (CBD) (Cousins et al. 2010). According to the CBD Article 2, sustainable use encompasses 'the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations'. This definition of sustainable use centres on the management and use of wild species and ecosystems within biologically sustainable limits (Hutton and Leader-Williams 2003). As such, sustainable use presents two challenges: (1) "to ensure that use increasingly becomes biologically sustainable"; and (2) "that wherever possible it serves as a conservation strategy to conserve specific resources and prevent the conversion of land to uses that are incompatible with biodiversity conservation" (Hutton and Leader-Williams 2003). The practice of intensive and selective breeding based on the findings of this assessment may not meet these criteria for sustainability.

It is concluded that intensive management and selective breeding of game poses a number of significant risks to biodiversity at landscape, ecosystem and species levels, as well as to other sectors of the biodiversity economy of South Africa, and may compromise the current and future contribution of the wildlife industry to biodiversity conservation. This assessment has identified several important direct risks and impacts on biodiversity at different scales, as well as indirect collateral negative impacts on conservation and the broader wildlife economy. Issues that were identified as key risks to biodiversity on an ecosystem and Threatened and Near Threatened species level, Ecosystems level only and Threatened and Near Threatened species level only are listed in the table below.

Table 1 : Key biodiversity risks identified on an ecosystem and species level.

Ecosystem and species level risks	Ecosystem level risks	Species level risks
10. Off-label use of pesticides and unlawful use of hazardous substances cause mortality of indigenous species resulting in changes in ecosystem functioning and increased threats to the conservation of threatened.	6.1. Fragmentation of the landscape through impermeable fencing restricts movement of free-ranging species and reduces habitat quality and quantity.	4.1. Expression of deleterious attributes that may lead to physical, behavioural and lethal outcomes.
7.1. The killing of predators and other conflict species may result in a reduction in population numbers which in turn may lead to a change in the conservation status of the species and thereby furthering the extinction risk.	6.2. Concentration of species in small areas with impermeable fences for intensive breeding purposes results in habitat degradation within such areas.	4.2. Loss of genetic diversity resulting in decreased fitness and reduced adaptive potential.
7.2. The disruption of social structures of species targeted for removal may exacerbate the conflict potential as a result of the constant removals of individuals. This in turn could result in a decline in the survival rate of the affected population. This constant removal of dispersers may also create a loss or disruption of dispersal opportunities, thereby increasing local extinction risk.	7.3. The removal of predators will at a certain scale disrupt predation as a natural process in the broader landscape/environment thereby affecting ecosystem functioning.	4.5. Domestication of wild species resulting in a loss of their natural ability to adapt to wild conditions.
		5.1. Changes in natural genetic composition, evolutionary trajectory and adaptive potential of wild populations through the introgression of captive population genetics wherein genetic changes in the captive population may lead to an altering genetic composition and/or evolutionary trajectory and/or adaptive

		potential of wild populations through deliberate and accidental introductions.
		5.3. The removal of wild specimens of naturally rare species or species with currently small population sizes, in South Africa or other African countries where sourcing is often cheaper, can lead to population declines resulting in a lower overall conservation status and a higher extinction risk for these species.
		8.1. Development of resistance to stock remedies and veterinary medicines resulting in microbes, helminths and ectoparasites that may start infesting free-roaming game and livestock on a large scale with conservation and economic consequences.
		8.2. Disruption of the process of natural selection in terms of host-parasite evolution with resulting loss of disease and parasite resistance within the game population.

A mix of regulatory, awareness-raising and incentive-based systems need to be implemented in order to mitigate the risks posed by this sub-sector of the wildlife industry. Given the challenges and costs of a regulatory approach, wherever possible, incentive-based approaches should be used as well as taking advantage of market forces to reward practices that are more compatible with biodiversity conservation and that are less risky to the biodiversity economy. However, the necessary enabling legislative framework for this needs to be created. Lastly, government and all role players in the wildlife economy, should take cognisance of potential far reaching implications of developing new ventures and sub-sectors within the wildlife sector. Principles of business and environmental sustainability as entrenched in NEMA and the King reports on governance that considers social, environmental and economic aspects within the current and future landscape of the country would be critical to ensure sustainable growth of the biodiversity economy to the benefit of all.

Figure 2: Unintended consequences of electric strands very low to the ground. Various invertebrate and vertebrate species electrocuted when trying to move underneath this fence at an intensive breeding facility in Limpopo Province.

CHAPTER 1: PURPOSE OF THE REPORT

The last decade has seen the emergence and unprecedented growth of a sub-sector of the wildlife ranching industry focusing on the intensive and selective breeding of wildlife for commercial purposes. Key elements of this sector involve intensive breeding of high value rare species and active selective for rare and unusual traits. What started as a small niche market is now mainstream, with 45% of game ranchers involved in this practice which covered an estimated 6% (1 000 000 ha) of the wildlife estate by 2015 (Taylor et al. 2015). Much of this is taking place in areas where extensive game ranching previously thrived. Moreover, this sector of the industry has been growing rapidly, and growth in investment in this sector was said to outperform investment portfolios listed on the Johannesburg Stock Exchange (Slabbert 2013).

As a result of persistent concerns being raised regarding the environmental impacts and full socio-economic implications of this new practice, the Minister of Environmental Affairs requested the Scientific Authority, as designated in the National Environmental Management: Biodiversity Act, 10 of 2004, to identify, investigate and assess the current and potential risks of the practice of intensive and/or selective breeding of indigenous antelope and predator species to the biodiversity heritage of the country (19 March 2010).

An expert task team consisting of scientists with a diverse range of skills and experience was established by the Scientific Authority in February 2013 to both identify and assess these risks to biodiversity and the biodiversity economy, and to compile a report for submission to the Scientific Authority, which would advise the Minister on appropriate policy and regulatory responses.

The purpose of this report is to provide a thorough assessment of the potential risks of the practices of intensive and selective breeding of game to biodiversity and the biodiversity economy. While not specifically part of the brief, some potential mitigation measures and recommendations are provided to assist decision makers. It is envisaged that the report will facilitate decision-making and the development of a coherent policy and regulatory framework within the broader environmental legal framework (embracing the concepts of sustainable use and responsible economic growth, thus ensuring that the needs of both present and future generations are met).

CHAPTER 2: BACKGROUND

Where wildlife has limited or no socio-economic value it is likely to dwindle and disappear from the landscape becuse it cannot compete with other economic activities such as mining and agriculture (Prins and Grootenhuis 2000). However, where wildlife has economic value it has proven to be a competitive land-use (Child et al. 2012, Di Minin et al. 2013b). In South Africa the promulgation of the Game Theft Act (No. 105 of 1991; as amended in Acts 18 of 1996 and 62 of 2000), which grants conditional ownership of wildlife to private landowners that obtain a Certificate of Adequate Enclosure (CAE), consolidated the foundations of an economically viable wildlife industry (Carruthers 2008). The right of ownership of wildlife, combined with a growing understanding that wildlife ranching was ecologically and financially sustainable, significantly reduced subsidies for conventional agriculture. Increasing financial incentives for commercial wildlife ranching has led to a tremendous increase in game numbers over the past 30 years and the establishment of a formal wildlife sector in South Africa. However, at present knowledge on the true scale and scope of wildlife ranching across South Africa is very limited, but there is agreement that the industry is growing at probably more than 6% per annum (Cloete et al. 2015, Taylor et al. 2015). Estimates on the scale and scope vary between various reports and documents (NAMC 2006, Bothma and Von Bach 2010, Cloete et al. 2015, Taylor et al. 2015). A recent report estimated that there are approximately 8 979 wildlife ranches in the country, covering an estimated total land area of 1 221 000 km^2 (Taylor et al. 2015). This means that the area covered by commercial wildlife ranchers comprises 14% of the country, which is more than double the area covered by South African state protected areas (78 100 km^2, or 6.4% of the country's surface area) (Taylor et al. 2015).

Game numbers in South Africa are at an all-time high (Cloete et al. 2015). However, an increase in numbers alone does not necessarily constitute a significant contribution to biodiversity conservation. It is only when wildlife areas contribute to all the components of biodiversity from genetic to functional diversity that these areas/ranches contribute to the conservation of biodiversity as a whole. Genetic diversity reflects the evolutionary history of the organisms within a community and is often quantified from phylogenetic relationships. Functional diversity, on the other hand, reflects the phenotypic variation of a community that is directly linked to specific ecosystem functions. Therefore, it reflects contemporary ecosystem performance. The establishment of extensive wildlife ranches over the past number of years has contributed to the conservation of most components of biodiversity and thus made a significant positive contribution to biodiversity conservation in South Africa. However, there are some concerns related to habitat degradation (overstocking), the effect of fencing in causing mortality and on gene flow, and the inappropriate introduction, mixing of genetic varieties and/or extralimital species, (Pienaar et al. 2017).

In recent years, there has been a trend towards intensification with an increasing focus on selective and/or intensive breeding of high value game species, which warrants further investigation of both the causes and effects this may have. Concerns about the impacts of these activities on biodiversity conservation have been raised by several organisations and institutions and question the contribution made by this sector to biodiversity conservation (Castley et al. 2001, Cousins et al. 2010, Dalerum and Miranda 2016, Pienaar et al. 2017). In 2010, the Scientific Authority of South Africa stated that "the breeding of genetically inferior recessive colour morphs does not further the conservation of South Africa's wild biodiversity and therefore cannot be supported". It was further stated that due to the limited scale of the practice and the species affected at the time, it was considered a low risk threat (Donaldson 2013). However, the Scientific Authority made several recommendations in order to monitor the potential risks posed by this practice (Donaldson 2013). Unfortunately, none of the recommendations were implemented and no monitoring of the extent of the practice, species affected, or potential biodiversity risks was conducted. Since the Scientific Authority correspondence of 2010, the popularity of intensive and selective breeding as a diversification

strategy or direct business decision has increased significantly (Cloete et al. 2015). Results from a survey conducted by Cloete *et al*. (2015) revealed that the introduction of high value species was among the medium term plans of at least 69% of ranchers who do not already have a high value breeding component. Moreover, 59% of these ranchers are considering the breeding of colour variants. According to Taylor *et al*. (2015) the total area of wildlife ranches under camps in South Africa (i.e. properties fragmented into sub-cadastral portions) was estimated to be 10 228 km^2, which represented 6.0% of the total area of the ranching industry and approximately half the size of the Kruger National Park. Desmet et al. (2017) showed that for an area in the upper Limpopo Valley near Thabazimbi, the number of properties assessed as being intensive breeding operations increased from 8 properties (4% of total, n = 208) in 2006 to 17 (8%) in 2010, reaching 51 (25% of total) in 2015. However, the recent rapid decline in the auction prices for colour variants raises questions about the long-term sustainability of this sector. In 2017, the golden wildebeest market price dropped by 86.19% for males and 59.72% for females (Spoorex 2017). In the case of black impala males and females the drop was 76.89% and 57.16% respectively (Spoorex 2017). For both species, this is a dramatic plunge that by all measures can be considered a collapse (Spoorex 2017).

The intensive breeding of game is a relatively recent phenomenon, certainly when measured in ecological or evolutionary timeframes. Therefore, many of the impacts may not yet have appeared or measured, or may not yet be measurable. Assessing the risks is not just a case of documenting existing case studies, but requires some anticipation of impacts based on a broader understanding of biology and ecology and/or extrapolation of examples from other species or environments (including agriculture). However, there is one indigenous species, the ostrich, which has been intensively farmed in South Africa since the late 1800s (approximately 1866) and may give some insight into the possible trajectory and impacts resulting from intensification of production of other species.

The trend of ostrich farming has been away from extensive systems to semi-intensive or intensive production systems, resulting in severe habitat degradation and loss of ecosystem services (some of the worst veld condition in the Little Karoo is in ostrich camps), plus the loss of large areas of habitat to grow feed (Cupido 2005, Reyers et al. 2009). The process of domestication of ostrich has progressed very far, with sophisticated scientific and genetic methods being used to maximise production, resulting in birds that are genetically, morphologically, physiologically and behaviourally different to the ancestral wild population. Further, the quest for maximising production in ostrich has resulted in deliberate hybridization across subspecies and even species boundaries, with subsequent concerns and evidence of introgression back into wild populations in at least South Africa and Kenya (Freitag and Robinson 1993, Turner 2010). In the case of Kenya, collapse of the ostrich farming industry in the early 1900s resulted in domesticated (hybridized) varieties being released into the wild and existing wild populations may now be compromised (Turner 2010).

A second example worth reviewing is that of the management of grouse and grouse habitat in Scotland, recently reviewed by Wightman and Tingay (2015).

For economic reasons in recent years the management for grouse has intensified significantly with higher levels of intervention on both the habitat and the population of red grouse. Grouse moor owners and managers have introduced a more aggressive and intensive approach to management designed to increase grouse yields and overcome natural fluctuations in numbers (Wightman and Tingay 2015). Electric fencing, significant road construction, prophylactic use of medication, culling of other competing species such as mountain hares and persecution of predators are all features of a management framework that has intensified with very little government oversight or public scrutiny (Wightman and Tingay 2015). A parasitic worm is known to play a role in population fluctuations of red grouse. In order to encourage a consistently high population density of grouse available for hunting, one of the intensification methods adopted over the past 20 years has been the use of medication to reduce the incidence of the worm. Grouse naturally ingest mineral grit to assist the digestion

of heather. Coating grit with medication derived from Flubendazole and spreading it on the moors has been the principal means of administering the medicine, resulting in variable and uncalibrated dosages being administered. Other than the effect this has had on natural ecological and evolutionary processes of grouse, signs of worm resistance to the medication have appeared leading to the use of higher-strength medications (up to twenty times the concentration of the original deworming drug).

The authors of the grouse assessment argue for increased government oversight as well as the introduction of measures to reduce negative environmental impacts associated with the recent and rapidly changing trend of intensive grouse production. Both the ostrich and grouse examples provide insight into some of the possible impacts of intensive breeding of game and provided a framework for some of the issues that are evaluated in this report.

As highlighted by these and other case studies involving the breeding of wild species for commercial purposes, there appears to be a real risk of significant social, animal welfare, economic and environmental impacts. This observation is particularly relevant given that strategic and long-term sustainability assessments that address the economic and environmental sustainability of selective and intensive breeding were not adequately considered prior to embarking on the activity. In this, concern is raised as to whether the legislative framework of South Africa provided by NEMA and its principles, as well as the concepts of sustainable use and justifiable development, has been adequately applied by the breeders or government.

South Africa cannot ignore the risks associated with ethics, integrity or reputation and the “triple bottom line” of economic, environmental and social issues, especially not in today’s world of immense and instant market reaction. Intensive and selective breeding of indigenous game is a growing sector of the wildlife industry that has not been assessed in terms of its cumulative environmental impacts. A lack of understanding of the full spectrum of social, biodiversity and economic impacts of these practices may result in significant costs to government and society and thus require further investigation.

CHAPTER 3: BRIEF LEGAL CONTEXT

THE DECISION-MAKING ROLE OF GOVERNMENT

The regulation of wildlife in South Africa is affected by an array of national and provincial statutes, which ultimately coalesce to fulfil the provisions of the Constitution of the Republic of South Africa, 1996 (Constitution). This array guides both the government and the private sector as how the country's wildlife resource is to be utilised and conserved. The following statues have direct and overarching relevance in respect to the objectives and intent of this Report:

National Environmental Management Act 107 of 1998 (NEMA) and

National Environmental Management Biodiversity Act 10 of 2004 (NEMBA)

The relevance of these statutes speaks primarily to how the government, and therein, a presiding official, ought to make decisions on the use of the environment and therein wildlife and the achievement of the Environmental Right (section 28) in the Bill of Rights within the Constitution, namely:

Everyone has the right-

- *(a) to an environment that is not harmful to their health or well-being; and*
- *(b) to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that-*
  - *(i) prevent pollution and ecological degradation;*
  - *(ii) promote conservation; and*
  - *(iii) secure ecologically sustainable development and use of natural resources while promoting justifiable economic and social development.*

The Bill of Rights may be partitioned into two sets of rights, namely those that are absolute in nature and those that are progressively achieved by government. The achievement of rights to education, housing, social security, etc., are dependent on the capacity of government and the availability of financial resources (the fiscus) to deliver on these rights. Hence, these rights would be progressively achieved and are termed a 'progressive right'. By way of contrast, the Environmental Right as with the right to life, right to equity, etc., is absolute in nature in that it cannot be compromised by decisions (or indecisions) taken by Government or another person. This perspective on rights was set in place by Justice Yacoob in the founding Constitutional Court case of the *Government of the Republic of South Africa v Grootboom* (2001).

The Environmental Right provides the foundation for environmental law in South Africa. It is for this reason that each environmental statute requires that statute to be implemented in a manner that achieves this right. This legislation, and in particular the National Environmental Management Act 107 of 1998 (NEMA) and National Environmental Management Biodiversity Act 10 of 2004 (NEMBA), enacts the public trust doctrine as the omnibus to achieve the Environmental Right.

Use of the Principles in Environmental Management and Decision-making

NEMA specifies a number of environmental management principles that all organs of state are bound to apply in making decisions concerning the environment. In applying these principles, the state must uphold the environmental right in the Constitution, which includes the duty to ensure that the environment is protected in the public's interest. This obligation is

binding on all organs of state, and the officials therein, exercising the provisions of both NEMA and any other piece of legislation, policy or guideline that may affect the environment.

In addition to this obligation, the NEMA empowers the public to hold the government accountable for the correct and timely application of the environmental management principles. This empowerment includes lodging an appeal or seeking relief from the courts to direct officials or set decisions or authorisations (permits, licences, etc., instituting civil action), and avoiding a cost order should the courts not grant the relief sought.

Of the array of environmental management principles housed in NEMA, the following are seen to be key and serve to give insights into the importance of their application to selective and intensive breeding of South Africa's biodiversity. A full explanation of the application of the principles in respect to biodiversity is given in Blackmore (2015).

Principle of Public Trust and Biodiversity Conservation

The public trust doctrine is explicitly stated, as part of and central to an array of environmental principles, in the NEMA (Blackmore 2015, 2017),namely:

'[t]he environment is held in public trust for the people, the beneficial use of environmental resources must serve the public interest and the environment must be protected as the people's common heritage' (Section 2(4)(o)).

Following from the Environmental Right in the Constitution, and in keeping with the provisions of the White Paper on environmental management and biodiversity, this inclusion of the doctrine entrenches into South African environmental law the common law duty placed on government that the environment is to be held in trust for all people. Thus, the government – represented by the various organs of state – is obligated to act as a trustee and therein ensures that the environment and its biodiversity are safeguarded by its decisions (Wood 2013, Sand 2014). In essence, the public trust principle as well as the broader environmental management principles, serve to place the official considering an application for the use of biodiversity as a trustee of the biodiversity to be used and the biodiversity that may be indirectly impacted upon. Therefore, the official must not place himself or herself in a position that is inconsistent or in conflict with the interests of the biodiversity (the trust entity) and the appropriate application of the environmental management principles (Dernbach 2015).

The fiducial or trustee obligations of the official making a decision that may impact on the integrity of the country's biodiversity is equivalent to those pertaining to trustees of common law trusts (e.g. testamentary trusts, asset-protection trusts, charitable trusts, etc.). Here, the trustee's duties must be primarily focussed on administering the trust (biodiversity) solely in the interests of its beneficiaries (the public including and importantly future generations). Any decision taken by the trustees (authorising official, authorising authority) must be demonstrably and cautiously taken and with prudence. In circumstances where there is a serious breach of trust, the trustees may be compelled (by the beneficiaries or a person representing them) to ensure that the trust is compensated for any loss and make good on the breach (Blackmore 2015, 2017). Furthermore, liability for the breach is considered joint and several. As such, the trustees may be held personally responsible for the harm to the trust.

The same circumstance applies to authorities and government officials. The provision of the public trust doctrine in South Africa's common (including traditional customary) law as well as its inclusion into the country's constitutional, environmental and biodiversity statute law compels authorities and government officials to fulfil the legal requirements of a trustee. The duties and obligations of the trustee are binding on those officials and authorities (jointly and severally) authorising activities that may harm the country's biodiversity.

Needs of People

The NEMA reinforces the anthropogenic persuasion of the Constitution, but also emphasises the focus of broad-scale or holistic management on the health and wellbeing of people. This is underwritten by the following environmental principle:

'Environmental management must place people and their needs at the forefront of its concern, and serve their physical, psychological, developmental, cultural and social interests equitably' (Section 2(2)).

This principle, given the provisions of the Environmental Right, embraces the plurality of the term 'people' and hence it would be inappropriate for a government official to narrowly apply this principle solely to the needs of an individual, organisation or a limited group of people. To do so would risk being at the expense of a component of biodiversity (the trust entity) or the basic needs and aspirations of people for a better life in the broader society (Brundtland Commission 1987). Further, this principle embraces both present and future generations and therein brings to the fore inter- and intra-generational equity in decision-making. Inter-generational equity applies the notion that access to natural resources is balanced between current and future members of society. Therefore, decision-making by government must strive to achieve a reasonable balance between satisfying current people's needs and ensuring that sufficient biodiversity resources remain to provide for potentially greater future needs (Weiss 1992). A person or a select group of people, cannot not be afforded a right that may prejudice other's rights to fair enjoyment of that resource (*Fuel Retailers v Director-General Environmental Management* (2007)). This principle, as with the other principles in the NEMA, purposefully places a constraint on a person's perceived right to use biodiversity (Blackmore 2015).

Sustainable use of Biodiversity

The principle 'development must be socially, environmentally and economically sustainable' (Section 2(3)) sets in place the 'three pillars' that comprise sustainability (See for example: *Fuel Retailers v Director-General Environmental Management* (2007)). Any use of biodiversity must ensure that all three pillars, both collectively and individually, remain unchallenged. Thus, the objective principle is to ensure that one dimension or pillar is not prejudiced or over emphasised when the government is concerned or considering potential damage to biodiversity; the pillars are thus to be treated equally without bias in decision-making. This principle dispels a notion that the environment could be significantly compromised in favour of a parochial economic interest or the person wanting to damage the biodiversity for a 'national' or 'partisan' interest. This notion was reinforced in the *Fuel Retailers Association of Southern Africa v Director-General Environmental Management, Department of Agriculture, Conservation and Environment, Mpumalanga Province (2007)* where Justice Ngcobo stated that the concept of 'sustainable development will ensure that socio-economic developments remain firmly attached to their ecological roots and these roots are protected and nurtured so that they may support future socio-economic developments.' An official, therefore, adjudicating any application to use that will damage biodiversity without the necessary mitigation and remediation of these impacts, does not have the administrative power to grant such application in favour of the social, economic or other considerations.

Further, this principle requires that the official considering an application that was submitted on economic grounds must ensure that such grounds are sustainable. As such, it is incumbent on the government official to ensure that the arguments regarding economic benefits would reasonably manifest in a clear benefit to a broader economy beyond that of the applicant. It stands to reason that standard economic principles would need to be fulfilled in order for an applicant to sustainably use biodiversity. Given the concerns raised in this report and elsewhere, it would be common cause for the drivers of intensive and selective breeding of wildlife to be able to demonstrate a clear market of such use and, in particular, a clear

consumer of the animals bred. The test of economic sustainability would naturally be based on (a) sustained use of biodiversity (including other aspects of the environment impacted upon) without a net loss in its abundance and integrity of both the biodiversity and the receiving environment of the activity, and (b) a reasonable and substantive economic interest by the consumer of the intensive and selectively-bred wildlife.

Risk-Averse Approach to Decision-making

The precautionary principle to environmental decision-making as defined by NEMA requires a risk-averse and cautious approach that when applied, considers the limits of current knowledge about the consequences of decisions and actions (Section 2(4)(a)(*vii*)). The purpose of this principle is to anticipate and prevent environmental harm when there is a significant degree of uncertainty regarding the potential impact on biodiversity (Trouwborst 2006). In the instance where there is the reasonable concern or a potential of significant harm to the environment, the official must ensure that precautionary or risk-averse measures are applied in decision-making (*HTF Developers (Pty) Ltd v Minister of Environmental Affairs and Tourism*, (2006)). Within this scenario, the decision may vary from restricted use that would rely on monitoring the impacts to delaying the action by way of moratoria or a 'no go' or 'no destruction of biodiversity' option (White Paper on Environmental Management; (Myhr and Traavik 2002)). The core concept of the risk-averse principle may be viewed as an instrument to deter the government from favouring a presumption which favours the development and erosion of biodiversity, where there is an absence of clear evidence of, or investigation into, its impacts (Cooney 2004).

Defining the legal Context of Biodiversity

The public trust doctrine is explicitly stated in Section 3 of the National Environmental Management Biodiversity Act 10 of 2004 (NEMBA) as a means to achieve the Environmental Right Bill of Rights of the Constitution. The White Paper on the Conservation and Sustainable Use of South Africa's Biodiversity from which these two statutes are derived, draws on the Convention of Biological Diversity (CBD) as prevailing international policy. Thus, the interpretation of the NEMBA must fall within the scope and purpose of the fiduciary duties founded in the public trust doctrine and the CBD as well as other relevant multilateral agreements discussed below.

The NEMBA and the Regulations thereto (at the time of drafting this report) does not expressly prevent or limit intensive or selective breeding. The Act, therefore, when read in isolation may also be construed to facilitate such use. For example, the NEMBA imports the CBD definition of 'biological diversity' (Article 5) namely:

'the variability among living organisms from all sources including, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part and also includes diversity within species, between species, and of ecosystems.'

It is common cause that this definition is silent on the importance of areas of low species count, of the genetic makeup of a species in an area, when contrasted to the same elsewhere, is genetically homogeneous or have a low allelic diversity compared to similar populations of the same species elsewhere. In the absence of appropriate regulations this observation brings to the fore the concern that the definition may be a platform to justify the potential loss of the variability (or the absence of such) the Act requires the state to safeguard. For example, the Act does not dispel the misguided argument that that 'increased or enhanced biodiversity' is superior to traditionally or naturally occurring lower diversity (Sax and Gaines 2003, Thomas 2013b, Blackmore 2017). As is argued below, the selective and intensive breeding of wildlife may qualify as an activity regulated in terms of the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the Convention on Biological Diversity. In so doing the concern regarding the simplistic

misinterpretation of the term ‘biological diversity’ is that it may be argued that selectively and intensively bred wildlife may add to the biodiversity of the country in a manner similar to that derived from agricultural breeding programmes (Blackmore In press). In view of this, it is recommended that NEMA be amended in a manner that prevents the potential conflation of agricultural diversity with that natural diversity of South Africa.

Ensuring that biodiversity is safeguarded in decision-making

In order for an official considering authorising a particular activity, particularly when there is a potential to damage biodiversity or there is uncertainty whether the use will have an impact in the short, medium or long-term, it would be prudent for the official to be guided by a framework of probing questions to determine the significance of any risk that may arise. This framework may comprise of the following questions:

- Is the biodiversity involved rare, unique, endangered, or does it have significant historical significance?
- Is that component of biodiversity used or impacted upon easily replaced?
- Will the proposed action or decision have any significant consequential effect on other actions or initiatives that provide for the conservation of biodiversity?
- Are the negative consequences of an action or decision realistically or reasonably reversible?
- Can damages or costs for mitigation and amelioration of negative consequences be reasonably recovered from those responsible for the damage to biodiversity?
- Have the cumulative impacts of human activities on the elements of biodiversity under consideration not exceeded any sustainable-use threshold rendering any further use unsustainable?
- Is there sufficient confidence (i.e. defendable information) that the country’s biodiversity will not be damaged?
- Will the impacts of the activity be reasonably mitigated or remediated within a meaningful timeframe, or in the foreseeable future, or within a period that ensures strict compliance with the conditions of the decision? and
- Will the realisation of potential economic and social benefits that require the safeguarding of biodiversity, be compromised? (Blackmore 2017)

In answering these questions, while affirmatively considering the needs and expectations - not only the current generation, but those who are yet to be born - the government official is enabled to: ‘(1) consider the potential adverse impacts of any proposed activity over which it has administrative authority; (2) allow or grant permission to undertake such activities that do not substantially impair biodiversity; (3) continually monitor the impacts of an approved activity to ensure preservation of the country’s biodiversity; and (4) bring suit under the *parens patriae* doctrine to enjoin harmful activities and/or to recover for damages to the country’s biodiversity (Musiker et al. 1995), and importantly be able to defend the decision should it be challenged (Blackmore 2017).

The relevance of the Nagoya Protocol in Selective and Intensive Breeding of Game

The Nagoya Protocol, to which South Africa is a signatory, was primarily set in place to regulate, inter alia, commercial benefits from genes. It is conceivable that the purposeful manipulation of the genetic make-up of wildlife through selective breeding and intensive management to produce enhanced physical traits or uncommon colour variations (the ‘derivatives of genes’ - see Article 2(e) of the Protocol) for commercial gain, would fall within the ambit of the Protocol. It is thus common cause that such activities would be regulated in fulfilment of the provisions of this Protocol (Blackmore 2017). A key requirement of the Protocol is for the official considering an application for the selective and intensive breeding of wildlife to ensure that this use (of genetic resources) would reasonably contribute to the

conservation of biodiversity (Objective 1). Thus, the Protocol extends the NEMA and public trust requirement from 'no net harm to biodiversity', to ensuring that a positive and measurable conservation outcome accrues from the proposed use. This requirement also extends the common law principle on the burden of proof placed on those parties intending to make use of the genetic resources – to not only show beyond a reasonable doubt that no or insignificant (negligible) harm would result (Ellis 2006, Blumm and Guthrie 2012), but also demonstrate the contribution to biodiversity or the conservation thereof. These two requirements (burden of proof and a net gain to biodiversity) are particularly relevant in South Africa where conservation is an emerging sector, and where surveillance and compliance enforcement of the Protocol would be a significant additional cost to potentially already stressed wildlife management resources (Morgera 2014, Blackmore 2017).

In permitting selective and intensive breeding of wildlife, the Protocol requires the organ of state regulating such activity to set in place legal mechanisms which will give effect to the objectives and provisions of the Protocol. These include prohibitions of non-sustainable or harmful uses of genetic resources and a legal mechanism to monitor the use of genetic resources (Article 17). Such regulation may be applied via the application of 'applicable national legislative, administrative or policy measures' (Article 13(2)) and compliance thereto (Article 15). The key outcome of these measures would be the decision to decline the application or grant the same by way of a conditional permit. Such a decision would constitute an 'international certificate of compliance' with the Protocol (Article 17(2)). Because this being an uniquely identifiable permit, it must include the necessary information to enable any domestic or international official to assess compliance with the Protocol – as well as any restrictions or limitations the country of issue or any other affected country may have put in place (Article 17(4)) ((Morgera 2014, Blackmore 2017).

CHAPTER 4: APPROACH AND METHOD

Acknowledging that intensive and selective breeding of game is an emerging industry within the broader wildlife sector, it is recognised that there is currently little known about its full nature and extent within South Africa. Two broad categories of potential risks were identified, namely Biodiversity Risks and Biodiversity Economy Risks. Slightly different approaches and methodologies were used for these two categories to make provision for the integrated nature of potential risks to the biodiversity economy that often entails social, economic and environmental aspects. The Science-Policy Platform on Biodiversity and Ecosystem Services' (IPBES) is not well suited for this purpose.

BIODIVERSITY RISKS

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services' (IPBES) conceptual framework was used to assess potential risks to South Africa's biodiversity heritage resulting from the practice of intensive and selective breeding of game. The assessment is based on relevant existing scientific literature and local knowledge and will consider all knowledge systems to inform our recommendations and eventual decision making (IPBES 2014). Further, by following this approach it is anticipated that new knowledge will be generated through research that will fill existing knowledge gaps. Lastly and in line with the IPBES conceptual framework, capacity will have to be developed for future research and policy implementation.

Seven potential biodiversity risks/issues relating to the practice of Intensive and Selective Breeding of game were identified using the best available published scientific literature, information obtained from members of the wildlife sector, experts and the national dialogue process (Njobe and King 2016). From these seven potential issues (risks)**,** 17 potential **impacts** (harms/stressors) are described with specific c**oncerns** highlighted under each impact statement. The impact statement thus describes the current understanding associated with each anticipated impact.

Evidence in relation to each potential impact is then provided and briefly summarised with a list of key scientific publications, reports and other relevant documentation and listed in the reference list. The lack of substantial evidence is also highlighted where information is limited. The **quality of the evidence** is evaluated in terms of the extent of evidence available and the level of agreement in the literature on a specific impact. The extent of evidence includes local and international scientific publications in peer reviewed journals as well as other relevant reports and documentation. For each of the key issues, external experts were identified and assisted in developing the evidence section of each of the impacts.

The quality of the evidence was evaluated for scientific rigour using the 'uncertainty approach' used by the UK National Ecosystem Assessment (IPBES 2014). The 'uncertainty approach' consists of a set of uncertainty terms derived from a 4-box model (Fig. 1) and complemented, where possible, with a likelihood scale (see below). The X-axis in the 4-box model describes the extent of the evidence available for a specific impact ranging from limited to significant, while the Y-axis is an indication of whether there is high or low agreement on the potential impact. The following certainty terms were used, defined according to the relationship between the available evidence and the level of agreement in the literature:

1. Well established: high agreement based on significant evidence
2. Established but incomplete evidence: high agreement base on limited evidence
3. Competing explanations: low agreement, albeit with significant evidence
4. Speculative: low agreement based on limited evidence

An evaluation of both the extent of the evidence provided on a specific impact and the level of agreement in the literature on the specific impact places the impact in one of the four quarters of the 4-box model. For example, an impact with a significant amount of evidence but limited agreement will fall within the 4th quarter of the model – Competing explanations and receiving a score of 3.

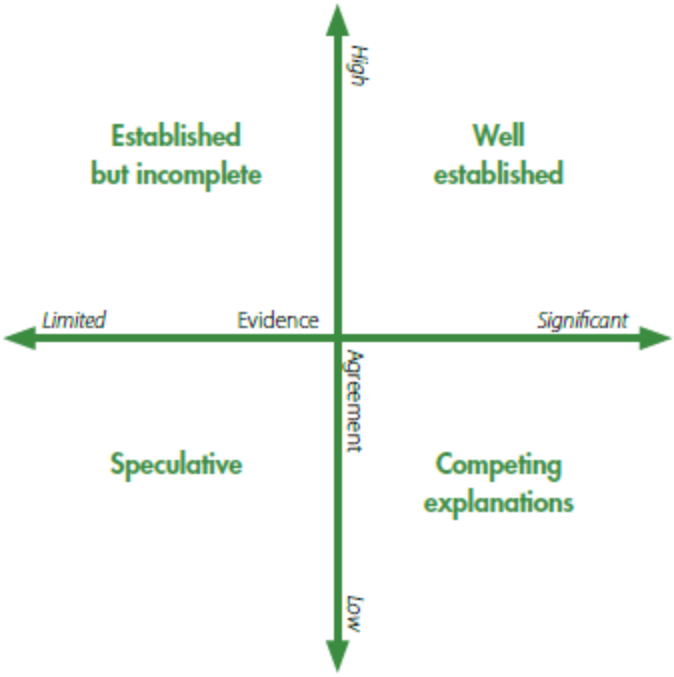


Figure 3: Diagram of the "uncertainty approach" of the UK National Ecosystem Assessment consisting of a set of uncertainty terms from a 4-box model.

Estimates of certainty are derived from the collective judgement of authors, observational evidence, modelling results and/or theory examined for each impact assessment.

The following likelihood scale was used:

Virtually certain (1):	≥80% probability of occurrence
Likely (2):	60-79% probability
About as likely as not (3):	40-59% probability
Unlikely (4):	20-39% probability
Very unlikely (5):	<20% probability

The task team, as a group assessed and scored each impact on the quality of scientific evidence available (4-box diagram), the probability that the impact is occurring within the industry, as well as the likelihood that the specific impact will have an effect on an ecosystems level, threatened and near threatened target species and the individual animal subjected to intensive and selective breeding. For the purposes of this report, the target species is defined as a species subjected to intensive and/or selective breeding. Assessments were only done for species with IUCN Threat status of Near Threatened and above. All Least Concerned species were omitted as *a priori* assumption was made that the impact on these species is likely to be unlikely. However, it is acknowledged that over time, depending on the scale, these risks and concerns may affect even these species.

A hierarchical ranking method was used to rank the impacts from highest to lowest risk. To determine the impacts with the highest potential risk at an ecosystem's level, impacts with a

score of 1 (Virtually certain) or 2 (Likely) were used. This was followed by ranking the selected impacts according to quality of evidence, only selecting impacts with a score of 1 or 2, and then lastly on the probability of occurrence within the industry. A similar process was followed for assessing the risk to Threatened or Near Threatened species. Potential mitigation measures that could reduce the risk or negative impact were identified and recommended for each impact.

Whilst this is a science-based approach of an existing body of available evidence, it is not intended to be either a comprehensive literature review or a full evidence-based assessment, but rather a process that uses available literature and consultation with experts to highlight potential risks.

BIODIVERSITY ECONOMY RISKS

Although the focus of this report is primarily on direct risks to biodiversity, the biodiversity economy and its potential in addressing socio-economic challenges of the country, it is an important focus area of government. Social and economic risks related to the biodiversity economy, may pose indirect risks to biodiversity and its contribution to the broader economy.

One issue and two impacts relating to the practice of intensive and selective breeding of game on the biodiversity economy were identified based on current events within the biodiversity economy. A similar process to IPBES was used to gather evidence and consider level of agreement. Due to the integrated nature of Biodiversity Economy Risks, it could however not be weighted using similar criteria as for the Biodiversity Risks.

At the onset, it is recognised that very few formal publications are available on the extent of socio-economic impacts of intensive and selective breeding on other sub-sectors of the biodiversity economy, including hunting because many of the impacts discussed are still emerging. In view of the lack of substantive published research on the nature and extent of impacts, the authors have sourced expert opinions, considered well documented and researched economic and business principles, trends and economic reviews expressed by experts and published in the popular media, as well as other relevant recent information and deductive reasoning. Chapter 11 thus serves as a compendium of the specialist knowledge and recorded statements by the major role players in the hunting industry and respected industry practitioners, conservation agencies and other key role-players. It further does not attempt to provide an in-depth strategic environmental analysis of all social, economic and environmental factors and their interplay in the wildlife sector, nor does it attempt to assess the extent of impacts. It is aimed at identifying if there is reasonable cause for concern and a need for interventions related to social, economic and environmental risks and potential future impacts associated with the current unregulated policy environment and growth in intensive and selective breeding of indigenous wildlife.

CHAPTER 5: BIODIVERSITY ISSUES

ISSUE 1: INTENTIONAL BREEDING FOR SELECTED TRAITS

IMPACT STATEMENT 1: EXPRESSION OF DELETERIOUS ATTRIBUTES THAT MAY LEAD TO PHYSICAL, BEHAVIOURAL AND LETHAL OUTCOMES

Concerns

1. Breeding practices, such as inbreeding, line breeding and artificial selection for specific phenotypic traits, which increase the physical expression of rare alleles, may lead to conditions that could compromise the wellbeing of the individual animal.
2. Where deleterious co-segregating traits are linked to selected genes, i.e. colour genes or horn length, and are non-lethal, they can be transmitted to other individuals within the subpopulation, and ultimately to several subpopulations as a consequence of translocations, increasing their occurrence within the broader population (see Chapter 5). These deleterious traits could lead to lower reproductive potential, as well as an altered ability to adapt to environmental change.

Evidence

Natural selection is the differential reproductive success of different phenotypes, a process which slowly changes gene frequencies in populations. Those individuals with phenotypes that are selected against are less successful than others and, eventually, their genes are eliminated from the population (Ruggiero et al. 2000). Thus, natural selection cannot be relied upon to produce significant numbers of rare phenotypes such as colour variants.

The objective of commercial breeding programmes is to maximize the rate of genetic change for economically important traits. Where these traits are only expressed in recessive phenotypes, inbreeding or line breeding are often used to maximize the genetic progress towards these traits. The benefits of inbreeding are increased uniformity, increased prepotency (ability to pass on traits to offspring) and “fixing” of desired traits and breed type. The main objective of inbreeding is thus to increase homozygosity at desirable loci in order to increase the frequency of homozygous individuals in the population, which can also be used to express recessive phenotypes. However, alleles responsible for the expression of these recessive phenotypes, such as coat colour, are seldom independent (Hofreiter and Schöneberg 2010). The determination of coat colouration is a complex process in which several genes, as many as 300 loci and more than 150 genes have been suggested to interact (Cieslak et al. 2011). More often than not these alleles are linked with various others. Pleiotropy is the phenomenon where different traits may be influenced by the same genes (Andersson 2001). In such a case, for example, increasing the frequency of alleles that cause faster growth, may at the same time cause a modification in developmental, behavioural, physiological, or immunological traits under the influence of the same genes. The action of one gene may also be influenced by the interaction with other genes, which is known as epistasis (Carlborg et al. 2003). Again, breeding for one or a few traits controlled by genes that have epistatic effects may then affect an array of other genes, related to other phenotypical characters than those selected for. For both mechanisms (pleiotropy and epistasis), side effects on traits that are not controlled for by selection indices are expected (Jensen and Andersson 2005).

At present little is known about the genes expressing specific traits in wildlife such as coat colour and/or patterns, or horn size, or linkages between these genes and other genes in the animal genome (Cieslak et al. 2011, Needham and Hoffman 2013), although much can be learnt from decades of research on domesticated and other model mammal species (Hofreiter

and Schöneberg 2010, Cieslak et al. 2011). When selecting for specific rare traits through breeding practices, such as inbreeding, there may be unintended consequences. Selective breeding for such traits through inbreeding, line breeding or artificial selection of specific phenotypic traits, may lead to a variety of negative consequences also documented for severe inbreeding, such as increasing the genetic load, congenital defects, and overall decline in fecundity and increase in morbidity (e.g. Laikre (1999)). "Fitness" is a measure of an animal's ability to produce offspring that can contribute genetically to the next generation. Inbreeding reduces fitness, and the decrement in fitness is proportional to the degree of inbreeding. This reduced fitness is called "inbreeding depression". The cost and benefit of inbreeding are thus directly proportional to the coefficient of inbreeding (COI). For example in Standard poodles, dogs with inbreeding less than 6%, live 4 years longer than those with higher COI (Armstrong 1998, Kosowska et al. 2005). Similar the risk of bloat (Canine gastric dilatation volvulus) is roughly proportional to the increase in COI - a 10% increase in COI elevates the risk of bloat by about the same amount (D. Schellenberg et al. 1998). However, it should be noted that species differ in terms of their inbreeding tolerance.

In the Wildlife Ranching South Africa (WRSA) code of best practice – Genetic Engineering (draft 7), the organisation recognises that line breeding is the cornerstone of selective breeding and argues that this practice is acceptable if used in line with sound management and the use of available scientific technology (Dry 2016). The document further states that colour variants are natural phenomenon and that game breeders select for homozygous recessive individuals for breeding to ensure that the rare coat colour is conserved. Currently more than 30 different colour variants from 10 antelope species are offered at auctions. Of these only a handful i.e. golden wildebeest, black impala (*Aepyceros melampus*), white springbok (*Antidorcas marsupialis*), have been recorded in wild populations in the past.

Line breeding is a mating system that is designed to maintain a substantial degree of relationship to a highly regarded ancestor or group of ancestors without causing high levels of inbreeding, but is still considered a mild form of inbreeding (Bourdon 2000). In order to avoid the negative consequences of inbreeding, detailed records of the pedigrees of each individual need to be maintained and a large number of founders are required (Kristensen et al. 2015). The number of founders required depends on the underlying alleles linked to the trait. Genetic studies on black and white springbok have failed to reveal the pattern of inheritance of the unusual colour morphs, though some believe that the colour variation results from a double recessive combination (Kruger 1976, Bezuidenhout 2012). Skead (2007) further indicates that the black colour morph was not recorded historically and appears to have arisen through inbreeding of an enclosed population. However, there are historical records of white springbok (Skead 2007). A recent study on impala has shown that a single base-pair deletion in the *ASIP* gene is associated with the black phenotype in this species (Miller et al. 2016b).

A long list of potentially harmful recessive traits exists for domesticated species that are associated with either substantial physical or behavioural impairment and/or even lethal outcomes such as a drastically shortened lifespan (e.g., (Laikre 1999, Kirkwood 2010, Reissmann and Ludwig 2013)). For example, taillessness in some duck breeds can lead to lowered reproductive rates and shorter lifespan as well as problems with copulation and egg laying (Stucki et al. 2008 as cited by AZA (2011)). Dwarfism in rabbits has been associated with problems in teeth positioning and eating, as well as thermoregulation and reproduction. Indeed most dwarf forms of a variety of species seem to be more prone to general infections and may show compromised immunity (Not et al. 2008 as cited by AZA (2011)). In cats, breeds such as the Manx cat and tailless cat are associated with locomotive disorders, dispositioning of the vertebral column, difficulties defecating, and a loss of about a quarter of offspring when breeding for the trait of “taillessness”. The “dominant white” trait in domestic cats (the individuals are homozygous for this recessive colour morph) has been associated with increased occurrence of deafness. Similarly in dogs, the “Merle factor” (e.g. Blue Merle Collie, or Merle Bobtail) has been associated with a disposition to deafness and eye disorders (Schmutz and Berryere 2007, Marsden et al. 2016). The list is lengthy and several

comprehensive reviews have documented the various problems associated with intentional breeding for rare and recessive alleles in a wide variety of species including domesticated and wild species such as white tigers (*Panthera tigris*) (Begany and Criscuolo 2009, Mongera and Dooley 2013) and white lions (*Panthera leo*) (Scaglione et al. 2010). Within the wildlife sector similar trends are starting to emerge. A recent comparison made on testicular circumference between colour variant versus common or wild type impala indicates that colour variants displayed underdeveloped testicles in relation to age/developmental stage with questionable fertility (Luther, Unpublished data, 2016).

Within inbred populations lethal deleterious mutations will be exposed quickly and possibly purged from the population (Amos and Balmford 2001). However, in a controlled environment deleterious alleles with less selective effects will be effectively invisible to selection and may be subject to genetic drift and fixation (Keller and Waller 2002). Even limited gene flow will not prevent drift load from accumulating and it may take several generations before these alleles are expressed phenotypically. These deleterious mutations could thus accumulate within a population and be transmitted to other populations through translocation of these individuals and impacting the broader population (refer to Chapter 5).

Extent of evidence

The genetic reasons for coat colour variation in game, specifically antelope species, have been poorly studied and little scientific evidence is available in terms of basic coat colour inheritance and the identification of allelomorphic series in game. To date, of all the colour variants available on the market, only the molecular basis of the black phenotype in impala has been identified (Miller et al. 2016b). Even though there has been little work done on the genetic basis for colour transmission in African game species, there is a large body of scientific literature showing that the selection of specific traits through a process of inbreeding may lead to the expression of recessive deleterious alleles.

Level of agreement

High level of agreement occurs in the scientific literature that the intentional selection for specific traits through inbreeding in a species, e.g. horn size or coat colour, may lead to the expression of deleterious traits. This has been well established in domesticated species and some wild species such as white tigers and white lions.

Key findings

Many genes control complex mammalian traits such as coat colour and ornaments. The expression of these genes, the interactions among them, as well as genotype-phenotype-environment interactions need to be investigated for the novel colour variants in the South African wildlife industry. This is imperative for the well-being of the individual animals and for understanding the short-term and long-term consequences for the industry and for the wildlife species involved.

Even though there has been little work done on the genetic basis for colour transmission in African game species, it is well established that the selection of specific traits through a process of inbreeding or otherwise is very likely to lead to the expression of recessive deleterious attributes that may lead to physical, behavioural and lethal outcomes. It is further virtually certain that breeding practices, such as inbreeding, line breeding and artificial selection for specific phenotypic traits, for example colour variants, are taking place within sectors of the wildlife industry. The extent of the impact is likely to be limited to the individual and the specific population if no translocation or selling of individuals occurs. However, where individuals, that carry deleterious genes, are translocated to other populations the extent of the impact may increase to species level. This will depend on the pattern of inheritance and the number of animals in a larger population that may possess a specific colour allele.

Breeding with the wild type phenotype and unrelated individuals will decrease homozygosity at other loci (and potentially at the loci for e.g. colour) but depending on the breeding success of the animal with the trait, the trait may still persist in the new population. Theoretically we predict that there will be dilution.

The level of impact of these concerns will depend on the potential of the affected individual to reproduce. Where the impacts of the deleterious trait are such that it prevents or significantly lowers the potential of the individual to reproduce, the impact on the broader population would be insignificant but the impact on the individual may be high.

Potential mitigation measures

The following is suggested to improve breeding practices and reduce inbreeding:

1. Maintain proper stud books with detailed records of the pedigrees of each individual selectively bred. Breeding with close relatives (e.g. between first order relatives) should be discouraged.
2. Keeping of accurate pedigree information and records of movement of animals should be mandatory.
3. Exchanging breeding animals, which are not related, between different breeders to prevent inbreeding.
4. Development of scientifically-based best practice guidelines for the breeding of game species based on conservation breeding principles.
5. Openly reporting any incidences of deleterious effects.
6. Support of current research efforts and commissioning of research into under-investigated aspects such as the genetic basis for colour transmission in African game species.
7. The expression of undesirable traits and its impact upon wild populations need to be quantified.

IMPACT STATEMENT 2: LOSS OF GENETIC AND ALLELIC DIVERSITY RESULTING IN DECREASED FITNESS AND REDUCED ADAPTIVE POTENTIAL

Concerns

1. Removal of the process of natural selection, including mate selection and selection by differential mortality will reduce the evolutionary potential of populations to adapt to environmental change, especially in light of environmental and climate change.
2. Using a small subset of the available gene pool (number of founders) and deliberate inbreeding thereafter can result in the fixation of certain genetic traits. Deleterious mutations will tend to accumulate, because selection is less effective in small populations and likely to be less effective in captive populations protected from natural selection pressures. Both the founder effect and inbreeding result in the loss of allelic diversity (loss of rare alleles).
3. In the short term, inbreeding depression can affect birth weight, survival, reproduction and resistance to disease, predation and environmental stress. In the long term it reduces the evolutionary potential of populations to adapt to environmental change.

Evidence

Small and isolated populations are inherently more vulnerable to external environmental perturbations and chance fluctuations in local survival and fecundity (Keller and Waller 2002). This is a general principle that equally applies to antelope species in reserves and national parks. Genetic diversity is the raw material for evolutionary change within wildlife populations; it can be considered as the life insurance policy for a species or population. It allows

populations to evolve in response to environmental change, whether that is new or changed diseases, pests, parasites, competitors or predators or greenhouse warming, ozone layer depletion, or pollution (Frankham 1996). A strong correlation between population size and genetic diversity exists, meaning that a reduction in population size would compromise the ability of a population to adapt to changing environments (Frankham 1996, O'Ryan et al. 1998). Selecting a small number of founders, for example only homozygous individuals for a rare phenotype, and the deliberate breeding of these individuals with each other will lead to the loss of rare alleles from the population.

Inbreeding leads to inbreeding depression in virtually all species studied thus far (e.g., (Sewall-Wright 1977, Charlesworth and Charlesworth 1987, Frankham 1995, Lynch and Walsh 1998, Crnokrak and Roff 1999, Hedrick and Kalinowski 2000). The deleterious consequences of inbreeding depression have been widely reported and impact on all aspects of reproduction and survival, including sperm production, mating ability, female fecundity, juvenile survival, mothering ability, age at sexual maturity and adult survival in animals ((Saccheri et al. 1996, Newman and Pilson 1997, Crnokrak and Roff 1999) (see Hedrick and Kalinowski (2000) for a recent review). Levels of inbreeding depression vary across taxa, populations and environments, but are usually substantial enough to affect both individual and population performance (Keller and Waller 2002). Heterozygosity levels are linked directly to reduced population fitness via inbreeding depression (Reed and Frankham 2003). Heterozygosity, population size, and quantitative genetic variation are further positively and significantly correlated with population fitness (Reed and Frankham 2003). There is no doubt that genetic diversity is related to population size and that small population size reduces the evolutionary potential of wildlife species (Frankham 1996).

Inbreeding depression has an immediate impact, while loss of genetic diversity typically impacts over the long term, associated with environmental change (Frankham 2005). The loss of adaptive genetic variation and inbreeding depression put wildlife populations at an increased risk of extinction. This increase can occur as a result of reduced reproductive fitness due to inbreeding depression, or from a failure to track the changing abiotic and biotic environment of the population as a result of the loss of genetic variation through drift (Reed and Frankham 2003).

These impacts are already observed in the wildlife breeding sector. A recent comparison made on testicular circumference between colour variant versus common or wild type impala indicates that colour variants displayed underdeveloped testicles in relation to age/developmental stage with questionable fertility (Luther, Unpublished data, 2016).

Extent of evidence

The link between genetic diversity and fitness and the impacts of small population size are aspects that have been investigated since the development of the field of Conservation Biology in the 1970s (Frankham 1996, Reed and Frankham 2003, Leroy 2011, Leffler et al. 2012). In virtually all species studied thus far inbreeding has led to inbreeding depression (e.g., (Sewall-Wright 1977, Charlesworth and Charlesworth 1987, Frankham 1995, Lynch and Walsh 1998, Crnokrak and Roff 1999, Hedrick and Kalinowski 2000). The deleterious consequences of inbreeding depression have been widely reported and impact on all aspects of reproduction and survival, including sperm production, mating ability, female fecundity, juvenile survival, mothering ability, age at sexual maturity and adult survival in animals ((Saccheri et al. 1996, Newman and Pilson 1997, Crnokrak and Roff 1999) (see Hedrick and Kalinowski (2000) for a recent review).

Level of agreement

High level of agreement exists in the scientific literature that the loss of genetic and allelic diversity specifically in small populations will result in decreased fitness and a reduced adaptive potential.

Key findings

It is well established with a high level of agreement in the scientific literature that a loss of genetic diversity is highly likely to result in decreased fitness and in the long term reduces the evolutionary potential of populations to adapt to environmental change. It is further virtually certain that inbreeding and line breeding are used in the wildlife breeding sector as methods to increase certain rare phenotypic characteristics in animal populations and that several colour variant populations were established from very small founder populations (Needham and Hoffman 2013). However, the extent and severity of the impact will be related to the proportion of animals of a particular species that are in intensive breeding facilities versus the wild. The risk is especially high for species with low population numbers in the wild, but much lower for common or Least Concern species. The highest level of impact will be on the individual exposed to these practices.

We cannot accurately predict how species will respond to future challenges. The implementation of sound practices should safeguard populations or species in future. Monitoring of aspects such as genetic diversity, disease risks and outbreaks, etc. should be implemented.

Potential mitigation measures

The following can be suggested to reduce the loss of genetic diversity:

1. Maintain proper stud books with detailed records of the pedigrees of each individual selectively bred. Keep records of translocations.
2. Development of scientifically-based best practice guidelines for the breeding of game species based on conservation breeding principles.

IMPACT STATEMENT 3: THE MIXING OF GENES FROM NATURALLY SEPARATED[1] GENE POOLS LEADING TO THE BREAKDOWN OF NATURAL EVOLUTIONARY PROCESSES AND/OR POSSIBLY LEADING TO OUTBREEDING DEPRESSION

Concerns

1. Animals that may be less adaptable to the current environment due to the loss or gain of genetic traits.
2. Hybrid subpopulations may have a greater probability of extinction as they may be less adaptable to their current environment.
3. Hybrid subpopulations may have negative impacts on native species through introgression.

Evidence

In conservation planning, the importance of natural variation is often given inadequate consideration (Ruggiero et al. 2000). However, ignoring the implications of variation within species may result in conservation strategies that jeopardize, rather than conserve, target species (Ruggiero et al. 2000). Natural variation exists between individuals in a population

[1] Naturally separated could mean a species, sub-species or other evolutionary significant population or at extremes of clinal variation.

and between populations of a species. Differences among populations generally increase with distance and/or isolation. It is unwise to assume that behavioural or genetic attributes exhibited by one population are within the range of behavioural or genetic potential of another (Ruggiero et al. 1988). When this assumption is false, it can result in failed reintroduction efforts or a net loss of genetic or behavioural variability (Storfer 1999).

Humans have the potential to alter patterns of gene flow and introgression among wild animal populations and species in two main ways (Crispo et al. 2011). First, genetic exchange between groups of individuals requires that their breeding ranges overlap. Human alterations of the physical landscape and species' distributions, e.g. translocation of species outside of their natural distribution range, can affect gene flow and introgression by influencing the degree of contact between groups of individuals (von Brandis and Reilly 2007, Grobler et al. 2011). Second, genetic exchange relies on successful breeding among groups of individuals. Thus, humans can also alter rates of gene flow and introgression through any activity, such as the accidental or intentional introduction of different conservation units on the same property (Allendorf et al. 2001), which affects the integrity of reproductive barriers. Translocation of species within and outside their natural distribution range within South Africa is widespread with few control measures in place to prevent either hybridization or genetic drift, apart from the standard legislative controls, such as the Threatened and Protected Species Act (TOPS), provincial Acts and ordinances (Castley et al. 2001, Spear and Chown 2009, Taylor et al. 2015). Genetic drift is unlikely to be an important factor in open and large populations, but is a major issue within fenced populations (Mysterud 2010). Recent examples in South Africa include the mixing of bontebok and blesbok (van Wyk et al. 2017), blue wildebeest (*Connochaetes taurinus*) with black wildebeest (*Connochaetes gnou*) (Grobler et al. 2017)and West African roan with Eastern and Southern roan populations (Alpers et al. 2004).

Presently, there are huge economic incentives for landowners to maximise the number of game species on one property for hunting and tourism purposes, and this is the ultimate driver for having both species of wildebeest together on one property. Spear and Chown (2009) note that it may be problematic for conservationists to make a case against activities such as extralimital translocations when these activities result in substantial economic benefit. There is thus an economic incentive to keep both species of wildebeest together. However, Grobler et al. (2011) state that reproductive isolation may disappear under the conditions typically found on small properties. Small fenced areas often have insufficient suitable habitat for one of the species, resulting in them sharing the same areas (Grobler et al. 2011). In addition, unnatural sex ratios resulting from hunting and translocations may result in mate-choice changing where there is an absence of species-specific mates. The unidirectional hybridization between blue wildebeest bulls and black wildebeest cows as described by Grobler et al. (2011) is a good example of the above-mentioned.

Human-induced increases in genetic exchange can have direct consequences for biodiversity (Crispo et al. 2011). Increased rates of genetic exchange can impact the fitness of populations and species and can influence their persistence. It can further result in the loss or creation of entire taxa (Loehr et al. 2008). Population fitness can be negatively affected when gene flow between locally adapted populations results in decreased adaptation due to the introduction of locally maladaptive alleles and the swamping out of locally beneficial alleles, a consequence referred to as migration load. A good example is the introduction of West African roan into Eastern and Southern roan populations (Alpers et al. 2004). Similarly, a recent study on the genetic structuring of oribi (*Ourebia ourebi ourebi*) populations in South Africa found no geographic pattern, probably as a result of significant movement of animals through translocation (Vuuren et al. 2017). When migration load on a population is strong enough, the reduction in fitness can lead to reduced population growth and eventually to the extirpation of the population, a process referred to as migration meltdown (Bolnick and Nosil 2007). Gene flow between two locally adapted populations can further lead to outbreeding depression when adaptive combinations of alleles get broken down or when genetically incompatible alleles are

recombined into the same genome (Hedrick et al. 2001, Edmands 2007, Crispo et al. 2011). Outbreeding depression may be difficult to detect as it is often only expressed after the second (or subsequent) generation of interbreeding after the two parental genomes recombine (Hedrick et al. 2001, Edmands 2007).

Several examples illustrate the problems inherent in translocating animals from one ecosystem to another such as the attempted introduction of wild turkeys (*Meleagris gallopavo*) from an arid region of Texas to the wetter areas of eastern Texas (Evans and Williamson 1976). Although the dry-adapted birds survived, high humidity prevented them from nesting successfully, and the introduction failed. In a translocation of individuals from two British-Columbia (B.C.) woodland caribou (*Rangifer tarandus*) populations to northern Idaho to augment a small remnant population, the augmentation failed because individuals had a tendency to retain the movement and feeding behaviour of their original population. Caribou from west-central B.C. eat mainly terrestrial lichens in winter, whereas caribou from east-central B.C. eat arboreal lichens because the snow is too deep to dig through. After translocation, more east-central caribou emigrated from the transplant area, in keeping with the tendency of the parent population to emigrate more frequently than west-central caribou. Most west-central caribou died, and apparently starved while digging for lichens in winter, ignoring abundant arboreal lichens in the new area (Warren et al. 1996). Populations of the Tatra mountain ibex (*Capra ibex ibex*) in central Europe were supplemented with ibex from Turkey and the Sinai during the 1960s (Templeton et al. 1986). The hybrids of these ibex rutted in early autumn instead of in winter (as the native ibex did), and the females gave birth in February in Czechoslovakia during the coldest time of the winter. The entire population went extinct because of the mixing of these groups.

A recent review of 62 anthropogenic hybridization cases found that nearly all had identified negative consequences, especially for mammals (Piett et al. 2015). Van Wyk *et al*. (2017) examined hybridization between bontebok (*Damaliscus pygargus pygargus*) and blesbok (*D. p. phillipsi*) and a sample of nearly 3 000 animals from across South Africa determined that approximately 25% of tested animals were hybrids. Importantly, hybrids (admixed individuals) were found in two-thirds of the locations tested and few populations on private land can be considered as nonadmixed or pure. They concluded that as the game breeding and hunting industry advances, translocation and hybridization (intentional and unintentional) rates will likely increase. Most recent hybridization is taking place between pure (nonadmixed) and hybrid (admixed) animals, and not between pure blesbok and pure bontebok. Failure to remove hybrids and prevent further hybridization could ultimately result in swamping of the bontebok gene pool. The optimal threshold of admixture (the optimal balance between reduction in admixture and prevention of loss of genetic diversity in bontebok) was determined, necessitating the removal of 20 - 60% of the population, which is a significant undertaking. The legacy of the hybridization that took place, other than that there will always be blesbok genes in the bontebok population, is a long term administrative burden of regulation, including testing and certification of herds and trophies.

Augmenting populations with captive-bred or intensively-managed animals, or introducing these individuals into the wild, is likely to be problematic when the principles of conservation breeding are ignored. Intensively-managed animals may have lower survival rates, a reduced flight response to predators, and may lack other behaviours necessary to survive in the wild (Stahl 1981, Reinert 1991). In addition, adaptation to intensive conditions can occur allowing those genotypes best suited for surviving and reproducing under intensive conditions to increase in frequency, while those genotypes best suited for surviving and reproducing in the wild dwindle (Kohane and Parsons 1988, Allendorf 1993). Further, the mixing of diverging gene pools, i.e. intensively-bred individuals mating with their wild counterparts may result in genetic homogenization and lead to maladaptation to the local environment and potentially outbreeding depression. The converse is that admixed genotypes may be generated that are more adapted than the parental populations, resulting in a local increase in species invasiveness (Canu et al. 2014). This notion is supported by Garcia *et al.* (2011) in which it is

argued that the invasive potential of wild boar (*Sus scrofa*) populations in Uruguay has been as a result of introgressive hybridization with domestic pigs.

Harrison and Larson (2014) suggest there are numerous examples of introgression, yet very few have resulted in speciation events i.e. a resultant new species. They suggest that the semi-permeable genetic boundary (genetic exchange between species being possible) that exists between species may have important implications for contact between species as a result of the accidental or intentional introductions thereof. For example, such interactions between similar species may result in changes to coat pattern/colour, which may not be a desirable trait and thus have a negative impact upon the species concerned. The interaction between domestic sheep and their wild counterparts has led to adaptive changes in their coat colour and pattern (Loehr et al. 2008).

Positive effects of gene flow on population fitness can include the rescue of small populations from the loss of genetic diversity due to drift and from the perils of inbreeding (Hedrick et al. 2001, Hedrick 2013). Gene flow also allows the exchange of beneficial mutations among gene pools. These effects promote biodiversity persistence by preventing extinction and increasing genetic variation upon which natural selection, and thus the response to environmental change, is dependent. Such positive effects are expected when gene flow between populations is moderate. However, the effects of gene flow on fitness are poorly understood, and the relative importance of positive and negative effects in nature remains an open question (Crispo et al. 2011). In addition to the effects on population fitness, gene flow or introgression between species can further lead to the extinction or creation of entire species. Hybrid swarms, in which unique combinations of alleles in the parental genomes are lost as genomes get mixed, is a common outcome of hybridization (Crispo et al. 2011). This process is facilitated when there are no mating barriers in place and no selection against hybrids. The many examples of introgression following hybridization with invasive species, such as White-headed duck (*Oxyura leucocephala*) and Ruddy duck (*Oxyura jamaicensis*), serve as examples of this process (Rhymer and Simberloff 1996).

Extent of evidence

Outbreeding depression has been documented in the wild, but is not as easily quantifiable as inbreeding (Edmands 2007, Frankham et al. 2011). However, several examples illustrate the problems inherent in translocating animals from one ecosystem to another such as the attempted introduction of wild turkeys from an arid region of Texas to the wetter areas of eastern Texas (Evans and Williamson 1976), the translocation of two British-Columbia woodland caribou to northern Idaho to augment a small remnant population (Warren et al. 1996) and the introduction of Tatra mountain ibex from Turkey into central Europe (Templeton et al. 1986). Translocations are thus most likely to succeed between populations whose environments and ecological relationships are similar, especially with regard to climate, habitat, and community composition (Ruggiero et al. 2000). A large body of scientific evidence exists that anthropogenic hybridization has negative consequences, especially for mammals. There are several examples within the South African wildlife industry such as the hybridization of bontebok and blesbok, blue and black wildebeest and the introgression of West African roan into southern African roan populations. Thus, the concept has been established in the literature, but there is still a lack of evidence.

Level of agreement

From the literature it is clear that there is evidence for the changes associated with translocation and introgression, but the direction of change is not clear i.e. whether an individual is less fit or more adaptive/invasive.

Key findings

There is good evidence, that with the expansion of the wildlife industry over the past few decades, there has been increased human-mediated movement (translocation) of animals, within and outside their natural distribution ranges, with unclear consequences for the species themselves or their impact on other biota (Castley et al. 2001, Spear and Chown 2009, Taylor et al. 2015).

Despite many decades of research, scientists are only now coming to grips with understanding local adaptation and species responses to various impacts (e.g. (Pertoldi et al. 2007, Reusch and Wood 2007). Thus, the concept has been established in the literature but there is still a lack of evidence. However, until empirical evidence is available, one should always use a precautionary principle – some actions cannot easily be undone (e.g. introgressive hybridization leading to extinction of a parental gene pool). The duration of this impact, as evidenced by the bontebok (van Wyk et al. 2017) and wildebeest (Mackey 2009, Grobler et al. 2011) examples, can be considered permanent and will impact the entire species.

Potential mitigation measures

1. No properties should be permitted to keep species that are likely to hybridize together in the same area.
2. Keeping wild and intensively-bred animals separate including by prohibiting intentional release into the wild (i.e. outside of intensive breeding facilities) for hunting etc.
3. Requirement to distinguish between intensively-bred and wild in marketing/hunting etc. This should include the establishment of a stud book and micro chipping register.
4. Guidelines for the management of hybrids of rare and threatened species.
5. Further research is required to understand
    a. how species evolved and how distinct they are;
    b. what the potential consequences are of natural and human-mediated hybridization between distinct species;
    c. whether former subspecies descriptions based on morphological traits were accurate and biologically meaningful;
    d. what the consequences are of outbreeding (at multiple levels of distinctiveness among taxa);
    e. how distinct geographically separated populations are and whether they should be mixed or managed separately;
    f. if there is gene flow between populations, how could that be best maintained across the transformed landscape or between separate protected areas or game ranches; and
    g. whether we can predict adaptability of species or populations.

IMPACT STATEMENT 4: PHYSIOLOGICAL STRESS AS A RESULT OF POORLY ADAPTED ANIMALS TO THEIR CURRENT ENVIRONMENT

Concerns

1. Colour patterns of many species assist in their ability to adapt to their environment and play a role for example in camouflage and thermoregulation. The artificial selection for colour variants, such as black springbok in arid environments, may lead to increased physiological stress as a result of the increased cost of thermoregulation.
2. Movement of animals to habitats outside of their natural environmental tolerance may lead to physiological stress and lower performance.

Evidence

The coat and skin colouration in mammals is driven by three broad evolutionary forces namely concealment from predators or prey, communication within or between species and diverse physiological considerations (Caro 2005). Within mammals the principle evolutionary driver of colouration is background matching to avoid predation (Caro 2013). For example, many antelope species have dark dorsal and light ventral fur assumed to be a form of self-shadow concealment. Patches of coloured fur on the faces, ears, legs, and tails of mammals are used for intraspecific signalling, but the understanding of the content of these signals or understanding what observers they target (e.g., predators, prey, or potential mates) is not fully understood (Caro 2005). Further, populations may have very different ratios of different colour morphs, indicating the adaptive significance of coat colour to specific ecological conditions e.g. leopard (*Panthera pardus*), and Asian golden cat (*Pardofelis temminckii*) (Eizirik et al. 2003, Allen et al. 2010). The physiological consequences of different coat hues are also very poorly understood. Coat colours may also affect heat loading (Walsberg 1983), but whether this impacts maximum or minimum body temperatures and the seasonality of such effects is only beginning to be understood.

Hetem *et al.* (2009) were the first to complete a study comparing the body temperature and behavioural thermoregulation of three colour morphs of a single wild African mammalian species, under the same environmental conditions. The study reported that black springbok farmed in the Eastern Cape tended to tolerate cold winter temperatures better than the other colour variants. This may be due to the fact that black springbok spent less time foraging, which may reflect a lower cost of thermoregulation in winter. However, this is likely to be a disadvantage in hotter climates, such as the Karoo. In hot climates, the black springbok displayed a higher peak body temperature and a faster rate of body temperature rise than white springbok and the springbok with natural colouration (wild-type). The black variant also displayed the lowest diurnal activity and since the major contribution to this activity is foraging, they ate less during the winter. This may represent a lower metabolic cost of thermoregulation since they gain heat from solar radiation more readily. White springbok had lower minimum body temperatures than the other springbok as a result of not being able to maintain a positive energy balance in cold conditions. The coat reflectance of the black and white springbok was consistent with that reported for black goats and white cattle; the black coats had a significantly higher conductance than others and thus absorb more radiant energy. The study concluded that there are most probably selective pressures acting against both the black and white springbok at different times of the year, while the wild-type tends to be the compromise between these two morphs.

Natural selection is the differential reproductive success of different phenotypes, a process which slowly changes gene frequencies in populations. Those individuals with phenotypes that are selected against are less successful than others and, eventually, their genes are eliminated from the population (Ruggiero et al. 2000). According to Butler in an article published in the Farmer's Weekly (Bezuidenhout 2012) constant artificial selection could influence sexual selection in various ways. Mate selection, which usually depends on the female, has a great influence on sexual selection. Female mate choice can be influenced by several factors. Secondary sexual traits such as ornaments, displays and colour could influence mate selection (Bailey and Moore 2012). In Stone's sheep (*Ovis dalli stonei*), and possibly springbok, coat colour influences sexual selection (Loehr et al. 2008). Similar, in the Himalayan tahr (*Hemitragus jemlahicus*) males with a lighter coloured ruff dominated over darker ruffed ones, in both aggressive interactions and access to oestrus females (Lovari et al. 2009). In other instances, offspring reference the physical characteristics of their parents and may choose a mate that resembles those parents (Bezuidenhout 2012). Young females may also observe and choose rams that have successfully mated with other females and thus base their choice on the experience of older ewes. In male to male competition physical characteristics, such as size, are generally important in determining which individual will win

mating rights. When a colour morph i.e. a white springbok, happens to be more dominant than the wild-type, individuals expressing the recessive white gene would out-compete other males for the best territories and mate with the most females, transferring the trait to the next generation. At this stage these impacts are still very theoretical. Practical evidence and a better understanding of imprinting and mate choice are required.

When specific traits, such as visual appeal to humans, are selected for artificially, the adaptive value of the trait is seldom considered. This may have unforeseen consequences and is likely to counter natural selection pressures that adapt an animal to its environment. As highlighted above colour variation may influence an animal's thermoregulation. But colouration may also influence camouflage and social interactions such as mate selection.

Extent of evidence

The evidence is limited. The study by Hetem *et al*. (2009) was the first to compare the body temperature and behavioural thermoregulation of three colour morphs of a single mammalian species, under the same environmental conditions.

Level of agreement

There seems to be a general agreement that colouration may influence an animal's thermoregulation as shown by the study of Hetem *et al*. (2009). However, the physiological consequences of different coat hues are poorly understood and both positive and negative consequences have been documented depending on the environmental conditions. Similarly, the influence of colouration on both social interactions such as mate selection and camouflage and its impact on predation risk are still poorly understood and require further investigation.

Key findings

When specific traits, such as coat colour, are selected using artificial selection, the adaptive value of the trait is seldom considered. This may have unforeseen consequences and is likely to counter natural selection pressures that adapt an animal to its environment. It has been established in the scientific literature that colour variation is likely to influence an animal's thermoregulation. However, the physiological consequences of different coat hues are poorly understood and both positive and negative consequences have been documented depending on the environmental conditions. It has also been established that colouration may influence camouflage and social interactions such as mate selection. Evidence to support this however is still limited and further research is recommended. The probability of colouration affecting the thermoregulation of an individual and as a consequence the productivity of the animal is likely for as long as the animal is kept outside of its natural environmental tolerance.

Potential mitigation measures

1. Research on the effect of coat colour selection on level of predation, territoriality and dominance and mate selection is required to fully understand the impact of coat colour selection on an animal's adaptability to its environment.
2. Physiological stress and behavioural differences between the novel phenotypes and wild types should be investigated in a wide variety of landscapes and across different species.

IMPACT STATEMENT 5: DOMESTICATION OF WILD SPECIES RESULTING IN A LOSS OF THEIR NATURAL ABILITY TO ADAPT TO WILD CONDITIONS

Concerns

1. Process of domestication that in the short term leads to the habituation of animals to humans, but in the long term leads to the selection for more timid animals that adapt better to a captive environment and might be less adaptable to wild conditions.
2. Erosion of the social structure and behaviour of intensively-bred animals over time resulting in a loss of their natural ability to adapt to wild conditions i.e. predator naivety.
3. Resource selection and the inability to adapt to changing environmental conditions i.e. droughts.

Evidence

Even though domestication has been studied for centuries (Larson and Burger 2013, Marshall et al. 2014), there is still little consensus about its definition (Price 1999, Dobney and Larson 2006, Zeder 2015, Teletchea 2017). The definition of domestication can be unpacked as a long and endless process during which intensively managed animals become gradually adapted to both humans and captive conditions (Teletchea 2015). This is a much more complex process than taming, because it includes genetic modifications of the species in question by intensive directed selection for preferred phenotypic traits and their underlying genotypes (Cieslak et al. 2011). As controlled environments differ from wild ones, the genetic variants favoured under these conditions differ from those favoured in natural environments. Genetic adaptations to captivity have been shown to be overwhelmingly deleterious when captive populations are returned to the wild and have been documented in mammals, fish, insects, plants and bacteria (Frankham 2008, Champagnon et al. 2012). According to Price (1997) three processes are central to domestication. First, there is a relaxation of certain natural selection factors, such as predation and starvation. Second, there is an intensified selection of traits preferred by humans resulting in animals morphologically different to the ancestral population. Third, there is selection for characters and characteristics adapting animals for a life in captivity. Teletchea (2017) follows a similar approach but divides the evolution of animals into five main genetic processes namely two uncontrolled processes – inbreeding and genetic drift, two partially controlled processes – natural selection to captivity and relaxed natural selection and one controlled process – active selection, because changes are directional. Frankham (2008) further notes that the extent of genetic adaptation to captivity depends upon selection intensity, genetic diversity, effective population size and number of generation in captivity. Alleles that were previously rare and deleterious in the wild, but favoured in captivity appear to form the basis of genetic adaptation to captivity (Frankham 2008). Waples (1999) stated that domestication selection is inevitable in a captive population. Even though there is limited experimental research on the evolution of different traits during domestication there is sufficient evidence based on comparative studies of domestic stock and their wild ancestors to identify typical domestication changes. The evidence includes the following aspects (see Jensen and Andersson (2005), Larson and Burger (2013) and Wilkins *et al*. (2014) for reviews on the topic and examples):

1. External morphological changes such as altered fur and plumage colours (mainly an increased frequency of white and spotted colour morphs), changes in body size and growth pattern, and changes in relative size of different body parts (including brachycephaly, the shortening of skulls, and chondrodystrophy, the shortening of legs);
2. Internal morphological changes, such as an overall decrease in brain size, and modified relative sizes of other internal organs, for example, intestines;
3. Physiological changes, such as changes in endocrine responses and reproductive cycles;

4. Developmental changes, such as earlier sexual maturity and changes in the length of sensitive periods for socialization; and
5. Behavioural changes, such as reduced fear, increased sociability, and reduced anti-predator responses.

Figure 4: Where the objective is to intensively manipulate animals, “tameness” may be preferred and are included in the marketable features of the animals displayed in auction brochures. (Names of auctioneers and breeders disguised on purpose).

One of the longest standing domestication projects focusing on tameness in foxes found that through intensive artificial selection the offspring exhibiting the aggressive and fear avoidance responses were eliminated from the experimental population in just two to three generations of selection (Trut 1999, Hare et al. 2005, Trut et al. 2009). Not only did the foxes became tame but their behaviour, morphology and reproductive habits started changing. Trut states that “by 2005-2006, almost all the foxes were playful, friendly and behaving like domestic dogs. The foxes could "read" human cues and respond correctly to gestures or glances. The vocalisations they made were different to wild foxes”. In addition, many new morphological traits appeared in the tame foxes. The domesticated foxes had floppier, drooping ears, curlier tails and their coat colour started to change (Trut et al. 2009). At the more advanced steps of selection, changes in the parameters of the skeletal system began to arise including shortened legs, tail, snout, upper jaw, and widened skull. Their reproductive habits also changed (Hare et al. 2005). The domesticated foxes became sexually mature about a month earlier than non-domesticated foxes. Their mating season was longer, and they could breed out of season. On average, their litters had one more cub.

One of the first modifications during domestication is behaviour (Price 1999). Yet, behaviour traits did not appear or disappear, but the threshold of their expression changed (Price 1999). Behaviour is a central part of the mechanisms allowing animals to adapt to their social and physical environments. Therefore, selection side effects on behaviour may have serious effects on the welfare of animals. If genes that are under selection pressure during breeding for a specific trait simultaneously affect behaviour, the adaptive capacity of the selected animals may be affected (Jensen and Andersson 2005). The relationship between temperament (a behavioural trait) and individual fitness is complex. However, several studies have shown an association between temperament traits and fitness traits both in captivity and in the wild (see (McDougall et al. 2006). Furthermore, the existence of sets of correlated

temperament traits, or of some links between temperament and morphological or physiological traits, is common to many species (McDougall et al. 2006).

The history of ostrich farming is interesting as it represents one of the first attempts at truly domesticating a wild indigenous species in Africa (Roth and Merz 1997). Breeders have generally applied animal husbandry methods, which are characteristic of domestication, focusing initially on characteristics of feather quality and particular feather features. Various distinct strains of domesticated ostrich emanated, which were entered into a special Ostrich Section of the South African Studbook and which attained a higher degree of hereditary purity through inbreeding. Due to changing market demands and circumstances feather quality is no longer the prime selection factor and now selection is aiming to achieve maximum meat and hide productivity, including by using advanced genetic and breeding techniques.

The process of domestication can be divided into four stages (Roth and Merz 1997):

Stage I - Kept captive or with occasional breeding;

Stage II - Kept captive with breeding, beginning genetic isolation;

Stage III - Kept captive or herded, selective breeding, full genetic isolation, semi-domesticated; and

Stage IV - Fully domesticated, docile, genetic changes, breeds.

Prolonged breeding alone in controlled environments (be it in cages or larger enclosures or camps), constitutes only ‘semi-domestication’ and rarely renders animals completely docile. Full domestication (Stage IV) is achieved only by long-term controlled breeding with isolation from wild species and the application of various degrees of animal husbandry. Selective breeding and husbandry aim at the promotion of distinct anatomical and physiological characteristics culminating in the formation of different breeds.

Domestication exists on a continuum and is thus difficult to measure. Teletchea (2017) states that wild and domesticated represent two extremes of a process and not a simple dichotomy. In some cases this process can be reversed, a process known as feralisation (Daniels and Bekoff 1989, Price 1999). This implies that feral animals, which are no longer exposed to artificial selection by humans or natural selection imposed by the captive environment, will therefore evolve through generations to become “wild” once more (Price 1999). Depending on the species and the number of generations in captivity, feralization might not be possible (animals will die rapidly in nature) or will take a long period of time for animals to return to “wild” form; yet they will not go back to the original “wild” ancestor genotype and phenotype. One of the best example is the house cat (*Felis catus*) (Driscoll et al. 2009), whose domestication started about 4000 years ago from the African wildcat (*Felis silvestris lybica*) and that establishes numerous feral populations worldwide (Nogales et al. 2004).

The popularity of intensive and selective breeding as a diversification strategy or direct business decision in the wildlife ranching sector has increased significantly (Cloete et al. 2015). Results from a survey conducted by Cloete *et al*. (2015) revealed that the introduction of high value species was among the medium term plans of at least 69% of ranchers who do not already have a high value breeding component. Moreover, at the time 59% of these ranchers were considering the breeding of colour variants. Desmet *et al*. (2017) showed that for an area in the upper Limpopo Valley near Thabazimbi the number of properties assessed as being intensive breeding operations increased from eight properties (4% of total, n = 208) in 2006 to 17 (8%) in 2010, and reaching 51 (25% of total) in 2015. Ostrich farming in South Africa reached Stage III about 50 years ago (Roth and Merz 1997), and it could be argued is now entering Stage IV with the registration of distinct strains. Based on the above many game species would be considered to be at Stage II but are now entering Stage III with sophisticated selection taking place for desirable characteristics. The recent ability for the registration of

breeds under the Animal Improvement Act is potentially providing an enabling environment for "speeding up" the process of domestication of game towards Stage IV i.e. full domestication.

Extent of evidence

There is a large body of evidence showing that domestication results in diverse phenotypic and behavioural changes to wild animals, including decreased flight responses, increased sociality, earlier reproduction, and modification of endocrine and metabolic systems (Roth and Merz 1997, Jensen and Andersson 2005, McDougall et al. 2006, Larson and Burger 2013). Valuable lessons must thus be drawn from conservation or captive breeding, e.g. conservation breeding of cheetah, breeding programmes in zoos, etc.

Level of agreement

There is a high-level agreement within the scientific literature that the process of domestication will be initiated when animals are moved from the wild into a controlled environment and that over time this will lead to genetic, morphological and behavioural changes.

Key findings

It is well established in the scientific literature that over time domestication results in diverse phenotypic and behavioural changes to wild animals, including decreased flight responses, increased sociality, earlier reproduction, and modification of endocrine and metabolic systems. The probability that the process of domestication will take place within intensive breeding facilities is virtually certain and the impacts or effects of domestication are likely to be permanent with respect to the individuals within intensive breeding facilities. However, the severity of this activity will be related to the proportion of animals of a specific species that are intensively bred versus the wild (more severe for rare animals), as well as the time frame that the individuals are subject to these conditions.

Potential mitigation measures

1. Keeping wild and domesticated/intensively bred animals separate including by prohibiting intentional release into the wild (i.e. outside of intensive breeding facilities) for hunting etc.
2. Requirement to distinguish between domesticated and wild in marketing/hunting etc. This should include the establishment of a stud book and micro chipping register.
3. Conscious prevention of domestication through regulation of breeding practices or adoption of best practice following conservation breeding guidelines.
4. Development of a clear policy framework with respect to domestication of wild species

Figure 5: An example of blue wildebeest colour variants currently available in the market.

ISSUE 2: IMPACTS ON WILD POPULATIONS THROUGH UNSUSTAINABLE MOVEMENT OF ANIMALS FROM THE WILD INTO CONTROLLED ENVIRONMENTS, INTRODUCTION AND GENETIC INTROGRESSION OF GENETICALLY ALTERED ANIMALS INTO WILD POPULATIONS, AND INCREASED RISK OF INTRODUCTION OF SPECIES TO HABITATS WHERE THEY DO NOT NATURALLY OCCUR

IMPACT STATEMENT 1: THE NATURAL GENETIC COMPOSITION, EVOLUTIONARY TRAJECTORY AND ADAPTIVE POTENTIAL OF WILD POPULATIONS ARE COMPROMISED AS A RESULT OF INTRODUCTIONS OF INTENSIVELY-BRED ANIMALS THAT HAVE UNDERGONE GENETIC CHANGES.

Key concerns

1. It is expected that intensively-bred specimens will differ from wild specimens as a result of different selection pressures (see Chapter 4). Animals that escape from these intensive breeding facilities could have direct and indirect negative impacts on wild populations. It is further expected that these farmed specimens will have severely reduced fitness compared to wild counterparts.
2. The number of animals with selected traits may become dominant in the wild through sheer mass of presence as the industry grows. The morphological changes that are present within intensive and/or selective-bred individuals do not allow for the species to survive within the wild. These may be as a result of the reduced ability to feed correctly or survive in sub-optimal conditions in comparison to captivity.
3. This change in the genetic composition may result in populations that are unable to adapt to environmental changes and consequently face an increased extinction risk.

Evidence

After a review of 233 indexed, scientific papers dealing with the potential impacts that intensive and selectively bred populations may have on wild populations, Champagnon et al. (2012) concluded that impacts on wildlife include among others: changes in behaviour, morphology, demography in recipient populations, as well as enhancement of pathogen spread. Negative genetic effects on recipient populations include homogenisation, introduction of non-native genes, and loss of local adaptation, affecting the genetic integrity of wild populations. The review demonstrates that restocking practices may and do cause significant disruptions of natural patterns in wild recipient populations (Champagnon et al. 2012). The resultant effect may limit the settlement success of these animals in the wild (various authors cited in (Blanchet et al. 2008)). Lynch and O'Hely (2001) recommended that the selective pressures within the captive environment need to be closely managed and resemble those in the wild, failing which genetic transformations will occur resulting in a population that is unlikely to sustain itself under wild conditions.

Slade *et al.* (2014) suggested that significant differences exist in phenotypic and genetic measures between captive-bred and wild adult mice. In addition, offspring produced post release shared the same parentage, which inferred pronounced assortative or non-random mating i.e. captive-bred individuals were more likely to breed with one another than with their wild conspecifics. This in turn led to the disruption of natural selection within the wild population.

Behavioural differences within species arise from a variation in the temperament traits of such species. These differences in turn can significantly affect the manner in which individuals interact with the environment, thereby influencing Darwinian fitness. The temperament of a species refers to relatively consistent individual traits, such as activity, tameness, aggressiveness, exploration, sociability and boldness, that underlie and modulate the expression of behaviour (Gosling 2001, Sih et al. 2004). These traits are the result of combined

influences of genetic, epigenetic (developmental) and environmental effects (McDougall et al. 2006). This concept implies a consistency in character over time and across different behavioural contexts and ecological situations. These differences in temperament can result in individual differences in behavioural patterns relevant to conservation, such as anti-predator, foraging and exploratory behaviour (McDougall et al. 2006). Humans may deliberately attempt to modify animal temperaments, but human-induced changes are often an unexpected, unplanned and a detrimental side-effect of human activities. Belyaev (1979) demonstrated that selective breeding in foxes (*Vulpes vulpes*) produced major shifts in morphology, physiology and temperament. Additional studies cited by McDougall *et al.* (2006) supports the notion that selective breeding produced such major shifts through directed and methodical selection. Managing temperament is critical if intensively-bred individuals are to survive in the wild, yet currently very little is known of such in terms of indigenous game species in South Africa.

Oldfield mice (*Peromyscus poloniotus*) kept in captivity for several generations no longer demonstrated the appropriate anti-predator reactions. Captive-bred individuals exhibited less caution, spent more time in an open area and remained within a shelter for shorter periods after exposure to a predator. This suggests that the differences in anti-predator behaviour could be related to selection for bolder individuals in captivity (McPhee 2003). Selection in captivity based on temperament may also alter the appropriate life history, morphological and physiological traits required for life in the wild. For wild species, these are likely to be functionally related and shaped by the long-term action of natural selection (Sih and Maurer 2000). Further, Marliave *et al.* (1993) suggests that the selective pressures present in captivity and the effects of associated inbreeding can break the functional links between specific traits of species, which in turn leads to non-adaptive trait combinations. As an example, Coonstripe shrimp (*Pandalus danae*) which were selected for tameness, had subsequent generations inadvertently characterised by a loss of pigment and an increased growth rate, a trait which was not initially selected for. The available knowledge as to which temperament traits are the most important for survival is limited (McDougall et al. 2006).

Figure 6: Example of sable antelope *(Hippotragus niger)* being intensively bred in camps smaller than their natural home ranges, where social structure is manipulated and animals are provided with supplementary feeding in small buckets fixed to the fence on a regular basis.

O'Regan and Kitchener (2005) list the following factors that will have an impact upon species housed in captivity and which will lead to morphological differences to their wild counterparts. These differences include:

- Diet – The provision of additional food sources and food high in nutrient value lead to increased horn and body size in certain species, the Dall's sheep (*Ovis dalli*) and alpine ibex (*Capra ibex*) are good examples. Further, Ohlsson & Smith (2001) cited in O'Regan and Kitchener (2005) found that the protein levels in the first three weeks of life had a permanent effect on the length and asymmetry of the tarsometatarsus in captive birds.
- Mechanical properties of food – Diets of captive-bred animals are designed to facilitate growth to the extent that they may lack the necessary texture and interest which are required for other behavioural and biological processes. The absence of abrasion in captive diets has led to reduced and abnormal tooth wear in for example elephants (*Loxodonta africana*) leading to problems of malocclusion.
- Biomechanics of the skull – The authors suggest that not all areas of the skull react equally to circumstances of captivity. Various studies have found the skulls of captive animals to be significantly different from their wild conspecifics, yet in other instances dimensions remained the same. Many of these changes appear to be related to feeding apparatus and this may be as a result of the mechanical properties of the food. The incorrect positioning of feeding trays, inappropriate foodstuffs and stereotypical behaviour may all impact upon skull differences. The extent of such differences being hereditary is uncertain as a result of the limited life history of captive animals. Meers (1996) cited by the author found the skulls of captive America alligators (*Alligator mississippiensis*) to be different from their wild counterparts. These differences related to the biomechanics of the feeding apparatus. Furthermore, as a result of the desire for maximum growth in the minimum time period, captive individuals often exhibit a smaller skull than those of wild counterparts of similar body size. This suggests that the cranium cannot match the growth rate of the body. As a result, animals with larger body sizes bred in captivity are likely to outcompete wild counterparts as their larger body size would give them an advantage when competing for resources. However, the smaller skulls which result may place them at a disadvantage in holding onto carcasses (especially in the case of predators and scavengers) and swallowing food which may result in their inability to feed properly in the wild.
- Changes to soft tissues and physiology - Rich, nutritious and easily digestible captive diets may have a significant effect on gut morphology and physiology. The concentration of the required nutrients in a highly nutritious artificially formulated diet meant that captive capercaillies (*Tetrao urogallus*) developed shorter intestines and caecae and lighter hearts, livers and gizzards than those of their wild counterparts (Liukkonen-Anttila, Saartoala & Hissa, 2000 cited in O'Regan and Kitchener (2005)). This led to captive birds being less able to digest natural food particles and thus less likely to survive in the wild. Grey-Ross *et al.* (2009) suggests that captive-bred oribi released into the wild suffered as a result of not being able to adjust to natural foraging regimes after being fed a diet of pellets whilst in captivity.
- Age, activity levels and arthropathies – Grey-Ross *et al.* (2009) analysis of captive-bred oribi released into wild identified human habituation as a major concern. This habituation also increased the risk of predation upon release and as such a number of animals released were killed as a result of predation. Many animals live longer in captivity than they do in the wild. However, it is not clear as to whether changes such as arthropathy, which have been observed in captive animals, are normal age-related changes or as a result of lack of activity, different nutrition, chronic infection or other environmental factors.
- Brain size and complexity – One of the key requirements for defining a domestic mammal is the reduction in brain size (Clutton-Brock 1999). Brain size can however also be affected by environmental enrichment. This is the process of encouraging

captive animals to display more of their normal behavioural repertoire and responding to appropriate environmental stimuli (Young 2003). The ability to develop spatial learning and memory is encouraged through environmental enrichment and thus those animals which have not had such an opportunity will be disadvantaged upon release into the wild (Menzel and Beck 2000 cited by O'Regan and Kitchener (2005)). O'Regan and Kitchener (2005) list several examples of captive breeding programmes in which the subjects have smaller brain sizes compared to their wild counterparts. A study of captive reared hatchery trout showed that the areas of the brain which exhibited the greatest differences were linked to aggression, feeding behaviour and reproduction. Morphological changes as a result of captivity may be reversed with the appropriate husbandry techniques (nutrition, environmental enrichment, etc.). The inclusion of the necessary environmental enrichment may further ensure that captive animals are not disadvantaged when released into the wild. The lessening of sexual dimorphism in captive animals (as a result of selection) is a concern as this is central to most reproductive strategies. These changes may result in males not normally being selected for reproduction having an equal opportunity to breed (O'Regan and Kitchener 2005).

Domestic gene introgression into wild populations is a problem throughout the world. It should be noted, however that certain traits selected for during intensive breeding/domestication, may be preferentially selected for when domesticated animals cross back with wild populations. An example of this is provided with the mixing of free-living Soay sheep (*Ovis aries*) of St Kilda and more modern breeds (Feulner et al. 2013). The haplotype carrying the domesticated light coat colour allele was favoured by natural selection after crossing, this haplotype however was also associated with decreased survival. The authors concluded that admixture has the potential to facilitate rapid evolutionary change, as evidenced by the presence and maintenance of coat polymorphisms in the Soay sheep population.

Concerns have been expressed about the release of captive-bred mallards (*Anas platyrhynchos*) in terms of the potential to introduce gene-linked traits that reduce fitness under natural conditions to populations of wild mallards (DGIF 2007). Although largely speculative, the concern is that captive-reared mallards from various game-farm stocks may interbreed with wild mallards and adversely affect the characteristics and survival of the wild population. Studies comparing these different mallard strains indicated that differences in egg production, fertility, growth rates, and body weights may be linked to genetic differences. Game farm mallard hens began egg-laying earlier, laid for a longer time period, laid larger clutches, and had greater incubation time than wild hens bred in captive breeding situations (Prince et al. 1970, Greenwood 1975). O'Regan and Kitchener (2005) state that captive animals exhibit marked differences from their wild counterparts relatively quickly. This is as a result of the small founder populations and the marked difference in selection pressures.

An analysis of the movement of game animals through auctions within South Africa by African Wildlife Auctions illustrates an increase in number of game sold during the period 2010 to 2015, with a slight decrease in 2016 (Figure 4A). This increase year on year indicates considerable growth within the industry and thus the availability of animals for movement into extensive systems. Even though this in itself does not point to an increase in the number of intensively-bred animals sold, figures for the number of high value rare game species, mainly bred in intensive systems, sold shows an increase from 291 animals in 2001 to 6 673 animals in 2016 (Figure 4B). According to the same auctioneers the number of roan sold increased from 11 in 2001 to 208 in 2016. Similar the number of sable sold increased from 87 in 2001 to 1 096 in 2016.

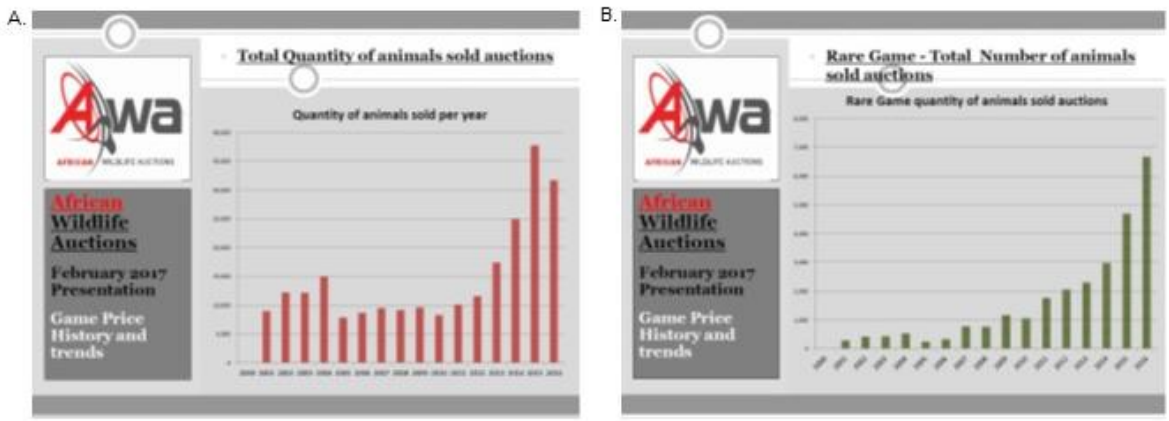


Figure 7: The quantity of all game animals sold (A) and the quantity of rare game animals sold (B) on auction per annum between 2000 and 2016 as recorded by African Wildlife Auctions (AWA 2017). Summary and analysis of auction statistics for 2016 auction calendar year for the wildlife industry (February 2017 presentation).

Extent of evidence

There are a limited number of publications, predominantly internationally, which include peer-reviewed scientific journals, popular articles and webpages that have investigated the impact of released captive individuals into wild populations. The available evidence is not considered sufficient to fully understand the implications that the release of captive/intensively bred animals may have on indigenous game populations within South Africa. However, it is clear that market trends depict an increasing supply of game animals for the wildlife industry suggesting that captive-bred game may be released into extensive systems. In comparison, there are a number of publications, both locally and internationally, that address the impacts of captive or intensive breeding and trait selection, which provide sufficient evidence to understand the possible implications that trait selection will have on game populations within South Africa. This would still require additional research in understanding the temperament of intensively-bred game populations. In terms of the impact of intensive breeding and associated genetic and morphological changes, suitable evidence is available to understand the implications that such will have on game populations within South Africa, yet further literature assessments could be undertaken, more specifically related to the genetic changes.

Level of agreement

Slade *et al*. (2014) highlights the fact that there is limited literature available on the differences between the mating patterns of captive-bred, or other wild translocated individuals and existing wild populations. The available literature makes various assumptions as to what will happen should captive-bred animals be released into the wild. These assumptions are considered sufficient to support the concerns of the effect of captive-bred animals being released into the wild. It must be noted that the captive-bred and subsequent wild animal interaction would also extend to ex situ conservation breeding programmes.

There is sufficient consensus that animals do undergo morphological changes as a result of being born and raised in captivity.

Key findings

The evidence to support the validity of this impact is established but incomplete as it relates to the wildlife industry. The assortative mating between captive and wild animals may mean that the genetic, behavioural and morphological differences exhibited in captive-bred

specimens are not necessarily passed on to wild counterparts. However, it may result in captive-bred individuals becoming more dominant within the population and thus wild counterparts are eventually reduced. It is a virtual certainty that animals are and will be introduced from captive facilities into extensive systems (the stated intention is to hunt such animals), however the impact of these introductions are uncertain at this stage as well as the scale at which these may occur. The evidence presented suggests that introductions of captive-bred specimens, which have an altered genetic composition, is unlikely to impact the broader biodiversity but has the potential to impact rare and threatened species over time. The individual welfare concerns are considered high and such will need to be monitored over time, especially when the evidence confirms that individuals bred in captivity do undergo certain changes in relation to behaviour, morphology and physiology and thus captive-bred game species may exhibit different traits to their wild counterparts. The direction in which these changes occur is also influenced by the selective and methodical management actions undertaken at each captive facility. Appropriate interventions from permitting authorities within the various provinces will be required in order to ensure that such changes do not negatively impact the species.

Furthermore, this low frequency of breeding between captive and wild individuals may also mean that any deleterious genetic changes acquired in captivity and expressed in their offspring, may result in a reduced fitness and thus lower probability of survival of these specimens in the wild. The potential for genetic issues to manifest themselves when captive-bred animals are released into the wild is probable and thus a degree of caution and understanding is required. This is an area which would require further research and investigation in terms of game populations in South Africa.

It is noted that animals undergo morphological changes as a result of captivity. The impact of small founder populations, often as a result of the cost associated with establishing an initial population, and the selective management actions associated with such, results in an exacerbated time scale. These morphological changes result in animals which may no longer be able to survive in the wild and thus the concern of such being released into the wild being no longer relevant. This impacts upon the individual's contribution to the conservation status of the species. For such species to be released into the wild successfully, suitable management actions need to be established within captive facilities to limit these morphological changes.

Mitigation measures

A detailed research programme should be initiated to understand the relationship between individuals bred under controlled environments and their wild counterparts. To ensure that no unknown impacts occur in the interim, all captive-breeding facilities should be required to maintain a comprehensive breeding register and stock record system. Any releases and or fatalities should be accounted for and monitored through the legislative channels and the following should be implemented through a standard permitting control across all provinces:

1. All captive-bred animals should be clearly marked with an approved marking system that is durable and weather resistant.
2. Captive-bred animals should not be allowed to be released into the wild without permission and under authority of a permit.
3. Wild specimens of rare or endangered species may not be introduced into a captive facility/breeding operation unless the facility forms part of an approved conservation breeding programme or is a registered commercial exhibition facility.
4. Prior to the establishment of any breeding facility or the holding of animals that exhibit any colour variation, a review of all game species present on the neighbouring properties must be undertaken.
5. Animals with colour variations or originating from captive facilities must be made known at the time of application.

6. Captive-bred individuals should not be allowed to occur on properties with existing wild populations of the same species.

Research is required to ascertain the threshold at which captive-bred individuals will become dominant in the wild. Thus, any movements of animals need to be regulated and monitored accordingly.

A complete husbandry manual needs to be compiled for captive-breeding facilities which will need to be regulated through a permitting system. Furthermore, all breeders should be required to be audited annually to ensure compliance to best husbandry practices. Such auditing should be undertaken by an independent accredited institution.

IMPACT STATEMENT 2: OUT OF RANGE INTRODUCTIONS INTO HABITATS WHICH ARE SUITABLE COULD LEAD TO SPECIES BECOMING ESTABLISHED WITHIN THE LANDSCAPE THEREBY IMPACTING UPON THE HABITAT AND/OR LOCAL NATIVE SPECIES.

Key concerns

1. The introduction of species outside of their natural distribution range may have the potential to outcompete indigenous species for available resources thereby contributing to the localised extinction of such species.
2. The habitat into which extralimital species are introduced may be impacted upon to such an extent that irreversible changes occur within the structure and composition of the habitat leading to a reduction in veld condition. This in turn may impact upon indigenous antelope species.
3. Out of range introductions may also result in unintentional hybridizations.

Evidence

Large game mammals have been introduced outside of their range for various reasons, *inter alia* game farming, hunting and conservation. Such species, although intentionally introduced have the ability to become naturalised and disperse within the surrounding area (Swanepoel et al. 2016b). Non-native species may present a threat to both the natural and modified habitats, thereby incurring environmental and economic costs. Non-native species present an array of threats to local biodiversity (genetic, species or ecosystem diversity) (Spear and Chown 2009) and have been recorded to increase human-wildlife conflict. Economic losses as a result thereof are in the magnitude of billions of U.S. dollars per year (Schlaepfer et al. 2010). Pimentel *et al.* (2005) estimate that the introduction of the Indian mongoose (*Herpestes auropunctatus*) into Puerto Rico and the Hawaiian islands has resulted in approximately US$50 million in damages each year based on public health damages, killing of poultry, extinctions of amphibians and reptiles and the destructions of native birds. This has resulted in conservation agencies across the globe undertaking an array of activities to prevent or limit the effects of the introductions of non-native species. Introductions of non-native species can however also realise positive ecological and economic benefits for example through hunting opportunities and mitigation to climate change (Spear and Chown 2009).

Native and non-native game species are regularly introduced and/or translocated across South Africa as part of the game farming industry, where such industry involves both extensive and intensively managed parcels of land. The common warthog (*Phacochoerus africanus*) have been introduced outside of their known distribution, which has resulted in increased conflict with landowners as well as impacting upon the natural as well as agricultural environment (Swanepoel et al. 2016b). Giraffe (*Giraffa camelopardalis giraffe*) introduced into Ithala Game Reserve in KwaZulu-Natal has had a notable impact upon the reserves woodland composition through the elimination of sensitive species and had it not been for inaccessible

refuge areas within the reserve, *Acacia davyi* and other species intolerant of browsing would be extinct (Bond and Loffell 2001). Pienaar (2013) found that nyala (*Tragelaphus angasii*) in the Eastern Cape exert browsing pressure within localised areas of the habitat types, which they occupy instead of across the entire habitat gradient and the stocking rate of such do need to be carefully managed, failing which nyala exert a greater impact than indigenous species in the area. Spear and Chown (2009) highlight that the rate of extralimital introductions in South Africa is considerably high, yet the extent to which the perceived risks of such introductions become evident is lacking. This lack of evidence does however not indicate that such risks do not exist and the theoretical reasons that the risks associated with translocation are indeed valid.

Rowe-Rowe (1994) suggested that nyala outcompete bushbuck (*Tragelaphus sylvaticus*) for available resources as they are able to browse at a greater height above ground level. This suggestion is supported by Coates and Downs (2005). Further competition is experienced between nyala, in high densities, and small antelope such as blue (*Philantomba monticola*) and red duiker (*Cephalophus natalensis*) and suni (*Neotragus moschatus*). This competition and localised extinctions have been reported by numerous landowners within southern KwaZulu-Natal, an area into which nyala were introduced, and once home to substantial bushbuck populations.

Extent of evidence

The publications, both local and international, which include peer-reviewed scientific journals, popular articles and webpages are considered sufficient to understand the impact of released non-native species into the wild and to make appropriate recommendations thereto.

Level of agreement

The publications purveyed are all in agreement in that non-native species do have an impact upon the local biodiversity. However, the extent thereof varies, especially if such introductions are not managed correctly. Schlaepfer *et al.* (2010) argues that certain introductions may be of some conservation value and should not be excluded in its entirety.

Key findings

The evidence of this impact is established but incomplete because the impacts have not yet been fully investigated within South Africa and currently rely on theoretical hypotheses. It is virtually certain that out of range introductions are taking place, the scale of which is uncertain. Most of the animals from intensive breeding facilities that are sold are moved to other intensive breeding facilities where they are kept in controlled environments. These controlled environments may be out of range, but because they are not released into the wild it is not really a problem at this stage. However, should these animals be released from these facilities into the wild, it may significantly increase out of range introductions. The intensive breeding of species is thus unlikely to have any additional net impact over and above that caused by existing out of range translocations between extensive and semi-extensive properties at this stage. The evidence does suggest that the introduction of non-native species presents a real threat to biodiversity, which may be limited to a cadastre level only, especially when such are not managed accordingly. Introductions can provide a degree of conservation value when managed correctly. It is not likely to impact upon the species level, except in instances where out of range introductions result in welfare concerns. This uncertainty should however invoke the precautionary principle to some extent and any introductions should only take place after careful consideration and with the necessary interventions available (research into impacts, the ability to remove, etc.) to limit irreparable damage. It is suggested that conservation experts do need to look into the future to address climatic and environmental changes and manage for such as opposed to looking backwards as to which species were present in an

area and trying to replicate such. This presents a number of ethical dilemmas which are beyond the scope of this report.

Mitigation measures

1. The movement of game within South Africa outside of their natural distribution ranges should be managed and controlled through a strict permitting procedure. The easing of permitting requirements within natural distribution ranges could be utilised as an incentive to discourage out of range introductions.
2. Further research is required to identify the extent of competition between native and non-native species, in terms of the South African game industry context and to whether or not intensive breeding facilities are encouraging out of range introductions.

IMPACT STATEMENT 3: THE REMOVAL OF WILD SPECIMENS OF NATURALLY RARE SPECIES OR SPECIES WITH CURRENTLY SMALL POPULATION SIZES, IN SOUTH AFRICA OR OTHER AFRICAN COUNTRIES, CAN LEAD TO POPULATION DECLINES RESULTING IN A LOWER OVERALL CONSERVATION STATUS AND A HIGHER EXTINCTION RISK FOR THESE SPECIES

Concerns

1. A number of high value game species are being captured from the wild and brought into intensive breeding facilities. For species with small population sizes in the wild or rare species, the continuous sourcing or “leakage” from wild populations will reduce wild population sizes and can increase extinction risk of the species.
2. For species that are not very successful breeders in intensive facilities, new wild-caught individuals must be brought into intensive breeding facilities regularly. This has a negative impact on the free-roaming or wild populations.
3. For certain high value species, it is cheaper to source animals from other African countries for intensive breeding in South African breeding facilities. This may have a negative impact on populations in those countries as well as the overall conservation status of those species.

Evidence

Introduction

It is well known that the unsustainable sourcing of animals from the wild as witnessed in the pet trade, which includes using wild sourced breeding stock for intensive breeding operations, can have a detrimental impact on wild populations (Bush et al. 2014). There are many examples related to the collection of reptiles and birds for the pet trade (Collar and Juniper 1992, Wirminghaus et al. 1999, Schlaepfer et al. 2005). The World Wildlife Fund (WWF), in their policy statement on captive breeding (2007) identified that one of the major risks is that species that are popular as pets such as reptiles or birds have sometimes been labelled as “captive-bred” but have since been discovered to have been laundered and removed from the wild unsustainably, thus damaging wild populations. WWF also stated that the greatest conservation risks associated with captive breeding are when threatened or endangered species are bred for commercial purposes for profit. This is particularly risky when individual animals or their parts and products are of high value, while at the same time the animals are highly endangered in the wild (WWF Policy Statement on captive breeding, 2007). A recent survey of green python traders and breeding farms in Indonesia found that a large number of green pythons are illegally wild-caught and exported annually. They are being laundered through breeding farms as “captive-bred” and this continuous harvesting from the wild has resulted in the decline in certain populations (Lyons and Natusch 2011).

At the same time ex situ breeding programmes have contributed significantly to the conservation of many species such as the California condor (*Gymnogyps californianus*), the Arabian oryx (*Oryx leucoryx*), whooping crane (*Grus americana*) and black-footed ferret (*Mustela nigripes*) to name a few (Maunder and Byers 2005). However, in some cases such as the Arabian oryx a major threat to the reestablishment of the species in the wild has been their illegal capture and sale to private captive facilities (Ostrowski et al. 1998). In other cases, well-intended removal of rare or threatened species from wild populations for captive breeding and release programmes may actually have a negative impact on wild populations (McCleery et al. 2014).

The source of animals for intensive breeding can come from either extensive systems or other intensive breeding facilities, and this can vary depending on the species. There are numerous wildlife breeders in South Africa that manage and breed sable antelopes in small intensive systems (Kriek 2005), and therefore availability of intensively bred animals should not be a problem. Wild-caught nyala are said to tame very well (Pfitzer and Kohrs 2005), and this offers little disincentive to use wild caught nyala in intensive breeding operations. In contrast to nyala, wild bushbuck are shy and skittish, and as such Pfitzer and Colenbrander (2005) advise that for intensive breeding of bushbuck, it is better to source captive-bred breeding stock.

Species that are difficult to breed in captivity

Cheetah are considered a prime example of a species that is under threat because there is a high export demand for captive-bred animals (CITES trade data), while they are not easy to breed in captivity, particularly wild caught cheetahs (Marnewick et al. 2007). The high numbers of live captive cheetah exported each year from various captive facilities in South Africa despite the fact that cheetahs are difficult to breed in captivity, and data collected by the Endangered Wildlife Trust, suggest that it is highly likely that wild cheetahs are routinely moved or sold into captivity (Non-detriment finding for *Acinonyx jubatus* (cheetah), issued by the Scientific Authority of South Africa). Marnewick *et al*. (2007) identified the illegal capture of free-ranging cheetahs on ranch lands, for sale to captive breeding institutions as a serious threat to the species in South Africa.

Sourcing animals from other African countries

There have been a number of news articles reporting on cases of smuggling of sable from Zimbabwe and Zambia to South Africa. In September 2015, police in southern Zimbabwe arrested three South Africans trying to smuggle 29 sable across the border. In October 2015, more South Africans were arrested for attempting to smuggle sable out of Zambia using an aircraft (News article on https://www.enca.com/south-africa/sa-men-arrested-zambia-smuggling).

Populations of wild sable in Zimbabwe are extremely vulnerable, as are those in South Africa (Parrini et al. 2016), and any losses from these wild populations, particularly adult females, can have disastrous effects on these populations (Capon et al. 2013). Although Capon *et al*. (2013) identified predation by lions as the main culprit for population declines, additional losses from the population by live capture will have a compounding negative effect.

In 2013, SAHGCA responding as an interested and affected party for the risk analysis process relating to the importation of 120 eastern Lord Derby eland (*Tragelaphus derbianus gigas*) from the Republic of Cameroon to South Africa as a commercial enterprise, indicated that this import would pose a high risk to indigenous biodiversity through the alien species' ability to cross-breed with indigenous eland, but in addition raised the concern that such a large removal could have a potential negative impact on the source population in Cameroon. SAHGCA also highlighted that this could result in the South African wildlife industry being seen as impacting

negatively on wild populations in their natural environment in the interest of financial benefit (Nel, E.J., pers. comm.)[2].

Other Evidence from South Africa

The recent regional (South Africa, Swaziland and Lesotho) red list assessment for roan (*Hippotragus equinus*) has listed the species as Endangered, with the wild and free-roaming population within the natural distribution range considered to be declining and less than 250 mature individuals (Kruger et al. 2016). The assessment identified that one of the threats to roan antelope is their removal from the wild and into captive breeding systems. It also highlighted that given the expansion of private wildlife enterprises, this threat could become more severe in the future, especially as wild roan antelope dwindle in the assessment region and thus increase their financial value. Currently, according to the assessment there are at least 1 756 individuals on 77 wildlife ranches and/or in breeding camps across South Africa, but it is estimated that only 0.8-5% of these individuals can be considered wild and free-roaming.

Breeding of small game in South Africa has recently gained momentum. There has been an increase in the number of applications to remove blue duiker from the wild into captive facilities (Ezemvelo KZN Wildlife, Unpubl. data). However, the recent non-detriment finding (NDF) for blue duiker showed that legal local and international trade in live animals and the export of hunting trophies at present poses a moderate to high risk to the survival of this species in South Africa and is detrimental to the species in the wild (Non-detriment finding for *Philantomba monticola* (blue duiker), issued by the by the Scientific Authority of South Africa). Another species that appears to be gaining popularity with small game breeders is oribi (*Ourebia ourebi ourebi*), as there has recently been a significant increase in the number of oribi moved from wild to captive facilities (Oribi Working Group, 2018)[3]. This species was listed as Endangered in the 2016 regional red list assessment, in which conservation recommendations focus on conserving the species in their natural grassland habitat, while it is also mentioned in the assessment that intensive captive breeding have been unsuccessful due to spatial requirements associated with male territoriality (Shrader et al. 2016). In this case, it is quite likely that removal of wild specimens into captivity will have a negative impact on the population and species conservation status.

Extent of evidence

There is much evidence to show that in general the practice of continuous unsustainable harvesting of animals from the wild for captivity, can have a negative impact on wild populations, to varying degrees. However, when looking specifically at species in the South African wildlife industry, the evidence is relatively limited, and is mainly applicable to higher value rare species (e.g. roan, sable, blue duiker, oribi and cheetah) rather than colour variations of common Least Concern species. There is some evidence, although limited, that some high value game species are being illegally sourced outside South Africa.

Level of agreement

There is general agreement that removal of animals from the wild for keeping in captivity can be detrimental to wild populations if done at unsustainable levels. There is also recognition that in exceptional circumstances only, captive populations can contribute to the recovery of threatened populations.

[2] Nel, E.J., Manger Conservation, SAHGCA. Response letter as an interested and affected party to the introduction of Lord Derby eland from Cameroon to South Africa, 27 January, 2014.
[3] Oribi Working Group minutes, 14 June 2018.

Key findings

The evidence on this impact, specifically with regard to species in the South African wildlife industry (e.g. roan, sable, blue duiker, oribi and cheetah) is established but incomplete (high agreement based on limited evidence). There is a certain level of continuous “leakage” of wild animals into captive facilities (for commercial purposes), which may have an impact on some species where removals are at unsustainable levels. Some species that are rare or threatened due to small population size, are very sensitive to even small removals from the wild populations. The probability of this activity/impact occurring in the industry is virtually certain, but the impact on broader diversity is unlikely because it is a scale issue and will depend on the current status of the species in the wild and the sensitivity of the population to removals. It is highly likely that there will be an impact at the species level and where species disappear from the system completely, through the disruption of the targeted species’ ecological function in the landscape.

Potential mitigation measures

1. Limit/restrict the sourcing of wild animals for commercial intensive breeding facilities, especially for rare and threatened species.
2. Introduction of wild specimens of threatened species into captivity should only be undertaken as part of a conservation breeding programme with a re-introduction plan in line with IUCN guidelines.
3. Invest in research and development of better breeding practices to improve breeding success of species considered difficult to breed under captive conditions, such that there will be less reliance on constant sourcing of wild animals for breeding operations.
4. Implement active in situ conservation programmes to ensure survival of wild populations of the species that are targeted for intensive and selective breeding.
5. Provide incentives to private game farmers to form conservancies or own larger properties where wild and free roaming populations of game can be re-established and maintained.
6. Undertake research on the viability of re-introduction of intensively-bred animals into extensive systems.
7. Rating of production systems and labelling systems implementation, so that consumers can distinguish whether products are from a controlled or wild environment.

ISSUE 3: SIGNIFICANT INCREASES IN THE EXTENT OF IMPERMEABLE FENCES WITH ASSOCIATED NEGATIVE BIODIVERSITY IMPACTS

IMPACT STATEMENT 1: FRAGMENTATION OF THE LANDSCAPE THROUGH IMPERMEABLE FENCING RESTRICTS MOVEMENT OF FREE-RANGING SPECIES AND REDUCES HABITAT AVAILABILITY

Concerns

1. Intensification of impermeable fences fragments the landscape and has a range of negative ecological impacts.
2. Impermeable predator proof fences for high value game species reduce the habitat available for free ranging populations of threatened species such as wild dog (*Lycaon pictus*), cheetah and pangolin (*Smutsia temminckii*).
3. Impermeable predator-proof fences restrict the free movement of predators resulting in an increase in the level of predation and the subsequent persecution of predators.
4. Impermeable fences are often electrified and designed in a way that lead to the unintentional mortality of non-target species.

Evidence

In South Africa, Botswana and Namibia fencing is required for land owners to legally own wildlife. In most other countries in the world the status of wildlife is res nullius (Lindsey et al. 2012). The use of fences elsewhere in the world often focus on protection of reintroduced threatened species, predator exclusion (Hayward et al. 2009, Cavalcanti et al. 2012, De Tores and Marlow 2012), exclusion of pest species (Burns et al. 2012), elimination of human wildlife conflict (East et al. 2012) and disease control (Gadd 2012). Fences used in the intensive game breeding industry in South Africa are usually erected to be impermeable for target species such as small and large ungulates often with unintentional impacts on smaller taxa such as chelonians, other reptiles as well as small mammal species i.e. pangolins. This situation seems to be unique to South Africa.

Intensification of impermeable fences fragments the landscape and has a range of negative ecological impacts.

A recent case study of an area located in the north-west of Limpopo Province, in the upper Limpopo River valley, showed that there has been a very rapid growth in the number of intensive game breeding properties and an associated rapid increase in internal impermeable fencing over a nine year period (Desmet et al. 2017). This study showed that between 2006 and 2015 all indicators assessed, pointed to a rapidly expanding industry with exponential growth in the density of highly impermeable fences (Desmet et al. 2017). The number of properties assessed with intensive breeding camps increased from 4% in 2006 to 25% in 2015. Internal fences increased by 22% from 3 702 km to 4 517 km with the rate of change increasing by 64% in the second period from 69 km new fence/year (2006-2010) to 108 km new fence/year (2010-2015). Desmet *et al*. (2017) found that the conversion from extensive to intensive operations is associated with a reduction in camp size and a decrease in fence permeability to wildlife. Median camp size decreased by 51.5 % from 35.5 ha in 2006 to 17.2 ha in 2015 and the number of camps increased by 46% from 3 129 to 4 582. The level of intensification (i.e. total area comprising camps less than 50 ha in size) observed is greater (23%) than the current estimate for the industry (6%) Desmet et al. (2017).

This trend is also supported by studies conducted by Cloete *et al*. (2015) and Taylor *et al*. (2015). In a recent survey of wildlife ranches in South Africa, 44.6% of surveyed ranches conducted intensive breeding of game (Taylor *et al*. 2015). Intensive breeders breed either high value species such as buffalo (*Syncerus caffer*), sable or roan (38% of surveyed properties) or colour variants (23% of surveyed properties). Interestingly, a small proportion of the properties surveyed (~10%) kept colour variants under extensive conditions. On the properties that conduct some form of intensive breeding, the study found that 10.9% of the land was divided into camps. A small proportion of ranches was completely (100%) subdivided into camps. Cloete *et al.* (2015) reported that there is an ongoing shift from ranching to the intensive breeding of game animals, which they refer to as “game farming”. This report also provides evidence for this stronger focus on breeding, as approximately half of the respondents in the survey derived the largest part of their income from live sales of game animals that include higher value and/or colour and morphological variants (Cloete *et al*. 2015). Cloete *et al.* (2015) suggest that this trend is expected to continue as long as the financial performance of these activities remains superior to the other wildlife-based alternatives. As long as this trend persists, it can be expected that intensification of fencing will continue and that fragmentation of the landscape into small camps will increase.

Fences used for conservation purposes can yield conservation benefits, but may also threaten biodiversity (Hayward and Kerley 2009, Woodroffe et al. 2014). The impacts associated with impermeable fences such as interference with behaviour, movement patterns and fragmentation of populations of smaller species are acknowledged by de Tores *et al*. (2012) and Hayward and Kerley (2009). Dickman (2012) states that fences may exacerbate problems such as inbreeding, genetic adaptation to confinement and predator naivety.

Fences limit the movement of large herbivores in the landscape. Therefore, fenced areas become fragments within the broader landscape. Fenced pieces of land eliminate herbivore access to heterogeneous forage patches and may reduce the carrying capacity of a landscape irrespective of habitat loss (Boone and Hobbs 2004, Cousins et al. 2008, Lindsey et al. 2012). Fences can prevent wildlife from accessing key resources, and at a landscape scale, have been known to disrupt the ecological processes of emigration and immigration of wildlife (Boone and Hobbs 2004). Conversely, in an unfragmented heterogeneous landscape, herbivores have a wider range of choice of resources relative to those landscapes where movement is restricted (Boone and Hobbs 2004, Lindsey et al. 2009). Boone and Hobbs (2004) used ecosystem modelling to quantify the decrease in herbivore stocking rate as a 300 km^2 parcel of land was purposefully fragmented. They showed that when the parcel was fenced as 10 km^2 sub-parcels, 19% fewer cattle could be supported, compared to the unfenced parcel.

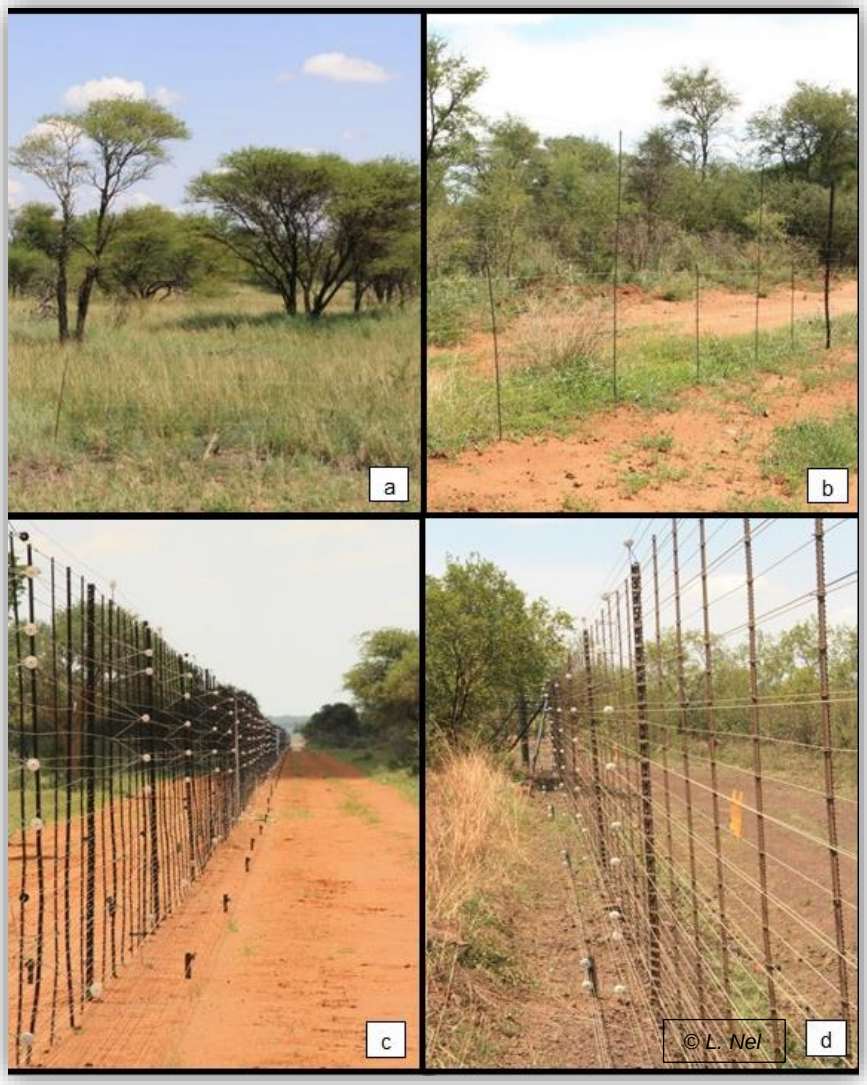


Figure 8: Examples of a) typical cattle fence, b) game fence, c) and d) impermeable predator proof fencing used by many intensive breeding facilities, some with more than 13 electrical strands.

Impermeable predator proof fences for high value game species reduce the available habitat for free-ranging populations of threatened species such as wild dog, cheetah and pangolin

Cheetahs are regionally classified as Vulnerable by the IUCN Red List (van der Merwe *et al*. 2016). The decline in range for the species across Africa is estimated to be 89%, while the overall population abundance and extent of occurrence of cheetah is strongly suspected to have declined by at least 30% over the last three cheetah generations (Durant *et al*. 2015). In southern Africa, which is considered a regional stronghold for cheetah, the species is known to occur in only 22% of their historical range (Durant *et al*. 2015). According to the IUCN, the decline in range of the species is primarily due to habitat loss and fragmentation; killing and capture of cheetahs due to livestock depredation; and loss of prey (Durant *et al*. 2015).

African wild dogs have also disappeared from much of their former range and their populations have declined to such an extent that they are considered Endangered by the IUCN (Woodroffe and Sillero-Zubiri 2012). Habitat fragmentation, together with conflict with human activities and infectious diseases, are considered the principal threats to survival of African wild dogs (Woodroffe and Sillero-Zubiri 2012).

In South Africa, wildlife ranchers are increasingly constructing predator proof impermeable fencing to protect their high value antelopes and are thus reducing habitat availability for free-ranging populations of threatened species such as wild dogs and cheetahs. For example, the study area where Desmet et al. (2017) found a rapid increase in intensive breeding and associated intensification of fencing, namely the Limpopo Waterberg, is an area where wild dog and cheetah are known to occur as free-roaming populations.

Impermeable predator – proof fences restricts the free movement of predators resulting in an increase in the level of predation and the subsequent persecution of predators.

In a study on wildlife predation in the Limpopo Province of South Africa (Schepers et al. 2018) black-backed jackal and cheetah were identified as priority predators for large antelope species such as kudu and blue wildebeest. Caracal and wild dogs were identified as priority predators of scarce and colour-variant antelope species. In their study the use of electric fences was identified as a non-lethal control method and contrary to expectation the use of electric fences had an increasing effect on the level of predation (percentage of wildlife losses) for large antelope species. They concluded that the enclosed perimeter means that the predators are limited to the enclosed area resulting in increased predation. This would most probably result in the persecution of such predators.

Impermeable fences are often electrified and designed in a way that leads to the unintentional mortality of non-target species.

The scientific evidence describing the impact of fences and specifically the mortalities associated with electrified fences is mostly based on work done on veterinary fences (e.g. Gadd, (2012)) and game fences around protected areas (Van Dyk and Slotow 2003, Ferguson *et al*. 2012), and more extensive conservation areas (Lindsey et al. 2012, Slotow 2012). Very little scientific evidence exists on the impact of fences for smaller areas used for intensive game breeding purposes.

The negative impacts of fences (Lindsey et al. 2012) can briefly be summarised as: 1) Impacts on ecological processes at various scales; 2) Animal welfare implications; 3) Social impacts; and 4) Financial impacts.

For the purpose of this discussion the focus will be on the animal welfare implications and specifically mortalities associated with electrified fences. The impact on species not intended for containment is discussed although some of the issues raised may also be relevant to species kept for breeding purposes. Mortality is usually due to starvation, dehydration, entanglement and/or electrocution (Beck 2010, Hanks 2010, Pieterson 2013, Pieterson *et al*. 2014). Electrocution has been reported as the most important cause of mortality of pangolin

by Pieterson (2013) and Pieterson *et al*., (2014). In South Africa it has been recorded that at least 33 species (16 mammals, 14 reptiles, 3 amphibians) have been either killed or injured as a result of electrical fences (Beck 2010, Lindsey et al. 2012). Chelonians (Hayward and Kerley 2009) seem to be particularly vulnerable, especially the leopard tortoise *Geochelone pardalis* (Lindsey et al. 2012). Of the 43 tortoise species that occur across the world 33% are found in South Africa (Arnot and Molteno 2017). The vast majority of the country's tortoises are found on commercial and game ranches making them extremely vulnerable to the effects of electrical fences (Arnot and Molteno 2017).

Design of fences (example single strand vs diamond mesh) and particularly the configuration (example number and height of electric strands) for the electrification of fences differ depending on the purpose of the fence. In South Africa, provincial conservation authorities differ in the prescripts for fencing with most authorities only setting minimum standards. This leaves the door open for private fencing companies to erect fences that exceed the minimum requirements (Beck 2010). Lindsey *et al*., (2012) and Arnot and Molteno (2017) recommended the re-design of fences to reduce entanglement and electrocution of non-target species.

Figure 9: Various invertebrate and vertebrate species electrocuted when trying to move underneath this fence at an intensive breeding facility in Limpopo Province.

In conclusion, within South Africa the negative impacts of fencing and particularly the impact of electrified fences has been relatively well documented. The major causes of mortality are entanglement and electrocution. Most of the impacts associated with fencing (extensive areas) would be relevant to scenarios where electrified fences are used for intensive breeding purposes. However, a detailed scientific study of the impact of fences on non-target species in an intensive game breeding scenario has not been conducted. Detailed information on the extent of fences related to intensive breeding systems is largely unknown although it can be

assumed that it is on the increase if the number of new applications received by conservation authorities for such projects is used as an indicator (Van Wetten, pers. comm.)[4].

Figure 10: Unintended consequences of electric strands very low to the ground. Number of beetles electrocuted at an intensive breeding facility in Limpopo Province.

Extent of evidence

Evidence suggests that fencing for conservation purposes in general may have positive outcomes for the effective conservation of biodiversity worldwide. Evidence of the impact of fences is largely based on fences around protected areas and larger conservation areas. The impact of veterinary fences is also well documented. There is substantial evidence for the negative effects of fragmentation as a result of fencing in general. Evidence from South Africa illustrates that the extent of impermeable fencing is on the increase. The negative impacts of impermeable fencing on free-ranging species are well documented. Impermeable fences, often as a result of electrification, are generally acknowledged as undesirable. Very little research has been done on the impact of fences for smaller more intensively managed areas although there would be no reason why the impacts would be any different from other fences mentioned above.

Level of agreement

Although conservation fencing in general may have positive outcomes relative to the purpose for which it is used (disease control, minimizing human wildlife conflict) there is a high level of agreement in the literature that fences contribute to the fragmentation of the landscape and that especially impermeable fencing restricts movement of free-ranging species and reduces habitat availability for such species.

Within the South African context authors seem to agree that the extent of impermeable fencing is on the increase in South Africa. There is currently no agreement on the percentage of the

[4] Van Wetten, A.A., Deputy Director, CITES and Permit office, LEDET. 8 March 2017.

landscape affected by it (6% - 23%) nor is there a minimum threshold beyond which affected landscapes will become a critical issue.

There is a high level of agreement that impermeable fences have negative impacts on free-ranging species such as cheetah, wild dog, pangolins, various reptile species and even Scarabaeoidea species. There is agreement on the major causes of mortality in free-ranging species caused by impermeable fencing.

Key findings

Fencing in general may have both positive and negative outcomes. Fencing whether for game farming or agricultural purposes is a permanent fixture in the South African landscape from a local (property) to a national scale. Impermeable fencing is widely regarded as undesirable and the evidence for that is well established. It is virtually certain that fencing for intensive game farming practices is on the increase, and with increased impermeable fencing comes greater fragmentation of the landscape. For intensive breeding projects (smaller parcels of fenced land) the length of fences per unit area is higher than for extensive areas. The full extent of such fences (smaller parcels of fenced land) is not fully known at this stage. It is highly likely that fragmentation will have a negative impact on broader biodiversity and especially on free-ranging threatened species such as cheetah, wild dog and pangolin, which could lead to reduced population performance, possible local extinctions and a declining conservation status of the species. Poorly designed electrified fences have been identified as a key contributor to the mortality of non-target species. Electrocution and entanglement are the main causes of mortality. Scientific studies on the impact of impermeable fences, specifically for smaller parcels of fenced land, do not exist.

Mitigation measures

1. Apply a land-use planning process to restrict or limit intensive breeding in camp systems in sensitive and threatened ecosystems (Critical Biodiversity Areas).
2. The implementation of a conservation spatial planning approach that will, for example, identify corridors for movement of free-ranging species.
3. Reduce fencing to the minimum by promoting large extensive areas, e.g. through tax and other financial incentives for landowners that opt not to sub-divide into small camps.
4. Allowing for a portion of the property to remain extensive and ensuring that external fencing is permeable.
5. Re-design intensive breeding facility fences to be more permeable for non-target species.
6. National Norms and Standards for fencing for different land-uses, focussing on maximum specifications instead of minimum specifications.

IMPACT STATEMENT 2: HIGH CONCENTRATION OF ANIMALS IN SMALL AREAS WITH IMPERMEABLE FENCES FOR INTENSIVE BREEDING PURPOSES RESULTS IN HABITAT DEGRADATION WITHIN SUCH AREAS

Concerns

1. Unusually high densities of animals in small breeding camps may cause overstocking that result in overgrazing and increased trampling effects, leading to habitat degradation and loss of plant species diversity.
2. Overgrazing and trampling in small camps may result in soil compaction or crusting, erosion, and loss of soil.

3. Severe grazing patterns may result in an increase of undesirable woody species and some poisonous plant species.
4. Transformation of natural veld to planted pastures with a homogenised structure and composition.
5. Excessive or complete removal of certain vegetation strata such as the woody component from intensive breeding camps.
6. Land intensification practices negatively affect ecosystem functioning and services.

Evidence

Unusually high densities of animals in small breeding camps may cause overstocking that leads to overgrazing and increased trampling effects, and ultimately to habitat degradation and loss of plant species productivity and diversity.

By definition, intensive breeding or farming of game entails the confinement of game in small to medium-sized fenced camps or enclosures, protected from predators and provided with most or all of their food and water (Carruthers 2008, Taylor *et al*. 2015). Cloete *et al.* (2015) reported that intensive breeding practices were conducted in camps ranging from 25 ha to 50 ha each, while Taylor *et al*. (2015) found the median size of camps to be 50 ha. It was also estimated that 6% of the area of all ranches in South Africa is under intensive breeding (i.e. area under camps) (Taylor *et al*. 2015). Desmet *et al*. (2017) found that there is a general trend of decreasing camp sizes and a growth in the number of medium and small camps <50 ha. Even though intensive breeding involves variable levels of supplemental feeding, there is concern that confinement to small camps can lead to impacts related to overabundance and overstocking.

In considering the issue of overabundance or overstocking, the ecological capacity and stocking rate or density must be taken into account. The “ecological capacity” is the animal population density when it is in equilibrium with the resources of the environment that is likely to exist in large unmanaged natural areas (van Rooyen 2002). The “stocking rate” or “stocking density” is the density at which wildlife populations are kept through the management practices of landowners (number of animals per unit area) and is adjusted according to the aims of management. Stocking animals at higher densities than the ecological capacity (i.e. overstocking) is expected to lead to negative environmental consequences. Taylor *et al*. (2015), in their survey, compared potential versus actual stocking densities of game farms, and found that 32% of properties were over the recommended stocking density, while 5% of the total had more than twice the recommended biomass of grazers.

In a synthesis of available empirical information, O’Connor *et al.* (2005) showed that stocking rate, amongst other factors such as fire intervals, can have an impact on plant diversity. In this study that focused on forb species in grasslands, it was found that increased grazing pressure (stocking rate) inevitably results in a loss of forb species over time (O'Connor 2005, Uys 2006, Martindale 2007, Scott-Shaw and Morris 2008). Some forb species increase and others decrease in response to increasing grazing pressure (Scott-Shaw and Morris 2008).

O’Connor *et al.* (2010) also used a multi-criteria approach known as the analytic hierarchy process (AHP) to formalise expert opinion into a quantitative statement on the relative impact of different grazing systems on grassland biodiversity integrity. Their results showed that high-intensity grazing systems had a substantially greater negative impact on biodiversity integrity than either conventional or continuous systems. The greater impact of high-intensity systems was attributed to the greater degree of transformation and fragmentation associated with additional infrastructure and the effect of higher stocking rates on both landscape composition and ecosystem functioning.

Stocking rate has an immediate effect on the quantity of forage available to herbivores thereby affecting animal performance, but there are also long-term effects of stocking rate on vegetation, which affect the composition and productivity of the veld. Overgrazing as a result

of high stocking rates reduces the ability of vegetation to produce herbage, while prolonged overgrazing leads to a change in plant species composition of the veld (Tainton *et al*. 1999). In general, these changes occur in the form of a reduction in the palatable and productive plant species and their replacement by unpalatable less productive grasses and forbs (Tainton *et al*. 1999). The nature and extent of composition change due to overstocking may vary across different vegetation types, but productivity of the veld is usually reduced as a result of such changes (Tainton *et al*. 1999).

A consistent pattern of heavy grazing is the reduction and replacement of perennial grasses by shorter lived biennials or even annuals that produce less, fail during drought years, thereby increasing the variability of forage supply over time. Fynn and O'Connor (2000) showed that heavy stocking (of cattle) in a semi-arid savannah resulted in large decreases in certain tufted perennial grasses with relative increases in annuals and weakly tufted perennials. The effect of degraded composition on primary production has been shown to apply elsewhere on the globe over a range of climatic conditions (Milchunas and Lauenroth 1993).

The number of ostriches in the Little Karoo has fluctuated radically over time due to outbreaks of avian diseases and changes in demand; however, the deregulation of the industry in 1996, along with growing demand for ostrich meat, resulted in a dramatic increase in the number of ostriches. At present, ostrich numbers alone total more than five times the total potential carrying capacity of the Little Karoo (Reyers *et al*. 2009). Ostriches have a significant impact on rangeland vegetation because of their feeding method, trampling and territorial displays leading to soil compaction, the removal of the biological soil crust, and the formation of pathways that channel surface water (Cupido 2005, Esler *et al*. 2006, Le Maitre *et al*. 2007). Ostrich farming had negative and ongoing repercussions for the region's ecosystem services (Reyers *et al*. 2009). In 2005, about 60% of Little Karoo farms were overstocked, and rangeland condition was poorest in ostrich camps and near livestock watering and feeding points (Cupido 2005). Further expansion of the ostrich farming industry in the Little Karoo is likely to pose a greater threat in the future if no mitigating measures are taken (Low and Rebelo 1996, Cupido 2005).

Agricultural practices in the Little Karoo region have shifted away from traditional crop and livestock production to the intensive production of ostriches and their main feed, lucerne (Cupido 2005, Reyers et al. 2009). Therefore, the impact is not just on site, but extends to areas being cultivated for feed (Nongwe 2008). Ostrich farming and associated lucerne production are two of the major causes of vegetation degradation of the Little Karoo.

Overgrazing and trampling in small camps can result in soil compaction or crusting, erosion and loss of soil.

According to Snyman (1999), one of the most important factors to which high rates of soil loss in South Africa can be ascribed, is the degradation of plant cover and composition resulting from overgrazing and/or poor grazing practice. The maintenance of plant cover and veld condition appear to be the two most important factors controlling runoff and soil loss (Snyman 1999). Snyman (1999) cited a wide range of studies which showed that plant cover offers the best protection against surface runoff and soil loss and furthermore, that veld in poor condition has higher runoff, lower infiltration rates and lower effective use of rainfall (production of dry matter/ha/mm) than veld in good condition.

Trampling and hoof action of grazing animals can play an important part in initiating erosion (Snyman 1999). Piospheres are bare tramped-out areas that develop around water-points and other areas of animal concentration. The speed with which such piospheres develop and the size which they attain, are directly related to the number of animals and the size of the camp (Roux and Opperman, 1986, cited in (Snyman 1999). Hoof action of large animals alters the structure of the soil surface by loosening or chipping, by compacting or by deforming, depending on soil type and soil moisture (Tainton et al. 1999). Soil compaction is a form of physical degradation resulting in densification and distortion of the soil where biological

activity, porosity and permeability are reduced, strength is increased, and soil structure partly destroyed. Compaction can reduce water infiltration capacity and increase erosion risk by accelerating run-off. Compaction can be initiated by wheels, tracks, rollers or by the passage of animals (European Commision). Therefore, overgrazing and hoof action can seriously impact on the soil surface depending on the soil type and soil moisture content (Tainton et al. 1999).

Specific research on the impacts of intensive game farming on soil erosion is limited. However, an example from New Zealand showed that intensive deer farming activities have resulted in increased erosion and soil loss (de Klein *et al*. 2003).

Severe grazing patterns may result in an increase of undesirable woody species and some poisonous plant species.

In their review paper on the changes and causes of bush encroachment in southern Africa, O'Connor, Puttick and Hoffman (2014), showed that, among multiple causes of bush encroachment, severe grazing by livestock or wildlife could promote bush encroachment (i.e. increase in undesirable woody species) by reducing the herbaceous fuel load (lower intensity or frequency of fire) or by reducing grass competition.

Heavy grazing can also promote a number of undesirable poisonous plants. Tainton (1972) found that overgrazing in tall grassland led to the establishment of weeds and geophytes, of which *Senecio retrorsus* was prominent and poisoned the animals that grazed in the area. Furthermore the density of species such as *Senecio retrorsus* appears to increase as the severity of grazing increases (Du Toit and Aucamp 1985).

Transformation of natural veld to planted pastures with a homogenised structure and composition.

Wildlife ranching, which includes intensive breeding, is being conducted on a large scale in South Africa with an estimated 9 000 privately owned wildlife properties covering a surface area of 17 million hectares according to Taylor *et al*. (2015) and 20 million hectares according to Cousins, *et al*. (2005). Although positive conservation impacts such as maintaining "natural" or semi-natural habitat and providing resources and land for species introduction programmes are attributed to private wildlife ranches (Taylor *et al*. 2015), there is a number of practices within the ranching industry today that conflict with conservation principles, including; conversion of natural veld to pastures and manipulating natural vegetation such as excessive bush clearing and complete removal of all woody vegetation (Cousins et al. 2010). In game ranching, cultivated pastures are a regular feature, and are considered to ease the grazing pressure on over-utilised veld (van Rooyen 2002), but game farmers are also cautioned against destroying natural vegetation to establish such pastures (Coetzee 2002). For the intensive production of a number of species, for example eland (Du Toit 2005b), nyala (Pfitzer and Kohrs 2005) and buffalo (Du Toit 2005a), irrigated pastures or irrigation to improve forage production is considered essential for the overall success of breeding operations. In terms of ecological principles, the 'habitat heterogeneity hypothesis' assumes that structurally complex habitats may provide more niches and diverse ways of exploiting the environmental resources and thus increase species diversity ((Bazzaz 1975) cited in (Tews et al. 2004)). Conversely, the process of homogenising habitat through, for example, planting pastures, should lead to fewer niches and effectively lower species diversity. Tews *et al*. (2004) review of literature revealed that the majority of studies found a positive correlation between habitat heterogeneity/diversity and animal species diversity. In addition to the direct loss of above-ground habitat as a result of cultivated pastures, Swanepoel *et al*. (2015) pointed out how the mismanagement of intensive cultivated pastures can cause degradation and loss of soil quality. Practices such as continuous tillage, improper grazing management, injudicious application of fertilizers, and poor irrigation management were highlighted as the main causes. According to the concept of ecological resistance more diverse (species rich) plant communities would prevent the establishment of new species such as invasive species into

the already species rich community due to lower resource availability for an invading species (Symstad 2000). Reducing species richness of natural plant communities, by for instance converting to mono-specific pastures, may result in invasive species becoming more successful.

Excessive or complete removal of certain vegetation strata such as the woody component from intensive breeding camps.

Tews *et al*. (2004) also introduced the concept of a "keystone structure ecosystem", using an example of a South African savanna, which illustrated how one predominant vegetation structure creates structural diversity essential for a wide array of species groups. This particularly emphasises the importance of trees in a savanna ecosystem structure and thus the potential loss of species diversity if, for example, there is excessive or complete removal of the woody component in an attempt to improve grazing capacity.

Tews *et al*. (2004) cited the following evidence in support of the example of trees in arid and semi-arid savannas being a keystone structure ecosystem: large solitary trees, scattered in a grassy matrix are focal points for animal activity because they supply nest sites, shade and scarce food resources (Barnes *et al*. 1997, Dean *et al*. 1999). They also provide shade for ungulates resting in the sub-canopy of adult trees (Milton and Dean 1995), nests for arboreal rodents (Eccard and Meyer 2001), perches and nesting sites for raptors, owls and vultures (MacLean 1970), or nest sites for different bird species in the crowns of the trees (Milton and Dean 1995). Faeces, fallen nest material and carcass remains left below trees elevate levels of nutrients available to plants in the otherwise poor soil (Dean *et al*. 1999).

The importance of maintaining heterogeneous habitats rather than homogeneous habitats has also been demonstrated in studies on centipedes, millipedes and scorpions by Druce (2000), and for woodland birds and ants by Cumming *et al*. (1997).

According to Hoekstra *et al.* (2005) habitat conversion exceeds habitat protection by a ratio of 8:1 in temperate grasslands and Mediterranean biomes, and 10:1 in more than 140 ecoregions.

Intensive game breeding often associated with habitat conversion as found in South Africa does not really exist elsewhere in the world. Ludwig, De Kroon and Prins (2008) found that in an east African savanna system, large trees play an important ecological role. They showed that the forage quality of the herbaceous layer is much higher under trees compared to open grassland areas with no trees. Grasses growing under tree canopies contained lower fibre content and the highest concentrations of protein and calcium and had higher digestible organic matter content. Reductions in tree cover could have serious consequences if trees have a positive effect on herbivore food quality and availability. They also concluded that the negative effects of reduced tree numbers initially go unnoticed providing a false sense that it has no negative impact.

Land intensification practices negatively affects ecosystem functioning and services.

Pekin and Pijanowski (2012) state that human land-use activities are creating a biodiversity crisis. It is unclear how species respond to different intensities of land-use at a global scale. Urbanisation and high intensity crop farming cause significant threats to mammal populations worldwide (Foley et al. 2005, Pekin and Pijanowski 2012). Flynn *et al.* (2009) showed that agricultural land-use intensification, can greatly reduce species richness and ecosystem functioning, which potentially has an impact on ecosystem services. The land-use categories used in the study ranged from "natural" (largely unaffected by agricultural activity) to "semi-natural" (largely dominated by natural vegetation but have been modified indirectly for agricultural activities or are directly adjacent to agricultural lands) to "agricultural" (directly managed for agricultural production including row crops and pastures). They found that for birds and mammals, both species richness and functional diversity declined significantly with

land-use intensification, but that the loss of functional diversity was more dramatic than would be predicted by the observed loss of species richness.

A review by Fleischner (1994) showed that livestock grazing can have profound impacts on ecosystem composition, function and structure. The overall ecological costs can include the loss of biodiversity, lowering population densities for a variety of taxa, disruption of ecosystem functions such as nutrient cycling and succession, change in community organisation and change in physical characteristics of habitats. In one of the studies cited in Fleischner (1994), plant community architecture changed from midgrass/tallgrass to shortgrass, which changed grasshopper species composition (Quinn and Walgenbach 1990).

Although this evidence applies to domestic livestock, intensive grazing by indigenous species is likely to have similar impacts. High densities of browsing ungulates are also likely to change the vegetation structure and as such affect biodiversity. A number of studies have shown that native browsing ungulates can have a substantial impact on woody vegetation structural dynamics (Augustine and McNaughton 2004, Scholtz *et al*. 2016).

Extent of evidence

There is a large amount of information relating to general game ranch management and veld management. There are a number of books and many scientific papers dealing with the consequences of overstocking and intensive grazing practices. There is documented evidence of the impact of intensive ostrich farming on veld condition and ecosystem services; however, there are limited studies about intensive game breeding activities. There is no reason why evidence pertaining to effects of heavy stocking cannot be directly extrapolated to this issue, especially since the research from Taylor *et al.* (2015) found that 32% of properties (wildlife ranches) were over the recommended stocking density, while 5% of the total had more than twice the recommended biomass of grazers.

Evidence on the mostly negative effects of homogenisation of habitats is well documented. There are several international papers dealing directly or indirectly with the effects thereof. Most of the evidence exists for extensive areas and often focusses on the functioning of ecosystems. There is virtually no documented evidence on the negative impacts of homogenised habitats perceived to be associated with the intensive game breeding industry in South Africa. Many (if not most) applications for intensive breeding received by conservation authorities specify planting of pastures, planting/removal of trees or other habitat manipulation as part of their management plans (Rushworth, pers. comm.)[5]. South African papers dealing with the private game ranching industry focus more on extensive areas and the contribution of this sector to conservation and sustainable utilisation. The fact that very little scientific evidence exists on the concerns raised, does not mean that there should be no concern. Efforts should be made to test the hypotheses associated with homogenisation of habitats in an intensive breeding camp setup.

Level of agreement

There is a high level of agreement on the impact of overstocking and overgrazing on natural veld. There is general agreement on the impacts associated with the degradation and homogenisation of natural habitats. Anthropogenic influences are usually the cause of habitat conversion. This happens at a global, regional, national and local scale. It affects almost all taxa and can be seen as a threat to biodiversity in general.

The level of agreement at the scale of intensive game breeding projects cannot be determined due to the very limited research that has been done in this field. At this point, it can be argued

[5] Rushworth, I., Manager Ecological Advice West, Ezemvelo KZN Wildlife, 13 February 2018.

that the concerns raised with specific reference to intensive breeding facilities are speculative and needs to be verified through proper research.

Key findings

A large proportion of game ranches in South Africa practise intensive breeding of game where high value species or colour variants are confined to relatively small camps. It is estimated that about 6% (or 1 022 785 ha) of game farms are divided into such camps and it is likely to increase in the future. At present, because the activity is largely unregulated, little is known about the number of animals and camp sizes in which high value animals are kept. Although there appears to have been no monitoring of the impacts in these small camp systems in South Africa, there is well established evidence to show that animals kept at densities higher than ecological capacity have a negative impact on veld condition and productivity, habitat integrity and species diversity. It is not clear to what extent supplemental feeding mitigates the impacts related to high grazing intensity, but this will not mitigate the impacts of trampling and hoof action of animals kept at high densities.

It is anticipated that a similar trend to what has happened in the Little Karoo with ostrich will happen in other biomes in relation to the intensive production of game, with degradation of habitat in game camps (already estimated to cover more than one million hectares with exponential growth forecast), and with additional areas being required over and above those used for traditional agriculture for the production of feed for the rapidly growing captive population.

Some breeders of colour variants keep their animals in more extensive systems, and from a veld management perspective, there should be no negative impacts if this is within the ecological capacity of those environments.

It is highly likely that homogenisation, and therefore degradation of habitats, irrespective of the way in which it occurs, generally will have a negative impact on biodiversity. To date very little research has been conducted in this field at the scale where concerns have been raised but it is likely that sensitive and threatened habitats and non-target species will be adversely affected. Target species farmed with are unlikely to be affected.

Mitigation measures

1. Game farmers must manage densities of game in camps to a level that is ecologically sustainable. A set of best practice norms and standards should be developed to guide stocking rates of camps.
2. Apply land-use planning process to restrict or limit intensive breeding in camp systems in sensitive and threatened ecosystems (Critical Biodiversity Areas).
3. Limit the proportion of a property that can be used (fenced) for intensive breeding.
4. Leave natural vegetation in place (herbaceous and woody).
5. Design camps to include transformed areas with low biodiversity to minimise impact.
6. Design game breeding camps to be as large as possible.
7. Avoid where possible mono-specific stocking of animals.

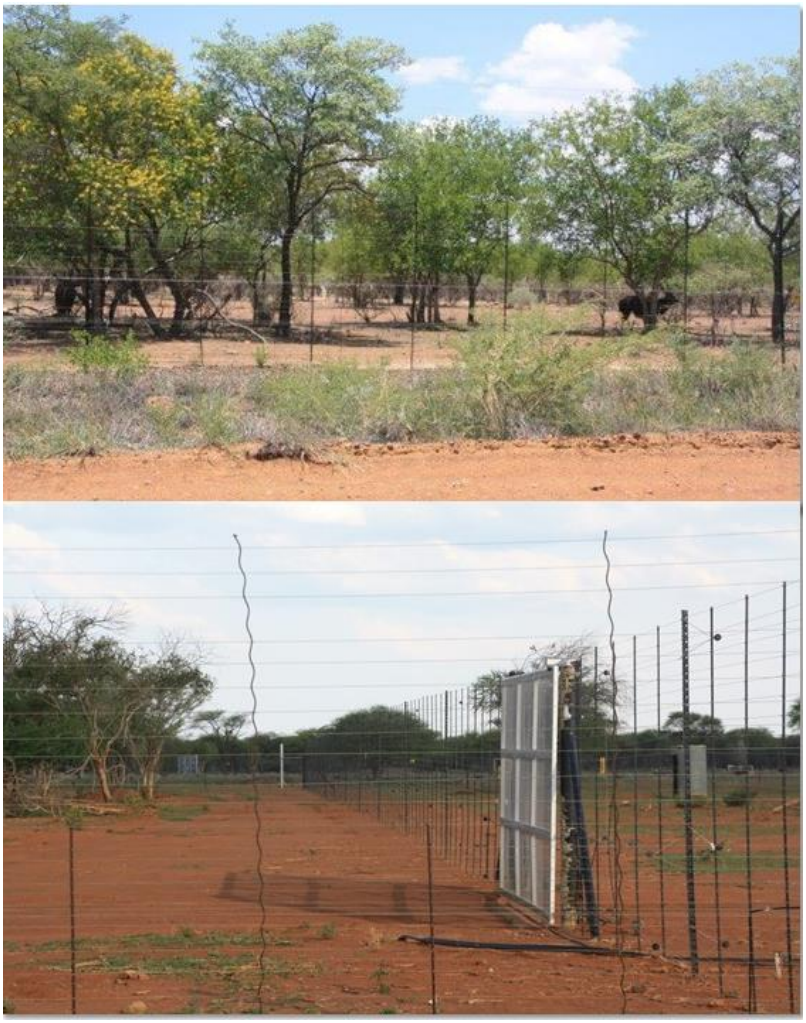


Figure 11: Examples of overgrazed habitat within intensive breeding camps.

ISSUE 4: THE INTENSIFICATION OF MANAGEMENT PRACTICES AND SUBSEQUENT CONTROL OF SPECIES THAT ARE LIKELY TO IMPACT NEGATIVELY ON THE COMMERCIAL OBJECTIVE OF BREEDING PROGRAMMES

IMPACT STATEMENT 1: THE KILLING OF PREDATORS AND OTHER CONFLICT SPECIES MAY RESULT IN A REDUCTION IN POPULATION NUMBERS OR ELIMINATION FROM CERTAIN AREAS WITH LIMITED OPPORTUNITY FOR RECOLONISATION, WHICH IN TURN MAY LEAD TO A CHANGE IN THE CONSERVATION STATUS OF THE SPECIES AND THEREBY FURTHERING THE EXTINCTION RISK OF THESE SPECIES.

Key concerns

1. The high economic value of species bred in intensive systems is likely to perpetuate the problem of indiscriminate killing of perceived damage-causing animals.
2. Predators are not the only species deemed to increase human-wildlife conflict within the game ranching sector. Species that may be the cause of increased management interventions and potentially lower profit margins may also be targeted through systematic control and removal.

3. The population sizes of large apex predators are naturally small and as such the indiscriminate and uncontrolled killing of these species is likely to result in population declines, thereby furthering the extinction risk of the species.
4. Declining apex predator populations limit sustainable utilisation opportunities and thereby reduce the incentives for landowners to conserve such predators.
5. Removal of top-order predators from systems can have effects at an ecosystem level, cause trophic cascades and meso-carnivore release. This in turn can cause increased human-carnivore conflict and increased losses, in particular to sheep and goat farmers that are more vulnerable to meso-carnivore pressure.

Evidence

Large predators that occur within the agricultural landscape not only prey upon domestic livestock but are likely to prey on game species. This can lead to potential conflict with humans and the resulting persecution can lead to a decline in such species (Thorn et al. 2012, Lindsey et al. 2013). Globally, conflict related killing is the leading threat to large carnivores. This can reduce the benefits of game ranching to conservation through the protection of biodiversity and associated habitats (Lindsey et al. 2009).

Large carnivores are particularly vulnerable to persecution-related killing due to their large home ranges that cross multiple land-uses, low reproductive rates, low population densities and their carnivorous diet that results in predation of animals with high economic or recreational value (St John et al. 2011). The conservation of predators outside formal protected areas receives limited support from the agricultural community (Boast 2014), as a result human-mediated deaths are high and are a primary threat to the survival of large carnivores (Swanepoel et al. 2014).

The presence of commercial game species represents an important agricultural asset, which generates substantial financial revenue and return for landowners (Pitman et al. 2016). A consequence of this is that landowners/ranchers are unlikely to tolerate wildlife, especially free-ranging predators that pose a threat to these assets. This intolerance is further compounded by an unstable socio-economic climate that affects revenue generation (Van der Merwe et al. 2004). The financial implications stemming from the loss of game animals as a result of predation is predicted to be greater than that of the loss of domestic livestock killed in a similar manner (Boast 2014). The Botswana Wildlife Producers Association (BWPA) has estimated that 50% of the game stock calf crop will be lost to predation in its first year (since calving) if predators are present in the landscape, a scenario which may be considered financially unsustainable, especially for emerging landowners (BWPA 2005 cited in Boast 2014). The presence of predators does not only equate to direct losses of animals. The potential loss of future offspring and subsequent financial losses when breeding individuals are killed or the expenses related to the erection of 'predator proof' fencing and similar mitigation measures all contribute to the indirect costs of predator prevalence. It is suggested that the indirect financial costs associated with predator interactions can be equal to or more than the direct costs resulting from predator interactions (Boast 2014).

The trend within the game ranching sector to shift from high numbers of common game species requiring very little management intervention to the breeding of high value game species has resulted in increased management actions (Lindsey et al. 2009). This intensification to ensure increased profitability has resulted in an increase in predator proof fencing and the lethal control of nuisance wildlife through legal and possibly illegal destruction (Cousins et al. 2008, Taylor 2016). Pitman *et al*. (2016) suggest that the increase in lethal control is disproportionate to the increase in intensive game farming methods, thereby suggesting that intolerance of predators is increasing. This is likely to have significant and detrimental impacts on species persistence and the functioning of ecological systems (Ripple et al. 2014). Schroder and Reilly (2013) indicates that on a property undertaking semi-extensive breeding of roan antelope, all predators are excluded from breeding camps. The

article does not indicate the manner in which this exclusion is undertaken. Bradley and Pletscher (2005) determined that various factors contribute to depredation on livestock in a study in Montana and Idaho, United States of America. These factors included the size of the property, distance to human habitation and presence of other game species. Combined with these factors, the increase in vegetation cover resulted in an increase in predator depredations, a notion supported by Thorn *et al*. (2012). Thus landowners would be required to assess their properties holistically in order to address predator conflict and not focus on either lethal or translocation control only.

St John *et al.* (2011) indicate that landowners (farmers) are likely to persecute predators to minimise actual or perceived losses. This notion is supported by Thorn *et al.*, (2012) who conclude that human-wildlife conflict is driven more by social and environmental factors i.e. abundance of a particular predator rather than the actual damage caused by a predator or the actual economic losses. It is suggested that 19% of the farmers surveyed within the St John *et al.*, (2011) study area had killed leopard on their particular properties within the preceding 12 months and a similar percentage of respondents killed carnivores (pesticides) without the required permits. Six out of 12 cheetahs which were monitored in a collaring project in Thabazimbi, Limpopo, were shot without any permit authorisation (Marnewick and Somers 2015). Furthermore, many disregarded the restrictions that apply to the use of stock remedies. The deliberate misuse of stock remedies may also impact upon other non-target species such as large raptors, more notably vulture populations (see chapter 10). This is evident in the recent poisoning of baboons and white-backed vultures (*Gyps coprotheres*) by a landowner in Limpopo Province (Phillips 2015).

Pitman *et al.* (2016) demonstrated that legal mortalities of leopard in the Limpopo province are not sustainable, an assertion confirmed through camera-trap surveys conducted during and after the study period which indicated population declines. The Thorn *et al.*, (2013) study highlights the extent of carnivore persecution in Limpopo. Forty-two farmers in the Waterberg, Limpopo reported that they killed a total of 303 carnivores in the preceding year. Annual species persecution extrapolated over the whole study area equates to the killing of 1 096 black-backed jackal (*Canis mesomelas*), 129 brown hyenas (*Hyaena brunnea*) (8% of the SA population), 77 leopards (9% of the Waterberg/Mpumalanga population), 67 caracal (*Caracal caracal*) and 72 African wild dogs (72% of the free roaming population or 16% of the national population).

The perceptions of landowners are a key driver of predator control and possibly the level of tolerance to predator presence is unsustainable. Thorn *et al.*, (2013) indicated that the species most widely blamed for predation are not the same ones that are least tolerated. “Only 23% of respondents said that they would tolerate African wild dogs if they were regularly present at their farm, compared with 58% for cheetahs, 73% for leopards, 78% for jackals, 79% for caracals, and 92% for brown hyenas. Of 78 people who answered the question, 42% said that they would cross the threshold from tolerance of carnivores to intolerance if ≤1% of their stock were preyed upon. Another 23% reached that point after losing between 1 and 5% of their stock and 17% gave a tolerance threshold of between 5 and 35% of stock holdings. A further 18% felt that predation losses were a natural and acceptable hazard no matter how many animals were preyed upon. The median percent loss that farmers with high value stud or game animals were willing to tolerate was 83% lower than famers without high value animals (W = 2103.5, P\0.001, n = 78) and two thirds of respondents who said they were willing to tolerate losing ≤1% of their stock were farming with high value animals.” The intolerance towards predators is thus compounded by the presence of high value game species.

It is not just predators that are persecuted as a result of increased predator intolerance. Since 2008, there has been an increase in the number of permit applications to control problem elephants in the Limpopo province. This coincides with the increase in predator proof fencing. Pitman *et al.*, (2016) suggest that the increase in predator proof fencing has not only failed to mitigate conflict with problem predators, but also resulted in a decreased tolerance for

elephant. Elephant, lion and leopard are considered the most desirable for non-consumptive tourism yet are the top three species targeted as supposed problem animals (Di Minin et al. 2013a, Maciejewski and Kerley 2014).

Extent of evidence

A number of scientific publications, both locally and internationally (Bradley and Pletscher 2005, Lindsey et al. 2005, St John et al. 2011, Thorn et al. 2012, Thorn et al. 2013, Marnewick and Somers 2015, Pitman et al. 2016), address the impact of predator management on specific predator species. There is sufficient evidence available to understand the implications that intensified predator management will have on predator populations within South Africa.

Level of agreement

Within the scientific literature, there is agreement that human-wildlife conflict is increasing within the non-protected area landscape (Bergman et al. 2013, Pitman et al. 2016). The transition from traditional agriculture to intensive game ranching, coupled with the instability of the socio-economic climate has resulted in an increasing intolerance towards predators and thus increased persecution.

Key findings

It is well-established and virtually certain that the destruction of predators and other 'problem animals' is unlikely to decrease within the current economic climate and high value of intensively and selectively bred game. The return on investment in intensive breeding and breeding for colour variations is still such that entrants into the market will continue to invest and thereby ensure that predators or other species likely to impact upon animals considered as an investment, are removed from the system. Bothma (2005) cited in Boast (2014) further suggests that the low breeding rates and decreased survival of colour phenotypes may result in blame being attributed to carnivore damage and thus increased intolerance towards these species. It is well-established that the intensive management and control of predators is occurring within the wildlife industry and will continue as long as these activities are undertaken. However, the impact on the target species will be evident long after the activities are no longer considered viable. It is further likely that these actions will occur throughout the areas in which these activities are undertaken. The removal of predators or considerable reduction in numbers will probably have an impact upon the conservation status of these species. However, it is the responsibility of both conservation agencies and landowners alike to ensure that this is avoided. Pitman *et al*. (2016) indicates that the current leopard population in the Limpopo province is declining as a result of various factors – illegal offtake (snaring, poisoning and illegal hunting), hunting and those removed as damage causing animals. It is not feasible to conserve large predators only within the confines of formal protected areas and thus landowners within the broader agricultural landscape need to become custodians of such species rather than adversaries.

Mitigation measures

1. A concerted effort is required by conservation agencies to highlight the plight of predators and the importance of their existence within the landscape.
2. Landowners should be made aware of the possible options to prevent losses: fertility control, appropriate fencing, habitat modification (although this may be controversial and impact on other elements of biodiversity conservation), use of repellents, guard dogs, and aversion conditioning. These options depend on the species being targeted and should be managed accordingly within the broader landscape.
3. The perception that predators present a direct impact upon the economic viability of game farming enterprises, despite Thorn *et al.* (2013) concluding that losses as a result of predation are limited, would suggest that any facility/property undertaking intensive

or selective breeding should ideally be located outside areas in which large apex predators occur. The placement of such facilities should be planned accordingly prior to authorisation.

4. A holistic approach to land-use management is required to address predator conflict and management and may be best managed through an approved management plan for the property, wherein such management plans are viewed across the entire landscape and not just per cadastre property.
5. Enforcement of the illegal persecution of predators needs to be improved but requires the cooperation of both conservation agencies and private landowners. Private landowners are to be encouraged to report illegal activities, while conservation agencies are to ensure that such activities are investigated thoroughly.
6. There is no proverbial silver bullet in addressing predator conflict and avoidance. It is recommended that the suitable regulatory measures are implemented to ensure the mitigation measures mentioned above are incorporated into an approved management plan. In addition, a collective dialogue between the wildlife industry and conservationists to investigate incentives for predator occupancy needs to be undertaken, where each other's contribution is duly acknowledged.

IMPACT STATEMENT 2: THE DISRUPTION OF SOCIAL STRUCTURES OF SPECIES TARGETED FOR REMOVAL MAY EXACERBATE THE CONFLICT POTENTIAL, AS A RESULT OF THE CONSTANT REMOVALS OF INDIVIDUALS AS WELL AS WITHIN THE RECEIVING ENVIRONMENTS[6]. THIS IN TURN COULD LEAD TO A DECLINE IN THE SURVIVAL RATE OF THE AFFECTED POPULATION. CONSTANT REMOVALS OF DISPERSERS MAY FURTHER LEAD TO A LOSS OR DISRUPTION OF DISPERSAL OPPORTUNITIES, THEREBY INCREASING LOCAL EXTINCTION RISK. RELOCATIONS MAY INCREASE CONFLICT AND REDUCE REPRODUCTIVE PERFORMANCE AND INCREASE MORTALITIES.

Key concerns

1. The constant removal of predators has the potential to disrupt social hierarchies of other predator species. This in turn may exacerbate conflict leading to further declines of predator populations.
2. The survival of removed problem predators may be low in the receiving environment and may result in increased intraspecific competition as well as increased losses in the receiving environment with the decreasing availability of safe areas for relocations.
3. The constant removal of individual predators from within a particular area may result in the disruption of a species' social behaviour over a broader area.
4. The continued translocation of problem animals, either predator or other species impacting upon an intensive breeding operation, may disrupt the ecological processes within the receiving environment. Existing populations within the receiving environment may be forced to utilise more resources defending a particular territory than it would have prior to such introductions. This increased resource demand may impact upon breeding success.

Evidence

The removal of large carnivores has been associated with meso-predator release (Prugh et al. 2009). Boast (2014) suggests that the result of the removal of lions and spotted hyenas (*Crocuta crocuta*) from systems has led to an increase in the densities of cheetah populations to unnatural levels. Furthermore, the removal of cheetah and leopard from a system results in increased black-backed jackal and caracal populations that in turn become the top predators

[6] The receiving environment is the area into which translocated predators are released.

(Klare et al. 2010) cited in (Boast 2014)). Where these species are persecuted the numbers of bat-eared foxes (*Otocyon megalotis*), common genets (*Genetta spp.*) and cape foxes (*Vulpes chama*) also increase (Blaum et al. 2009).

Translocated problem predators demonstrate low post release survival rates (Weilenmann et al. 2010) in addition to translocations often being ineffective at reducing losses and have the potential to increases losses within the receiving environment (Massei et al. 2010, Weilenmann et al. 2010, Boast 2014). Bradley *et al.* (2005) found that more than 25% of translocated problem wolves (*Canis lupus*) continued to prey on livestock after release and thus created additional conflicts in the areas of release. The removal of any predator prior to the identification of whether it is responsible for actual mortalities will ultimately have little impact upon actual losses (Massei et al. 2010). In fact, losses may increase as the predators from surrounding territories move in to fill vacated spaces, thereby creating sink zones within the landscape. This will have an impact upon the species' social dynamics over a much larger area and such removals have been known to reduce predator populations within adjoining protected areas which are often source zones for these sink areas. Boast (2014) further states that the removal of individuals whose social structures are a key component of their hunting strategies, i.e. African wild dog, lions or male cheetah coalitions can reduce the hunting success of the remaining groups.

The removal of territorial individuals also results in the increase in the number of sub-adult or transient individuals within an area. This may result in an increased conflict potential and subsequent losses resulting in an increase in the removal of predators (Boast 2014). Problem cheetahs often exhibit physical or behavioural abnormalities and thus are weaker and less dominant than other predators and thus survival in the receiving environment is low.

Translocated individuals also have far greater home ranges as a result of their attempts to return to their natal area. This leads to greater post-release movements. Homing has been recorded in a number of species translocated to reduce conflict, of which many returned to their natal areas within a few weeks (Smallidge et al. 2008). This includes wolves in Minnesota that had to be moved more than 70 km to ensure that they did not return to their capture area. Two leopards relocated into Etosha National Park immediately returned to the capture area and begun to kill livestock again, while an individual relocated 60 km returned to the capture site and commenced killing livestock. Eight lions relocated in Zimbabwe produced varying results: two moved 27 km from the capture site, returned and resumed killing livestock within five months. Six individuals relocated 45 km did not return, either staying within the reserve (two) or never being observed again (4) (Linnel et al. 1997). Conversely, some species do not exhibit such homing tendencies, with movements of translocated racoons (*Procyon lotor*) being limited to a few weeks post release (Massei et al. 2010). Movement may be reduced as a result of the presence of offspring.

The lethal control of territorial males of species that practice infanticide (leopard and lion) may result in such spaces being occupied by individuals who are likely to kill the previous male's offspring (Balme and Hunter 2013). This will in turn affect the predator population on a broader scale (Boast 2014). However, solitary predator species are more susceptible to infanticide when compared to social species because they cannot rely on a cooperative defence mechanism (Balme and Hunter 2013).

Fatal territorial fights have been recorded during reintroductions between male cheetah both within fenced ranches in South Africa (Hofmeyr and Van Dyk 1998, Bissett and Bernard 2007) and in the Serengeti (Caro 1994). Territorial fights thus contribute to the poor survival rates of translocated males either through direct interaction or the species exhibiting increased ranging behaviour to avoid conflict and thereby exposing itself to increased mortality threats (Boast 2014).

The survival of translocated individuals increases when competition within the receiving environment is low which would suggest that competition is a limiting factor in post translocation success (Massei et al. 2010).

Extent of evidence

There are a numerous publications, both locally and internationally, including peer-reviewed scientific publications and popular articles that address the impact of predator removal (Bradley et al. 2005, Weilenmann et al. 2010, Boast 2014). There is sufficient evidence available to understand the implications that intensified predator removal will have on predator populations within South Africa. Additionally, in South Africa there are limited opportunities for the relocation of predators. Invariably fenced reserves, which are willing to accept predators, are all saturated and moving into other existing ranges present the problems as discussed above.

Level of agreement

There is consensus that once large predators are removed from the system, meso-predator release occurs that could lead to increased conflict (Prugh et al. 2009). Furthermore, the disruption of predator populations as a result of removals are well documented.

Despite the limited literature reviewed, there is agreement that the translocation of predators does have an impact on both the receiving and donating environment if not managed correctly.

Key findings

The negative attitude towards predators within the agricultural landscape has resulted in previous predator removals and translocations being undertaken as a form of control, a scenario that still occurs with well-established evidence to support this impact. These 'control' options of predators from within areas in which intensive game ranching takes place is unlikely to decrease and will probably increase (Pitman et al. 2016) considering that predator management has been identified as a key intervention towards ensuring the return on investment. Thus, the occurrence within the industry is virtually certain. The impact on broader biodiversity is likely as predator removal may create an unnatural predator hierarchy. The mesopredator release theory is well established and the probability of increased conflict between these predators and landowners is high. The impact of predator removal on the species and individuals is highly likely because a new competitive environment is created.

Translocation of problem predators may not always be the most desired option. However, public perception is often the deciding factor. If problem predators are to be translocated, success should be measured by resolving the problem that the animal caused and not necessarily the survival of the animal post release, settlement in the recipient area and reproductive output (Massei et al. 2010) as success may depend on the age and sex of the individual, which are factors that are often outside the control of mitigation relocations. Post release monitoring at the release and capture sites will assist in determining whether the problem has been resolved; if additional problems have been created; and ensuring that the same or additional problem has not been created in the release area. It is important to implement suitable management actions to ensure that the translocation of predators is not detrimental to both environments. However, it is unlikely that private landowners will undertake translocations because they might prefer to initiate lethal control. Therefore, the impact of such will last as long as such actions are not coordinated correctly. The potential for conservation agencies and NGOs to manage the translocation of predators and associated management actions, will rely on the necessary funding, the source of which is not always known or sustainable. Lindsey *et al.* (2005) compared the cost of maintaining wild dog populations on ranchland in relation to establishing a reintroduced population. Although the costs for ranchland can be offset against its eco-tourism value, reintroduction costs are often

prohibitive. Thus, without the necessary funding, translocations will not take place, or the impact thereof will not be managed. The impact of inappropriate predator translocations and/or lethal removals may only manifest themselves long after the event and possibly when further interventions are no longer possible.

Potential mitigation measures

1. It is suggested to undertake research into the impact of predator removal and subsequent mesopredator release within the wildlife ranching landscape.
2. This is concurrent to an intensive landowner awareness campaign to highlight the impact of predator removal and mesopredator management.
3. Further investigation is required into current methods to manage small predators within the game farming landscape. Currently, initiatives are being implemented to deploy guard dogs within the agricultural landscape to assist in lowering stock losses. The viability of such, and other possible non-lethal control methods, in the game stock industry must be assessed.
4. The translocation of any animal should only be undertaken with the necessary permits and to dedicated release sites. Such sites should be pre-approved for relocation to limit any disruptions to natural processes.
5. It is suggested that each provincial agency maintains a database of suitable release areas.
6. Considering the need to understand the effect of predator removal within the wildlife ranching landscape, it is recommended that the criteria required to validate an area as a suitable release site be compiled and distributed to all provincial conservation agencies.

IMPACT STATEMENT 3: THE REMOVAL OF PREDATORS WILL AT A CERTAIN SCALE DISRUPT PREDATION AS A NATURAL PROCESS IN THE BROADER LANDSCAPE/ENVIRONMENT THEREBY AFFECTING ECOSYSTEM FUNCTIONING AND NON-TARGET SPECIES

Key concerns

1. The disruption of natural predation interactions, through the exclusion of apex-predators from the agricultural landscape and the resultant increase of smaller to medium-sized predators. This in turn may lead to a disproportionate impact upon smaller prey species and potentially increasing human-wildlife conflict.
2. The removal of predators from a system may also impact on the prey's inherent fear of predation over time and reduce their ability to avoid predators.
3. The removal of key species may disrupt certain ecological processes that assist in maintaining equilibrium in the system. This in turn leads to an increased management intervention that places further strain on the system. These increased interventions will impact on non-target species.
4. Ecological processes take place within a broader landscape context and affect more than the cadastral boundary of any breeding operation. Thus, increased management intensity at one site will have cascading ecological effects on the broader landscape.
5. Historically, areas where large livestock (predominantly cattle) production was maintained, contributed to predator conservation objectives, yet the increase of intensive and selective breeding operations occurs in the same areas, thereby reducing the ability to contribute to predator conservation as a result of changing management actions.

Evidence

Several examples of the impact of predator removal have been documented for North America. The removal of wolf populations from the Rocky Mountains resulted in a general degradation of riparian habitat and a decrease in neo-tropical migrant songbirds. Similar impacts have been recorded for mountain lions (*Puma concolor*) in Zion National Park (Ripple and Larsen 2000, Ripple and Beschta 2006). Predators play a fundamental role in ecosystems and their removal has been linked to both herbivore and mesopredator release. This in turn disrupts food webs, herbivore and vegetation communities that have far-reaching effects on ecosystems (Estes et al. 2011). The increase in herbivore abundance may result in a reduction of available food resources and the need for supplementary feeding. Furthermore, the loss of predators within an ecosystem allows for the successful establishment of invasive and undesirable species (Wallach et al. 2010). In the absence of predators, native herbivores display the tendency of invasive herbivores resulting in almost identical degradation of the environment (Winnie. Jr and Creel 2016).

The removal of predators may also impact upon non-target individuals or species. The inappropriate use of stock remedies or snares can result in the death of other species (vultures) and innocent species such as aardwolf (*Proteles cristatus*) and bat-eared foxes that are often mistakenly associated with livestock losses. Furthermore, predators are an integral part of ecosystem functioning and the removal of these species has been linked to herbivore and mesopredator release. This in turn disrupts food webs, herbivore communities and vegetation resulting in a much broader impact upon the ecosystem (Boast 2014). Predators are able to limit the densities of non-migratory species. This in turn reduces the foraging impact on plant species, thereby maintaining vegetative biodiversity that can buffer the impact of drought and climate change on herbivorous prey populations. The decline in predator populations has been linked to increased intraguild competition and aggression at lower trophic levels, resulting in one particular species or group of species becoming more dominant or displacing others (Boast 2014). Studies in the western United States of America indicate that recruitment of deciduous trees is ongoing with the presence of large predators, however a dramatic reduction was recorded within protected areas (parks) following the extirpation or displacement of large predators. This reduction occurred irrespective of management attempts to reduce ungulate populations or climatic fluctuations (Beschta and Ripple 2009). It is further reported that other studies in western North America demonstrate that the loss of large predators precedes a significant plant community change.

Populations of olive baboons (*Papio anubis*) in West Africa have increased as a result of the decrease of apex predators. This increase in turn resulted in the decrease in small ungulates and other primates as well as increasing the potential for human-wildlife conflict in that baboons pose the biggest risk to livestock and crops. Furthermore, children are removed from school to guard planted crops from marauding troops of baboons ((Brashares et al. 2010) cited in (Ripple et al. 2014)). Thorn *et al.* (2012) suggest that the increase in abundance and distribution of middle-ranked predators is a result of the decline in density of the larger, socially dominant predators which are often only resident in formally protected reserves. This presence of middle-sized predators results in a preference of small to medium sized prey, often avoiding adult sized animals.

Predation has a strong effect on the selection pressure on most ungulate behaviour, physiology and life history. This ability to replicate such pressure through human harvesting is under debate as the selectivity of human harvesting and natural predation by carnivores are likely to differ (Mysterud 2010). Thus, the removal of predators from not only intensive and selective breeding facilities but also from the broader landscape will impact on the remaining ungulate population. Predators whose hunting strategies involve chasing, tend to target sick or injured prey (Pole et al. 2004). This preference allows for the removal of weaker individuals and may improve the species' genetics and limits disease transmission in prey populations. Therefore, the removal of predators may result in prey populations being exposed to disease,

genetic outbreeding and increased competition (Packer et al. 2003, Pole et al. 2004, Hayward 2006). Studies have begun to demonstrate the deleterious impacts that mesopredators inflict on prey species following the removal of dominant predators. Mesopredator release can itself result in counterintuitive effects – modelled impacts demonstrate that the removal of cats (dominant predator) from an island, instead of allowing for the increase in the resident seabird colony, actually leads to their decline as a result of increase predation by rats (Courchamp et al. 1999, Sutherland et al. 2010). Mesopredator release has also been reported for non-mammalian predators such as reptiles as a result of mammalian carnivore management.

To conclude, Terborgh *et al.* (2006) captures the state succinctly - the presence of a viable predator association is essential in maintaining biodiversity within the landscape. Ripple *et al.* (2014) the impact of predator population declines across the globe and the impact of such on ecosystem functioning. Whilst certain species benefit as a result of predator decline, other species also decline as a result.

Extent of evidence

Publications which are available (Beschta and Ripple 2009, Thorn et al. 2012) to provide evidence are varied but include peer-reviewed scientific publications, popular articles and webpages. There is sufficient evidence available to discuss the perceived impact statement, but such literature is limiting when evaluating the long-term impacts of predator removal, especially within the South African context. International examples of trophic cascades are well documented but limited for southern Africa.

Level of agreement

The mesopredator release theory is well documented throughout the literature, yet there is no consensus on the effect it has on the ecosystem as both positive and negative impacts have been recorded. Furthermore, the impact may only be detectable at a much later stage. The majority of articles reviewed support the theory that the removal of apex predators from the system results in mesopredator release and subsequent impact. Furthermore, such literature agrees on the impact of predator removal and subsequent trophic release. What is not quantifiable is the extent to which this occurs in southern Africa and more specifically in areas where game ranching is the predominant land-use.

Key findings

The evidence is well established to support this impact, yet the extent to which the removal of predators impact on ecosystem functioning in the South African context requires further investigation, but it is highly likely that removals will impact biodiversity in general. The removal of predators from the system is virtually certain to occur as highlighted by Pitman *et al.* (2016). The removal of individual predators is unlikely to have an impact, but it is reasonable to suggest that the impact on the species will be present for as long as predators are not tolerated and a holistic approach to predator management is not followed. The result of the removals of predators from the environment will result in further conservation interventions being required to ensure that population declines are halted – the moratorium of leopard hunting in 2016 and 2017 being an example of such. This in turn has the potential to further increase the landowners' intolerance towards the remaining predator populations. The ability of predators to assist in the removal of weak and or injured animals from a system can be considered beneficial to landowners that operate within extensive systems, yet the benefits thereof (such removals result in little or no impact upon healthy individuals) are often not promoted. The impact of predator removal, either apex or mesopredator, and trophic release is a key research question that should be undertaken as a matter of urgency. It is most probable that such is occurring within game ranching areas, but the extent is not quantifiable. The potential risk for ecosystem health and functioning is high with possible long-term consequences for not only conservation but also the economic viability of agriculture.

Potential mitigation measures

1. It is suggested that a national predator monitoring initiative is implemented in order to understand the current population dynamics of predator populations (more specifically those that impact upon commercial agriculture and wildlife ranching). The intensity of leopard monitoring as a result of the moratorium on hunting in 2016 and 2017 has increased, which bodes well for the population and impacts thereon.
2. In addition, an awareness programme should be initiated to highlight the impact of predator removal on ecosystem functioning in areas where wildlife ranching increases.
3. Increased law enforcement to reduce illegal hunting of predators.
4. Creation of an incentive-based conservation programme to encourage landowners to conserve predators.
5. Furthermore, a national assessment of the spatial locations and timing of the onset of intensive and selective breeding facilities must be implemented concurrently with increased law enforcement and incentive-based opportunities to assess the impact of the change in management actions/interventions – the move from large stock/extensive systems to smaller more intensively managed areas, similar to small stock farming areas.

ISSUE 5: IMPROPER USE OF STOCK REMEDIES (ANIMAL HEALTH PRODUCTS) AND VETERINARY MEDICINES RESULTING IN THE DEVELOPMENT OF BOTH RESISTANCE TO THESE PRODUCTS BY PARASITES AND TO THE LOSS OF DISEASE AND PARASITE RESISTANCE IN HOST POPULATIONS

IMPACT STATEMENT 1: DEVELOPMENT OF RESISTANCE TO STOCK REMEDIES AND VETERINARY MEDICINES RESULTING IN MICROBES, HELMINTHS AND ECTOPARASITES THAT MAY START INFESTING FREE-ROAMING GAME AND LIVESTOCK ON A LARGE SCALE WITH CONSERVATION AND ECONOMIC CONSEQUENCES.

Concerns

1. Resistance to stock remedies, such as anthelmintics, ectoparasiticides and anti-microbials is of global concern. The liberal and improper use of these commodities in the intensive breeding industry (by both game breeders and veterinarians) is likely to cause a rapid genetic shift to resistance in worm, ectoparasite and microbe populations. Should these resistant populations spill over to livestock the economic consequences for the livestock industry are likely to be significant. There could be a regulatory response leading to increased restrictions on the movement of animals which would affect both the intensive and extensive wildlife farming/ranching industry, and which may therefore have negative impacts on the conservation contribution of this industry.
2. As one of the objectives of intensive and selective breeding is to produce animals for the hunting industry this necessitates releasing intensively-bred animals, with high probability of containing resistant parasites, into wild conditions with other wild indigenous species. It is likely that the resistant strains would be transferred to wild animal populations and it is possible that these animals may be more impacted by resistant worm, ectoparasite and/or microbes as they have not evolved with such organisms. These wild populations may also directly or indirectly transfer resistant parasites to livestock.

Evidence

Introduction

The World Health Organization (1965) developed a definition of resistance in broad terms as "the ability of a parasite strain to survive and/or to multiply despite the administration and absorption of a drug given in doses equal to or higher than those usually recommended, but within the limits of tolerance of the subject". Resistance develops because some individual parasites have an innate ability to withstand a given drug and this ability is heritable.

There is a direct relationship between concentration of the drug and degree of resistance. A strain controlled by one dose of a drug may develop resistance when a lower concentration of the same drug is administered (Mitchell 1996). This may allow for the selection of initially resistant mutants to lower levels of the drug (Faza et al. 2013). Continued exposure to an acaricide (anti-tick treatment) results in removal of the susceptible members of the population with a concomitant increase in the proportion of the resistant strains, i.e. a process of selection for resistance (Abbasa et al. 2014). High frequency of acaricide application has been shown to be a positive risk factor for the emergence of resistant strains of ticks (Roush 1993, George et al. 2004, Abbasa et al. 2014).

Some cattle breeds carry fewer ticks than others under the same environmental management; such natural resistance is because of animals' abilities to respond immunologically to tick infestation (Roberts 1968). However, parasites which induce effective immunity in their hosts will be under weaker selection pressure for resistance because immunity affects parasites irrespective of drug-resistance status. This reduces the chance of resistant parasites surviving and reproducing (Abbasa et al. 2014). It has been shown that wildlife translocation can inadvertently introduce foreign parasites into new habitats (e.g. (Fernandez-de-Mera et al. 2003).

Evidence in southern Africa

A recognised difference between intensive livestock production and intensive game production is the difficulty to apply individual and calibrated doses of anthelmintics, ectoparasiticides and anti-microbials in the latter. The evidence about the application of uncalibrated doses lies in popular game ranching magazines where adverts abound for automatic dipping apparatus for ectoparasiticides and articles that advocate the prophylactic treatment of game against helminths, ectoparasites and microbes. From the extent of advertising, plus observations by conservation officials in the field, it appears that there is likely to be large scale use of unregistered products and off-label use of registered products taking place, but the extent has not been quantified. It is clear that the ad hoc use of a 'medicated' lick block to control internal parasites will not only have a minimal chance of success, but can also easily exacerbate the parasite problem (see http://wildboere.com/game-industry/animal-diseases/internal-parasites-and-game/).

There are few scientific articles that have investigated the development of resistance within the game ranching industry, but a number of parasite and disease veterinary specialists have raised grave concerns about the liberal use of ecto- and endoparasites and anti-microbials (Gerhard Verdoorn, pers comm.)[7].

In one of the few published accounts in the region, Schroder & Reilly (2013) report on a survey undertaken to determine the species composition and abundance of ticks in an intensive roan breeding camp where ticks were controlled versus a control-free area on the same game ranch. In the intensive breeding camps an acaricide was applied using two application methods simultaneously: feed bins with an extended outer rim holding the acaricide (this was undertaken with every feed) and the tick-off pressure plate system at the entrance to the waterhole. While numbers of most tick species were reduced in the intensive breeding camp

[7] Verdoorn, G., President Griffon Poison Information Centre, Email communication, 24 February 2016

where acaricide was used, the numbers of *Rhipicephalus (Boophilus) decoloratus* were significantly (45%) higher. The authors concluded that the reason for the numbers being so high in the controlled area was bad management of acaricide treatment, resulting in this tick species becoming resistant to the acaricide treatment. The reduction in numbers, but not elimination, of seven other species of ticks may well indicate that strong selective pressures for resistance are being applied. The type of acaricide application method being applied on the property in this study is being used extensively, both on intensive game breeding camps and semi-extensive systems. In another study Schroder *et al*. (2006) recommended that acaricides should not be used continuously on intensive breeding farms due to the potential for development of resistance.

Severe wireworm infestations in sable in small camps in Machado have been reported (Drs. Harris, Klopper and Jacobs in Livestock Health and Production Group of the South African Veterinary Association, 2014). Veterinarians are "of the opinion that it is very possible that in the near future too much resistance to the available products for deworming may develop which will prohibit effective parasite control" (http://wildboere.com/game-industry/animal-diseases/internal-parasites-and-game/).

Evidence from elsewhere in the world

Game animals in highly diverse and small reserves may suffer from higher than normal rates of parasite infection, underscoring the importance of considering parasite transmission dynamics in the management of small, fenced areas, such as intensive breeding camps (Ezenwa 2004).

Babesiosis, often a fatal disease of cattle endemic to Mexico, for which U.S. cattle have no resistance, is transmitted by a tick *Rhipicephalus* (*Boophilus) microplus*. Mexican strains of *Rhipicephalus* (*Boophilus) microplus* have become highly resistant to acaricides. Eradication of a future widespread outbreak of babesiosis in the USA could be greatly hampered by this acaricide resistance, rapid transportation of tick-infested animals, and the presence of increased wildlife populations potentially capable of serving as alternative hosts, resulting in "a difficult and prolonged, if not impossible, eradication effort" (George et al. 2002, Temeyer et al. 2004). Similarly, the wildlife ranching industry in South Africa could be contributing to the development of resistance (through inappropriate use of anthelmintic and ectoparasiticide treatments), rapid transfer of resistant parasites (through game sales and translocations), and the creation of larger host populations or alternative hosts (indigenous and exotic ungulates) that can act as a reservoir during any control efforts. Fears that the presence of alternative hosts within an infested area might allow sufficient reproduction and tick survival to prevent disease eradication have been used as justification for depopulation (culling) of potential alternative hosts in the USA (Shillinger 1938, Marshall et al. 1963). Whilst the indigenous deer do not seem to act as an alternative host for *Rhipicephalus* (*Boophilus) microplus*, the importation and establishment of large numbers of exotic ungulates in south Texas present the potential that ungulate populations may allow sufficient reproductive cycling of these ticks to "significantly delay or prevent eradication through vacation, strongly suggesting the need for development of strategies and technologies to treat wild ungulates" (Temeyer et al. 2004).

Once resistance to stock remedies has been developed by parasites, reversion to susceptibility seems to be so slow as to be of little practical value as a management strategy (Van Wyk 2001).

Two key recommendations relating to the management of resistant ticks in an agricultural context are to reduce the adverse effects of disseminating resistant strains through controls of the movement of livestock and to avoid contact between ticks and livestock with low concentrations of residual acaricide (George et al. 2004). The sale of sheep with resistant worms has assisted in the spread of resistant worms to other areas (Van Wyk 2001). The trade and transport of wildlife is largely unregulated from a parasite perspective, increasing

the risk of spreading resistant strains, particularly from those areas managed in such a way as to promote parasite resistance.

Under the intensified rearing conditions in agriculture, very frequent treatments with anthelmintic drugs are practiced in response to the high helminth infection risk, resulting in the earliest and worst cases of anthelmintic resistance recorded in ruminant livestock e.g. (Sykes et al. 1992); the intensive breeding of wildlife would result in similar opportunities for anthelmintic resistance to develop.

A common livestock precaution is to move animals to pastures that have been rested sufficiently long to reduce parasite burdens; in intensive breeding systems animals may remain in the same camps for extended periods, promoting build-up of parasite loads and necessitating regular and ongoing treatment (leading to increased chance of developing resistance). Studies of the population dynamics of deer when kept at high stocking density showed high susceptibility to parasite infection (Maublanc et al. 2009, Pato et al. 2013); animal densities are kept high in intensive breeding camps in South Africa.

A generalist species of nematode (*Haemonchus contortus*) has been demonstrated, under experimental conditions, to be able to be passed between cattle, sheep and white-tailed deer, and back to cattle and sheep from deer (Mcghee et al. 1981). The transfer of resistant parasites to wildlife populations and back to livestock has been experimentally demonstrated, but requires additional work (Chintoan-Uta et al. 2014). It has been suggested that comprehensive in-field studies should assess whether nematode cross-transmission between wild animals and livestock occurs and contributes, in any way, to the development of resistance on livestock farms.

In Kenya, resistance to anthelmintics was detected in a cattle population previously not treated with anthelmintics; this was speculated to be as a result of the import of cattle from areas where resistance to anthelmintics had already developed (Mungube et al. 2015). This finding raises concerns relating to the large scale and long-distance translocation of live animals from intensive breeding operations, either to other intensive breeding camps or into wild populations.

In the Scottish grouse shooting industry the use of parasite control in semi-extensive systems has evolved in an unregulated way in recent years, largely as it has in the wildlife industry in South Africa. Medicating the wild red grouse population appears to have had some success in reducing mortality and ensuring high numbers of grouse. However, signs of resistance have appeared leading to the use of higher super-strength medications (up to twenty times the concentration of the original wormer drug) (Osborne 2013). The conclusion of a recent assessment of this practice was that no further medication should be administered in the open air before a full environmental impact assessment has been carried out (Wightman and Tingay 2015).

Controlling the spread of resistant strains of ticks is a costly and administratively burdensome process of imposing quarantines with relatively low chance of success (George et al. 2004). If the need arose to manage the spread of resistant strains of ticks this would affect both livestock and game farming.

Extent of evidence

There is a large body of scientific literature from around the world on development of resistance, although the advice on appropriate management strategies can be conflicting. There is a strong correlation between management practices considered to be risk factors for development of parasite resistance and how intensive breeding of game in South Africa takes place. A number of parasite and disease veterinary specialists have raised concerns about current intensive breeding practices in terms of development of parasite resistance, and the only study that could be found to specifically measure effects of what is largely representative

of current practices concluded that tick control practices in an intensive breeding operation had contributed to the development of resistance in a species of tick. Studies investigating the actual development of resistance specifically related to intensive breeding are rare (not entirely surprising given that this is a relatively new practice), and much of the evidence at present resides in anecdotal or unpublished sources. There are likely to be economic implications for agriculture of increased parasite resistance, but this has not been predicted or quantified. For example, it is not known to what extent presence of resistant strains would affect any subsequent conversion of land-use back to livestock production.

Level of agreement

It is not entirely clear what range of different stock remedies and veterinary medicines are being used in this industry, nor the full range of application methods. However, many that are advertised and known to be commonly used are likely to result in under- or over-dosing and consequent development of resistance.

There is some debate and disagreement in the scientific literature about appropriate treatment systems for preventing the development of resistant parasites, much of this stemming from different objectives of short term efficacy versus long term sustainable control. In general, under-dosing (the use of less than the therapeutic dosage level recommended by the manufacturers) and over-dosing seem to be significant risk factors, as is prophylactic treatment of all animals as opposed to only treating those with high infection rates. It is clear that effective parasite control requires a good understanding of, and working with, natural selection and evolution.

Key findings

The evidence of development of resistance to stock remedies and veterinary medicines is well established in the agricultural and biological literature. However, limited studies have been undertaken in the context of intensive breeding, although at least one study in South Africa has demonstrated the development of resistance under treatment regimens commonly used and various experts and industry role players have expressed concern about the development of parasite resistance. It is virtually certain that the risk factors for the development of parasite and disease resistance occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. Whilst there has been no assessment nor current evidence of the role of intensive breeding on the spread of resistant parasites to the broader agricultural and natural systems it appears likely that this could happen. It is unknown whether resistant parasites would be more damaging to natural populations of wildlife than non-resistant strains, but it is likely to have direct negative production and economic consequences for livestock agriculture and other intensive breeding operations, and (in the absence of interventions) could have indirect negative consequences for the broader wildlife economy and biodiversity conservation through increased agricultural regulatory oversight and restrictions placed on the movement of game. It is possible (about as likely as not) that game species, where a large proportion of the population is contained in intensive breeding facilities, could be affected through the development and spread of resistant parasites, but unlikely to be an issue for species where a significant wild population undergoing natural selection still exists (unless resistant parasites and diseases have a more negative effect than non-resistant strains). It is very likely that individual properties where parasite resistance has established will be more difficult or costly to continue intensive breeding, or to get back into agricultural production should the need arise.

Potential mitigation measures

The nature of intensive breeding means that it will invariably promote development of resistance. However, at least the following should be implemented:

1. Much stricter regulation of stock remedy and veterinary medicine use by private game breeders and veterinarians is required;
2. Use of automatic ectoparasiticide applicators must be prohibited (and replaced with individual-based monitoring and management) unless such devices have been tested and approved by the SABS; and
3. The Veterinary and Para-veterinary Act should restrict veterinary use of these products to only crisis management and not prophylactic treatment.
4. More needs to be done on general awareness and training, both of farmers and veterinarians, in terms of drug resistance and the use of parasite management approaches that minimise the development of resistance.

IMPACT STATEMENT 2: DISRUPTION OF THE PROCESS OF NATURAL SELECTION IN TERMS OF HOST-PARASITE EVOLUTION WITH RESULTING LOSS OF DISEASE AND PARASITE RESISTANCE WITHIN THE GAME POPULATION.

Concerns

1. In nature, parasites and host animals are in a continual evolutionary 'arms race', and parasites and disease are very important components of natural selection. Host animals in the wild and in agricultural production systems will always be exposed to parasites and diseases. In many cases early exposure to parasites and diseases results in the development of individual and herd immunity. In nature, differential mortality (natural selection) results in perpetuation of genetic lineages of species more resistant to such diseases and parasites. The nature of intensive breeding (confined areas, desire to minimise mortality and produce bigger and faster growing animals) necessitates and incentivises intensive and ongoing control of helminths and ectoparasites.
2. The control of helminths and ectoparasites in intensive breeding operations compromises the ability of game animals in both the short and long term to cope with parasitic organisms, leaving the individuals and populations vulnerable to a continuous shift towards lack of resistance to parasites. The process of inbreeding, used to maximise the expression of human-desired traits, may reduce the genetic diversity and hence disease and parasite resistance of animals. In intensive breeding operations natural selection for disease and parasite resistance will almost certainly cease to operate. Such game animals are likely to suffer high levels of mortality when/if released into the wild, and will have to be farmed like livestock with continuous disease and parasite management, further entrenching the domestication process.

Evidence

Evidence in southern Africa

No specific studies on loss of disease or parasite resistance in the context of intensive breeding of game in southern Africa were found. However, a study on the 'indigenous' Nguni cattle demonstrated that there is a genetic basis for host genetic resistance to ticks, with certain areas of the genome being identified related to cattle tick resistance (Mapholi et al. 2016).

While reliable quantitative data on the extent of use of stock remedies and veterinary medicines for control of helminths, ectoparasites and diseases in intensive breeding operations is difficult to find, it is widely acknowledged that this is taking place on a large scale. Many advertisements for these products, and well as advice on their use, exist on the internet, magazines and books.

Evidence from elsewhere in the world

Although there are some anecdotal and published studies that infer a relationship between genetic variability and disease resistance in the short term, this is an area that requires more work. However, inbred hosts with lower genetic variability may suffer serious handicaps in the host-parasite arms race, in the long term at least (Lyles and Dobson 1993). A process of inbreeding is used to get expression of recessive alleles coding for coat colour or pattern in intensive and selective breeding operations in South Africa (see Chapter 4).

Parasites have coevolved with the host, so under natural conditions there is a balance (Fowler 2001). Lyles and Dobson (1993) state that "Proper management of the dynamic relationship between animals and the organisms that infect them mandates that we understand the relationships that we presume to manage, and that we integrate this understanding, along with detailed genetic considerations, to produce more holistic (management) plans". It is safe to say that large scale parasite management is taking place in intensive breeding operations in the absence of a good understanding of these relationships or the evolutionary consequences of such interventions.

The proportion of (actively or passively) resistant individuals in a host population determines the level of herd immunity. If a high proportion of individuals have no resistance then a disease can establish and spread rapidly (Lyles and Dobson 1993). This is a concern with the release of intensively bred animals, treated for parasites and not exposed to diseases and hence with no immunity, back into wild situations. It is not known to what extent there may be links between lack of resistance and other heritable traits, and whether there is any risk of lack of resistance being transferred to wild populations linked to other successful traits. In general it is likely that animals with lack of resistance would suffer higher rates of mortality on release into the wild, and this would reduce the potential conservation value of any animals produced under intensive conditions. There is anecdotal evidence that roan antelope produced in intensive conditions suffer high rates of mortality when released into wild conditions, and some of this could be related to lack of resistance to parasites.

Extent of evidence

Intensive breeding of game is a relatively new practice in South Africa and therefore it is not surprising that there are no published studies of the risks associated with protection from exposure to diseases and parasites. There is however a fairly large published literature from elsewhere in the world investigating this issue.

Level of agreement

There is a large degree of agreement in the agricultural and conservation literature that ''protection'' of animals from exposure to diseases and parasites ultimately makes them less adapted for life in the wild, necessitating ongoing management.

Key findings

The evidence for loss of disease and parasite resistance within the game population is established but incomplete. There is a strong theoretical basis for expecting the loss of resistance over time based on the removal of natural selective processes. It is virtually certain that the risk factors for the development of loss of resistance at an individual and herd level to parasites and diseases occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. It is unlikely that loss of resistance in populations kept under intensive conditions will have broader biodiversity or species impacts as any animals that escape or are released into wild conditions under natural selection are likely to suffer higher rates of mortality. However, translocation of intensively-bred animals onto other properties is very likely to result in receiving properties implementing the same parasite and disease controls that initially lead to both parasite

resistance to stock remedies and veterinary medicines and loss of disease and parasite resistance in game populations (see Chapter 9), thereby further exacerbating and accelerating both of these impacts. It is highly likely that species where a large proportion of the population is contained in intensive breeding facilities will lose resistance and be at risk, but unlikely to be an issue for species where a significant wild population undergoing natural selection still exists, unless selected phenotypic traits are positively linked to fitness in the wild, while having linked to genes resulting in reduced resistance (unknown but unlikely). However, natural selective processes are increasingly being removed or moderated on properties outside of intensive breeding properties, in which case it is more likely that non-resistant individuals will pass on their genes to wild populations making the broader population more susceptible to periodic disease or parasite outbreaks.

Potential mitigation measures

High densities of game in intensive breeding operations will invariably result in parasite and disease concerns, and ongoing treatment in these conditions is consequently going to be almost essential. It is therefore not easy to mitigate this risk or introduce any breeding systems to maximise host genetic resistance to ticks. Maximising genetic diversity and ensuring some exposure to diseases and parasites may go some way towards preventing the total loss of resistance, but this may come at a cost of short term profit.

An important mitigation measure is to ensure that there are large enough wild populations being exposed to natural selection and that are not exposed to ongoing parasite treatment. This may require an expansion of the protected area estate for species, particularly habitat specialists, where current populations are relatively small e.g. roan antelope, tsessebe (*Damaliscus lunatus*).

It would be risky for conservation strategies for threatened species to rely on the contribution of animals from intensive and selective breeding facilities to achieve conservation objectives of successful establishment of wild populations. However, intensively-managed populations could become more useful for contributing to conservation programmes if proper conservation breeding principles, especially in relation to genetic diversity and parasite resistance, were adopted and implemented.

ISSUE 6: IMPROPER USE OF STOCK REMEDIES (ANIMAL HEALTH PRODUCTS) AND VETERINARY MEDICINES RESULTING IN RISKS TO CONSUMERS

IMPACT STATEMENT 1: VENISON FROM INTENSIVELY-BRED GAME MAY BE CONTAMINATED BY ANTIMICROBIALS, ECTOPARASITICIDES, ANTHELMINTICS AND/OR ANTI-INFLAMMATORY AGENTS, THEREBY POSING A HEALTH RISK TO HUMANS; INTENSIVELY PRODUCED VENISON MAY NOT BE AS HEALTHY AS WILD VENISON, POTENTIALLY DAMAGING THE BRAND IMAGE AND VALUE OF EXTENSIVELY PRODUCED VENISON.

Concerns

1. Game animals in intensive game breeding operations are exposed to anthelmintic and ectoparasiticide treatments, and sometimes antimicrobials that are very seldom necessary and rarely administered according to prescribed dosage rates. In addition, the desire for animals with larger bodies and horns (current) and faster growth rates (linked to plans for maximising meat production) may promote the use of growth enhancing supplements in game feed.
2. This may result in undesired (qualitative and quantitative) residues of such chemicals in venison that is marketed locally and internationally. Should undesired residues be

identified in foreign markets it could severely jeopardise South Africa's position as an exporter of venison, placing further risk on the extensive wildlife ranching industry.

3. The stated intention of the industry is to move towards intensive venison production. The high stocking rates and high levels of physiological stress associated with such intensification will result in the ongoing need for use of stock remedies; there is some evidence that growth supplements such as beta-agonists are already being used to increase body and horn size. These may pose a risk to human health unless appropriately managed and monitored.
4. An indirect risk to extensive game farmers is that they may be subject to increased regulatory oversight (and hence costs) as a result of the need to manage intensively-bred game meat entering the market; the value of their brand as ''untainted by modern farming practices'' will almost certainly be at risk in the medium to long term.

Evidence

There is a large domestic and growing international market for venison. Presently there is little to no use of intensive production systems focussed on venison production taking place in South Africa; the main market of intensive systems presently is live trade and hunting. However, there are still opportunities for venison from animals produced in intensive systems to be consumed. In future it is the intention to specifically produce venison from intensive production systems (Initiative 10 Wildlife Economy Lab Outcomes).

Concerns of the general public regarding chemical residues in meat may tarnish their perceptions of food safety and environmental health when the husbandry of cattle includes frequent use of acaricides to manage ticks (Mapholi et al. 2016). As a result of real concerns and perceptions of meat safety there are well researched, legislated and strictly applied application rates and withdrawal periods for stock remedies and veterinary medicines for livestock. At present the guidelines and regulatory framework for withdrawal periods for venison production are not in place, and products are often applied in an uncalibrated manner and hence not according to recommended application rates. This could be a risk to the branding value of the health benefits of game meat which should be marketed as free from medications, antibiotics, growth stimulants and vaccines (Reilly et al. 2003).

The evidence for risk of residues of stock remedies (animal health products) and veterinary medicines lies in popular game ranching magazines where adverts abound for automatic dipping apparatus for ectoparasiticides and articles that advocate the prophylactic treatment of game against helminths, ectoparasites and microbes. It is also available from the Griffon Poison Information Centre. A number of advertisements and talks promoting the use of growth enhancing additives to game feed, including beta-agonists, can be found on the internet (see for example http://safarifeeds.co.za/products accessed 4 July 2017)[8].

In the USA, where commercial farming of venison takes place, a number of concerns have been raised, including the problems of fragmented legislation and regulatory oversight, lack of transparency for the consumer, false labelling of products, risk of illegal bush meat entering the market under the guise of farmed venison, and disease transfer (e.g. see http://www.foodsafetymagazine.com/magazine-archive1/december-2004january-2005/game-meat-a-complex-food-safety-and-animal-health-issue/).

In South Africa, Hoffman and Wiklund (2006) cautioned that care should be taken that some of the traits of venison, such as ''untainted by modern farming practices'', is not lost as the production of these animals is intensified to meet the market demand. They also noted that there is still information missing particularly regarding the interaction between production systems and meat quality. They point out that the lipid composition of venison, similarly to

[8] It is probable that some of the large body and horn size claimed to be the result of 'superior genetics' is in fact simply due to enhanced nutrition, food supplements and control of parasites, with no or limited genetic basis.

meat from other ruminants, is related to the animals' diet and that an intensive production system including feeding with grain-based feed mixtures will negatively affect the desirable fatty acid profile. This is another reason for being careful when introducing new production techniques for these alternative meat species (Hoffman and Wiklund 2006).

Tourists visiting South Africa enjoy game meat, know it well and are aware of the health benefits associated with it (Hoffman et al. 2003). Game meat is seen as organic, and tourists seem to perceive it as safe and healthy. It is important to protect this brand if there is a switch to venison production using intensive farming methods.

There is a growing trend of releasing male animals from intensive breeding operations onto properties for hunting (e.g. Ezemvelo KZN Wildlife permit applications). It is quite possible that animals may be shot, and the meat consumed (by hunters and/or staff and/or public) prior to what would be considered a safe withdrawal period.

Extent of evidence

No quantitative studies could be found specific to this issue in South Africa; this is not surprising given that intensive meat production from game is in its infancy. However, because of these same meat quality concerns, the livestock industry has specific and rigorous guidelines on dosage rates and withdrawal times for stock remedies (animal health products) and veterinary medicines, which do not exist yet for game. However, there are groups working on developing guidelines for venison safety and traceability.

Level of agreement

N/A

Key findings

The evidence for health risks to humans or damage to the brand image of venison is established but incomplete. It is virtually certain that current management practices adopted within a proportion of the industry could contribute to this risk. At present, it is presumed that little venison is entering the formal market from intensive breeding sources, although increasing volumes of meat from intensively-bred animals offered for put-and-take hunts may be increasing the health risk for hunters and staff. In future, there are plans to produce significant volumes of venison under intensive conditions for the national and international markets; if issues of meat safety are not properly managed it is likely that the damage to the brand of South African venison will have a negative effect on the profitability of extensive game ranches (largely compatible with biodiversity objectives) thereby indirectly likely to have negative biodiversity impact. There are unlikely to be any species or population level impacts, other than through the broader indirect biodiversity impact already discussed.

Possible mitigation measures

The following mitigation measures should be considered:

1. Much stricter regulation of stock remedy and veterinary medicine use by private game breeders and veterinarians is required. Use of automatic ectoparasiticide applicators must be prohibited unless such devices have been tested and approved by the SABS. Prophylactic use of anthelmintics in game animals should be prohibited. The Veterinary and Para-veterinary Act need to produce regulations to restrict veterinary use of these products to only crisis management and not prophylactic treatment.
2. While there are studies published on meat properties of wild harvested venison compared to livestock, or farmed venison compared to livestock, there is a dearth of studies comparing intensively farmed venison (under current conditions) to wild harvested venison in terms of both meat properties and also in terms of residues of

stock remedies and veterinary medicines. It is suggested that this should be an area of research.

3. Regulation should be made for venison labelling to distinguish between “free-roaming, wild harvested” or “intensively-bred/farmed” to allow for consumer choice.
4. The rules and regulations that apply to livestock meat should also be applied to intensively-bred game, including rules for handling, slaughter, product labelling and transportation. Government driven surveillance of game meat and meat products for drug residues should be instituted. This kind of surveillance is carried out in many other countries around the world and is a strong incentive for producers to use pharmaceutical products responsibly. This may require a review of relevant legislation as well as increased regulatory oversight.

ISSUE 7: INCREASED USE AND MISUSE OF PESTICIDES INCLUDING HERBICIDES, INSECTICIDES AND ACARICIDES AS DEFINED BY THE FERTILIZERS, FARM FEEDS, AGRICULTURAL REMEDIES AND STOCK REMEDIES ACT, ACT NO. 36 OF 1947 AND HAZARDOUS SUBSTANCES (TOXINS) AS DEFINED BY THE HAZARDOUS SUBSTANCES ACT, ACT NO. 15 OF 1973 AND EXCLUDED FROM ACT NO. 36 OF 1947

IMPACT STATEMENT 1: OFF-LABEL USE OF PESTICIDES AND UNLAWFUL USE OF HAZARDOUS SUBSTANCES CAUSE MORTALITY OF INDIGENOUS SPECIES RESULTING IN CHANGES IN ECOSYSTEM FUNCTIONING AND INCREASED THREATS TO THE CONSERVATION OF THREATENED SPECIES.

Concerns

The following concerns have been highlighted in relation to off-label use of pesticides and unlawful use of hazardous substances, but also in relation to increasing scale of use of products (which may be done according to product label) within the industry:

1. The misuse of pesticides and hazardous substances to control damage causing animals in agriculture and wildlife ranching is common; as the trend of intensive breeding increases, there is concern that there will be an increase in the off-label use of pesticides and unlawful use of hazardous substances to protect investments in high value animals. There is also concern that as the trend towards breeding small antelope increases that direct persecution of an additional suite of predators, including large raptors, may increase. Concerns have also been expressed that the use of Avermectin-type anthelmintics, especially in game feed, poses an additional severe risk to dung beetles.
2. The increased use of herbicides to eradicate trees and shrubs in overgrazed areas, along fence lines and in woodlands/bushveld where landowners endeavour to change the vegetation composition in breeding camps, is an increasing trend. Soil sterilant herbicides with a very long half-life and significant potential to leach laterally, such as bromacil and tebuthiuron, may be used by individuals without adequate knowledge of the potential impacts. Eradication of vegetation is also done with non-selective systemic herbicides such as glyphosate in sensitive areas like riparian zones and steep slopes. The scale of use and extent of habitat manipulation is expected to increase with the increase in area allocated to intensive breeding and may in some instances actually trigger the requirement for environmental authorisation.

Evidence

Use and misuse of poisons[9] is a significant risk to wildlife in South Africa and across Africa (Fourie 1996, Ogada 2014). In a recent survey across the wildlife ranching sector in South Africa, more than half of the respondents stated that they practised some form of active predator control (Taylor et al. 2015), using both non-lethal (e.g. live capture/translocation) and lethal methods (e.g. selectively shooting known individuals, culling or using non-selective methods such as poisoning).

The misuse of pesticides and hazardous substances to control damage causing animals including predators, birds of prey, primates and warthogs is thought to be common practice within certain sectors of the game breeding industry (Dr Gerhard Verdoorn, pers. comm.). Carbamate and organophosphate insecticides are generally misused, while sodium monofluoroacetate is increasingly being used unlawfully. This not only threatens mammalian predators and scavengers but also avian predators and scavengers. Chemicals implicated are aldicarb, carbofuran, methomyl, cadusafos, fenamiphos and sodium monofluoroacetate.

The secondary effect of pesticide and hazardous substance misuse in connection with poisoning of animals is that of poisoning non-target animals. Scavenging raptors, other birds that scavenge such as Marabou storks (*Leptoptilos crumeniferus*), southern ground hornbills (*Bucorvus leadbeateri*) and white storks (*Ciconia ciconia*), and scavenging mammals such as hyenas, aardwolf (*Proteles cristatus*) (that scavenge maggots), small predators and even rock monitors (*Varanus albigularis*) have been poisoned. Vultures of all species, Bateleur eagles (*Terathopius ecaudatus*), tawny eagles (*Aquila rapax*), Verreaux's eagles (*Aquilla verreauxii*) and to a lesser extent martial eagles (*Polemaetus bellicosus*) are species that are either poisoned directly or indirectly as a result of pesticide and hazardous substance misuse.

Based on an understanding of the scale and rate of growth of the intensive and selective breeding component of the wildlife industry (e.g. (Taylor et al. 2015)), there is a growing concern amongst conservationists that intensive and selective breeding of game will add a significant additional source of poisoning. There are scientific (peer reviewed) publications in South Africa about wildlife poisoning, but none that could be found specifically assessing the use of poisons in intensive game breeding; by the same token no credible publications could be found stating that it is not taking place. Some publications and popular literature refer to management practices of removing or excluding predators: "predators have been excluded from the breeding camps to limit mortalities" (Schroder and Reilly 2013). However, it is not clear whether this is done by shooting, trapping or poisoning, or a combination thereof.

Because poisoning is illegal it is generally not reported or discussed openly and is difficult to study and quantify. However, there are many cases of the misuse of pesticides and hazardous substances that have been reported to the Griffon Poison Information Centre, including a large number of calls from civil society reporting poisoning, requesting advice on predation management, advice on vegetation management and complaints about wildlife and habitat poisoning (Dr Gerhard Verdoorn, pers. comm.). The IUCN SSC Vulture Study Group, Endangered Wildlife Trust Birds of Prey Programme and Mabula Ground Hornbill Project also maintain records of poisoning incidents, and the latter group identified misuse of poisons in intensive breeding operations as a threat to conservation of the southern ground hornbill after a bird was poisoned.

There is at least one case where a game breeder has been successfully prosecuted for illegal use of poisons directly in relation to the practice of intensive and selective breeding (Phillips 2015). In this case 26 baboons and two Critically Endangered African white-backed vultures were poisoned in an attempt to protect feed for intensively-bred high value antelope species.

[9] Off-label use of pesticides and unlawful use of hazardous substances

However, the fine imposed was insignificant compared to the value of the game subsequently sold, indicating that current fines are unlikely to be an effective deterrent.

Red-billed oxpeckers (*Buphagus erythrorhynchus*) are on occasion targeted with direct attempts to poison them when they impact on game in small enclosures; organophosphate acaricides are implicated in these unlawful actions (Dr Gerhard Verdoorn pers. comm.). There is other unverifiable, circumstantial and 'off the record' evidence for the use of poisons for removal of 'damage causing animals' relating to intensive and selective breeding operations.

The small size of breeding camps results in a large edge to area ratio (see Chapter 6) and consequently large area being treated with herbicides (usually non-selective) along fence lines to protect the fence, to increase visibility and/or to act as firebreaks. While these herbicides may be applied according to guidelines the net result at a landscape scale can be a significant area of indigenous vegetation being eradicated. Eradication of vegetation is also done with non-selective systemic herbicides such as glyphosate, sometimes in sensitive areas like riparian zones and steep slopes. The use of herbicides to eradicate or reduce the density of trees and shrubs where landowners endeavour to change the vegetation structure using herbicides and/or mechanical clearing, appears to be an increasing trend based on a review of internet articles and advertisements. Soil sterilant herbicides with a very long half-life and significant potential to leach laterally, such as bromacil and tebuthiuron, are promoted and used.

Extent of evidence

The evidence is limited in terms of published or court accounts relating specifically to use of poisons in intensive and selective breeding (at least one confirmed case); however, there is a large body of circumstantial and 'off the record' evidence pointing to predator poisoning, supported by knowledge that the prices of colour varieties and so-called scarce game are extremely high, and by evidence of persecution of species such as porcupines (*Hystrix africaeaustralis*) that dig under fences thereby providing easier access to predators. The extent of use of herbicides for modifying vegetation has not been quantified but use of non-selective herbicides for keeping vegetation away from electric fences is common.

Level of agreement

Instances of the off-label use of pesticides and unlawful use of hazardous substances have been documented, reported or strongly suspected. While there is a general perception and much circumstantial evidence in some quarters that poisoning and persecution of predators (and other 'problem species') takes place to protect high value game species, there is strong denial of this by at least a portion of the industry. No scientific publications could be found providing evidence that off-label use of pesticides and unlawful use of hazardous substances is *not* taking place. Objective quantification of the extent and impact of poisoning is lacking and would be difficult to obtain. There is generally open admission of use of herbicides for vegetation modification, but the extent of this practice has not been quantified, although seemingly common.

Key findings

The evidence for negative biodiversity impacts resulting from the off-label use of pesticides and unlawful use of hazardous substances is well-established. Owing to the illegal nature of the activity it is however difficult to assess the probability and extent of occurrence of this issue within the intensive and selective breeding sector of the wildlife industry, but instances of this are have been documented, reported or strongly suspected. The frequency of occurrence is likely to be proportional to the growth of the industry, but the likelihood of it taking place at an individual property level is unknown. It is also known that off-label use of pesticides and unlawful use of hazardous substances takes place in extensive wildlife and agricultural

contexts, and it is unknown whether the extent of occurrence is any higher in intensive and selective breeding operations, although there is good reason and some evidence to suspect that intolerance of predators is higher due to the higher values of the game (see Chapter 7), and hence it is anticipated that the use of poisons (as well as other methods of controlling predators) will be higher, but there are competing explanations. It is known from other work that only a very small proportion of carcasses in the landscape (0.4 – 0.7%) need to contain poisons in order to result in massive population declines and even extinction of vultures (Green et al. 2004). It is therefore very likely that any off-label use of pesticides and unlawful use of hazardous substances in the sector will have severe negative impacts on biodiversity and species, particularly on rare or threatened species and on the predator and scavenger guilds e.g. leopards, vultures.

The clearing of vegetation, through chemical and/or mechanical means, to enhance production could be having significant impacts on biodiversity especially where this takes place in threatened or under-protected ecosystems. In many cases the extent of modification could actually constitute indigenous vegetation 'clearance' in terms of the NEMA Listing Notice 3. It is essential that it is clear at what point vegetation modification triggers the regulations and where environmental authorisation is required, especially where it is motivated as managing 'bush encroachment'.

Possible mitigation measures

Given the financial losses possible as a result of the high monetary value of intensively- and selectively-bred species resulting from predation, coupled with the difficulty of preventing predation through fencing, and the difficulty of targeting specific 'damage causing animals', it is likely that poisons will continue to be used as long as intensive and selective breeding continues or as long as predators remain. However, the following measures should, at a minimum, be implemented:

1. Penalty clauses in Act No. 36 of 1947 need to be amended significantly to match the nature of offences. Currently the maximum penalty under Act 36 is R1 000 and/or two years imprisonment, which is totally inadequate, especially given the massive biodiversity risks posed by poison misuse. It is specifically recommended that an amendment to Act 36 of 1947 to repeal and replace Section 18 (1) with a new sub-sub-section detailing substantially increased penalties is published.
2. Law enforcement and investigation needs to improve drastically. It is imperative for provincial and national conservation authorities to investigate poisoning cases professionally, obtain evidence, perform toxicological analysis and charge offenders using the provisions of the legislation (The possession of banned (prohibited) agricultural remedies is unlawful; this is according to the Regulation no. R862 of 29 July 2016);
3. A comprehensive and intensive awareness campaign needs to be launched to warn game breeders about the negative impacts of pesticide and hazardous substance misuse; and
4. Clarity needs to be provided and communicated as to when habitat clearing for fence and road construction, and for altering habitat structure/composition, constitutes 'clearance' in terms of the NEMA Listing Notices.

CHAPTER 6: BIODIVERSITY ECONOMY RISKS

ISSUE 1: INTENSIVE AND SELECTIVE GAME BREEDING PRACTICES MAY HAVE COLLATORAL NEGATIVE IMPACTS ON CONSERVATION AND THE BROADER WILDLIFE ECONOMY

IMPACT STATEMENT 1: UNCLEAR DISTINCTION BETWEEN WILD AND INTENSIVELY AND SELECTIVELY-BRED GAME IN THE HUNTING SECTOR MAY CONTRIBUTE TO REPUTATIONAL DAMAGE AND CONSEQUENT NEGATIVE ECONOMIC AND CONSERVATION OUTCOMES FOR THE BROADER WILDLIFE INDUSTRY.

Concerns

1. Shooting of intensive and selectively bred game is perceived negatively by important stakeholders and poses a broader reputational risk to hunting and other sectors of the South African wildlife industry.
2. Reputational damage resulting from shooting of intensive and selectively bred game may undermine the economic and conservation contributions of the hunting industry.
3. Absence of mechanisms to communicate credible market information on the conservation contribution of game populations and hunting activities can compound the reputational risks to responsible hunting and game ranching.

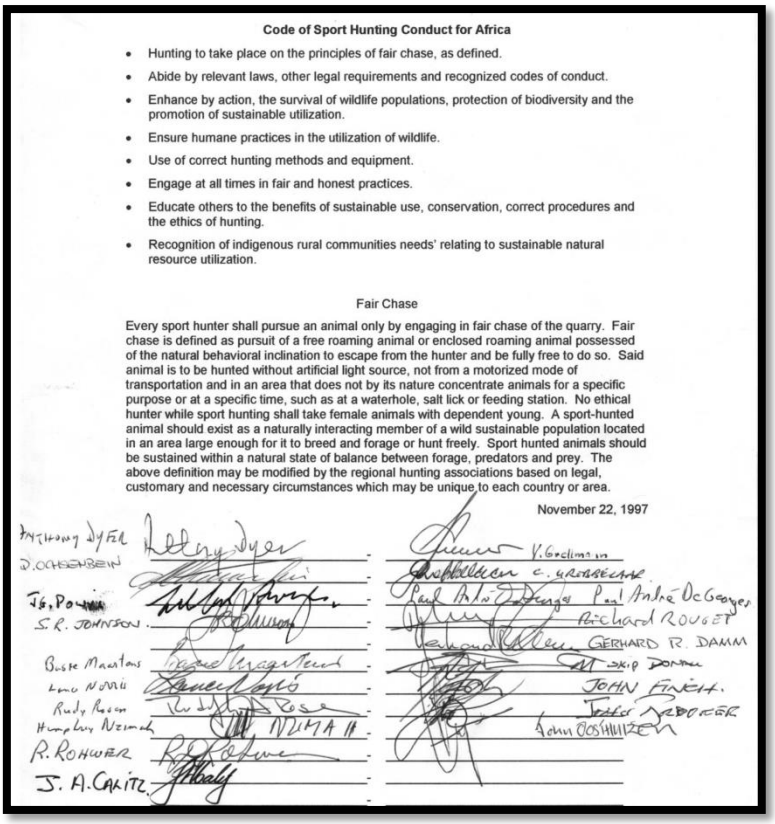
Evidence

Shooting of intensively- and selectively-bred game is perceived negatively by important stakeholders and poses a broader reputational risk to hunting and other sectors of the South African wildlife industry

People are becoming increasingly aware of the threats to the environment, including over- and irresponsible use of natural resources and there is increasing pressure on all forms of tourism to become more sustainable (APO 2009, Dodds *et al.* 2010, UN 2012, Blue & Green Tomorrow 2014). Sustainability here refers to resilience over time and the ability to generate income and benefits without significant deterioration of the environment and natural resources, whilst also addressing social responsibility (Brundtland Commission 1987, IoDSA 2009, De la Paix and Eugène-Rigot 2017, WebFinance 2017, Financial Times Undated). This is especially important for the future of hunting as part of wildlife-based tourism, as it is coming under increased scrutiny globally (IUCN 2016).

People's perceptions about performance on social, economic and environmental responsibility are critical in earning and maintaining a social licence to operate (IoDSA 2009, De la Paix and Eugène-Rigot 2017). This is very relevant for the reputational management of game ranchers, game breeders and hunters. According to Eccles *et al*. (2007) reputation is a matter of perception and it is separate from the actual character or behaviour of an enterprise. Based on the theory of reputational management, the perception of how game populations are managed and hunted ultimately impacts on the sustainability of all role players in the full value chain, individually and as a collective.

There is a substantial literature base from both the business and environmental sectors that provides guidelines to the hunting sector on requirements for sustainability. A number of initiatives have been launched within the hunting fraternity in this regard, including the development of a code for ethical and responsible sports hunting in Africa as early as 1997 (Figure 9) (DeGeorges and Reilly 2008).



Code of Sport Hunting Conduct for Africa

- Hunting to take place on the principles of fair chase, as defined.
- Abide by relevant laws, other legal requirements and recognized codes of conduct.
- Enhance by action, the survival of wildlife populations, protection of biodiversity and the promotion of sustainable utilization.
- Ensure humane practices in the utilization of wildlife.
- Use of correct hunting methods and equipment.
- Engage at all times in fair and honest practices.
- Educate others to the benefits of sustainable use, conservation, correct procedures and the ethics of hunting.
- Recognition of indigenous rural communities needs' relating to sustainable natural resource utilization.

Fair Chase

Every sport hunter shall pursue an animal only by engaging in fair chase of the quarry. Fair chase is defined as pursuit of a free roaming animal or enclosed roaming animal possessed of the natural behavioral inclination to escape from the hunter and be fully free to do so. Said animal is to be hunted without artificial light source, not from a motorized mode of transportation and in an area that does not by its nature concentrate animals for a specific purpose or at a specific time, such as at a waterhole, salt lick or feeding station. No ethical hunter while sport hunting shall take female animals with dependent young. A sport-hunted animal should exist as a naturally interacting member of a wild sustainable population located in an area large enough for it to breed and forage or hunt freely. Sport hunted animals should be sustained within a natural state of balance between forage, predators and prey. The above definition may be modified by the regional hunting associations based on legal, customary and necessary circumstances which may be unique to each country or area.

November 22, 1997

Figure 12: Code of sport hunting conduct for Africa developed in 1997.

Other internationally accepted codes of conduct and charters that provide guidelines, principles, criteria and indicators to guide the management of hunting practices include the European Charter on Hunting and Biodiversity adopted under the Bern Convention on the Conservation of European Wildlife and Natural Habitats (Brainerd 2007); the IUCN SSC Guiding Principles for Trophy Hunting as a Tool for Conservation Incentives (IUCN SSC 2012); the Sustainable Hunting Tourism scheme by the International Council for Game and Wildlife Conservation (CIC) (Damn 2008); with the latest development relevant to Africa, the Charter for Conservation, Habitat Protection and Hunting in Africa (MET 2017). The latter is supported by several countries in Africa. South Africa is however still in a process of consultation with the wildlife sector on its response to the Charter.

Some of the criteria for hunting to be sustainable and socially responsible, taken from these codes and charters, include that hunting must:

- be biologically sustainable;
- not substantially alter processes of natural selection and ecosystem function;
- maintain wild populations of indigenous species with adaptive gene pools;
- not contribute to substantially manipulating ecosystems or their component elements in ways that are incompatible with the objective of supporting the full range of native biodiversity;
- ensure a net conservation benefit for wildlife habitat on which the cost of management and conservation of biological resources are internalised within the area of management;
- generate benefits for retention, enhancement or rehabilitation of habitats; and
- adopt business practices that promote long-term economic sustainability.

Considering the findings of other sections of this report (See Issues 1, 2, 3 and 4), intensive and selective breeding of game, including hunting of animals released from such facilities, appears to be incompatible with several of the above-mentioned requirements for hunting to be ecologically and socially responsible.

Intensive and selectively bred game may become increasingly tame with an increased risks of domestication (IUCN 2016) (See issue 1.5). This would undermine the fair chase foundations of hunting game, which most local and international hunting organisations endorse (See Appendix II). Although definitions of 'fair chase' vary slightly, this principle highlights aspects such as wildness and the ability of the animal hunted, to be able to evade the hunter. This relates to the conditions under which animals are bred and then hunted. In line with the theory of reputational management, the latter plays an important role in the public's attitudes towards hunting (Cooney et al. 2017, Gamborg and Jensen 2017).

As early as 2006, the National Agricultural Marketing Council (NAMC), produced a report on opportunities and challenges relating to the sustainable development of South Africa's wildlife ranching fraternity (NAMC 2006). It was reported that "wildlife which is hunted and the land where the hunt takes place, have to allow for the wild character of game (not tame); that fair chase hunting makes a positive economic and conservation contribution; and that "canned" and "put-and-take" shooting severely damages the reputation of hunting and its sustainability into the future (NAMC 2006).

The initial notion of "canned" hunting as framed in the Cook Report in 1997, involved the "callous execution of tranquilized lions" or "shooting of lions in small camps" (SAPA 2017b). As one of the first associations in South Africa that represents the intensive game breeding sector (in this case predators), SAPA responded after the release of a documentary *"Blood Lions: Bred for the Bullet",* which highlighted controversial aspects of the lion breeding industry, that the narrow definition of "canned" hunting has changed in the public mind as referring to all hunting of intensive/captive-bred lions and not only those hunted in small enclosures (SAPA 2017b).

Irrespective of "canned" hunting being illegal and hunting of "intensively bred" game being legal in South Africa, there are growing negative perceptions about both practices from the public (SAPA 2015), the majority of reputable hunting organisations worldwide (See Appendix II), as well as 72 countries and 409 national and international non-governmental organisations that supported the motion to terminate hunting of captive-bred lions and other predators and captive breeding for commercial, non-conservation purposes (IUCN 2016). All the major international hunting associations in North America and Europe have expressed concern that hunting captive and selectively-bred game is seen as "tame" and not "fair-chase" hunting. More than 93% of international hunters that visited South Africa in 2015 came from North America and Europe (DEA 2015).

The two most prestigious record books will also not accept animals that were specifically bred and manipulated for trophy hunting (Roland Ward 2017). A small number of trophies for colour variants have previously been recorded in the Roland Ward Trophy Book (Roland Ward 2017). However, these were naturally occurring animals. Rowland Ward will not accept animals that are specifically bred with the goal of establishing a separate colour-based category for trophy hunting (Roland Ward 2017). The SCI Record Book committee does not support procedures or practices with wildlife that produce non-typical colour variants, horns, antlers, or body size (Boretsky 2015).

Since 2016, Germany's leading hunting show has instituted a moratorium on advertising or selling of canned or captive-bred hunts and species bred as colour variations (Jagd and Hound 2016). The Nordic Safari Club, representing the second biggest trophy hunting market of South Africa, has also removed all South African lion trophies from its record books and banned all advertisements from operators offering hunts of captive-bred animals in their magazine or any editorial material relating to the practice (NSC 2017).

The Namibian government does not support hunting of captive-bred lion or game from intensive and selective breeding facilities and according to the Ministry of Environment and Tourism of Namibia “those who breed domesticated wildlife and put wildlife that was manipulated and bred intensively in captivity up for sale, were putting hunting and conservation at risk” (NAPHA 2015).

SA Hunters and Game Conservation Association, the biggest local hunting and conservation organisation in South Africa, expressed concerns over the reputational risk associated with shooting of intensive and selectively bred game as early as 2014. Both canned and trophy hunting in general came under increased scrutiny with several cases of poorly conducted and regulated hunting being exposed in the media in years to follow (IUCN 2016).

Despite warnings from several hunting organisations worldwide, including the CIC and 17 USA based hunting associations about the reputational risks of supporting the practice of captive-bred lion shooting (CPHC 2017), both the Confederation of Hunting Associations of South Africa (CHASA) and Professional Hunters Association of South Africa (PHASA) expressed qualified support for the practice and shooting of game from these operations. They have consequently met with significant criticism from local and international hunting and conservation organisations and several of these publicly disassociated themselves from them and those supporting the practice (OPHAA 2017a, WSF 2017, ZPHGA 2017). Organisations in South Africa that represent intensive and selective breeders include Wildlife Ranching South Africa, the Wildlife Producers Association of South Africa (WRSA 2016) and SAPA (SAPA 2017b, a).

Outside of the hunting fraternity, there is also a growing public antipathy towards trophy hunting in Western countries which is also gaining traction within the conservation community (Batavia et al. 2018).This escalated after the shooting of Cecil, a black-maned lion that was a well-known tourism attraction in Zimbabwe. It was initially wrongfully reported that it was an illegal hunt and that the lion was allegedly lured out of a sanctuary and then shot and wounded with an arrow (Baldus 2016). Although Cecil’s hunter was exonerated by the Zimbabwean government of any wrongdoing, the incident was used by anti-hunters to denigrate hunting in all of Africa (Baldus 2016). It has generated a considerable amount of negative publicity and is considered a divisive issue for the hunting industry, both within and outside South Africa. The impacts reached far beyond the individual occurrence of the specific incident and the specific species as confirmed by the Zimbabwe Professional Hunters and Guides Association (ZPHGA).

The release of the *Blood lions* documentary shortly after the Cecil incident, was capitalised on by protectionist non-governmental organisations to taint perceptions about trophy hunting and inflict reputational damage to hunting in general (Blood lions 2016). According to a social media content analyst, the massive public backlash resulted in trophy hunting as a form of pro-conservation sustainable utilisation, losing its legitimacy (Botha and Antonites 2016). More information on the extent and reach of the social media-driven campaigns is reflected in Appendix II. In line with reputational management theory, these incidents also affected the reputation of other role players in wildlife-based tourism as well as ‘Brand South Africa’ as a whole (Polley 2014, Brophy 2015, Genever 2016, Steyn 2018).

Reputational damage resulting from shooting of intensive and selectively bred game may undermine the economic and conservation contributions of the hunting industry

Excluding photographic tourism, hunting is the biggest contributor to the wildlife economy (DEA 2016). Meat hunting contributed R8.6 billion to the economy in 2015 (TREES 2017), while trophy hunting, the most profitable form of consumptive wildlife utilisation (Crosmary et al. 2015), generated approximately R1.7 billion in 2015 (DEA 2016). The portion generated from visiting international hunters also contributes to addressing the countries’ trade deficit. However, the economic risks associated with reputational damage may affect this economic contribution of hunting to the biodiversity in the future.

Reputational damage to hunting as discussed above, growing public antipathy towards hunting in Western countries, and the social media campaigns against hunting such as those after the *Cecil* and *Blood Lions* incidents, resulted in several Western governments imposing stricter domestic measures against the import of hunting trophies. In some cases resulting in complete import bans relating to certain species (EU 2016, Nowak 2016). Although specific attention was given to lions, policy changes were directed at other species and trophy hunting in general (Table 1).

Table 2: Some of the countries that have implemented trade bans and restrictions

COUNTRY	BAN/RESTRICTIONS
France	Lions
Australia	Big five
Netherlands	All trophies
USA	Lions and others

At least one of these instances of stricter domestic measures could be seen to specifically target the hunting of captive-bred lions; a change in US policy toward the import of trophies specifically from South African captive-bred lions, on the grounds that such hunting does not enhance the conservation of wild lion populations.

Mounting negative perceptions further led to more than nine commercial passenger and cargo airlines change their policies to stop the transport of hunting trophies of several species of animals hunted in Africa, irrespective of the hunts being legal or within the provisions of international conventions such as CITES (Bloch 2015, Cach 2015). Within South Africa, the financial sector also responded to the breeding of game for hunting with Nedbank taking a decision "not to finance any activity constituting captive breeding of mammalian predator species for hunting or the exotic pet trade" (Mosupi 2016).

The consequences of these actual and proposed policy changes were perceived as significant enough for two ex-secretary generals of CITES to draft a public response stating that 'these embargoes by airlines and marine shipping companies will ultimately prove damaging to wildlife and to the livelihoods of those in poor communities" (CITES 2015). The IUCN also responded by drafting an information document on the benefits of trophy hunting for European Union decision-makers to guide their decision-making in an attempt to prevent the potential socio-economic impact that these trade restrictions could have on conservation and livelihoods of communities in affected countries (IUCN 2016).

At the 2016 CITES Conference of Parties a resolution was passed focussing specifically on hunting trophies (CITES 2016). Several governments however already implemented stricter domestic requirements than CITES for the importation of hunting trophies or have banned the importation of hunting trophies from African countries altogether (PHASA 2016, UK 2017).

With such added trade restrictions on hunting trophies from several countries and the trade ban from the USA (as the biggest importer of lion trophies), it was expected that the entire value chain would be affected. A preliminary assessment by SAPA after a period of nine months after the implementation of export ban, indicated a loss of at least 320 lion hunts, which equates to a direct loss of income of approximately R78 million (Van de Vyver, pers. comm. 2016)[10].The cumulative impacts of this loss of revenue were devastating to the sector (SAPA 2017a). With the drop in international demand, lion breeders started offering cheap

[10] Van de Vyver, C. (2016). E-mail to SA Hunters on: Statistics on lion breeding facilities.12/5/2016.

lion hunting packages for locals (Lombaard 2016) further reducing income generated from the practice. Although shooting of lions only contributes 24% of the income of breeding facilities, it enables multiple use strategies. Secondary income sources from products such as skins and bones contribute only 9% of the income of breeding facilities (TREES 2017). With the loss in income from shooting, facilities started struggling financially and incidents of animal neglect and welfare were reported. Hundreds of people lost their jobs (Van de Vyver, pers. comm. 2016).

Various South African professional hunters and outfitters have reported that marketing hunting packages internationally has been increasingly challenging in recent years. This notion is supported by the number of international hunters visiting South Africa, which has seen a dramatic decrease of 28% between 2011 and 2016 (DEA 2011-2016, Table 2). South Africa was the preferred destination of 88% of hunting tourists to Africa prior to 2008 (Lindsey et al. 2007), and with the closure of hunting on public lands in Botswana in the beginning of 2014, one would have expected an increase in the number of international hunters visiting the country. However, this was not the case and visiting hunter numbers decreased by 14% between 2014 and 2016 even though there was an increase in the number of international tourists to South Africa (StatsSA 2016).

Table 3: Number of international hunters visiting South Africa annually (DEA, 2011-2016)

Year	Number of hunters
2011	9 138
2012	8 387
2014	7 638
2015	6 633
2016	6 539

At an average spending of R261 762 per international hunter per trip (TREES 2017) South Africa lost nearly R287 million in potential income from trophy hunting in 2016 compared to 2015.

Numerous industry participants believe that the recent negative publicity surrounding South Africa's trophy hunting practices has driven business away to other hunting destinations. Similar concerns have been expressed in relation to intensive and selective breeding more generally, with reference to attributes such as small camp sizes, manipulated colour variation and trophy size, reflecting negative perceptions about wildness and animal welfare at South African hunting destinations (Dorrington 2015) Flack, pers. comm.[11] 2016; Burger, pers. comm. 2016[12]; Geldenhuys, pers. comm. 2017[13]). Notwithstanding these sentiments, the current CEO of WRSA indicated that administrative challenges in acquiring hunting permits as well as marketing South Africa as a hunting destination by government were contributing factors that require attention (Landbou Weekblad 2016).

[11] Peter Flack, lawyer, hunter, conservationist, ex game rancher, producer of the award-winning documentary, The South African Conservation Success Story and recipient of the Musgrave Award (CHASA), the Bateleur Award (SAHGCA) and the Selous Award (APHA). Several comments on his hunting blog confirm these sentiments including that of R. Murphy.
[12] Personal comment by Mr. Stan Burger. Previous president of PHASA, professional hunter and hunting outfitter that attended international hunting shows.
[13] Geldenhuys, N. (2017). International sentiments around colour variants. Interview and e-mail communication from the Managing Director of the international hunting magazine, African Outfitter.

The events above demonstrate the economic risks associated with reputational damage. As intensive breeding of lions for shooting is the forerunner in the intensive and selective breeding of game for shooting, it is important that policy-makers take cognisance of these risks when considering economic risks associated with intensive breeding of wildlife for purely commercial purposes. Additional information on a few case studies relating to the commercialisation of wildlife, including for lion and colour variants, are included in Appendix III.

If reputational damage to hunting results in a shrinkage of the number of local and international hunters then professional hunters, hunting outfitters, land-owners and other industry participants may experience reductions in income, the loss of which may ultimately compromise the socioeconomic contribution of hunting as a major role player in the biodiversity economy.

Responsible hunting has conservation benefits and these benefits are threatened by reputational damage to hunting

Although there are some differences of opinion concerning the extent and effectiveness of hunting as a conservation tool, there is a significant evidence base confirming the role of hunting as an incentive to conserve species and functional ecosystems (Potts 1992, Hitchcock 2000, Mateo-Tomás and Olea 2010, Child et al. 2012, Fischer et al. 2013b, Nelson et al. 2013, Winnercom 2014). According to the IUCN, hunting can generate much needed revenue and economic incentives for the management and conservation of target species and their habitats as well as supporting local livelihoods. This is especially the case for developing countries (Manfredo 2002, Di Minin et al. 2016, Naidoo et al. 2016) and areas where alternative wildlife-based land-uses such as ecotourism are not economically viable (Lindsey et al. 2006).

Although there is evidence of a few cases where unsustainable trophy hunting has contributed to declines of species in Africa (Loveridge et al. 2007, Packer et al. 2011), it is not considered a primary threat and is typically a negligible or minor threat to African wildlife populations (Lindsey et al. 2016). Based on a literature review and six case studies about the impacts of trophy hunting on conservation and livelihoods, Cooney *et al*. (2017) however concluded that outright bans on trophy hunting, as well as import or transport restrictions on high value species could end trophy hunting and that “in the absence of effective and sustainable alternatives, removing the incentives and revenue provided by trophy hunting would likely cause serious population declines for a number of threatened or iconic species.”

Research on the profitability of game ranching in South Africa indicated that 30% of wildlife ranchers derive the majority of their income from hunting (Cloete et al. 2015). With hunting increasingly being perceived as undesirable following, among other possible factors, the public outcry about hunting of intensively and selectively-bred game as discussed above, hunting as a revenue source may be excluded as an income stream for wildlife areas. Without the revenue from hunting a reduction of competitiveness of wildlife-based land-uses relative to ecologically unfavourable alternatives could be expected. Given that extensive wildlife areas managed by the private sector comprise 14% of the country and represent more than twice the area covered by state protected areas (Taylor et al. 2015), the escalating pressures towards land-use change and current trends in the agricultural sector - where the number of farms on sale has increased by 45% in one year (Bezuidenhoudt 2016) - the future contribution of these areas to conservation and ecosystem products and services would be highly uncertain. The concomitant result of this could be negative for the broader economy that is supported by ecosystem services from these functional ecosystems and it could increase the financial burden on government to achieve conservation targets.

Additional to the conservation implications highlighted above, a shrinking hunting sector and reduction in international hunters coming to South Africa as a result of reputational damage, may further impact on community-based approaches that allow local people access to and control over wildlife resources through hunting (Hitchcock 2000). The ban on safari hunting in Botswana in 2014, resulted in the loss of income generated by local communities and jobs

previously created from safari hunting. In less than a year, the community-based forum in the Okavango Delta lost R9.4 million and 200 jobs due to the hunting ban (Mbaiwa 2017). Other reported socio-economic impacts include reduction of income, employment opportunities, social services such as funeral insurance, scholarships and income required to make provision for housing for the needy and elderly (Mbaiwa 2017).

Hunting largely takes place in the rural underdeveloped areas that often have struggling economies. Some of the poorest provinces in South Africa, namely Limpopo and Eastern Cape, are the preferred provinces for hunting. Between 50% and 90% of the net revenues from hunting (excluding operator costs) are generally allocated to local entities, with the remainder going to government authorities, in developing countries (Cooney et al. 2017). Although contributions to local communities in South Africa are not as substantial as other countries in SADC, this may change in future as transformation initiatives are implemented through the national Biodiversity Economy Strategy of the country. Without hunting as a sustainable income stream, communities may struggle to establish and run economically viable enterprises in parts of the country that are not suitable for photographic tourism or other forms of agriculture.

Other positive impacts of hunting on poor rural communities include the contribution to food security, secondary economic activities where by-products from hunting are sold, and a reduction in poaching (Hofer et al. 1996, McCrindle et al. 2013, Mbaiwa 2017). Private landowners also use trophy hunting revenue to fund anti-poaching operations that can reduce poaching (Lindsey et al. 2007). An increase in poaching can be far more damaging in both scale and demographic impact than well-manned responsible hunting as is demonstrated by the fact that almost 20 times more African rhinos were poached in 2015 than those legally hunted (Emslie et al. 2016). In addition, the money from poaching flows to criminals whilst regulated hunting can contribute to the management of wildlife.

Reputational damage to hunting can also impede the ability of wildlife managers to manage game numbers, as not all areas are well suited to cost-effective live capture. The Bubye Valley Conservancy, one of Zimbabwe's largest wildlife reserves, recently announced that the controversy around the hunting of *Cecil* the lion potentially contributed to a dramatic decline in hunters, resulting in challenges in managing the park's predator numbers that were previously done through hunting. This negative impact was exacerbated by the loss in much needed revenue to manage the park (OUTDOORHUB 2015).

For sustainable contribution of hunting to conservation, it is critical that all role-players in the value chain of hunting, including game ranchers and breeders, consider the requirements for sustainability of hunting as discussed in the beginning of this chapter. Despite the claims by commercial breeders that their captive-bred game animals are required for reintroduction and species restoration, these are often not based on any scientific evidence (See Issues 1, 2, 3 and 4). Several research reports and international conservation organisations, including the African Lion Working Group, the United States Fish and Wildlife Service and SANBI, have questioned whether intensive and selective breeding of game for pure commercial purposes and 'captive-bred' hunting contributes to conservation (Geist 1992, Hunter et al. 2013, Nelson et al. 2013, UTC 2015, Van Der Merwe 2016).

Many hunting organisations agree that irresponsible hunting practices are undermining the work that the hunting fraternity is doing in overcoming challenges associated with negative public perceptions about hunting and advocating its positive contribution to conservation of game and their natural habitats (NAPHA 2017, OPHAA 2017b, PHAZ 2017, ZPHGA 2017, DSC 2018). The loss of hunting income would threaten numerous private and communal land conservation projects, with spill-over effects to state-protected areas (Lindsey et al. 2007, Naidoo et al. 2016).

Absence of mechanisms to communicate credible market information on the conservation contribution of game populations and hunting activities can compound reputational risks to responsible hunting and game ranching

Sustainability certification schemes are used by consumers to make informed decisions on the choice of purchases (APO 2009). Several international publications have highlighted the need for best practice guidelines and/or a certification system to inform responsible hunting (Baldus and Cauldwell 2005, Booth 2005, Child and Wall 2009, Miller et al. 2016a).

For the hunting sector to determine their performance on these and other criteria, information is required on wildlife management practices of hunting destinations because hunting cannot be separated from the management of game (Fischer et al. 2013a). However, the required information is not available for most of the products and services in the wildlife industry in South Africa (Child and Wall 2009, Brink et al. 2011). There is no segmentation and labelling system to distinguish between the various game farms and breeding operations based on sustainability criteria and the requirements discussed above for the hunting sector to demonstrate it is sustainable and socially responsible. Dalerum and Miranda (2016) confirmed that current valuation and demand relating to game on offer in South Africa is driven by imperfect market information. The lack of reliable market information and a mechanism for informed decision-making may impede the sustainable contribution of hunting as the biggest contributor to the wildlife economy.

To address risks for the broader tourism industry, South Africa developed a “Responsible Tourism” Guideline and manual to harness opportunities presented by the trend towards more responsible tourism business practices. It acts as a positive marketing tool, provided that claims of responsibility are credible and based on demonstrable delivery of responsible activities and objectives. It further states that the environmental, social and economic impacts of tourism developments must be assessed and monitored, with open disclosure of information (Spenceley et al. 2002). However, the guideline does not address the requirements of the hunting sector to make informed decisions on whether or not management practices associated with the game on offer for hunting aligns with their requirements for sustainability and a responsible hunt that contributes to conservation of wild populations and their habitats. As such, it cannot be used as a mechanism by either the game ranching, game breeding or the hunting sector to demonstrate performance on sustainability indicators or to reduce reputational risks for the various sectors and the industry.

The need for industry standards and a certification/labelling scheme specifically to reduce risks to the sustainable growth of the various sectors within the wildlife economy was identified during the Biodiversity and Tourism Lab of Department of Environmental Affairs (DEA 2016). This need has been confirmed by South African consumptive hunters that believe a “green certification system” will help the market to distinguish between responsible hunting and other forms (TREES 2017) and contribute to addressing the challenges associated with South Africa losing market share as preferred hunting destination. A certification system could create related comparative advantages for South Africa by capitalising on the changing consumer trends towards increasing environmental and social responsibility, and awareness in tourism (Sasidharan et al. 2002). It is notable that both Namibia and Botswana, that are known for large extensive wildlife areas that support their wildlife-based tourism industry, supersede South Africa on environmental sustainability indicators measured for competitiveness (LEDET 2013). According to the chief executive officer of NAPHA, the growth in international hunting in Namibia has a lot to do with the promotion of the country as a responsible hunting destination.

Most hunting associations have some form of code of good practice for their members on their websites, but the focus is predominantly on the activity of hunting and the required links to assess performance in terms of contribution to conservation of the resource base is lacking. WRSA and SAPA, which represent breeders, are developing internal standards for intensive and selective breeding operations for their members, with SAPA also having developed a

standard for hunting of captive-bred lions (SAPA 2017b). However, these standards fall short of addressing the international best practice criteria discussed above and those organisations have not consulted with the broader hunting sector to determine consumer perception.

The trophy hunting sector must address requirements of the CITES COP17 approved resolution on hunting trophies, which indicate that “countries should consider the contribution of hunting to species conservation and socio-economic benefits, and its role in providing incentives for people to conserve wildlife, when considering stricter domestic measures and making decisions relating to the import of hunting trophies” and “that trophy hunting activities relating to CITES Appendix I listed species should produce tangible conservation benefits for the species concerned” (CITES 2016). Work has already begun on a clear framework to “guarantee the sustainable and legal origin of hunting trophies of species listed in Appendix I or II, to ensure that trophy hunting is sustainably managed, does not undermine the conservation of target species and, as appropriate, provides benefits to local communities” (CITES 2016).

Extent of evidence

Past records and events; industry practice and formal positions; expert opinion; business and media reports; policy and legislative frameworks and reports; theory on reputation management; and case studies supported by scientific literature provide a large body of evidence that indicates the positions of the most prominent local and international role-players in the hunting and conservation sector in relation to reputational risks associated with shooting of intensively- and selectively-bred game. As supporting evidence, please refer to the figures and tables in the Appendix II.

There is well-established evidence that the provision and shooting of intensively and selectively-bred game does not align with several of the hunting codes of good practice, conduct and charters discussed above. Evidence has further been provided that evaluating hunting as a practice cannot be separated from how source populations are managed. Public perception of hunting is influenced by both practices in the value chain. This is in line with the theory that sustainability and reputational risks of one role-player in a value chain affects the sustainability and reputation of other role-players in the same value chain.

Although it is acknowledged that lions evoke stronger emotions than other game species, the captive-bred lion shooting case study, supported by published international case studies and research as discussed in this report, supports the notion that reputational damage resulting from negative perceptions about perceived irresponsible hunting practices and the intensive breeding and subsequent shooting of game, can result in trade restrictions with socio-economic implications for hunters, breeders and rural communities.

Although interviews and the chain of events indicate that reputational damage to the hunting sector in South Africa resulting from negative perceptions about the practice of intensive and selectively game breeding, the associated erection of small game-breeding camps and the subsequent shooting of these animals play a role in the country losing market share as a preferred hunting destination. The evidence is speculative and further research is needed to better understand and quantify this aspect.

The contribution of responsible hunting to conservation, livelihoods and the economy has been well documented for several countries in southern Africa. Additionally, several case studies and scientific publications have been published that highlight\ potential risks and actual impacts related to hunting being discounted as an incentive for conservation of wildlife areas and associated species. Although the nuances may differ from country to country, and the extent of some of the potential impacts may differ in South Africa, the extent of the evidence is sufficient to highlight that if reputational damage to hunting would result in a reduction in the responsible hunting in the country, it may have a significant impact on conservation in South Africa.

The importance of credible market information in a market economy and management of reputational risks are well researched and documented, including for wildlife-based tourism. No evidence of a comprehensive operational mechanism that provides credible market information on the conservation contribution of game populations and hunting activities in South Africa could be found.

Based on all of the above, it is concluded that there is significant evidence that shooting of intensively- and selectively-bred game may contribute to reputational risks for the hunting sector that can lead to negative economic and conservation outcomes for the broader wildlife industry.

Extent of agreement

Organisations that represent intensive and selective game breeding, i.e. Wildlife Ranching South Africa (WRSA), the Wildlife Producers Association (WPA), and South African Predators Association (SAPA), do not perceive intensive breeding for commercial purposes and the subsequent shooting of the animals negatively, provided that it is legal and complies to internal standards and protocols. Also, notwithstanding significant criticism against the practice by those not participating in it, both the Confederation of Hunting Associations of South Africa (CHASA) and Professional Hunters Association of South Africa (PHASA) expressed qualified support for the practice and shooting of game from these operations. There is neither consensus on the acceptability of shooting intensively- and selectively-bred game animals, nor on the extent and relevance of reputational damage (past or future) to the industry within South Africa.

The biggest local hunting and conservation association, SA Hunters and CPHC, the newly formed hunting organisation that represents professional hunters, agree with the most prominent international hunting and conservation organisations that shooting intensively- and selectively-bred game animals poses significant reputational risk to the hunting sector. These groups' practices are mostly aligned with internationally accepted principles for sustainability and social responsibility from both the business and conservation sector, and specific requirements of local and international codes of good practice for hunting to be seen as sustainable and responsible.

However, there is general consensus amongst all role-players that reputational damage can have significant negative consequences for individual operators (breeding and hunting); the sector; the industry at large, and its consequent contribution to conservation and livelihoods. This includes SAPA, which acknowledges that the international trade restrictions affecting the export of hunting trophies from intensively-bred game is a result of reputational damage that is having dire consequences for the intensive lion breeding and hunting sector (SAPA 2017a). There is further general agreement that commonly accepted standards for the various subsectors of the wildlife industry are lacking and that these could reduce risks to the growth of the individual sub-sectors and the entire wildlife industry.

The disagreement between the game breeders and most hunting and conservation stakeholders concerns the extent to which shooting of intensively- and selectively-bred game contributes toward reputational damage to hunting. The perception of most breeders is that local and international reputational damage to hunting is primarily a result of illegal hunting activities and consequent efforts of protectionist and animal rights activists seeking to ban hunting completely. However, the internal standards of organisations in South Africa that breed game animals intensively to release them for shooting do not align with local and international codes for sustainable and responsible hunting practices.

There is a high level of agreement among prominent international hunting organisations, conservation organisations, CITES parties, and IUCN members, that hunting of captive or intensively- and selectively-bred game animals - irrespective of whether these breeding facilities comply with certain standards (as in the case of SAPA) - contributes to reputational

risks for the responsible hunting sector, and threatens its contribution to conservation and livelihoods.

Most significantly, there is a high level of agreement from scientists and industry members that the development of industry standards, guidelines and/or mechanisms is needed to communicate credible market information to reduce risks associated with management of source populations and hunting. In summary, there is widespread agreement that the unclear distinction of intensively and selectively bred game in the hunting sector may contribute to reputational damage and consequent negative economic and conservation outcomes for the broader wildlife industry. To the extent that there is disagreement, it concerns the nature and extent of the damage and of necessary mitigation measures.

Key findings

Considering increasing pressures on natural resources, this assessment confirms that there is mounting pressure from the public at large that enterprises should better demonstrate that their practices are in line with international principles of sustainability and that they are socially responsible in utilising the natural resource base. However, this assessment demonstrates that the various sectors within the wildlife industry have different levels of understanding and acceptance of these realities and principles in evaluating their own activities for future growth.

It has been demonstrated that hunting is under increased public scrutiny. Animal rights groups use any possible opportunity to bring hunting in disrepute, with substantive risks associated with perceived irresponsible hunting practices for the individuals involved, enterprises and the sector as a whole. All role players agree that this reputational damage to hunting can have significant negative implications for its sustained contribution to conservation, livelihoods and socio-economic development. Policy-makers should be mindful that ignoring this risk can exacerbate the reputational damage that the industry already experiences with concomitant negative economic impacts for individuals, the various sectors and the wildlife industry.

The majority of prominent and relevant local and international role-players in the hunting and conservation fraternities have negative attitudes towards breeding game intensively and selectively for shooting. The major concerns include hunting ethics, reputational risk and lack of conservation contribution. This is in line with internationally accepted guidelines for hunting to be seen as sustainable and socially responsible, as included in several internationally accepted codes of good practice and guidelines developed by the IUCN. In all of these hunting codes, the management of source population and their habitat form an integral part of responsible hunting.

Irrespective of the overwhelming negative response of hunting organisations worldwide to the change in policy position of PHASA in support of captive-bred lion hunting where facilities comply to the norms and standards of SAPA, the intensive and selective game breeding subsector does not appear to accept that breeding game intensively and selectively for hunting poses a reputational risk to hunting. Illegal hunting and animal rights activism are seen as the major causes for reputational damage.

There is general agreement that the lack of a widely accepted mechanism to communicate credible market information on the practices of role-players in the value chain can reduce reputational risk. Theory on reputation management and this assessment suggest that such a mechanism should address both the conservation contribution and sustainability of management practices of source populations and hunting activities to reduce reputational risks to like-minded role-players in the same value chain.

Mitigation measures

1. Segmentation of “producers” within the wildlife management sector to distinguish between the types of “production” systems along the continuum of captive-bred to wild

populations with clear standards /codes of good practice for each sub-sector. Some work on this has been started during the Biodiversity Economy Lab where key role-players in the sector identified the need to develop and implement industry standards to ensure sustainability and reduce risks to biodiversity and growth of the sector as a priority.

2. Like-minded wildlife sector role-players within the game ranching and hunting value chain to collaborate in the development of mechanisms that communicate credible market information on the conservation contribution and sustainability of management practices of source populations and hunting activities in the same value chain, to consumers and the public, towards reducing shared reputational risks. This can form the basis for product specific labelling/certification and also inform "enhancement findings" by DEA as required for the export of certain hunting trophies from CITES-listed species.
3. Government to create an enabling environment that facilitates trade between role-players in the same production categories and those in same value chain, with increased self-administration between like-minded producers, e.g. from translocation from one captive-bred facility to another. Policy objective 5.2 of the Biodiversity White Paper states that government has to create and implement incentives that support the conservation and sustainable use of biological diversity to act as instruments and mechanisms to induce people to change their behaviour. The introduction of incentives by Government is an important way to motivate people to conserve and use biodiversity sustainably. Current policies that are counterproductive should also be identified and removed, according to the White Paper. This could include engaging international stakeholders such as CITES member states to recognise and incentivise role-players that demonstrate adherence to sustainability principles and internationally recognised codes of good practice for managing wildlife populations and hunting and preference in support programmes from government.
4. Government and the wildlife sector to partner in the development of a market-based industry support mechanism that would provide sound market and trade information for all sectors of the wildlife industry, to ensure informed decision-making throughout the industry. Adopting the user pays principle to allocate licencing fees from a standardised national hunting licence fee system could be beneficial in providing funds for socio-economic research, while proper recording and processing of permitting information would be an important source of information.
5. Research is needed on the nature, extent and interrelationships between the various subsectors of the wildlife industry and their stakeholder base. In particular, research is needed on: stakeholder perceptions and market demand in relation to hunting practices; the management of source populations, including reasons for reputational damage to hunting; South Africa losing market share as a preferred hunting destination; and the economic implications of trade restrictions on the captive-bred lion industry.

IMPACT STATEMENT 2: UNREGULATED CHANGE IN LAND-USE FROM CONSERVATION COMPATIBLE EXTENSIVE WILDLIFE AREAS TO INTENSIVE GAME BREEDING OPERATIONS REDUCES THE EFFECTIVENESS OF CONSERVATION INITIATIVES AND RAISES CONSERVATION COSTS TO GOVERNMENT AND SOCIETY

Concerns

1. There is a lack of legal and policy instruments that quantify the nature and extent of intensive and selective breeding operations and their location in sensitive landscapes.
2. The above-mentioned shortcoming impedes the ability of conservation agencies and government to factor in the impacts of the practice in conservation and land-use planning processes.

3. The result is uninformed decision-making with potential detrimental conservation and economic implications.

Evidence

Land-use change and habitat loss are well-known as major threats to biodiversity conservation. According to the National Biodiversity Assessment (South African National Biodiversity Institute 2011) there would be little natural vegetation left in KwaZulu-Natal, Gauteng and North-West province outside protected areas by about 2050 at the current levels of land-use change. Previous sections of this assessment discussed how the intensive and selective game breeding sector contributes to land-use change from extensive wildlife systems to intensive breeding operations. The latter does not require an environmental impact assessment (EIA), nor does the erection of highly impermeable fences for small camps associated with the practice. Information on the nature, extent, distribution and impact of these operations and their associated practices are not well-known, making it impossible to factor in the land-use change from extensive wildlife system to intensive breeding camps into conservation planning processes.

In addition, conservation planning processes in South Africa typically consider land cover as an input layer. While implications of intensive breeding operations on habitat integrity have been discussed in other sections in this assessment, habitat integrity of “natural” and “near natural” areas are not considered in most biodiversity conservation planning process in provinces currently. Even though the reduced permeability of the landscape to free range species movement and its utility as an ecological corridor and area for protected area expansion may be lost as a result of the intensification process, this will not be picked up by the conservation planning processes, including for protected area expansion. This poses a risk to conservation planning, adding to the fact that the rate of land-use change in certain landscapes is very high (Desmet et al. 2017) compared to the lifecycle of five years for review of most provincial conservation planning processes.

These shortcomings pose a risk for conservation planning and may result in costly conservation decisions being based on imperfect information. This would especially be problematic when conservation off-sets are considered. Because of the high cost associated with erection and electrification of breeding camps, it would further be extremely costly to purchase and rehabilitate these areas to achieve protected area expansion targets.

A study by Desmet et al. (2017) in the Waterberg district of Limpopo Province, confirms these risks and shows an exponential growth in intensive game breeding properties in the last five years (three-fold) in the area between three protected areas namely Madikwe, Atherstone and Marakele. The number of properties assessed with intensive breeding camps increased exponentially and the area is now highly fragmented. However, the same area is earmarked in the Limpopo Conservation Plan and Waterberg Bioregional Plan (LEDET 2016) as an ecological corridor linking the three protected areas. It also forms part of the Protected Area Expansion Strategy for the Province and a priority conservation area for wild dog and cheetah (Jackson et al. 2016). Three of the priority conservation areas identified for wild dog, that function as source populations for this endangered species according to Jackson et al. (2016), are being compromised by proliferation of intensive game breeding with highly impermeable fences that are interspersed in this sensitive landscape.

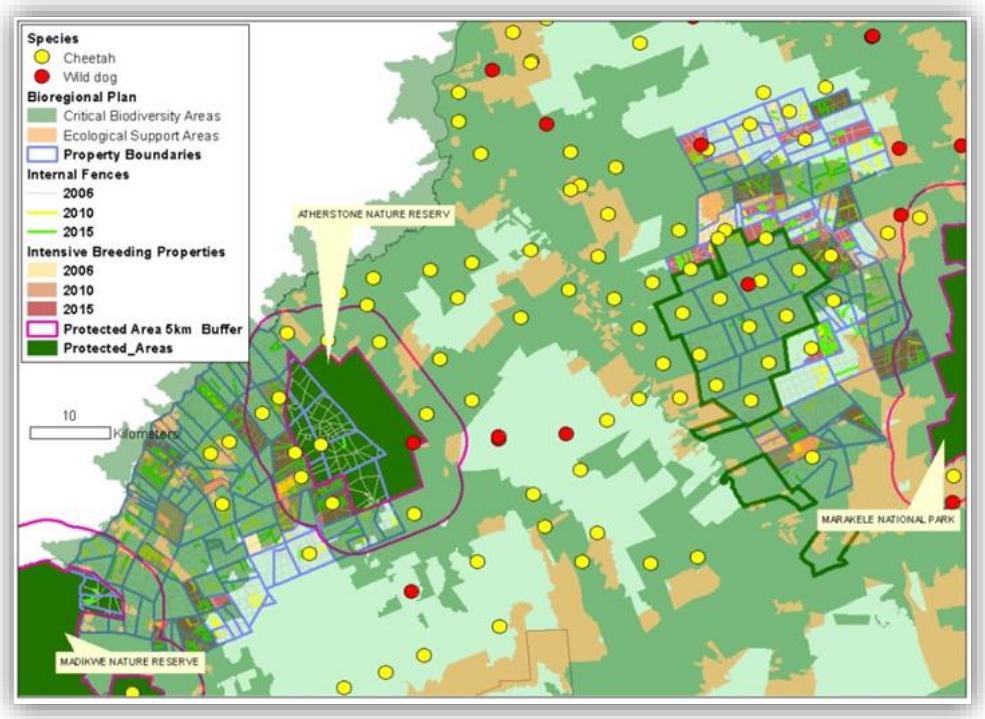


Figure 13: Priority conservation areas identified for wild dog and cheetah being compromised by the proliferation of intensive game breeding operations interspersed in extensive wildlife areas, fragmenting their habitat through impermeable fences (Jackson et al. 2016, Desmet et al. 2017).

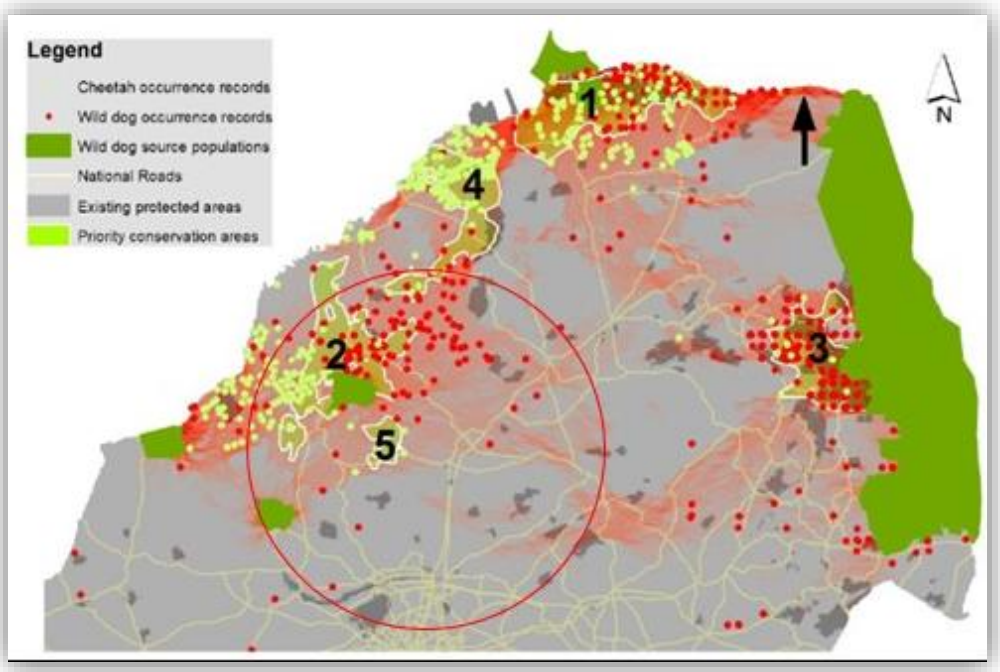


Figure 14: Priority conservation areas identified for wild dog that function as source populations for this endangered species (Jackson et al. 2016).

Critical Biodiversity Areas can also be transformed to intensive breeding operations without any mechanism currently to mitigate the associated impacts.

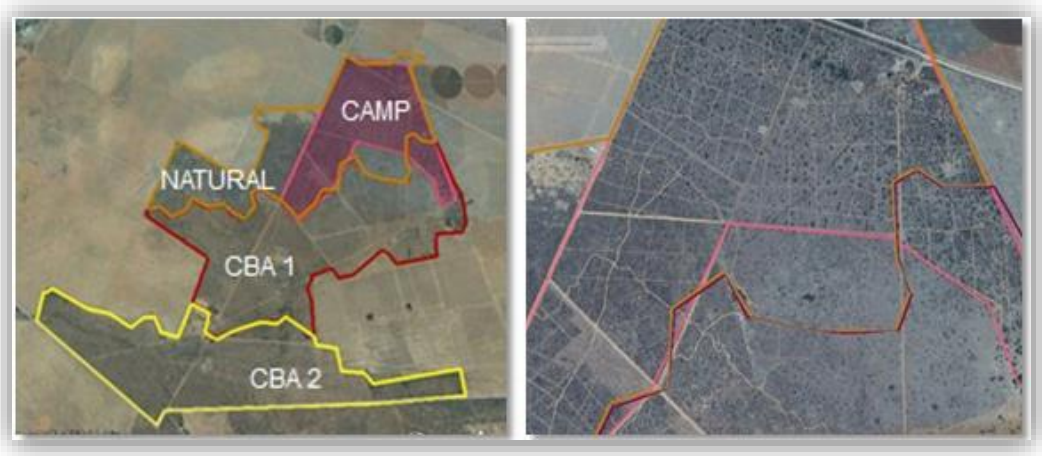


Figure 15: "Natural" and Critical Biodiversity Areas (CBAs) identified in the Limpopo Conservation Plan that are now fragmented through smaller camps.

Furthermore, because the erection of small game breeding camps does not trigger an EIA, these camps can be erected in sensitive areas on the borders of protected areas. Considering the impacts on biodiversity as discussed in other sections this report, having intensive and selective breeding operations on protected area borders can negatively impact on ecosystems of protected areas that depend on ecological processes within their neighbouring landscapes (Ferguson and Hanks 2010). The difference in predator tolerance between protected areas and intensive breeding facilities (discussed earlier), may further result in conflict between the adjacent landowners with very different land-use objectives, whilst it can also affect the integrity of animal populations in adjacent protected areas. The latter is also relevant when purchasing these areas as part of protected area expansion strategies.

Mokala National Park is a case in point where three different colour variants of springbuck have been reported as well as buffaloes with ear tags that were part of adjacent intensive game breeding operations (Nel, pers. comm., 2015). In addition to biodiversity consideration, this has an impact on visitor experience and sense of place. The importance of ambience in the tourism experience has been discussed earlier and having small camps with high electric fences will change the natural character of areas where parks are highly reliant on visitor numbers. Addressing these impacts also has cost implications for protected areas and conservation agencies and may tarnish the reputation of protected areas.

Figure 16: Different colour variants from adjacent intensive and selective game breeding operations inside Mokala National Park (Nel, pers. Comm., 2015).

Moreover, intensive breeding especially on park borders irreversibly changes land values that can severely constrain future protected area development, impacting on the ability of the State to achieve biodiversity conservation outcomes. More importantly, as protected areas can function as a driver in rural economic development, this may ultimately constrain rural economic development opportunities (van Ierland 2010).

Mechanisms to address at least some of the impacts do exist within several of the provinces' conservation policy frameworks. Provisions exist to exempt game ranchers from applying for individual land-use and wildlife utilisation permits for individual hunting, catching and other permits as an incentive for protecting viable wild populations of species and their habitats that could be utilised ecologically sustainable. The exemption policy of LEDET for example indicates that "when breeding camps/holding camps are erected on an exempted farm and the remaining part of the farm is less than 400 ha after the erection of the camps, that farm does not qualify for exemption anymore" (LEDET 2016). This should serve as a disincentive for erection of small camps within the extensive wildlife landscape, but confidential sources within LEDET indicate that the department does not have the resources to monitor compliance to this policy. Several game ranchers have also confirmed that renewal of exemption farms happens without any site visits.

Extent of evidence

The case studies presented demonstrate that the impacts of shortcomings associated with the lack of legal and policy instruments to quantify the nature and extent of intensive and selective breeding operations and their location in sensitive landscapes, are at least prevalent for one province. Although this has not been tested for all provinces, the absence of legislative mechanisms that can inform conservation planning processes is evident in the existing legislative framework that guides conservation and land-use planning, including the EIA regulations, Spatial Planning and Land-use Management Act (SPLUMA) and provincial conservation legislation and conservation plans for provinces. This has further been confirmed in the Biodiversity Economy Lab during 2016 (DEA 2016) and brought to the attention of SANBI.

Extent of agreement

The absence of legislative mechanisms to inform conservation planning processes is evident in the legislative framework that guides conservation and land-use planning in South Africa, and the potential risks to conservation and land-use planning processes have been confirmed in the Biodiversity Economy Lab during 2016 (DEA 2016) where industry, government and social society were represented. Agreement on the extent of the implications for all provinces are limited as it has not been assessed nationally.

Key findings

The impacts of intensive and selective breeding on the resource base and biodiversity have been discussed extensively in this assessment. It has been demonstrated that the ecological footprint of these game breeding operations is very different from traditional extensive wildlife areas, even if the landcover may still be close to natural in some cases.

Associated impacts such as habitat fragmentation, animals killed in fences, and reduced tolerance by intensive game breeders towards free-ranging threatened predators, are of greater concern in near natural and sensitive environments such as areas adjacent to protected areas and wildlife corridors, than in areas already modified or zoned for intensive agricultural practices.

The existing shortfalls in policy frameworks that govern land-use planning and environmental impact assessments result in a gap between the information used to inform policy processes and what is happening on the ground. This limits the ability of government to address the

impacts of intensive and selective breeding operations on biodiversity associated with natural landscapes and extensive wildlife areas that:

- form the basis of wildlife-based tourism and hunting, the biggest contributors to the wildlife economy;
- form the basis of wildlife corridors, private sector contribution to national conservation targets and the national protected area strategy, wherein protected areas can function as drivers in the rural economy; and
- generate ecosystem goods and services, which in South Africa, amounts to R73 billion per annum, or equivalent to 3% of the country's GDP (SANBI, 2010) and that supports the economy.

In the absence of a policy framework to prevent or at least mitigate these impacts, the direct and indirect costs to government may be significant.

Mitigation measures

1. Segmentation of the game ranching and game breeding sectors.
2. Development of standards and/or a "green/sustainability" certification/labelling system reflecting ecological footprint and performance on sustainability criteria.
3. Regulation of the practice to ensure the country can monitor and map the spatial distribution thereof to ensure its consideration in conservation and land-use planning processes. Specific aspects that can be considered include the following:
   - include the "controlled environments"/camps in a different land-use class as extensive wildlife systems and develop zoning standards to ensure they are clustered in areas away from Critical Biodiversity Areas, Protected Areas and important extensive wildlife systems to reduce risks or addressed in environmental impact assessments;
   - limited the percentage of a farm in an extensive wildlife area that can be transformed into controlled environments/breeding camps; and
   - develop a category of fences from highly impermeable to permeable (like national roads to district roads) to inform land-use planning decisions and to act as triggers for environmental impact assessments or an off-set requirement.
4. Develop incentives and disincentives promoting responsible wildlife management practices.

CHAPTER 7: SUMMARY, CONCLUSION AND RECOMMENDATIONS

Over the last three decades the South African wildlife industry has been largely compatible with conserving biodiversity and as such has made a significant contribution thereto (Child et al. 2012). However, in recent years, selective breeding and the intensive management of game has emerged as a new and growing sector within the broader private wildlife industry (Cloete et al. 2015, Taylor et al. 2015). Concerns have been raised about the long-term and potential consequences of the practice on other sub-sectors of the wildlife sector, as well as the country's biodiversity and biodiversity economy (Cousins et al. 2010, Dalerum and Miranda 2016, Pienaar et al. 2017). Following concerns raised within the Scientific Authority of South Africa in 2009 and the subsequent request from the Minister of the Department of Environmental Affairs, a task team consisting of scientists with a diverse range of skills and expertise was established by the Scientific Authority in February 2013. The purpose of the task team was to both identify and assess the full range of potential risks to biodiversity and the biodiversity economy, and to compile a report for submission to the Scientific Authority. The Scientific Authority, in accordance with section 61 of NEMBA, would in turn advise the Minister on appropriate, if required, policy and regulatory responses.

As the intensive breeding of game is a relatively recent phenomenon, certainly when measured in ecological or evolutionary timeframes, many of the impacts may not yet have appeared, measured, or not have yet been measurable. Therefore, assessing the risks is not just a case of documenting existing case studies, but requires some anticipation of impacts based on a broader understanding of biology and ecology and/or extrapolation of examples from other species or environments (including agriculture). Two broad categories of potential risks were identified, namely Biodiversity Risks and Biodiversity Economy Risks. Slightly different approaches and methodologies were used for these two categories to make provision for the integrated nature of potential risks to the biodiversity economy that often entails social, economic and environmental aspects. Using the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services' (IPBES) conceptual framework, the most prominent potential biodiversity risks of selective breeding and intensive management of game to South Africa's biodiversity were assessed. Seven potential risks/issues were identified using the best available scientific literature, information obtained from members in the wildlife sector, biodiversity conservation experts, and a national dialogue process (Njobe and King 2016). From these seven potential risks, 17 potential impacts (harms/stressors) were described with specific concerns highlighted under each impact. Subsequently, each impact was assessed and scored on the empirical quality of evidence available (extent of the evidence and level of expert agreement), the probability of occurrence within the industry, and the likely impact at an ecosystem and species level respectively. The quality of the evidence was evaluated for scientific rigour using the 'uncertainty approach' as used by the UK National Ecosystem Assessment. This approach consists of a set of uncertainty terms derived from a 4-box model and complemented, where possible, and placed on a 'likelihood of manifestation' scale (see Chapter 4). A hierarchical ranking method was used to rank the impacts on a gradient from highest to the lowest impact at an ecosystem and threatened species level respectively. All Least Concern species were omitted as the assumption was made *a priori* that present the impact on these species is unlikely. However, it is acknowledged that over time, depending on the scale, these risks and concerns may affect even these species. To determine the impacts with the highest potential risk at an ecosystems level, impacts with a score of 1 (Virtually certain) or 2 (Likely) were used. This was followed by ranking the selected impacts according to quality of evidence, only selecting impacts with a score of 1 or 2, and then lastly on the probability of occurrence within the industry. A similar process was followed for assessing the risk to Threatened or Near Threatened species.

Although the focus of this report is primarily on direct risks to biodiversity, the biodiversity economy and its potential in addressing socio-economic challenges of the country, it is an important focus area of government. Social and economic risks related to the biodiversity economy, may pose indirect risks to biodiversity and its contribution to the broader economy. One issue and two impacts relating to the practice of intensive and selective breeding of game on the biodiversity economy were identified based on current events within the biodiversity economy. A similar process to IPBES was used to gather evidence and consider level of agreement. Due to the integrated nature of Biodiversity Economy Risks, it could however not be weighted using similar criteria as for the Biodiversity Risks.

Each of the issues, impacts and findings identified in the report chapters are summarized in Table 3. The result of the hierarchical ranking is presented in table 5 in Appendix 1. In summary, it is concluded that the practice of selective breeding and intensive management of game pose several risks to South Africa's biodiversity.

Table 4: Summary of issues, impacts, concerns and key findings on the biodiversity risks related to the intensive and selective breeding of game species in South Africa.

Biodiversity risks			
Issue	**Impact**	**Concerns**	**Key findings**
1. Intentional breeding for selected traits	1.1 Expression of deleterious attributes that may lead to physical, behavioural and lethal outcomes	Breeding practices, such as inbreeding, line breeding and artificial selection for specific phenotypic traits, which increase the physical expression of rare alleles, may lead to conditions that could compromise the wellbeing of the individual animal. Where deleterious co-segregating traits are linked to selected genes, i.e. colour genes or horn length, and are non-lethal, they can be transmitted to other individuals within the subpopulation, and ultimately to several subpopulations as a consequence of translocations, increasing their occurrence within the broader population (see Chapter 5). These deleterious traits could lead to lower reproductive potential, as well as an altered ability to adapt to environmental change.	Many genes control complex mammalian traits such as coat colour and ornaments. The expression of these genes, the interactions among them as well as genotype-phenotype-environment interactions need to be investigated for the novel colour variants in the South African wildlife industry. This is imperative for the well-being of the individual animals and for understanding the short-term and long-term consequences for the industry and for the wildlife species involved. Even though there has been little work done on the genetic basis for colour transmission in African game species, it is well established that the selection of specific traits through a process of inbreeding or otherwise is very likely to lead to the expression of recessive deleterious attributes that may lead to physical, behavioural and lethal outcomes. It is further virtually certain that breeding practices, such as inbreeding, line breeding and artificial selection for specific phenotypic traits, for example colour variants, are taking place within sectors of the wildlife industry. The extent of the impact is likely to be limited to the individual and the specific population if no translocation or selling of individuals takes place. However, where individuals that carry deleterious genes are translocated to other populations the extent of the impact may increase to species level. This will depend on the pattern of inheritance and the number of animals in a larger population that possess a specific colour allele. Breeding with the wild type phenotype and unrelated individuals will decrease homozygosity at other loci (and potentially at the loci for e.g. colour) but depending on the breeding success of the animal with the trait, the trait may still persist in the new population. Theoretically we predict that there will be dilution. The level of impact of these concerns will depend on the potential of the affected individual to reproduce. Where the impacts of the deleterious trait are such that it prevents or significantly lowers the potential of the individual to reproduce, the impact on the broader population would be insignificant. However, the impact on the individual may be high.

	1.2 Loss of genetic and allelic diversity resulting in decreased fitness and reduced adaptive potential.	Removal of the process of natural selection, including mate selection and selection by differential mortality will reduce the evolutionary potential of populations to adapt to environmental change, especially in light of environmental and climate change. Using a small subset of the available gene pool (number of founders) and deliberate inbreeding thereafter can result in the fixation of certain genetic traits. Deleterious mutations will tend to accumulate, because selection is less effective in small populations and likely to be less effective in captive populations protected from natural selection pressures. Both the founder effect and inbreeding result in the loss of allelic diversity (loss of rare alleles). In the short-term inbreeding depression can affect birth weight, survival, reproduction and resistance to disease, predation and environmental stress; in the long term it reduces the evolutionary potential of populations to adapt to environmental change.	It is well established with a high level of agreement in the scientific literature that a loss of genetic diversity is highly likely to result in decreased fitness and in the long term reduce the evolutionary potential of populations to adapt to environmental change. It is further virtually certain that inbreeding and line breeding are used in the wildlife breeding sector as methods to increase certain rare phenotypic characteristics in animal populations and that several colour variant populations were established from very small founder populations (Needham and Hoffman 2013). The extent and severity of the impact will be related to the proportion of animals of a particular species that are in intensive breeding facilities versus the wild. The risk is especially high for species with low population numbers in the wild, but much lower for common or Least Concern species. The highest level of impact will be on the individual exposed to these practices. We cannot accurately predict how species will respond to future challenges. However, implementation of sound practices should safeguard populations or species in future. Monitoring of aspects such as genetic diversity, disease risks and outbreaks, etc. should be implemented.
	1.3 The mixing of genes from naturally separated gene pools leading to the breakdown of natural evolutionary processes and/or possibly leading to outbreeding depression.	Animals that may be less adaptable to the current environment due to the loss or gain of genetic traits. Hybrid subpopulations may have a greater probability of extinction as they may be less adaptable to their current environment. Hybrid subpopulations may have negative impacts on native species through introgression.	There is good evidence that with the expansion of the wildlife industry over the past few decades, there has been increased human-mediated movement (translocation) of animals within and outside their natural distribution ranges, with unclear consequences for the species themselves or their impact on other biota (Castley et al. 2001, Spear and Chown 2009, Taylor et al. 2015). Despite many decades of research, scientists are only now coming to grips with understanding local adaptation and species responses to various impacts (e.g. (Pertoldi et al. 2007, Reusch and Wood 2007). Thus, the concept has been established in the literature but there is still a lack of evidence. Until empirical evidence is available, one should always use a precautionary principle – some actions cannot easily be undone (e.g. introgressive hybridization leading to extinction of a parental gene pool). The duration of this impact, as evidenced by the bontebok (van Wyk et al. 2017) and wildebeest (Mackey 2009, Grobler et al. 2011) examples, can be considered permanent and will impact the entire species.

	1.4 Physiological stress as a result of poorly adapted animals to their current environment.	Colour patterns of many species assist in their ability to adapt to their environment and play a role for example in camouflage and thermoregulation. The artificial selection for colour variants, such as black springbok in arid environments, may lead to increased physiological stress as a result of the increased cost of thermoregulation. Movement of animals to habitats outside their natural environmental tolerance may lead to physiological stress and lower performance.	When specific traits, such as coat colour, are selected using artificial selection, the adaptive value of the trait is seldom considered. This may have unforeseen consequences and is likely to counter natural selection pressures that adapt an animal to its environment. It has been established in the scientific literature that colour variation is likely to influence an animal's thermoregulation. However, the physiological consequences of different coat hues are poorly understood and both positive and negative consequences have been documented depending on the environmental conditions. It has also been established that colouration may influence camouflage and social interactions such as mate selection. However, evidence to support this is still limited and further research is recommended. The probability of colouration affecting the thermoregulation of an individual and as a consequence the productivity of the animal is likely for as long as the animal is kept outside of its natural environmental tolerance.
	1.5 Domestication of wild species resulting in a loss of their natural ability to adapt to wild conditions.	Process of domestication that in the short term leads to the habituation of animals to humans, but in the long term leads to the selection for more timid animals that adapt better to a captive environment and might be less adaptable to wild conditions. Erosion of the social structure and behaviour of intensively-bred animals over time resulting in a loss of their natural ability to adapt to wild conditions i.e. predator naivety. Resource selection and the inability to adapt to changing environmental conditions i.e. droughts.	It is well established in the scientific literature that over time domestication results in diverse phenotypic and behavioural changes to wild animals, including decreased flight responses, increased sociality, earlier reproduction, and modification of endocrine and metabolic systems. The probability that the process of domestication will take place within intensive breeding facilities is virtually certain and the impacts or effects of domestication are likely to be permanent with respect to the individuals within intensive breeding facilities. However, the severity of this activity will be related to the proportion of animals of a specific species that are intensively bred versus the wild (more severe for rare animals), as well as the time frame that the individuals are subject to these conditions.
2. Impacts on wild populations through unsustainable movement of animals from the wild into captivity, introduction and genetic introgression of genetically altered animals into wild	2.1 The natural genetic composition, evolutionary trajectory and adaptive potential of wild populations is compromised as a result of deliberate or accidental introductions of captive populations/animals which have undergone genetic changes.	It is expected that intensively-bred specimens will differ from wild populations as a result of different selection pressures (see Chapter 4). Animals that escape from these intensive breeding facilities could have direct and indirect negative impacts on wild populations. It is further expected that these farmed specimens will have severely reduced life-time fitness compared to wild counterparts with intermediate hybrid fitness. It is not just the target species i.e. those that are intentionally bred, but also parasites associated with	The evidence to support the validity of this impact is established but incomplete as it relates to the wildlife industry. The assortative mating between captive and wild animals may mean that the genetic, behavioural and morphological differences exhibited in captive-bred specimens are not necessarily passed into wild counterparts. However, it may result in captive-bred individuals becoming more dominant within the population and thus wild counterparts are eventually reduced. It is a virtual certainty that animals are and will be introduced from captive facilities into extensive systems (the stated intention is to hunt such animals). However, the impact of these introductions are uncertain at this stage as well as the scale at which these may occur. The evidence presented suggests that introductions of captive-bred specimens, which have an altered genetic composition, is unlikely to impact the broader biodiversity but has the

populations and increased risk of introduction of species to habitats where they do not naturally occur.		these animals that may have undergone genetic modification. The release of such parasites into the wild may impact upon hosts, which are not adapted to these parasites (see Chapter 8). The number of animals with selected traits may become dominant in the wild through sheer mass of presence as the industry grows. The morphological changes that are present within intensive and/or selective-bred individuals do not allow for the species to survive within the wild. These may be as a result of the reduced ability to feed correctly or survive in sub-optimal conditions in comparison to captivity. This change in the genetic composition may result in populations that are unable to adapt to environmental changes and consequently face an increased extinction risk.	potential to impact rare and threatened species over time. The individual welfare concerns are considered high and will need to be monitored over time, especially when the evidence confirms that individuals bred in captivity do undergo certain changes in relation to behaviour, morphology and physiology and thus captive-bred game species may exhibit different traits to their wild counterparts. The direction in which these changes occur is also influenced by the selective and methodical management actions undertaken at each captive facility. Appropriate interventions from permitting authorities within the various provinces will be required to ensure that such changes do not negatively impact the species. Furthermore, this low frequency of breeding between captive and wild individuals may also mean that any deleterious genetic changes acquired in captivity and expressed in their offspring, may result in a reduced fitness and thus lower probability of survival of these specimens in the wild. The potential for genetic issues to manifest themselves when captive-bred animals are released into the wild is probable and thus a degree of caution and understanding is required. This is an area which would require further research and investigation in terms of game populations in South Africa. It is noted that animals undergo morphological changes as a result of captivity. The impact of small founder populations, often as a result of the cost associated with establishing an initial population, and the selective management actions associated with such, results in an exacerbated time scale. These morphological changes result in animals which may no longer be able to survive in the wild and thus the concern of such being released into the wild being no longer relevant. This does impact upon the individual's contribution to the conservation status of the species. In order for such species to be released into the wild successfully, suitable management actions need to be established within captive facilities to limit these morphological changes.
	2.2 The removal of wild specimens of naturally rare species or species with currently small population sizes in South Africa or other African countries can lead to population declines resulting in a lower overall conservation	A number of high value game species are being captured from the wild and brought into intensive breeding facilities. For species with small population sizes in the wild or rare species, the continuous sourcing or "leakage" from wild populations will reduce wild population sizes and can increase extinction risk of the species.	The evidence on this impact, specifically with regard to species in the South African wildlife industry (e.g. roan, sable, blue duiker, oribi and cheetah) is established but incomplete (high agreement based on limited evidence). There is a certain level of continuous "leakage" of wild animals into captive facilities (for commercial purposes), and this activity may have an impact on some species where removals are at unsustainable levels. Some species that are rare or threatened due to small population size, are very sensitive to even small removals from the wild populations. The probability of this activity/impact occurring in the industry is virtually certain, but the likelihood of an impact on

	status and a higher extinction risk for these species.	For species that are not very successful breeders in intensive facilities, new wild-caught individuals have to be regularly brought into intensive breeding facilities. This has a negative impact on the free-roaming or wild populations. For certain high value species, it is cheaper to source animals from other African countries for intensive breeding in South African breeding facilities. This may have a negative impact on populations in those countries as well as the overall conservation status of those species.	broader diversity is uncertain, as it is a scale issue and will depend on the current status of the species in the wild and the sensitivity of the population to removals. It is highly likely that there will be an impact at the species level, and where species disappear from the system completely, the impact is magnified through the disruption of the targeted species' ecological function in the landscape.
	2.3 The removal of wild specimens of naturally rare species or species with currently small population sizes, in South Africa or other African countries can lead to population declines resulting in a lower overall conservation status and a higher extinction risk for these species.	A number of high value game species are being captured from the wild and brought into intensive breeding facilities. For species with small population sizes in the wild or rare species, the continuous sourcing or "leakage" from wild populations will reduce wild population sizes and can increase extinction risk of the species. For species that are not very successful breeders in intensive facilities, new wild-caught individuals have to be regularly brought into intensive breeding facilities. This has a negative impact on the free-roaming or wild populations. For certain high value species, it is cheaper to source animals from other African countries for intensive breeding in South African breeding facilities. This may have a negative impact on populations in those countries as well as the overall conservation status of those species.	The evidence on this impact, specifically with regard to species in the South African wildlife industry (e.g. roan, sable, blue duiker, oribi and cheetah), is established but incomplete (high agreement based on limited evidence). There is a certain level of continuous "leakage" of wild animals into captive facilities (for commercial purposes), and this activity may have an impact on some species where removals are at unsustainable levels. Some species that are rare or threatened due to small population size are very sensitive to even small removals from the wild populations. The probability of this activity/impact occurring in the industry is virtually certain, but the likelihood of an impact on broader diversity is uncertain, because it is a scale issue and will depend on the current status of the species in the wild and the sensitivity of the population to removals. It is highly likely that there will be an impact at the species level, and where species disappear from the system completely the impact is magnified through the disruption of the targeted species' ecological function in the landscape.
3. Significant increases in the extent of impermeable fences with associated	3.1 Fragmentation of the landscape through impermeable fencing restricts movement of free-ranging	Intensification of impermeable fences fragments the landscape and has a range of negative ecological impacts. Impermeable predator proof fences for high value game species reduce the habitat available for free-ranging populations of threatened species such as	Fencing in general may have both positive and negative outcomes. Fencing in general whether for game farming or agricultural purposes is a permanent fixture in the South African landscape from a local (property) to a national scale. Impermeable fencing is widely regarded as undesirable and the evidence for that is well established. It is virtually certain that fencing for intensive game farming practices is on the increase, and with increased impermeable fencing

negative biodiversity impacts.	species and reduces habitat availability.	wild dog (*Lycaon pictus*), cheetah and pangolin (*Smutsia temminckii*). Impermeable fences are often electrified and designed in a way that leads to the unintentional mortality of non-target species.	comes greater fragmentation of the landscape. For intensive breeding projects (smaller parcels of fenced land) the length of fences per unit area is higher than for extensive areas. The full extent of such fences (smaller parcels of fenced land) is not fully known at this stage. It is highly likely that fragmentation will have a negative impact on broader biodiversity and especially on free-ranging threatened species such as cheetah, wild dog and pangolin, which could lead to reduced population performance, possible local extinctions and a declining conservation status of the species. Poorly designed electrified fences have been identified as a key contributor to the mortality of non-target species. Electrocution and entanglement are the main causes of mortality. Scientific studies on the impact of impermeable fences specifically for smaller parcels of fenced land do not exist.
	3.2 High concentrations of animals in small areas with impermeable fences for intensive breeding purposes result in habitat degradation within such areas.	Unusually high densities of animals in small breeding camps may cause overstocking resulting in overgrazing and increased trampling effects, leading to habitat degradation and loss of plant species diversity. Overgrazing and trampling in small camps may result in erosion and loss of soil. Severe grazing patterns may result in an increase of undesirable woody species and some poisonous plant species. Transformation of natural veld to planted pastures with a homogenised structure and composition. Excessive or complete removal of certain vegetation strata such as the woody component from intensive breeding camps. Land intensification practices negatively affect ecosystem functioning and services.	A large proportion of game ranches in South Africa practise intensive breeding of game. They confine these high value species or colour variants in relatively small camps, and it is estimated that the area of game farms currently under such camps is about 6% (or 1 022 785 ha), and it is virtually certain that it will increase in the future. At present, because the activity is largely unregulated, little is known about the number of animals and camp sizes in which high value animals are kept. Although there appears to have been no monitoring of the impacts in these small camp systems in South Africa, there is well established evidence to show that animals kept at densities higher than ecological capacity have a negative impact on veld condition and productivity, habitat integrity and species diversity. It is not clear to what extent supplemental feeding mitigates the impacts related to high grazing intensity, but this will not mitigate the impacts of trampling and hoof action of animals kept at high densities. It is anticipated that a similar trend to what has happened in the Little Karoo with ostrich will happen in other biomes in relation to the intensive production of game, with degradation of habitat in game camps (already estimated to cover more than one million hectares with exponential growth forecast), and with additional areas being required over and above those used for traditional agriculture for the production of feed for the rapidly growing captive population. Some breeders of colour variants do keep their animals in more extensive systems, and from a veld management perspective, there should be no negative impacts if this is within the ecological capacity of those environments. It is highly likely that homogenisation, and therefore degradation of habitats, irrespective of the way in which it occurs, generally will have a negative impact

			on biodiversity. To date very little research has been conducted in this field at the scale where concerns have been raised but it is highly likely that sensitive and threatened habitats and non-target species will be adversely affected. Target species farmed with are unlikely to be affected.
4. The intensification of management practices and subsequent control of species that are likely to impact negatively on the commercial objective of breeding programmes.	4.1 The killing of predators and other conflict species may result in a reduction in population numbers or elimination from certain areas with limited opportunity for recolonisation, which in turn may lead to a change in the conservation status of the species and thereby furthering the extinction risk of these species.	The high economic value of species bred in intensive systems is likely to perpetuate the problem of indiscriminate killing of perceived damage-causing animals. Predators are not the only species deemed to increase human-wildlife conflict within the game ranching sector. Species that may be the cause of increased management interventions and potentially lower profit margins may also be targeted through systematic control and removal. The population sizes of large apex predators are naturally small and as such the indiscriminate and uncontrolled killing of these species is likely to result in population declines, thereby furthering the extinction risk of the species. Declining apex predator populations limit sustainable utilisation opportunities and thereby reduce the incentives for landowners to conserve such predators. Removal of top-order predators from systems can have effects at an ecosystem level, cause trophic cascades and meso-carnivore release. This in turn can cause increased human-carnivore conflict and increased losses, in particular to sheep and goat farmers, which are more vulnerable to meso-carnivore pressure.	It is well-established and virtually certain that the destruction of predators and other 'problem animals' is unlikely to decrease within the current economic climate and high value of intensively- and selectively-bred game. The return on investment in intensive breeding and breeding for colour variations is still such that entrants into the market will continue to invest and thereby ensure that predators or other species likely to impact upon animals considered as an investment, are removed from the system. Bothma (2005) cited in Boast (2014) further suggests that the low breeding rates and decreased survival of colour phenotypes may result in blame being attributed to carnivore damage and thus increased intolerance towards these species. It is well-established that the intensive management and control of predators is occurring within the wildlife industry, and will continue as long as these activities are undertaken. However, the impact on the target species will be evident long after the activities are no longer considered viable. It is further likely that these actions will occur throughout the areas in which these activities are undertaken. The removal of predators or considerable reduction in numbers will likely have an impact upon the conservation status of these species. However, it is the responsibility of both conservation agencies and landowners alike to ensure that this is avoided. Pitman et al. (2016) indicates that the current leopard population in the Limpopo province is declining as a result of various factors – illegal offtake (snaring, poisoning and illegal hunting), hunting and those removed as damage causing animals. It is not feasible to conserve large predators only within the confines of formal protected areas and thus landowners within the broader agricultural landscape need to become custodians of such species rather than adversaries.
	4.2 The disruption of social structures of species targeted for removal may exacerbate the conflict potential, as a	The constant removal of predators has the potential to disrupt social hierarchies of other predator species. This in turn may exacerbate conflict leading to further declines of predator populations.	The negative attitude towards predators within the agricultural landscape has resulted in previous predator removals and translocations being undertaken as a form of control, a scenario that still occurs and thus the evidence is well established to support this impact. These 'control' options of predators from

	result of the constant removals of individuals as well as within the receiving environments. This in turn could lead to a decline in the survival rate of the affected population. Constant removals of dispersers may further lead to a loss or disruption of dispersal opportunities, thereby increasing the local extinction risk. Relocations may increase conflict and reduce reproductive performance and increase mortalities.	The survival of removed problem predators may be low in the receiving environment and may result in increased intraspecific competition as well as increased losses in the receiving environment with the decreasing availability of safe areas for relocations. The constant removal of individual predators from a particular area may result in the disruption of a species social behaviour over a broader area. The continued translocation of problem animals, either predator or other species impacting upon an intensive breeding operation, may disrupt the ecological processes within the receiving environment. Existing populations within the receiving environment may be forced to utilise more resources defending a particular territory than it would have prior to such introductions. This increased resource demand may impact upon breeding success.	areas in which intensive game ranching occurs, is unlikely to decrease and will probably increase (Pitman et al. 2016), considering that predator management has been identified as a key management intervention to ensure the return on investment is protected. Thus, the occurrence within the industry is virtually certain. The impact upon broader biodiversity is likely as predator removal may create an unnatural predator hierarchy and the mesopredator release theory is well established and the probability of increased conflict between these predators and landowners is high. The impact of predator removal upon the species and individuals is highly likely as a new competitive environment is created. Translocation of problem predators may not always be the most desired option. However, public perception is often the deciding factor. In the event that problem predators are to be translocated, success should be measured by the resolution of the problem caused by the animal and not necessarily the survival of the animal post release. Settlement in the recipient area and reproductive output (Massei et al. 2010) as success may depend on the age and sex of the individual, which factors are often outside the control of mitigation relocations. Post release monitoring at both the release and capture site will assist in determining if the problem has been resolved, or in fact additional problems created, as well as ensuring that the same or additional problem has not been created in the release area. In order to ensure that the translocation of predators are not detrimental to both environments it is important to ensure that suitable management actions are implemented. Translocations are unlikely to be undertaken by private landowners that may prefer to initiate lethal control, and thus the impact of such will last as long as such actions are not coordinated correctly. The potential for conservation agencies and NGOs to manage the translocation of predators, and associated management actions, will be reliant on the necessary funding, the source of which is not always known or sustainable. Lindsey et al. (2005) compared the costing of maintaining wild dog populations on ranchland in relation to establishing a reintroduced population. Although the costs for ranchlands can be suitably offset against the eco-tourism value of such, reintroduction costs are often prohibitive. Thus, without the necessary funding, translocations will not take place or the impact thereof will not be managed. The impact of inappropriate predator translocations and/or lethal removals may only manifest themselves long after the event and possibly when further interventions are no longer possible.

	4.3 The removal of predators will at a certain scale disrupt predation as a natural process in the broader landscape/environment thereby affecting ecosystem functioning and non-target species.	The disruption of natural predation interactions, through the exclusion of apex-predators from the agricultural landscape may result in an increase of smaller to medium-sized predators. This in turn may lead to a disproportionate impact upon smaller prey species, also potentially increasing human-wildlife conflict. The removal of predators from a system may also impact upon the prey's inherent fear of predation over time. This may result in the species' ability to avoid predators becoming diminished over time. The removal of key species may disrupt certain ecological processes that assist in maintaining equilibrium in the system. This in turn leads to an increased management intervention thereby placing further strain on the system. These increased interventions in turn impact upon non-target species. Ecological processes take place within a broader landscape context and thus affect more than the cadastral boundary of any one breeding operation. Thus, the management interventions increase throughout the landscape as a result of the disruption in one particular area. Historically, areas in which large livestock production was maintained, contributed to predator conservation objectives, yet the increase of intensive and selective breeding operations is taking place within these same areas, thereby reducing the ability to contribute to predator conservation as a result of changing management actions.	The evidence is well established to support this impact, yet the extent to which the removal of predators has on ecosystem functioning within the South African context does require further investigation, but is highly likely that removals will impact biodiversity in general. The removal of predators from the system is virtually certain to occur as has been highlighted by Pitman et al. (2016). The removal of individual predators is unlikely to have an impact, but it is reasonable to suggest that the impact on the species will be present as long as the intolerance of predators within the landscape is present and that a holistic approach to predator management is not employed. The result of the removals of predators from the environment will result in further conservation interventions being required to ensure that population declines are halted – the moratorium of leopard hunting in 2016 and 2017 being an example of such. This in turn has the potential to further increase the intolerance of landowners to the remaining predator populations. The ability of predators to assist in the removal of weak and or injured animals from a system can be considered beneficial to landowners that operate within extensive systems, yet the benefits thereof (such removals result in little or no impact upon healthy individuals) are often not promoted. The impact of predator removal, either apex or meso-predator, and trophic release is a key research question that should be undertaken as a matter of urgency. It most probably occurs in game ranching areas, but the extent is not quantifiable. The potential risk for ecosystem health and functioning is high with possible long-term consequences for not only conservation but also the economic viability of agriculture.
5. Improper use of stock remedies (animal health products) and veterinary	5.1 The development of resistance to stock remedies and veterinary medicines resulting in microbes, helminths and ectoparasites	Resistance to stock remedies, such as anthelmintics, ectoparasiticides and anti-microbials is of global concern. The liberal and improper use of these commodities in the intensive breeding industry (by both game breeders and veterinarians) is likely to	The evidence of development of resistance to stock remedies and veterinary medicines is well established in the agricultural and biological literature, but limited studies have been undertaken in the context of intensive breeding, although at least one study in South Africa has demonstrated the development of resistance under treatment regimens commonly used and various experts

medicines resulting in the development of both resistance to these products by parasites and to the loss of disease and parasite resistance in host populations.	that may start infesting free-roaming game and livestock on a large scale with conservation and economic consequences.	cause a rapid genetic shift to resistance in worm, ectoparasite and microbe populations. Should these resistant populations spill over to livestock the economic consequences for the livestock industry are likely to be significant. There could be a regulatory response leading to increased restrictions on the movement of animals which would affect both the intensive and extensive wildlife farming/ranching industry, and which may have negative impacts on the conservation contribution of this industry. As one of the objectives of intensive and selective breeding is to produce animals for the hunting industry this necessitates releasing intensively-bred animals with high probability of containing resistant parasites into wild conditions with other wild indigenous species. It is likely that the resistant strains would be transferred to wild animal populations and it is possible that these animals may be more impacted by resistant worm, ectoparasite and/or microbes as they have not evolved with such organisms. These wild populations may also directly or indirectly transfer resistant parasites to livestock.	and industry role-players have expressed concern about the development of parasite resistance. It is virtually certain that the risk factors for the development of parasite and disease resistance occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. Whilst there has been no assessment nor current evidence of the role of intensive breeding on the spread of resistant parasites to the broader agricultural and natural systems it appears likely that this could happen. It is unknown whether resistant parasites would be more damaging to natural populations of wildlife than non-resistant strains, but it is likely to have direct negative production and economic consequences for livestock agriculture and other intensive breeding operations, and (in the absence of interventions) could have indirect negative consequences for the broader wildlife economy and biodiversity conservation through increased agricultural regulatory oversight and restrictions placed on the movement of game. It is possible (about as likely as not) that game species, where a large proportion of the population is contained in intensive breeding facilities, could be affected through the development and spread of resistant parasites, but unlikely to be an issue for species where a significant wild population undergoing natural selection still exists (unless resistant parasites and diseases have a more negative effect than non-resistant strains). It is very likely that individual properties where parasite resistance has been established will be more difficult or costly to continue with intensive breeding, or to get back into agricultural production should the need arise.
	5.2 Disruption of the process of natural selection in terms of host-parasite evolution with resulting loss of disease and parasite resistance within the game population.	In nature, parasites and host animals are in a continual evolutionary 'arms race', and parasites and disease are very important components of natural selection. Host animals in the wild and in agricultural production systems will always be exposed to parasites and diseases. In many cases early exposure to parasites and diseases results in the development of individual and herd immunity. In nature, differential mortality (natural selection) results in perpetuation of genetic lineages of species more resistant to such diseases and parasites. The nature of intensive breeding (confined areas, desire to minimise mortality and produce bigger and faster growing animals) necessitates and incentivises	The evidence for loss of disease and parasite resistance within the game population is established but incomplete. There is a strong theoretical basis for expecting the loss of resistance over time, based on the removal of natural selective processes. It is virtually certain that the risk factors for the development of loss of resistance at an individual and herd level to parasites and diseases occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. It is unlikely that loss of resistance in populations kept under intensive conditions will have broader biodiversity or species impacts as any animals that escape or are released into wild conditions under natural selection are likely to suffer higher rates of mortality. Translocation of intensively-bred animals onto other properties is very likely to result in receiving properties implementing the same parasite and disease controls that initially lead to both parasite resistance to stock remedies and veterinary medicines and loss of disease and parasite resistance in game populations (see Chapter 9), thereby further exacerbating

		intensive and ongoing control of helminths and ectoparasites. The control of helminths and ectoparasites in intensive breeding operations compromises the ability of game animals in both the short and long term to cope with parasitic organisms, leaving the individuals and populations vulnerable to a continuous shift towards lack of resistance to parasites. The process of inbreeding, used to maximise the expression of human-desired traits, may reduce the genetic diversity and hence disease and parasite resistance of animals. In intensive breeding operations natural selection for disease and parasite resistance will almost certainly cease to operate. Such game animals are likely to suffer high levels of mortality when/if released into the wild and will have to be farmed like livestock with continuous disease and parasite management, further entrenching the domestication process.	and accelerating both of these impacts. It is highly likely that species where a large proportion of the population is contained in intensive breeding facilities will lose resistance and be at risk, but unlikely to be an issue for species where a significant wild population undergoing natural selection still exists, unless selected phenotypic traits are positively linked to fitness in the wild, while having linked to genes resulting in reduced resistance (unknown but unlikely). Natural selective processes are increasingly being removed or moderated on properties outside of intensive breeding properties, in which case it is more likely that non-resistant individuals will pass on their genes to wild populations making the broader population more susceptible to periodic disease or parasite outbreaks.
6. Improper use of stock remedies (animal health products) and veterinary medicines resulting in risks to consumers.	6.1 Venison from intensively-bred game may be contaminated by antimicrobials, ectoparasiticides, anthelmintics and/or anti-inflammatory agents, thereby posing a health risk to humans; intensively produced venison may not be as healthy as wild venison, potentially damaging the brand image and value of extensively produced venison.	Game animals in intensive game breeding operations are exposed to anthelmintic and ectoparasiticide treatments, and sometimes antimicrobials that are very seldom necessary and rarely administered according to prescribed dosage rates. In addition, the desire for animals with larger bodies and horns (current) and faster growth rates (linked to plans for maximising meat production) may promote the use of growth enhancing supplements in game feed. This may result in undesired (qualitative and quantitative) residues of such chemicals in venison that is marketed locally and internationally. Should undesired residues be identified in foreign markets it could severely jeopardise South Africa's position as an exporter of venison, placing further risk on the extensive wildlife ranching industry. The stated intention of the industry is to move towards intensive venison production. The high	The evidence for health risks to humans or damage to the brand image of venison is established but incomplete. It is virtually certain that current management practices adopted within a proportion of the industry could contribute to this risk. However, it is presumed that little venison is entering the formal market from intensive breeding sources, although increasing volumes of meat from intensively-bred animals offered for put-and-take hunts may be increasing the health risk for hunters and staff. In future there are plans to produce significant volumes of venison under intensive conditions for the national and international markets; if issues of meat safety are not properly managed it is likely that the damage to the brand of South African venison will have a negative effect on the profitability of extensive game ranches (largely compatible with biodiversity objectives) thereby indirectly likely to have negative biodiversity impact. Species or population level impacts are unlikely, other than through the broader indirect biodiversity impact already discussed.

		stocking rates and high levels of physiological stress associated with such intensification will result in the ongoing need for use of stock remedies; there is some evidence that growth supplements such as beta-agonists are already being used to increase body and horn size. These may pose a risk to human health unless appropriately managed and monitored. An indirect risk to extensive game farmers is that they may be subject to increased regulatory oversight (and hence costs) as a result of the need to manage intensively-bred game meat entering the market; the value of their brand as "untainted by modern farming practices" will almost certainly be at risk in the medium to long term.	
7. Increased use and misuse of pesticides including herbicides, insecticides and acaricides as defined by the Fertilizers, Farm Feeds, Agricultural Remedies and Stock Remedies Act, Act No. 36 of 1947 and hazardous substances (toxins) as defined by the Hazardous Substances Act, Act No. 15 of 1973 and excluded from Act No. 36 of 1947.	7.1 Off-label use of pesticides and unlawful use of hazardous substances cause mortality of indigenous species resulting in changes in ecosystem functioning and increased threats to the conservation of threatened species.	The misuse of pesticides and hazardous substances to control damage causing animals in agriculture and wildlife ranching is common; as the trend of intensive breeding increases, there is concern that there will be an increase in the off-label use of pesticides and unlawful use of hazardous substances to protect investments in high value animals. There is concern that as the trend towards breeding small antelope increases, direct persecution of an additional suite of predators, including large raptors, may escalate. Concerns have also been expressed that the use of Avermectin-type anthelmintics, especially in game feed, poses an additional severe risk to dung beetles. There is an increase in the use of herbicides to eradicate trees and shrubs in overgrazed areas, along fence lines and in woodlands/bushveld where landowners endeavour to change the vegetation composition in breeding camps. Soil sterilant herbicides with a very long half-life and significant potential to leach laterally, such as bromacil and tebuthiuron, could be used by individuals without adequate knowledge of the potential impacts. Eradication of vegetation is also done with non-	The evidence for negative biodiversity impacts resulting from the off-label use of pesticides and unlawful use of hazardous substances is well-established. Owing to the illegal nature of the activity it is difficult to assess the probability and scope of misuse within the intensive and selective breeding sector of the wildlife industry, but instances have been documented, reported or strongly suspected. The frequency of off-label use is likely to be proportional to the growth of the industry, but the likelihood of it taking place at an individual property level is unknown. It is also known that off-label use of pesticides and unlawful use of hazardous substances take place in extensive wildlife and agricultural contexts. It is unknown whether the extent of occurrence is any higher in intensive and selective breeding operations, although there is good reason and some evidence to suspect that intolerance of predators is higher due to the higher values of the game (see Chapter 7). Hence it is anticipated that the use of poisons (as well as other methods of controlling predators) will be higher, but there are competing explanations. It is known from other work that only a very small proportion of carcasses in the landscape (0.4 – 0.7%) need to contain poisons to result in massive population declines and even extinction of vultures (Green et al. 2004). Any off-label use of pesticides and unlawful use of hazardous substances in the sector will have severe negative impacts on biodiversity and species, particularly on rare or threatened species and on the predator and scavenger guilds e.g. leopards, vultures. The clearing of vegetation, through chemical and/or mechanical means to enhance production could have significant impacts on biodiversity especially

		selective systemic herbicides such as glyphosate in sensitive areas like riparian zones and steep slopes. The scale of use and extent of habitat manipulation is expected to rise with the increase in areas allocated to intensive breeding and may in some instances trigger the requirement for environmental authorization.	where this takes place in threatened or under-protected ecosystems. In many cases the extent of modification could actually constitute indigenous vegetation 'clearance' in terms of the NEMA Listing Notice 3. It is essential that it is clear at what point vegetation modification triggers the regulations and where environmental authorisation is required, especially where it is motivated as managing 'bush encroachment'.
Biodiversity economy risks			
1. Intensive and selective game breeding practices may have collateral negative impacts on conservation and the broader wildlife economy.	1.1 Unclear distinction between wild and intensively- and selectively-bred game in the hunting sector may contribute to reputational damage and consequent negative economic and conservation outcomes for the broader wildlife industry.	Shooting of intensively- and selectively-bred game is perceived negatively by important stakeholders and poses a broader reputational risk to hunting and other sectors of the South African wildlife industry. Reputational damage resulting from shooting of intensively- and selectively- bred game may undermine the economic and conservation contributions of the hunting industry. Absence of mechanisms to communicate credible market information on the conservation contribution of game populations and hunting activities can compound the reputational risks to responsible hunting and game ranching.	With increasing pressures on natural resources, this assessment confirmed that there is mounting pressure from the public at large that enterprises should better demonstrate that their practices are in line with international principles of sustainability and that they are socially responsible in utilising the natural resource base. This assessment demonstrates that the various sectors within the wildlife industry have different levels of understanding and acceptance of these realities and principles in evaluating their own activities for future growth. It has been demonstrated that hunting is under increased public scrutiny, with animal rights groups using any possible opportunity to bring hunting in disrepute, with substantive risks associated with perceived irresponsible hunting practices for the individuals involved, enterprises and the sector as a whole. Role-players agree that this reputational damage to hunting can have significant negative implications for its sustained contribution to conservation, livelihoods and socio-economic development. Policymakers should be mindful that ignoring this risk can exacerbate the reputational damage that the industry already experiences with concomitant negative economic impacts for individuals, the various sectors and the entire wildlife industry. The majority of prominent, relevant local and international role-players in the hunting and conservation fraternities believe that breeding game intensively and selectively for shooting may conflict with hunting ethics, reputational risk and lack of conservation contribution. This is in line with internationally accepted guidelines that hunting should be seen as sustainable and socially responsible as captured in several internationally accepted codes of good practice, including guidelines developed by the IUCN. In all of these hunting codes, the management of source population and their habitat form an integral part of responsible hunting.

			Irrespective of the overwhelming negative response of hunting organisations worldwide to PHASA's change in policy in support of captive-bred lion hunting where facilities comply to the norms and standards of SAPA, the intensive and selective game breeding subsector does not appear to accept that breeding game intensively and selectively for hunting poses a reputational risk to hunting. Illegal hunting and animal rights activism are seen as the major causes for reputational damage. There is general agreement that the lack of a generally accepted mechanism to communicate credible market information on the practices of role players in the value chain can reduce reputational risk. Theory on reputation management and this assessment suggest that such a mechanism should address both the conservation contribution and sustainability of management practices of source populations and hunting activities to reduce reputational risks to like-minded role-players in the same value chain.
	1.2 Unregulated change in land-use from conservation compatible extensive wildlife areas to intensive game breeding operations reduces the effectiveness of conservation initiatives and raises conservation costs to government and society.	There is a lack of legal and policy instruments that quantify the nature and extent of intensive and selective breeding operations and their location in sensitive landscapes. The above-mentioned shortcoming impedes the ability of conservation agencies and government to factor in the impacts of the practice in conservation and land-use planning processes. The result is uninformed decision-making with potential detrimental conservation and economic implications.	The impacts of intensive and selective breeding on the resource base and biodiversity have been discussed extensively in this assessment. It has been demonstrated that the ecological footprint of these game breeding operations is very different from traditional extensive wildlife areas, even if the landcover may still be close to natural in some cases. Associated impacts such as habitat fragmentation, animals killed in fences and reduced tolerance by intensive game breeders towards free-ranging threatened predators, are of greater concern in near natural and sensitive environments such as areas adjacent to protected areas and wildlife corridors, than in areas already modified or zoned for intensive agricultural practices. The existing shortfalls in policy frameworks that govern land-use planning and environmental impact assessments result in a gap between the information used to inform policy processes and what is happening on the ground. This limits the ability of government to address the impacts of intensive and selective breeding operations on biodiversity associated with natural landscapes and extensive wildlife areas that: form the basis of wildlife-based tourism and hunting, the biggest contributors to the wildlife economy;

			form the basis of wildlife corridors, private sector contribution to national conservation targets and the national protected area strategy, wherein protected areas can function as drivers in the rural economy; and generate ecosystem goods and services that in South Africa amounts to R73 billion per annum, or equivalent to 3% of the country's GDP (SANBI, 2010) and that supports the economy. In the absence of a policy framework to prevent or at least mitigate these impacts, the direct and indirect costs to government may be significant.

BIODIVERSITY RISKS

The following issues were identified as key risks to biodiversity on an ecosystem and Threatened and Near Threatened species level, Ecosystems level only and Threatened and Near Threatened species level only (Table 4).

Table 5: Key biodiversity risks identified on an ecosystem and species level.

Ecosystem and species level risks	Ecosystem level risks	Species level risks
10. Off-label use of pesticides and unlawful use of hazardous substances cause mortality of indigenous species resulting in changes in ecosystem functioning and increased threats to the conservation of threatened.	6.1. Fragmentation of the landscape through impermeable fencing restricts movement of free-ranging species and reduces habitat quality and quantity.	4.1. Expression of deleterious attributes that may lead to physical, behavioural and lethal outcomes.
7.1. The killing of predators and other conflict species may result in a reduction in population numbers which in turn may lead to a change in the conservation status of the species and thereby furthering the extinction risk.	6.2. Concentration of species in small areas with impermeable fences for intensive breeding purposes results in habitat degradation within such areas.	4.2. Loss of genetic diversity resulting in decreased fitness and reduced adaptive potential.
7.2. The disruption of social structures of species targeted for removal may exacerbate the conflict potential as a result of the constant removals of individuals. This in turn could result in a decline in the survival rate of the affected population. This constant removal of dispersers may also create a loss or disruption of dispersal opportunities, thereby increasing local extinction risk.	7.3. The removal of predators will at a certain scale disrupt predation as a natural process in the broader landscape/environment thereby affecting ecosystem functioning.	4.5. Domestication of wild species resulting in a loss of their natural ability to adapt to wild conditions.
		5.1. Changes in natural genetic composition, evolutionary trajectory and adaptive potential of wild populations through the introgression of captive population genetics wherein genetic changes in the captive population may lead to an altering genetic composition and/or evolutionary trajectory and/or adaptive potential of wild populations through deliberate and accidental introductions.
		5.3. The removal of wild specimens of naturally rare species or species with currently small population sizes, in South Africa or other African countries where sourcing

		is often cheaper, can lead to population declines resulting in a lower overall conservation status and a higher extinction risk for these species.
		8.1. Development of resistance to stock remedies and veterinary medicines resulting in microbes, helminths and ectoparasites that may start infesting free-roaming game and livestock on a large scale with conservation and economic consequences.
		8.2. Disruption of the process of natural selection in terms of host-parasite evolution with resulting loss of disease and parasite resistance within the game population.

Those issues that were identified as key biodiversity risks are summarised below.

Increased use and misuse of pesticides including herbicides, insecticides and acaricides (Issue 7)

Use and potential misuse of poisons across agricultural and extensive game ranching systems is a significant risk to wildlife in South Africa and across Africa (Chapter 10). There is a growing concern that intensive and selective breeding of game will involve a significant increase in the use of herbicides, insecticides and acaricides, and other poisons (i.e. to eradicate predators). The evidence for negative biodiversity impacts resulting from the off-label use of pesticides and unlawful use of hazardous substances in South Africa is well established. Owing to the illegal nature of the activity it is difficult to assess the probability and extent of occurrence of this issue within the intensive and selective breeding sector of the wildlife industry, but instances of this have been documented, reported or strongly suspected. The frequency of occurrence is likely to be proportional to the growth of the industry, but the likelihood of it taking place at an individual property level is unknown. It is also widely known, and hence accepted, that off-label use of pesticides and unlawful use of hazardous substances occur in extensive wildlife and agricultural contexts. It is unknown whether the extent of occurrence is any higher in intensive and selective breeding operations. There is good reason and some evidence to suspect that intolerance of predators is higher due to the higher values of the game (see Chapter 7), and hence it is anticipated that the use of poisons (as well as other methods of controlling predators) will be higher, but there are competing explanations. It is known from other work that only a very small proportion of carcasses in the landscape need to contain poisons in order to result in massive population declines and even extinction of vultures. It is very likely that any off-label use of pesticides and unlawful use of hazardous substances in the sector will have severe negative impacts on biodiversity and species, particularly on rare or threatened species and on the predator and scavenger guilds e.g. leopards, vultures.

The intensification of management practices and subsequent control of species that are likely to impact negatively on the commercial objective of breeding programmes (Issue 6)

The trend within the game ranching sector to shift from high numbers of common game species requiring very little management intervention to the breeding of high value game species has resulted in increased management actions (Lindsey et al. 2009), such as an increase in predator proof fencing and the lethal control of nuisance wildlife through legal and possibly illegal destruction (Cousins et al. 2008, Taylor 2016). The presence of commercial game species and specifically high value species, represent an important agricultural asset, which generates substantial financial revenue and return for landowners (Pitman et al. 2016). A consequence of this is that landowners/ranchers are unlikely to tolerate wildlife, especially free-ranging predators that pose a threat to these assets.

Large carnivores are particularly vulnerable to persecution-related killing due to their large home-ranges that cross multiple land-uses, low reproductive rates, low population densities and their carnivorous diet that results in the predation of animals with high economic or recreational value (St John et al. 2011). Pitman *et al*., (2016) suggested that the increase in lethal control is disproportionate to the increase in intensive game farming methods, thereby proposing that intolerance of predators is increasing. This is likely to have significant and detrimental impacts on species persistence and the functioning of ecological systems (Ripple et al. 2014).

Although there are no accurate figures on the number of predators killed by landowners specialising in the intensive breeding of high value game, a number of scientific studies within South Africa has highlighted a low level of tolerance towards predators (Balme et al. 2009, St John et al. 2011, Thorn et al. 2012, Thorn et al. 2013, Pitman et al. 2016). This intolerance towards predators is further compounded by the presence of high value game species (Thorn

et al. 2013, Pitman et al. 2016). This practice in accordance with the risk-averse principle should be assumed to be material and significant in nature. Furthermore, conservation interventions through the translocation of predators may not be the proverbial 'silver bullet' required to reduce human wildlife conflict (Massei et al. 2010, Weilenmann et al. 2010). The resultant disruptions of predator and damage causing animal eradication within the landscape is likely to result in unintentional consequences, and in particular ecosystem imbalances, both within the breeding and receiving landscapes (where damage causing animals are translocated to other areas). These imbalances have been documented to have long-term impacts on ecosystem functioning and the integrity of landscape biodiversity.

The removal of predators will have an impact upon the conservation status of the respective species that are removed. Pitman *et al*. (2016) indicates that the current leopard population in the Limpopo province is declining as a result of various factors – illegal offtake (illegal hunting, snaring and poisoning), hunting and those removed as damage causing animals. Furthermore, the 2016 Red List Assessment for leopard highlights *inter alia* the increasing wildlife ranching industry as a contributing factor to the genuine change in the species assessment from Least Concern to Vulnerable (Swanepoel et al. 2016a).

Significant increases in the extent of impermeable fences with associated negative biodiversity impacts (Issue 3)

A large proportion of game ranches in South Africa practise intensive breeding of game. According to Taylor *et al*., (2015), 44.6% of surveyed ranches conducted intensive breeding of game with the total area of wildlife ranches under camps in South Africa (i.e. properties fragmented into sub-cadastral portions) estimated to be 10 228 km^2, representing 6.0% of the total area of the ranching industry and approximately half the size of the Kruger National Park. In addition, Desmet et al. (2017) showed that for an area in the upper Limpopo Valley near Thabazimbi, the number of properties assessed as being intensive breeding operations increased from eight properties (4% of total, n = 208) in 2006 to 17 (8%) in 2010, reaching 51 (25% of total) in 2015. On these ranches high value game species or colour variants are confined in high densities in relatively small camps. According to Desmet et al. (2017), median camp size decreased by 51.5% from 35.5 ha in 2006 to 17.2 ha in 2015 and the number of camps increased by 46% from 3 129 to 4 568. The landscape-level of intensification (i.e. total area comprising camps less than 50 ha in size) observed is greater (23%) than that currently estimated for the industry (6%) and this increase in impermeable fencing precipitates marked fragmentation of the landscape.

For intensive breeding projects (smaller parcels of fenced land) the length of fences per unit area is significantly higher than for extensive wildlife areas. The full extent of such fences (on these smaller parcels of fenced land) is not fully known at this stage. It is highly likely that fragmentation will have a negative impact on broader biodiversity and especially on free-ranging threatened species such as cheetah, wild dog and pangolin, leading to reduced population performance, possible local extinctions and an accelerated decline in the conservation statuses of these species. The design of electrified fences to enclose breeding camps has been identified as a key contributor to the mortality of non-target species. Unfortunately, scientific studies on the extent of the residual impact on biodiversity by the impermeable fences (and importantly for the smaller parcels of fenced land) does not exist. Observations and tacit records suggest that this impact may be significant at both a local and landscape level.

Because intensive and selective breeding is largely unregulated, little is known about the number of animals per camp and the camp sizes. Taylor *et al*., (2015), in their survey, compared potential versus actual stocking densities of game farms, and found that 32% of properties were over the recommended stocking density, while 5% of the total had more than twice the recommended biomass of grazers. The full extent of the impacts of this industry on biodiversity remains to be quantified. Although there appears to have been no monitoring of

the impacts in these small camp systems in South Africa, there is evidence that animals kept at densities higher than ecological capacity have a significant negative impact on veld condition and productivity, habitat integrity and species diversity (Fynn and O'Connor 2000, O'Connor 2005, O'Connor et al. 2010). It is not clear to what extent supplemental feeding mitigates the impacts related to high grazing/browsing intensity, but this will not mitigate, and may exacerbate, the impacts of trampling and hoof action. It is anticipated that a similar circumstance to that observed with intensive breeding of ostrich in the Little Karoo (Cupido 2005, Le Maitre et al. 2007) will extend into other biomes where intensive production of game takes place. Here it was concluded that intensive breeding of game is likely to lead to (a) significant fragmentation of the natural habitat (b) displacement of traditional agriculture (c) homogenisation of the landscape and (d) loss of predators and other non-target species, for example pangolin and tortoises and thus presents a significant threat to biodiversity, especially where these practices occur within sensitive and threatened habitats.

Intentional breeding for selected traits (Issue 1)

Potential impacts related to the intentional breeding for selective traits such as colour, or increased horn or body size considered in this report were 1) the expression of deleterious genes, 2) the loss of genetic and allelic diversity, 3) outbreeding depression, 4) physiological stress, and 5) domestication. The objective of commercial breeding programmes is to maximize the rate of genetic change for economically important traits. Where these traits are only expressed in recessive phenotypes inbreeding or line breeding is often used to maximize the genetic progress towards these traits. The benefits of inbreeding are increased uniformity, increased prepotency (ability to pass on traits to offspring) and “fixing” of desired traits and breed type. It is thus virtually certain that breeding practices, such as inbreeding, line breeding and artificial selection for specific phenotypic traits are taking place within sectors of the wildlife industry (Dry 2016). Even though there has been little work undertaken on the genetic basis for colour transmission in African game species (Needham and Hoffman 2013), it is well established in the peer-reviewed scientific literature with a high level of agreement that the selection of specific traits through a process of inbreeding or otherwise is highly likely to lead to physical, behavioural and lethal outcomes (Laikre 1999, Jensen and Andersson 2005, Hofreiter and Schöneberg 2010, Cieslak et al. 2011). In addition, these practices are likely to lead to a loss of genetic and allelic diversity that in turn is highly likely to result in decreased fitness and in the long term reduce the evolutionary potential of populations to adapt to environmental change (Hedrick and Kalinowski 2000, Reed and Frankham 2003, Frankham 2005). When specific traits, such as coat colour, are selected using artificial selection, the adaptive value of the trait is seldom considered. This may have unforeseen consequences and is likely to counter natural selection pressures that adapt an animal to its environment. It has been established in the scientific literature that colour variation is likely to influence an animal’s thermoregulation (Hetem et al. 2009) and that this (altered thermoregulation behaviour) may influence camouflage and social interactions such as mate selection. However, further research is required to better understand impact of coat colour selection on an animal’s ability to adapt to its environment.

With the expansion of the wildlife industry over the past three decades, there has been concomitant increased human-mediated movement (translocation) of animals, within and outside their natural distribution ranges (Castley et al. 2001, Spear and Chown 2009, Taylor et al. 2015). The consequences of mixing of genes from naturally separated gene pools are poorly understood and both positive and negative consequences have been documented depending on the environmental conditions (Laikre et al. 2010). It has been well-established in the scientific literature that domestication results in diverse phenotypic and behavioural changes to wild animals, including decreased flight responses, increased sociality, earlier reproduction, and modification of endocrine and metabolic systems (Waples 1999, Trut et al. 2009, Teletchea 2017). The probability that the process of domestication will take place within intensive breeding facilities is virtually certain and the impacts or effects of domestication are

dependent on the number of generations in a controlled environment and the degree of animal husbandry applied.

The extent and severity of all the impacts described for this issue will depend on the potential of the affected individual to reproduce and the proportion of animals of a particular species that are exposed to these breeding activities versus the wild. The risk is thus especially high for species with low population numbers or where the largest number of the species are kept under intensive conditions i.e. roan antelope (*Hippotragus equinus*), but much lower for common or Least Concern species. The highest level of impact will be on the individual exposed to these practices.

Impacts on wild populations through unsustainable movement of animals from the wild into controlled environments, introduction and genetic introgression of genetically altered animals into wild populations, and increased risk of introduction of species to habitats where they do not naturally occur (Issue 2)

It is well established in the peer reviewed scientific literature that the unsustainable removal of wild specimens of naturally rare species or species with currently small population sizes into captive or intensively managed facilities is likely to lead to population declines in the wild, resulting in a lower overall conservation status and a higher extinction risk for these species. The probability of this activity/impact occurring in the industry is virtually certain. There is some evidence, however still incomplete, that the natural genetic composition, evolutionary trajectory and adaptive potential of wild populations may be compromised as a result of deliberate or accidental introductions of intensively managed populations or animals, which have undergone genetic changes. It is a virtual certainty that animals are and will be introduced from intensively managed facilities into extensive systems (the stated intention is to release into the wild for hunting such animals), however the impact of these introductions are uncertain at this stage as well as the scale at which these would occur.

The evidence presented suggests that the unsustainable movement of animals from wild populations into controlled environments as well as the introductions of intensively-bred specimens, which have an altered genetic composition, is unlikely to impact the broader biodiversity, but has the potential to impact rare and threatened species over time. The risk is thus especially high for species with low population numbers or where the largest number of the species are kept under intensive conditions i.e. roan antelope (*Hippotragus equinus*), but much lower for common or Least Concern species. The individual welfare concerns are considered high where sound research observation confirms that individuals bred intensively undergo certain behavioural, morphological and physiological changes. The direction in which these changes occur is also influenced by the selective and methodical management actions undertaken at each facility.

Improper use of stock remedies (animal health products) and veterinary medicines resulting in the development of both parasite resistance to these products by parasites and to the loss of disease and parasite resistance in host populations (Issue 6).

The evidence for development of resistance to stock remedies and veterinary medicines is well established in the agricultural and biological literature, but limited studies have been undertaken in the context of intensive breeding. However, at least one study in South Africa has demonstrated the development of resistance under treatment regimens commonly used and various experts and industry role-players have expressed concern about the development of parasite resistance. It is virtually certain that the risk factors for the development of parasite and disease resistance occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. There is a risk that intensive breeding could contribute to the spread of resistant parasites to the broader agricultural and natural landscape, and this would then likely have direct negative production and economic consequences for livestock agriculture and other intensive breeding operations.

Indirect negative consequences for the broader wildlife economy and biodiversity conservation could occur through increased agricultural regulatory oversight and restrictions placed on the movement of game.

The evidence for loss of disease and parasite resistance within the game population is established but incomplete. There is a strong theoretical basis for expecting the loss of resistance over time, based on the removal of natural selective processes. It is virtually certain that the risk factors for the development of loss of resistance at an individual and herd level to parasites and diseases occur as a result of management practices adopted within a large proportion of the industry based on a broad understanding of current practices. Translocation of intensively-bred animals onto other properties is very likely to result in receiving properties implementing the same parasite and disease controls that initially lead to both parasite resistance. It is highly likely that species where a large proportion of the population is contained in intensive breeding facilities will lose resistance and be at risk, but unlikely to be an issue for species where a significant wild population undergoing natural selection still exists. However, natural selective processes are increasingly being removed or moderated on properties outside of intensive breeding properties, in which case it is more likely that non-resistant individuals will pass on their genes to wild populations making the broader population more susceptible to periodic disease or parasite outbreaks.

BIODIVERSITY ECONOMY RISKS

Reputational risks, socio-economic and conservation impacts

In line with the environmental and economic policy frameworks of South Africa, this issue takes a more holistic approach in discussing the potential current and cumulative impacts of intensive and selective breeding. As the purpose of the report was neither to do a strategic environmental assessment, nor to do a comprehensive cost benefit analysis, two impacts were identified that suggest that a specific activity and its risks to biodiversity should be assessed within the broader environmental, economic and social context within which it operates. For this reason, the potential impact of intensive and selective breeding on the broader hunting industry was assessed, along with the potential implications for land-use and conservation planning.

Extensive research exists on the environmental and business principles that support responsible and sustainable growth of enterprises and the risks associated with poor reputational management. Although relevant to all sectors of the wildlife industry, it has been demonstrated that the hunting sector, in particular, is highly vulnerable to negative stakeholder and public perception.

Despite absence of comprehensive economic research on the interdependencies of the various sectors within the wildlife industry in South Africa, there is a substantial body of evidence from prominent local and international hunting organisations, international conservation organisations and members' of CITES and the IUCN that, in the absence of clearly defined product differentiation in the hunting market, intensive and selective breeding of game for hunting has a high risk of exacerbating negative perceptions about hunters, hunting in general and conservation in South Africa.

The assessment of this literature and the chain of events and incidents related to the captive lion breeding subsector, demonstrated that reputation is a matter of perception and not necessarily a reflection of actual behaviour. Although shooting of intensively-bred lions is legal in South Africa and was practised for years, stakeholder perceptions about the practice have changed over time as attitudes towards hunting have changed. Negative perceptions are not only towards "canned" hunting, but also "captive-bred" hunting, "put-and-take" hunting and "tame" hunting.

Hunting incidents and activities perceived as socially unacceptable, such as shooting of captive-bred lions, have been used by protectionist groups to tarnish the reputation of hunting in general. Over the past few years these have contributed to international policy changes that were not only geared towards illegal hunts or captive-bred lion hunts, but also legal hunts of other species and exports of trophies from hunts where a contribution to conservation could not be readily demonstrated (even though such contributions almost certainly do exist). The majority of key role-players within the hunting fraternity worldwide do not condone the shooting of intensively- and selectively-bred game, because it does not comply with the principles of fair chase hunting.

It is clear from the assessment that reputational risks are linked to both the activity of hunting as well as the management practice associated with source populations, suggesting that reputational risks and management should be addressed, taking the full value chain into account.

If reputational risks are not managed urgently, appropriately and holistically within the full value chain, the trend of negative social, economic and conservation implications that several sectors of the wildlife industry are already experiencing may continue. It will have far-reaching implications for growth of the wildlife economy and the private sector's positive contributions to rural economic growth, the well-being of communities, and conservation in general. This would increase the pressure on government to foot the bill for delivery on these national priorities.

There is a high level of agreement from scientists and industry members that development of industry standards, guidelines and certification/labelling, which address both the practice of hunting and management practices of affected source populations, can reduce associated risks and mitigate potential impacts within a market economy like South Africa. Aspects that have been identified as critical for sustainability of hunting as highlighted in this report should inform this process.

As far as implications for broader land-use and conservation planning is concerned, assessment of biodiversity impacts confirmed that the ecological footprint of game breeding operations is very different from traditional extensive wildlife areas, even if the land cover may still be near natural in some cases. Associated impacts such as habitat fragmentation, animals killed in fences and reduced tolerance by intensive game breeders towards free-ranging threatened predators, are of greater concern in near-natural and sensitive environments, such as areas adjacent to protected areas and wildlife corridors, than in areas already modified or zoned for intensive agricultural practices.

The existing shortfalls in policy frameworks that govern land-use planning and environmental impact assessments, result in a gap between the information used to inform policy processes and what is happening on the ground. This limits the ability of government to address the impacts of intensive and selective breeding operations on biodiversity associated with natural landscapes and extensive wildlife areas, which:

- form the basis of wildlife-based tourism and hunting, the biggest contributors to the wildlife economy;
- form the basis of wildlife corridors, private sector contribution to national conservation targets and the national protected area strategy, wherein protected areas can function as drivers in the rural economy; and
- generate ecosystem goods and services, which in South Africa, amount to an estimated R73 billion per annum, or equivalent to 3% of the country's GDP (SANBI, 2010) and therefore fuels the economy.

In the absence of a policy framework to prevent or at the least mitigate these impacts, the direct and indirect costs to government may be significant.

CONCLUSION AND RECOMMENDATIONS

The White Paper on the Conservation and Sustainable Use of South Africa's Biological Diversity, the main policy document pertaining to the use and conservation of biodiversity in South Africa, is modelled on the Convention on Biological Diversity (CBD) (Cousins et al. 2010). According to the CBD Article 2, sustainable use encompasses 'the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations'. This definition of sustainable use centres on the management and use of wild species and ecosystems within biologically sustainable limits (Hutton and Leader-Williams 2003). As such, sustainable use presents two challenges: (1) "to ensure that use increasingly becomes biologically sustainable"; and (2) "that wherever possible it serves as a conservation strategy to conserve specific resources and prevent the conversion of land to uses that are incompatible with biodiversity conservation" (Hutton and Leader-Williams 2003). The practice of intensive and selective breeding based on the findings of this assessment may not meet these criteria for sustainability.

It is concluded that intensive management and selective breeding of game poses a number of significant risks to biodiversity at landscape, ecosystem and species levels, as well as to other sectors of the biodiversity economy of South Africa, and may compromise the current and future contribution of the wildlife industry to biodiversity conservation. This assessment has identified several important direct risks and impacts on biodiversity at different scales, as well as indirect collateral negative impacts on conservation and the broader wildlife economy.

A mix of regulatory, awareness-raising and incentive-based systems need to be implemented to mitigate the risks posed by this sub-sector of the wildlife industry. Given the challenges and costs of a regulatory approach, wherever possible, incentive-based approaches should be used as well as taking advantage of market forces to reward practices that are more compatible with biodiversity conservation and that are less risky to the biodiversity economy. However, the necessary enabling legislative framework for this needs to be created. Lastly, government and all role-players in the wildlife economy, should take cognisance of potential far-reaching implications of developing new ventures and sub-sectors within the wildlife sector. Principles of business and environmental sustainability as entrenched in NEMA and the King reports on governance that considers social, environmental and economic aspects within the current and future landscape of the country would be critical to ensure sustainable growth of the biodiversity economy to the benefit of all.

Key recommendations

Specific recommendations to mitigate negative impacts on biodiversity and the biodiversity economy are provided at the end of each Impact Statement in Chapters 4 and 5. The key integrated and overarching recommendations are provided below; each of these recommendations may address several Impact Statements.

Regulate where intensive and selective breeding facilities are located

The location of intensive and selective breeding facilities plays a major role in determining the impact on biodiversity. It is recommended that any new intensive and selective breeding facilities be located outside sensitive environments such as Critical Biodiversity Areas as designated in bioregional plans or systematic biodiversity conservation plans, threatened ecosystems, buffer zones of protected areas, National Protected Area Expansion Strategy focal areas, or any other ecologically sensitive areas. It is suggested that the NEMA Listing Notice 3 could be amended to make establishment of any intensive breeding facility a listed activity that is prohibited in the sensitive environments listed above and that a basic assessment is required elsewhere. Any assessment process must consider the cumulative impacts of existing and planned facilities. Provincial authorities should consider only issuing

permits for the keeping of game after the environmental assessment process has been completed confirming that there would be no significantly negative impacts on the environment that cannot be reasonably mitigated. It is further recommended to investigate mechanisms to link permissions to the land-use zoning process whereby it is easier to get permission in more transformed agricultural landscapes and more difficult to get permission in untransformed or priority biodiversity areas.

Fencing

Current fence design contributes to mortality of wildlife, including that of threatened species, and results in landscape fragmentation. It is recommended that current regulations, policies and guidelines on 'minimum' fence standards be reviewed to include 'maximum' standards incorporating mitigation measures to reduce negative impact on non-target priority species. These standards could vary geographically depending on the distribution of priority species and issues. It is further recommended that where there is a lack of information on appropriate mitigation that an academic research programme on appropriate fence designs to mitigate impacts on biodiversity be commissioned. It is conceivable that new designs that are equally effective, but less costly and less damaging could be found. It should be noted that improvement in fence design affects the entire wildlife industry, but is especially important in the intensive and selective breeding sector due to the exponential increase in length of fencing per unit area.

Registration

There is currently little known about the full nature and extent of the practice of intensive and selective breeding within South Africa and landowners are not currently required to register or disclose this activity to management authorities. The impacts of intensive and selective breeding are however related to the location and scale of the activity. In order to monitor and record the nature and extent of the practice it is recommended that all intensive and selective breeding facilities are registered with government and/or an approved industry body, indicating inter alia the type of operation, species and numbers involved, and specific location and layout of fences. This will allow for better monitoring and reporting on the scale of the activity, and for a better assessment of potential impacts. It is important that any such records are subject to regular sample audit to verify completeness and accuracy. The lack of such information was a limitation in undertaking this assessment.

Wild to captive

The removal of specimens of Threatened species from the wild for introduction into intensive breeding facilities is likely to be detrimental to the wild populations. Currently animals for intensive breeding are sourced from either extensive systems or other intensive breeding facilities, and vary depending on the species. There are numerous wildlife breeders in South Africa that manage and breed sable antelopes in small intensive systems (Kriek 2005), and therefore availability of intensively bred animals should not be a problem. However the breeding of small game, such as blue duiker and oribi in South Africa, has recently gained momentum. For these species removal from the wild is likely to be detrimental. The removal from the wild of Threatened species for the purpose of intensive and specifically selective breeding for commercial purposes is not recommended.

Captive to wild

Release of intensive and selectively bred/managed animals into the wild results in several risks to biodiversity and the biodiversity economy. As a general principle the movement of animals from intensive and selective breeding facilities to the wild must be carefully regulated (whereas the movement between wild populations could possibly be de-regulated to some extent). Given the identified risks, permits for the release of animals from intensive and selective breeding facilities into the wild should only be allowed if the animals were managed according to certain minimum standards. These minimum standards must specifically address

the risks and impacts highlighted in this assessment and require biodiversity authority and expertise involvement. This will address some of the direct biodiversity concerns. However, to address some of the biodiversity economy risks, the simultaneous implementation of a wildlife certification system to allow for transparent market information to consumers, specifically hunters, tourists and consumers of venison, will be required (see below).

Wildlife certification system

We recommend that a powerful tool to incentivise and reward best practice compatible with responsible biodiversity management (and hence compatible with and contributing towards the nation's biodiversity objectives and obligations) could be through the development of a national certification scheme for wildlife enterprises. If correctly developed, and appropriately linked to other government policies (such as the Working For programmes), this would create additional economic incentives for biodiversity-compatible practices and land-uses. This would also allow for transparency in the market and accommodate consumer preference, particularly in the hunting, tourism and game meat sectors of the market. It is necessary to allow both the market and government to understand which extensive systems have had animals introduced from intensive systems. The certification system must clarify the distinction of wild versus captive populations in terms of the IUCN Red List process. It is important to understand that the proposed certification system, while it will incentivise best practice, is primarily a mitigation measure for the negative impacts of the industry on biodiversity and the biodiversity economy. The minimum standards, which requires the industry's involvement and support, must be established by the biodiversity authorities to ensure that they adequately address the risks identified in this report.

Governance

Industry is quick to capitalise on new economic opportunities. The rapid development of intensive and selective breeding, and the uncoordinated response from authorities to this new sector, demonstrated that a risk to biodiversity stems from the fact that regulators generally anticipate and respond too slowly to new risks that may arise from new practices. The biodiversity sector needs to find more effective ways of anticipating industry direction and preparing in terms of collection of information, anticipating risks, and preparing/amending policy and regulatory response if necessary.

Legislative review

Specific recommendations in relation to amendments to the NEMA Listing Notice 3 have been discussed above. In addition, the recent listing of indigenous mammals under the Animal Improvement Act would appear to entrench and exacerbate many of the risks highlighted in this report. It is recommended that a critical review of the implications of this listing is initiated and amendments made if necessary. In the interim, it is strongly recommended that no further indigenous species are listed under the Animal Improvement Act.

CHAPTER 8: REFERENCES

2001. Government of the Republic of South Africa and Others vs. Grootboom, (Grootboom) 2001 (1) SA 46 (CC), 2000 (11) BCLR 1169 (CC).

2006. HTF Developers (Pty) Ltd v Minister of Environmental Affairs and Tourism, 2006 (5) SA 512,.

2007. Fuel Retailers Association of Southern Africa v Director-General Environmental Management, Department of Agriculture, Conservation and Environment, Mpumalanga Province 2007 6 SA 4 (CC).

Abbasa, R. Z., M. A. Zaman, D. D. Colwell, J. Gillearde, and Z. Iqbala. 2014. Acaricide resistance in cattle ticks and approaches to its management: The state of play. Veterinary Parasitology **203**:6-20.

Africa Geographic. 2016. Photos emerge of malnourished lions on breeding farm. 8 July 2016. http://africageographic.com/blog/photos-emerge-of-malnourished-lions-on-breeding-farm/. Accessed August 2016. Africa Geographic.

African Wildlife Auctions. 2018. Rare & Hunting Game Auction, Vaalwater. AWA. https://www.awla.co.za/3-feb-2018-rare-and-hunting-game.

Allen, W. L., I. C. Cuthill, N. E. Scott-Samuel, and R. Baddeley. 2010. Why the leopard got its spots: relating pattern development to ecology in felids. Proceedings of the Royal Society of London B: Biological Sciences:rspb20101734.

Allendorf, F. W. 1993. Delay of adaptation to captive breeding by equalizing family size. Conservation Biology **7**:416-419.

Allendorf, F. W., R. F. Leary, P. Spruell, and J. K. Wenburg. 2001. The problems with hybrids: setting conservation guidelines. Trends in Ecology & Evolution **16**:613-622.

Alpers, D., B. J. van Vuuren, P. Arctander, and T. Robinson. 2004. Population genetics of the roan antelope (Hippotragus equinus) with suggestions for conservation. Molecular Ecology **13**:1771-1784.

Amos, W., and A. Balmford. 2001. When does conservation genetics matter? Heredity **87**:257-265.

Andersson, L. 2001. Genetic dissection of phenotypic diversity in farm animals. Nature Reviews Genetics **2**:130-138.

APO. 2009. Green Tourism: Certification Manual. Asian Productivity Organisation, http://www.apo-tokyo.org/00e-books/GP-18_GreenTourism.htm.

Armstrong, J. B. 1998. Longevity in the standard poodle.http://www.caninegenetics.com/lifespan.html.

Arnot, L., and S. Molteno. 2017. How to reduce tortoise electrocution mortalities. Farmer's Weekly **2017**:36-38.

Augustine, D. J., and S. J. McNaughton. 2004. Regulation of shrub dynamics by native browsing ungulates on East African rangeland. Journal of Applied Ecology **41**:45–58.

AZA. 2011. Welfare and conservation implications of intentional breedign fro teh expression of rate recessive alleles. Association of Zoos & Aquariums.

Bailey, N. W., and A. J. Moore. 2012. Runaway sexual selection without genetic correlations: Social environments and flexible mate choice initiate and enhance the Fisher process. Evolution **66**:2674–2684.

Baldus, R. 2016. Cecil the Lion – The Truth. Hunter´s Path. Hunting magazine.

Baldus, R., and A. Cauldwell. 2005. Safari hunting certification. Pages 12-13 African Indaba.

Balme, G. A., and L. T. Hunter. 2013. Why leopards commit infanticide. Animal Behaviour **86**:791-799.

Balme, G. A., R. Slotow, and L. T. B. Hunter. 2009. Impact of conservation interventions on the dynamics and persistence of a persecuted leopard (Panthera pardus) population. Biological Conservation **142**:2681–2690.

Barnes, R. D., C. W. Fagg, and S. J. Milton. 1997. Acacia erioloba Monograph and annotated bibliography. Oxford Forestry Institute, Oxford.

Batavia, C., M. P. Nelson, C. T. Darimont, P. C. Paquet, W. J. Ripple, and A. D. Wallach. 2018. The elephant (head) in the room: A critical look at trophy hunting. Conservation Letters:e12565.

Bazzaz, F. A. 1975. Plant species diversity in old-field successional ecosystems in southern Illinois. Ecology **56**:485–488.

Beattie, A. 2017. Market Crashes: The Tulip and Bulb Craze. http://www.investopedia.com/features/crashes/crashes2.asp. Accessed February 2017.).

Beck, A. 2010. Electric fence induced mortality in South Africa. University of the Witwatersrand, Johannesburg.

Begany, L., and C. Criscuolo. 2009. Accumulation of Deleterious Mutations Due to Inbreeding in Tiger Population. CL Criscuolo **27**.

Belyaev, D. K. 1979. Destabilizing selection as a factor in domestication. Journal of Heredity **70**:301-308.

Bergman, D. L., H. O. De Waal, N. L. Avenant, M. Bodenchuk, M. C. Marlow, and D. L. Nolte. 2013. The Need to Address Black-backed Jackal and Caracal Predation in South Africa.*in* Wildlife Damage Management Conferences, Proceedings. University of Nebraska-Lincoln. Paper 165.

Beschta, R. L., and W. J. Ripple. 2009. Large predators and trophic cascades in terrestrial ecosystems of the western United States. Biological Conservation **142**:2401-2414.

Bezuidenhoudt, J. 2016. Meer as 19000 plase in SA op die mark. Landbou.com. 20 Mei 2016. Media 24 http://landbou.com/nuus/meer-as-19-000-plase-in-sa-op-mark/ Accessed 1 June 2016.

Bezuidenhout, R. 2012. Colour under investigation. Farmer's Weekly, http://www.farmersweekly.co.za/article.aspx?id=25770&h=Colour--under--investigation.

Bissett, C., and R. T. F. Bernard. 2007. Habitat selection and feeding ecology of the cheetah (Acinonyx jubatus) in thicket vegetation: is the cheetah a savanna specialist? Journal of Zoology **271**:310-317.

Blackmore, A. 2015. The Relationship between the NEMA and the Public Trust Doctrine: The Importance of the NEMA Principles in Safeguarding South Africa's Biodiversity. South African Journal of Environmental Law and Policy **20**:89-118.

Blackmore, A. 2017. The Public Trust Doctrine, Research and Responsible Wildlife Management in South Africa. Bothalia **47**:a2217.

Blackmore, A. C. In press. Selective Breeding and Intensive Management Of Wildlife: A Legal Challenge for Namibian Conservation. African Journal of Wildlife Research.

Blanchet, S., D. J. Páez, L. Bernatchez, and J. J. Dodson. 2008. An integrated comparison of captive-bred and wild Atlantic salmon (Salmo salar): Implications for supportive breeding programs. Biological Conservation **141**:1989-1999.

Blaum, N., B. Tietjen, and E. Rossmanith. 2009. Impact of livestock husbandry on small and medium sized carnivores in Kalahari savannah rangelands. Journal of Wildlife Management **73**:60-67.

Bloch, S. 2015. Airlines send SA trophy hunting industry into tailspin with cargo ban. http://www.traveller24.com/News/Airlines-send-SA-trophy-hunting-industry-into-tailspin-with-cargo-ban-20150514. Traveller 24, 14 May 2015.

Blood lions. 2016. A year into the campaign 2015-2016. https://web.facebook.com/notes/blood-lions/blood-lions-a-year-into-the-campaign-2015-16/1018245948289768.

Blue & Green Tomorrow. 2014. Sustainable Tourism 2014. 3rd edition. http://blueandgreentomorrow.com/reports/the-guide-to-sustainable-tourism-2014/, Accessed 28 February 2016.

Blumm, M. C., and R. D. Guthrie. 2012. THE PUBLIC TRUST DOCTRINE: 30 YEARS LATER: Internationalizing the Public Trust Doctrine: Natural Law and Constitutional and Statutory Approaches to Fulfilling the Saxion Vision. UC Davis L. Rev. **45**:741-1945.

Boast, L. 2014. Exploring the causes of and mitigation options for human-predator conflict on game ranches in Botswana: How is coexistence possible? . Cape Town, Cape Town.

Bolnick, D. I., and P. Nosil. 2007. Natural selection in populations subject to a migration load. Evolution **61**:2229-2243.

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

Boone, R. B., and N. T. Hobbs. 2004. Lines around fragments: effects of fencing on large herbivores. African Journal of Range & Forage Science **21**:147-158.

Booth, V. 2005. International and regional best practice and lessons applicable to sport and recreational hunting in southern Africa. Report for Department of Environmental Affairs and Tourism, Pretoria, South Africa.

Boretsky, J. 2015. Presentation given by Mr. Boretsky at the 2nd National Dialogue Workshops on Selective and Intensive Breeding of Colour Variants. Department of Environmental Affairs, Pretoria. SA.

Botha, L., and C. Antonites. 2016. Analysis of online sentiments relevant to conservation and biodiversity management in the Lowveld Protected Area: Hunting., Internal report commissioned by the Lowveld Protected Area Group.

Botha, T. 2016. Waak teen prysborrel by wild- Janovski. Lanbou.com. Media 24. http://landbou.com/nuus/waak-teen-prysborrels-by-wild-janovs2522y/ Accessed 27 July 2016.

Bothma, J. D. P. 2005. It is time to end the debate on introducing exotic wildlife. South African Journal of Wildlife Research **35**:97-102.

Bothma, J. d. P., and H. J. N. Von Bach. 2010. Economic aspects of extensive wildlife production in southern Africa. Pages 83-96 *in* J. d. P. Bothma and J. G. Du Toit, editors. Game Ranch Management. Van Schaik, Pretoria.

Bourdon, R. M. 2000. Understanding animal breeding, Richard M. Bourdon.

Bradley, E. H., and D. H. Pletscher. 2005. Assessing factors related to wolf depredation of cattle in fenced pastures in Montana and Idaho. Wildlife Society Bulletin **33**:1256-1265.

Bradley, E. H., D. H. Pletscher, E. E. Bangs, K. E. Kunkel, D. W. Smith, C. M. Mack, T. J. Meier, J. A. Fontaine, C. C. Niemeyer, and M. D. Jimenez. 2005. Evaluating wolf translocation as a nonlethal method to reduce livestock conflicts in the northwestern United States. Conservation Biology **19**:1498-1508.

Brainerd, S. 2007. European Charter on Hunting and Biodiversity. Adopted by the Standing Committee of the Bern Convention at its 27th meeting in Strasbourg, 26-29 November 2007. http://fp7hunt.net/Portals/HUNT/Hunting_Charter.pdf.

Brashares, J. S., P. R. Prugh, C. J. Stoner, and C. W. Epps. 2010. Trophic Cascades: Predators, Prey, and the Changing Dynamics of Nature. Island Press, Washington DC.

Brink, M., M. Cameron, K. Coetzee, B. Currie, C. Fabricius, S. Hattingh, A. Schmidt, and L. Watson. 2011. Sustainable management through improved governance in the game industry. South African Journal of Wildlife Research **41**:110-119.

Brophy, S. 2015. Blood Lions: 6 Harsh truths about canned lion hunting. http://traveller24.news24.com/Explore/Bush/6-Harsh-truths-about-canned-hunting-as-revealed-by-Blood-Lions-20150826Traveller 24.

Brundtland Commission. 1987. Our Common Future. Oxford University Press, UK.

Burns, B., J. Innis, and T. Day. 2012. The use and potential of pest-proof fencing for ecosystem restoration and fauna conservation in New Zealand. Pages 65-90 *in* M. J. S. M. W. Hayward, editor. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.

Bush, E. R., S. E. Baker, and D. W. Macdonald. 2014. Global Trade in Exotic Pets 2006–2012. Conservation Biology **28**:663-676.

Cach. 2015. Airlines around the world place embargo on tropies. Campaign Against Canned Hunting. 20 April 2015. http://www.cannedlion.org/.

Canu, A., S. Costa, L. Iacolina, P. Paitti, M. Apollonio, and M. Scandura. 2014. Are captive wild boar more introgressed than free-ranging wild boar? Tow case studies in Italy. European Journal of Wildlife Research **60**:459-467.

Capon, S. D., A. J. Leslie, and B. Clegg. 2013. The use of population viability analysis to identify possible factors contributing to the decline of a rare ungulate population in south-eastern Zimbabwe. Koedoe **55**.

Carlborg, Ö., S. Kerje, K. Schütz, L. Jacobsson, P. Jensen, and L. Andersson. 2003. A global search reveals epistatic interaction between QTL for early growth in the chicken. Genome research **13**:413-421.

Caro, T. 1994. Cheetahs of the Serengeti Plains - Group living in an asocial species. The University of Chicago Press, Chicago, USA.

Caro, T. 2005. The adaptive significance of coloration in mammals. BioScience **55**:125-136.

Caro, T. 2013. The colours of extant mammals. Seminars in cell & developmental biology **24**:542-552.

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.

Castley, J. G., A. F. Boshoff, and G. I. H. Kerley. 2001. Compromising South Africa's natural biodiversity — inappropriate herbivore introductions. South African Journal of Science **97**:344-348.

Cavalcanti, S. M. C., P. G. Crawshaw, and F. R. Torato. 2012. Use of electric fencing and associated measures as deterrents to Jaguar predation on cattle in the Pantanal of Brazil.*in* M. J. Somers and M. W. Hayward, editors. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.

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

Charlesworth, D., and B. Charlesworth. 1987. Inbreeding depression and its evolutionary consequences. Annual Review of Ecology and Systematics:237-268.

Child, B., and B. Wall. 2009. The application of certification to hunting: a case for simplicity. Recreational hunting, conservation and rural livelihoods:341-360.

Child, B. A., J. Musengezi, G. D. Parent, and G. F. T. Child. 2012. The economics and institutional economics of wildlife on private land in Africa. Pastoralism: Research, Policy and Practice **2**:1-32.

Chintoan-Uta, C., E. R. Morgan, P. J. Skuce, and G. C. Coles. 2014. Wild deer as potential vectors of anthelmintic-resistant abomasal nematodes between cattle and sheep farms. Proc. R. Soc. B **281**:20132985.

Cieslak, M., M. Reissmann, M. Hofreiter, and A. Ludwig. 2011. Colours of domestication. Biological Reviews **86**:885-899.

CITES. 2015. Protecting CITES Against Irrelevance. Media release by two ex-secretary generals of CITES. Convention on International Trade in Endangered Species.

CITES. 2016. Resolution on: Trade in hunting trophies of species listed in Appendix I or II. https://cites.org/eng/cop/17/doc/index.php.*in* C. o. I. T. i. E. W. F. a. Flora, editor.

Cloete, F. 2016. Live-game trends. Game & Hunt.

Cloete, F. 2017. ABSA Agricultural outlook 2017; http://www.agtag.co.za/view_shared_post/14585.

Cloete, P. C. 2015. Plains game, colour variants and strange species. In: ABSA Agricultural Outlook 2015. www.agriconnect.co.za.

Cloete, P. C., P. van der Merwe, and M. Saayman. 2015. Game ranch profitability in South Africa. ABSA, South Africa.

Clutton-Brock, J. 1999. A Natural Hisotry of Domesticated Mammals. 2nd edition. Cambridge University Press, Cambridge.

Coates, G. D., and C. T. Downs. 2005. Survey of the status and management of sympatric bushbuck and nyala in KwaZulu-Natal, South Africa. South African Journal of Wildlife Research **35**:179-190.

Coetzee, K. 2002. The Fynbos and Renosterveld. Pages 629-632 *in* J. d. P. Bothma, editor. Game Ranch Management. Van Schaik, Pretoria.

Collar, N., and A. T. Juniper. 1992. Dimensions and causes of the parrot conservation crisis.*in* S. R. Beissinger and N. F. R. Snyder, editors. New world parrots in crisis: Solutions from conservation biology. Smithsonian Institution Press, Washington DC.

Cooney, R. 2004. The Precautionary Principle in Biodiversity Conservation and Natural Resource Management: An issues paper for policy-makers, researchers and practitioners. IUCN.

Cooney, R., C. Freese, H. Dublin, D. Roe, D. Mallon, M. Knight, R. Emslie, M. Pani, V. Booth, and S. Mahoney. 2017. The baby and the bathwater: trophy hunting, conservation and rural livelihoods. Unasylva **68**:3.

Courchamp, F., M. Langlais, and G. Sugihara. 1999. Cats protecting birds: modelling the mesopredator release effect. Journal of Animal Ecology **68**:282-292.

Cousins, J., J. Sadler, and J. Evans. 2010. The challenge of regulating private wildlife ranches for conservation in South Africa. Ecology and Society **15**.

Cousins, J. A., J. P. Sadler, and J. Evans. 2005. Exploring the role of private wildlife ranching as a conservation tool in South Africa: Stake holder perspectives. Ecology & Society **13**:43.

Cousins, J. A., J. P. Sadler, and J. Evans. 2008. Exploring the role of private wildlife ranching as a conservation tool in South Africa: stakeholder perspectives. Conservation and Society **13**:43-59.

CPHC. 2017. Report on changes in policy by PHASA on captive bred lion hunting. CPHC,, Pretoria, South Africa.

Crispo, E., J.-S. b. Moore, J. A. Lee-Yaw, S. M. Gray, and B. C. Haller. 2011. Broken barriers: Human-induced changes to gene flow and introgression in animals. Bioessays **33**:508–518.

Crnokrak, P., and D. A. Roff. 1999. Inbreeding depression in the wild. Heredity **83**:260-270.

Crosmary, W. G., S. Cote, and H. Fritz. 2015. The assessment of the role of trophy hunting in wildlife conservation. Animal Conservation **18**:136-137.

Cumming, D. H. M., M. B. Fenton, I. L. Rautenbach, R. D. Taylor, G. S. Cumming, M. S. Cumming, J. M. Dunlop, A. G. Ford, M. D. Hovorka, D. S. Johnston, M. Kalcounis, Z. Mahlangu, and C. V. R. Portfors. 1997. Elephants, woodlands and biodiversity in southern Africa. South African Journal of Science **93**:231-236.

Cupido, C. F. 2005. Assessment of veld utilisation practices and veld condition in the Little Karoo. Dissertation. University of Stellenbosch, Stellenbosch, Stellenbosch, South Africa.

D. Schellenberg, Q. Yi, N.W. Glickman, and L.T. Glickman. 1998. Influence of thoracic conformation and genetics on the risk of gastric dilatation-volvulus in Irish setters. Journal of the American Animal Hospital Association **34**:64-73.

Dalerum, F., and M. Miranda. 2016. Game auction prices are not related to biodiversity contributions of southern African ungulates and large carnivores. Nature Scientific Reports **6**:21922.

Damn, G. R. 2008. Recreational trophy hunting: What do we know and what should we do? Pages 5-11 *in* R. D. Baldus, G. R. Damm, and K. Wollscheid, editors. Best practices in Sustainable hunting - A guide to best practices around the world. International Council for Game and Wildlife Conservation, Hungary.

Daniels, T. J., and M. Bekoff. 1989. Feralization: the making of wild domestic animals. Behavioural Process **19**:79-94.

de Klein, C. A. M., D. J.J., N. J.W., S. M., C. R., M. R.W., and R. Muirhead. 2003. Environmental impacts of intensive deer farming in New Zealand-a review. In Deer Nutrition Symposium: The nutrition and management of deer on grazing systems.

Pages 79-87 *in* Proceedings of a New Zealand Grassland Association (Inc) Symposium. New Zealand Grassland Association. , Lincoln University, New Zealand.
De la Paix, M., and C. Eugène-Rigot. 2017. Sustainability and enterprise risk management: The first step towards integration., Geneva 21, Switzerland; https://www.wbcsd.org/Projects/Non-financial-Measurement-and-Valuation/Resources/Sustainability-and-enterprise-risk-management-The-first-step-towards-integration.
De Tores, P. J., and N. Marlow. 2012. The relative merits of predator-exclusion fencing and repeated fox baiting for protection of native fauna: Five case studies from western Australia. Pages 21-41 *in* M. J. Somers and M. W. Hayward, editors. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.
DEA. 2011. Trophy Hunting Statistics – 2011.*in* D. o. E. Affairs, editor., Pretoria.
DEA. 2014. Trophy Hunting Statistics – 2014.*in* D. o. E. Affairs, editor., Pretoria.
DEA. 2015. Trophy Hunting Statistics – 2015.*in* D. o. E. Affairs, editor., Pretoria.
DEA. 2016. Biodiversity and Tourism Lab briefing report- May 2016. Department of Environmental Affairs, Pretoria.
Dean, W. R. J., S. J. Milton, and F. Jeltsch. 1999. Large trees, fertile islands, and birds in an arid savanna. Journal of Arid Environments **41**:61-78.
DeGeorges, P. A., and B. Reilly. 2008. A Critical Evaluation of Conservation and Development in Sub-Saharan Africa: Last Chance Africa. Edwin Mellen Press: Lewiston, NY, USA.
Dernbach, J. C. 2015. The Potential Meanings of a Constitutional Public Trust. Environmental Law **257**:464–465.
Desmet, P., E. J. Nel, and S. Pillay. 2017. The Landscape Ecological Impact of Intensive Game Breeding in South Africa: Mapping the Rate of Change in Game Camp Fence Density at Two Sites in Limpopo Province from 2006-2015. South African Hunters and Game Conservation Association, Pretoria.
DGIF, D. o. G. I. F. 2007. An evaluation of captive-reared mallard releases in Virginia. Virginia Department of Game & Inland Fisheries, Virginia, USA.
Di Minin, E., I. Fraser, R. Slotow, and D. C. MacMillan. 2013a. Understanding heterogeneous preference of tourists for big game species: Implications for conservation and management. Animal Conservation **16**:249–258.
Di Minin, E., N. Leader-Williams, and C. A. J. Bradshaw. 2016. Banning trophy hunting will exacerbate biodiversity loss. . Trends in Ecology and Evolution.
Di Minin, E., D. C. Macmillan, P. S. Goodman, B. Escott, R. Slotow, and A. Moilanen. 2013b. Conservation businesses and conservation planning in a biological diversity hotspot. Conservation Biology **27**:808–820.
Dickman, C. R. 2012. Fences or ferals? Benefits and costs of conservation fencing in Australia. Pages 43 - 63 *in* M. J. Somers and M. W. Hayward, editors. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.
Dobney, K., and G. Larson. 2006. Genetics and animal domestication: new windows on an elusive process. Journal of Zoology **269**:261-271.
Dodds, R., S. Graci, and M. Holmes. 2010. Does the Tourist Care? Journal of Sustainable Tourism **19**:207-222.
Donaldson, J. 2013. Scientific evidence on colour mutation / variations. Letter to the Department of Environmental Affairs. Ref: 1/1/40/1.
Dorrington, S. 2015. Opinion piece by past president of the Professional Hunters' Association of South Africa. African Indaba, June 2015.
Driscoll, C. A., D. W. Macdonald, and S. J. O'Brien. 2009. From wild animals to domestic pets, an evolutionary view of domestication. Proceedings of the National Academy of Sciences of the United States of America **106**:9971-9978.
Druce, D. J. 2000. Factors affecting millipede, centipede and scorpion diversity in a savannah environment. University of Natal, Durban.
Dry, G. C. 2016. WRSA Code of best practise - Genetic Engineering draft 7. WRSA.

DSC. 2018. Dallas Safari Club Position on Captive Bred Lion Hunting. http://dscnewscenter.org/2018/01/dsc-position-on-captive-bred-lion-hunting/.
Du Toit, J. G. 2005a. The African savanna buffalo.*in* J. d. P. Bothma and N. v. Rooyen, editors. Intensive wildlife production in southern Africa. Van Schaik, Pretoria.
Du Toit, J. G. 2005b. The eland. Pages 108-124 *in* J. d. P. Bothma and N. v. Rooyen, editors. Intensive wildlife production in southern Africa. Van Schaik, Pretoria.
Du Toit, P. F., and A. J. Aucamp. 1985. Effect of continuous grazing in the dohne sourveld on species composition and basal cover. Journal of the Grassland Society of Southern Africa **2**:6-14.
Durant, S. M., N. Mitchell, A. Ipavec, and G. R. 2015. Acinonyx jubatus. The IUCN Red List of Threatened Species 2015: e.T219A50649567.
East, M. L., J. W. Nyahongo, K. V. Goller, and H. Hofer. 2012. Does the vastness of the Serengeti limit human-wildlife conflicts.*in* M. J. S. M. W. Hayward, editor. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes Springer, New York.
Eccard, J. A., and J. Meyer. 2001. Space and vegetation use, activity pattern, and mating system of the nocturnal tree rat Thallomys nigricauda. Zeitschrift für Säugertierkunde **66**:10.
Eccles, R. G., S. C. Newquist, and R. Schatz. 2007. Reputation and its risk. Harvard Business Review **Feb; https://hbr.org/2007/02/reputation-and-its-risks**
Economist. 2012. From feathers to funny colours, but the future is the same. https://economist.com.na/1792/editors-desk/from-feathers-to-funny-colours-but-the-future-is-the-same/ Accessed February 2017.
Edmands, S. 2007. Between a rock and a hard place: evaluating the relative risks of inbreeding and outbreeding for conservation and management. Molecular Ecology **16**:463-475.
Eizirik, E., N. Yuhki, W. E. Johnson, M. Menotti-Raymond, S. S. Hannah, and S. J. O'Brien. 2003. Molecular genetics and evolution of melanism in the cat family. Current Biology **13**:448-453.
Ellis, J. 2006. Overexploitation of a valuable resource? New literature on the precautionary principle. European Journal of International Law **17**:445-462.
Emslie, R. E., T. Milliken, B. Talukdar, S. Ellis, K. Adcock, and M. H. Knight. 2016. African and Asian rhinoceroses: status, conservation and trade. A report from the IUCN Species Survival Commission (IUCN SSC) African and Asian Rhino specialist groups and TRAFFIC to the CITES Secretariat pursuant to Resolution Conf. 9.14.
Ernst, and Young. 2016. Value of sustainability reporting. EY and Boston College Center for Corporate Citizenship. Ernst & Young Global Limited, US.
Esler, K. J., S. J. Milton, and W. R. J. Dean. 2006. Karoo veld ecology and management. Briza Publications, Pretoria.
Estes, J. A., J. Terborgh, J. S. Brashares, M. E. Power, J. Berger, W. J. Bond, S. R. Carpenter, T. E. Essington, R. D. Holt, and J. B. Jackson. 2011. Trophic downgrading of planet Earth. Science **333**:301-306.
EU. 2016. Written Declaration to the EU Parliament on “Trophy Hunting”. Dated 8 January 2016. European Union.
European Commision. Soil Compaction. DG Joint Research Centre, Institute for Environment and Sustainability.
Evans, P., and L. Williamson. 1976. Return of the eastern turkey. Texas Parks and Wildlife.
Ezenwa, V. O. 2004. Parasite infection rates of impala (Aepyceros melampus) in fenced game reserves in relation to reserve characteristics. Biological Conservation **118**:397–401.
Faza, A. P., I. S. Pinto, I. Fonseca, G. R. Antunes, C. M. Monteiro, E. Daemon, S. Muniz Mde, M. F. Martins, J. Furlong, and M. C. Prata. 2013. A new approach to characterization of the resistance of populations of Rhipicephalus microplus (Acari: Ixodidae) to organophosphate and pyrethroid in the state of Minas Gerais. Brazil. Exp. Parasitol. **134**:519-523.

Ferguson, K., L. Adam, and F. Jori. 2012. An adaptive monitoring programme for studying impacts along the western boundary fence of Kruger National Park, South Africa. Fencing for Conservation:105-123.

Ferguson, K., and J. Hanks. 2010. Fencing impacts: A review of the environmental, social and economic impacts of game and veterinary fencing in Africa with particular reference to the Great Limpopo and Kavango-Zambezi Transfrontier Conservation Areas. Pretoria: Mammal Research Institute.

Fernandez-de-Mera, I. G., C. Gortazar, J. Vicente, U. Hofle, and Y. Fierro. 2003. Wild boar helminths: risks in animal translocations. Veterinary Parasitology **115**:335–341.

Feulner, P. G., J. Gratten, J. W. Kijas, P. M. Visscher, J. M. Pemberton, and J. Slate. 2013. Introgression and the fate of domesticated genes in a wild mammal population. Molecular Ecology **22**:4210-4221.

Financial Times. Undated. Definition of business sustainability. http://lexicon.ft.com/Term?term=business-sustainability.

Fischer, A., V. Kereži, B. Arroyo, M. Mateos-Delibes, D. Tadie, A. Lowassa, O. Krange, and K. Skogen. 2013a. (De)legitimising hunting – Discourses over the morality of hunting in Europe and eastern Africa. Land Use Policy **32**:261-270.

Fischer, A., C. Sandström, M. Delibes-Mateos, B. Arroyo, D. Tadie, D. Randall, F. Hailu, A. Lowassa, M. Msuha, and V. Kereži. 2013b. On the multifunctionality of hunting–an institutional analysis of eight cases from Europe and Africa. Journal of Environmental Planning and Management **56**:531-552.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology **8**:629-644.

Flynn, D. F., M. Gogol-Prokurat, T. Nogeire, N. Molinari, B. T. Richers, B. B. Lin, N. Simpson, M. M. Mayfield, and F. DeClerck. 2009. Loss of functional diversity under land use intensification across multiple taxa. Ecology Letters **12**:22-33.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, and H. K. Gibbs. 2005. Global consequences of land use. Science **309**:570-574.

Fourie, L. 1996. Poisoning of wildlife in South Africa. Journal of the South African Veterinary Association **67**:74-76.

Fowler, M. E. 2001. Host and helminth parasites–an evolutionary perspective. Pages 7-21 *in* N. a. A. Chowdhury, A.A., editor. Helminths of Wildlife. Science Publ Inc, UK.

Frankham, R. 1995. Inbreeding and extinction: a threshold effect. Conservation Biology **9**:792-799.

Frankham, R. 1996. Relationship of genetic variation to population size in wildlife. Conservation Biology **10**:1500-1508.

Frankham, R. 2005. Genetics and extinction. Biological Conservation **126**:131–140.

Frankham, R. 2008. Genetic adaptation to captivity in species conservation programs. Molecular Ecology **17**:325–333.

Frankham, R., J. D. Ballou, M. D. Eldridge, R. C. Lacy, K. Ralls, M. R. Dudash, and C. B. Fenster. 2011. Predicting the probability of outbreeding depression. Conservation Biology **25**:465-475.

Freitag, S., and T. J. Robinson. 1993. Phylogeographic patterns in mitochondrial DNA of the ostrich (Struthio camelus). The Auk:614-622.

Fynn, R. W. S., and T. G. O'Connor. 2000. Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. Journal of Applied Ecology **37**:491-507.

Gadd, M. E. 2012. Barriers, the beef industry and unnatural selection: A review of the impact of veterinary fences on mammals in southern Africa. Pages 153-186 *in* M. J. Somers and M. W. Hayward, editors. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.

Gamborg, C., and F. S. Jensen. 2017. Attitudes towards recreational hunting: A quantitative survey of the general public in Denmark. Journal of Outdoor Recreation and Tourism **17**:20-28.

Garber, P. M. 1990. Famous First Bubbles. The Journal of Economic Perspectives **4**:35–54.
Garcia, G., J. Vergara, and R. Lombardi. 2011. Genetic characterization and phylogeography of the wild boar Sus scrofa introduced into Uruguay. Genetics and Molecular Biology **34**:329-227.
Geist, V. 1992. Deer ranching for products and paid hunting: threat to conservation and biodiversity by luxury markets. Pages 554-561 The Biology of Deer. Springer.
Genever, S. 2016. African safari operators call for an end to captive breeding & canned lion hunting. SA Breaking News. http://www.sabreakingnews.co.za/2016/05/03/african-safari-operators-call-for-an-end-to-captive-breeding-canned-lion-hunting/
George, J. E., R. B. Davey, and J. M. Pound. 2002. Introduced ticks and tick-borne diseases: the threat and approaches to eradication. Veterinary Clinics of North America: Food Animal Practice **18**:401-416.
George, J. E., J. M. Pound, and R. B. Davey. 2004. Chemical control of ticks on cattle and the resistance of these parasites to acaricides. Parasitology **129**:353-366.
Gosling, S. D. 2001. From mice to men: what can we learn about personality from animal research? Psychological bulletin **127**:45.
Graham, B. 2003. The Intelligent Investor. Revised edition. HarperCollins Publications, New York.
Green, R. E., I. Newton, S. Shultz, A. A. Cunningham, M. Gilbert, D. J. Pain, and V. Prakash. 2004. Diclofenac poisoning as a cause of vulture population declines across the Indian subcontinent. Journal of Applied Ecology **41**:793-800.
Greenwood, R. J. 1975. Reproduction and development of four mallard lines. Prairie Naturalist **7**:9-16.
Grey-Ross, R., C. T. Downs, and K. Kirkman. 2009. Reintroduction failure of captive-bred oribi (Ourebia ourebi). South African Journal of Wildlife Research **39**:34-38.
Grobler, J. P., I. Rushworth, J. S. Brink, P. Bloomer, A. Kotze, B. Reilly, and S. Vrahimis. 2011. Management of hybridization in an endemic species: decision making in the face of imperfect information in the case of the black wildebeest—Connochaetes gnou. European Journal of Wildlife Research **57**:997.
Grobler, P., A. M. v. Wyk, D. L. Dalton, B. J. v. Vuuren, and A. Kotzé. 2017. Assessing introgressive hybridization between blue wildebeest (Connochaetes taurinus) and black wildebeest (*Connochaetes gnou*) from South Africa. Conservation Genetics.
Hanks, J. 2010. Death by instinct. Page 72 Africa Geographic.
Hare, B., I. Plyusnina, N. Ignacio, O. Schepina, A. Stepika, R. Wrangham, and L. Trut. 2005. Social cognitive evolution in captive foxes is a correlated by-product of experimental domestication. Current Biology **15**:226-230.
Harrison, R. G., and E. A. Larson. 2014. Hybridization, Introgression, and the Nature of Species Boundaries. Journal of Heredity **105**:795-809.
Hayward, M., G. Hayward, D. Druce, and G. Kerley. 2009. Do fences constrain predator movements on an evolutionary scale? Home range, food intake and movement patterns of large predators reintroduced to Addo Elephant National Park, South Africa. Biodiversity and Conservation **18**:887-904.
Hayward, M. W. 2006. Prey preferences of the spotted hyaena (Crocuta crocuta) and degree of dietary overlap with the lion (Panthera leo). Journal of Zoology **270**:606-614.
Hayward, M. W., and G. I. H. Kerley. 2009. Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? Biological Conservation **142**:1-13.
Hedrick, P. 2013. Conservation genetics and the persistence and translocation of small populations: bighorn sheep populations as examples. Animal Conservation.
Hedrick, P., R. Fredrickson, and H. Ellegren. 2001. Evaluation of d^2, a microsatellite measure of inbreeding and outbreeding, in wolves with a known pedigree. Evolution **55**:1256-1260.
Hedrick, P. W., and S. T. Kalinowski. 2000. Inbreeding depression in conservation biology. Annual Review of Ecology and Systematics:139-162.

Hetem, R. S., B. A. de Witt, L. G. Fick, A. Fuller, G. I. Kerley, L. C. Meyer, D. Mitchell, and S. K. Maloney. 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.

Hitchcock, R. K. 2000. Traditional African Wildlife Utilization: Subsistence Hunting, Poaching, and Sustainable Use. Pages 389-415 Wildlife Conservation by Sustainable Use. Conservation Biology Series. Kluwer Academic Publishers, Netherlands.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts. 2005. Confronting a biome crisis: global disparities of habitat loss and protection. Ecology Letters **8**:23-29.

Hofer, H., K. L. Campbell, M. L. East, and S. A. Huish. 1996. The impact of game meat hunting on target and non-target species in the Serengeti. The exploitation of mammal populations **117**:146.

Hoffman, L. C., K. Crafford, N. Muller, and D. W. Schutte. 2003. Perceptions and consumption of game meat by a group of tourists visiting South Africa. South African Journal of Wildlife Research **33**:125-130.

Hoffman, L. C., and E. Wiklund. 2006. Game and venison – meat for the modern consumer. Meat Science **74**:197–208.

Hofmeyr, M., and G. Van Dyk. 1998. Cheetah introductions to two North West Parks: case studies from Pilanesburg National Park and Madikwe Game reserve. Page 71 Symposium on cheetahs as game ranch animals. South African Veterinary Association Wildlife Group, Onderstepoort, South Africa.

Hofreiter, M., and T. Schöneberg. 2010. The genetic and evolutionary basis of colour variation in vertebrates. Cellular and Molecular Life Sciences **67**:2591-2603.

Humphries, P. 2003. Risk management. 17 July 2003. http://www.risk.net/risk-management/1525705/reputation-biggest-risk-says-pwc. Accessed August 2016.

Hunter, L. T., P. White, P. Henschel, L. Frank, C. Burton, A. Loveridge, G. Balme, C. Breitenmoser, and U. Breitenmoser. 2013. Walking with lions: why there is no role for captive-origin lions Panthera leo in species restoration. Oryx **47**:19-24.

IoDSA. 2009. King Report on Governance for South Africa and the King Code of Governance Principles. Institute of Directors in Southern Africa, Sandton, South Africa.

IUCN. 2016. Informing decisions on trophy hunting. A Briefing Paper for European Union Decision-makers regarding potential plans for restriction of imports of hunting trophies. UCN CEESP/SSC Sustainable Use and Livelihoods Specialist Group, International Union for the Conservation of Nature, https://www.iucn.org/downloads/iucn_informingdecisionsontrophyhuntingv1.pdf.

IUCN SSC. 2012. Guiding principles on trophy hunting as a tool for creating conservation incentives. Ver. 1.0., International Union for the Conservation of Nature Species Survival Commission, Gland, Switzerland.

Jackson, C. R., K. Marnewick, P. A. Lindsey, E. Roskraft, and M. Robertson, P. 2016. Evaluating habitat connectivity methodologies: a case study with endangered African wild dogs in South Africa. Landscape Ecology **31**:1433-2447.

Jagd and Hound. 2016. E-mail to prospective exhibitors for Germany's leading hunting show.

Jensen, P., and L. Andersson. 2005. Genomics meets ethology: a new route to understanding domestication, behavior, and sustainability in animal breeding. AMBIO: A Journal of the Human Environment **34**:320-324.

Keller, L. F., and D. M. Waller. 2002. Inbreeding effects in wild populations. Trends Ecol Evol **17**:230-241.

Kirkwood, J. 2010. Introduction: Darwinian selection, selective breeding and the welfare of animals. Animal Welfare **19**:1-5.

Klare, U., Kamler, J.F., U. Stenkewitz, and D. W. Macdonald. 2010. Diet, prey selection and predation impact of black-backed jackal in South Africa. Journal of Wildlife Management **74**:1030-1042.

Kohane, M. J., and P. A. Parsons. 1988. Domestication: evolutionary change under stress. Evolutionary Biology **23**:31-48.

Kosowska, B., T. Strzała, M. Moska, J. Kujałowicz, E. Oryszczak, J. Piecyk, M. Sawczuk, J. Ślusarczyk, and K. Wasilewska. 2005. Life-span of long and rough coated Dachshunds in depedence of their inbreed level and sex. EJPAU **8**:71.

Kriek, J. C. 2005. The sable antelope. Pages 138-146 *in* J. d. P. Bothma and N. v. Rooyen, editors. Intensive wildlife production in southern Africa. Van Schaik, Pretoria.

Krige, D. 2012. Fields of Dreams, Fields of Schemes: Ponzi Finance and Multi-Level Marketing in South Africa. Popular Economies in South Africa **82**:69-92.

Kristensen, T. N., A. A. Hoffmann, C. Pertoldi, and A. V. Stronen. 2015. What can livestock breeders learn from conservation genetics and vice versa? Frontiers in genetics **6**.

Kruger, J., F. Parrini, J. Koen, K. Collins, E. J. Nel, and M. F. Child. 2016. A conservation assessment of Hippotragus equinus.*in* M. F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo, and H. T. Davies-Mostert, editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa, Pretoria.

Kruger, J. H. 1976. An investigation of certain dermatological, morphological and genetic aspects of the black springbok and the white springbok (Antidorcas marsupialis) [in Afrikaans]. University of Pretoria, Pretoria.

Laikre, L. 1999. Hereditary Defects and Conservation Genetic Management of Captive Populations. Zoo Biology **18**:81-99.

Laikre, L., M. K. Schwartz, R. S. Waples, N. Ryman, and G. W. Group. 2010. Compromising genetic diversity in the wild: unmonitored large-scale release of plants and animals. Trends in Ecology & Evolution **25**:520-529.

Landbou Weekblad. 2016. Namibié se jagbedryf troef Suid Afrika. Pages 79-82 Landbou Weekblad, Julie 2016.

Larson, G., and J. Burger. 2013. A population genetics view of animal domestication. Trends in Genetics **29**:197-205.

Le Maitre, D. C., S. J. Milton, C. Jarmain, C. A. Colvin, I. Saayman, and J. H. J. Vlok. 2007. Landscape-scale hydrology of the Little Karoo: linking ecosystems, ecosystem services and water resources. Frontiers in Ecology and the Environment **5**:261-270.

LEDET. 2013. Tourism strategy 2013/2014. Presentation by the Limpopo Department of Economic Development Environment and Tourism. Limpopo. Limpopo Department of Economic Development, Environment and Tourism.

LEDET. 2016. Waterberg District Bioregional Plan. Limpopo Department of Economic Development, Environment and Tourism, Polokwane.

Leffler, E. M., K. Bullaughey, D. R. Matute, W. K. Meyer, L. Segurel, A. Venkat, P. Andolfatto, and M. Przeworski. 2012. Revisiting an old riddle: what determines genetic diversity levels within species? PLoS Biol **10**:e1001388.

Leroy, G. 2011. Genetic diversity, inbreeding and breeding practices in dogs: results from pedigree analyses. The Veterinary Journal **189**:177-182.

Lindsey, P. A., R. Alexander, L. G. Frank, A. Mathieson, and S. S. Romañach. 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, P. A., G. A. Balme, P. J. Funston, P. H. Henschel, and L. T. Hunter. 2016. Life after Cecil: channelling global outrage into funding for conservation in Africa. Conservation Letters **9**:296-301.

Lindsey, P. A., J. T. du Toit, and M. G. L. Mills. 2005. Attitudes of ranchers towards African wild dogs Lycaon pictus: Conservation implications on private land. Biological Conservation **125**:113-121.

Lindsey, P. A., C. P. Havemann, R. M. Lines, A. E. Price, T. A. Retief, and T. Rhebergen. 2013. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx **47**:41-53.

Lindsey, P. A., C. L. Masterson, A. L. Beck, and S. Romanach. 2012. Ecological, Social and Financial Issues Related to Fencing as a Conservation Tool in Africa. Pages 215-234

in M. J. Somers and M. W. Hayward, editors. Fencing for conservatoin. Springer, New York.
Lindsey, P. A., S. S. Romanach, and H. T. Davies-Mostert. 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, P. A., P. A. Roulet, and S. S. Romanach. 2007. Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. Biological Conservation **134**:455-469.
Linnel, J. D. C., R. Aanes, and J. E. Swenson. 1997. Translocation of carnivores as a method for managing problem animals: a review. Biodiversity and Conservation **6**:1245-1257.
Loehr, J., J. Carey, H. Ylönen, and J. Suhonen. 2008. Coat darkness is associated with social dominance and mating behaviour in a mountain sheep hybrid lineage. Animal Behaviour **76**:1545-1553.
Lombaard, L. 2016. ShockWildlifeTruths: Captive bred lions will forever be victims of the hunting industry. News24, http://traveller24.news24.com/Explore/Green/shockwildlifetruths-captive-bred-lions-will-forever-be-victims-of-the-hunting-industry-20160628.
Lovari, S., B. Pellizzi, R. Boesi, and L. Fusani. 2009. Mating dominance amongst male Himalayan tahr: blonds do better. Behavioural Processes **81**:20-25.
Loveridge, A. J., A. W. Searle, F. Murindagomo, and D. W. Macdonald. 2007. The impact of sport-hunting on the population dynamics of an African lion population in a protected area. Biological Conservation **134**:548-558.
Low, A. B., and A. G. Rebelo. 1996. Vegetation of South Africa, Lesotho and Swaziland. Department of Environmental Affairs and Tourism, Pretoria.
Ludwig, F., De Kroon, H., and Prins, H.H.T. 2008. Impacts of savanna trees on forage quality for a large African herbivore. Oecologia **155**:487–496.
Lyles, A. M., and A. P. Dobson. 1993. Infectious Disease and Intensive Management: Population Dynamics, Threatened Hosts, and Their Parasites. Journal of Zoo and Wildlife Medicine **24**:315-326.
Lynch, M., and M. O'Hely. 2001. Captive breeding and the genetic fitness of natural populations. Conservation Genetics **2**:363-378.
Lynch, M., and B. Walsh. 1998. Genetics and analysis of quantitative traits. Sinauer Sunderland, MA.
Lyons, J. A., and D. J. D. Natusch. 2011. Wildlife laundering through breeding farms: Illegal harvest, population declines and a means of regulating the trade of green pythons (Morelia viridis) from Indonesia. Biological Conservation **144**:3073–3081.
Maciejewski, K., and G. I. H. Kerley. 2014. Understanding tourists' preference for mammal species in private protected areas: is there a case for extralimital species for ecotourism? . PLoS ONE **9**:e88192.
Mackay, C. 1841. Memoirs of extraordinary popular delusions and the madness of crowds. https://web.archive.org/web/20080331091055/http://www.econlib.org/library/mackay/macExContents.html. Accessed February 2017.
Mackey, R. L., Page, B.R., Grobler, D., and Slotow, R. 2009. Modelling the effectiveness of contraception for controlling introduced populations of elephant in South Africa. African Journal of Ecology **47**:747–755.
MacLean, G. L. 1970. An analysis of the avifauna of the southern Kalahari Gemsbok National Park. Zoologica Africana **5**:249–273.
Manfredo, M. J. 2002. Wildlife Viewing in North America: A Management Planning Handbook. Oregon State University Press, Corvallis, Oregon.
Mapholi, N. O., A. Maiwashe, O. Matika, V. Riggio, S. B. Bishop, M. D. MacNeil, C. Banga, I. F. Taylor, and K. Dzama. 2016. Genome-wide association study of tick resistance in South African nguni cattle. Ticks and tick-borne diseases **7**:487-497.
Marliave, J. B., W. F. Gergits, and S. Aota. 1993. F10 Pandalid shrimp – sex determination, DNA and dopamine as indicators of domestication and out crossing for wild pigment pattern. Zoo Biology **12**:435-451.

Marnewick, K., A. Beckhelling, D. Cilliers, E. Lane, M. G. L. Mills, K. Herring, P. Caldwell, R. Hall, and S. Meintjes. 2007. The status of the Cheetah in South Africa.*in* C. Breitenmoser and S. Durant, editors. The Status and Conservation Needs of the Cheetah in Southern Africa. Cat News Special Edition.

Marnewick, K., and M. J. Somers. 2015. Home ranges of cheetahs Acinonyx jubatus outside protected areas in South Africa. African Journal of Wildlife Research: Carnivore Special Edition **45**:223–232.

Marsden, C. D., D. Ortega-Del Vecchyo, D. P. O'Brien, J. F. Taylor, O. Ramirez, C. Vilà, T. Marques-Bonet, R. D. Schnabel, R. K. Wayne, and K. E. Lohmueller. 2016. Bottlenecks and selective sweeps during domestication have increased deleterious genetic variation in dogs. Proceedings of the National Academy of Sciences **113**:152-157.

Marshall, C. M., G. A. Seaman, and F. A. Hayes. 1963. A critique on the tropical cattle fever tick controversy and its relationship to white-tailed deer. Trans. N. Am. Wildl. Nat. Res. Conf. **28**:225-232.

Marshall, F. B., K. Dobney, T. Denham, and J. M. Capriles. 2014. Evaluating the roles of directed breeding and gene flow in animal domestication. Proceedings of the National Academy of Sciences **111**:6153-6158.

Martindale, G. 2007. Influence of livestock grazing on plant diversity of Highland Sourveld grassland in KwaZulu-Natal. University of the Witwatersrand, South Africa.

Massei, G., R. J. Quy, J. Gurney, and D. P. Cowan. 2010. Can translocations be used to mitigate human-wildlife conflicts? Wildlife Research **37**:428-439.

Mateo-Tomás, P., and P. P. Olea. 2010. When hunting benefits raptors: a case study of game species and vultures. European Journal of Wildlife Research **56**:519-528.

Maublanc, M. L., E. Bideau, D. Picot, J. L. Rames, M. Dubois, H. Ferte, and J. F. Gerard. 2009. Demographic crash associated with high parasite load in an experimental roe deer (Capreolus capreolus) population. Eur. J. Wildl. Res. **55**:621-625.

Maunder, M., and O. Byers. 2005. The IUCN Technical Guidelines on the Management of Ex situ Populations for Conservation: reflecting major changes in the application of ex situ conservation. Oryx **39**:95-98.

Mbaiwa, J. E. 2017. Effects of the safari hunting tourism ban on rural livelihoods and wildlife conservation in Northern Botswana. South African Geographical Journal:1-21.

McCleery, R., J. A. Hostetler, and M. K. Oli. 2014. Better off in the wild? Evaluating a captive breeding and release program for the recovery of an endangered rodent. Biological Conservation **169**:198-205.

McCrindle, C. M. E., M. Siegmund-Schultze, H. W. Heeb, A. Valle Zárate, and A. Ramrajh. 2013. Improving food security and safety through use of edible by-products from wild game. Environment, Development and Sustainability **15**:1245-1257.

McDougall, P., D. Réale, D. Sol, and S. Reader. 2006. Wildlife conservation and animal temperament: causes and consequences of evolutionary change for captive, reintroduced, and wild populations. Animal Conservation **9**:39-48.

Mcghee, M. B., V. F. Nettles, E. A. Rollor, A. K. Prestwood, and W. R. Davidson. 1981. Studies on cross-transmission and pathogenicity of Haemonchus contortus in white-tailed deer, domestic cattle and sheep. J. Wildl. Dis. **17**:353–364.

McPhee, M. E. 2003. Generations in captivity increases behavioural variance: considerations for captive breeding and reintroduction programs. Biological Conservation **115**:71-77.

MET. 2017. Charter for Hunting, Wildlife Conservation and Habitat Protection in Africa., Ministry of Environment and Tourism, Namibia, Windhoek, Namibia.

Milchunas, D. G., and W. K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecological Monographs **63**:327-366.

Miller, J. R., G. Balme, P. A. Lindsey, A. J. Loveridge, M. S. Becker, C. Begg, H. Brink, S. Dolrenry, J. E. Hunt, and I. Jansson. 2016a. Aging traits and sustainable trophy hunting of African lions. Biological Conservation **201**:160-168.

Miller, S. M., A. J. Guthrie, and C. K. Harper. 2016b. Single base-pair deletion in ASIP exon 3 associated with recessive black phenotype in impala (Aepyceros melampus). Animal genetics **47**:511-512.
Milton, S. J., and W. R. J. Dean. 1995. How useful is the keystone species concept, and can it be applied to Acacia erioloba in the Kalahari Desert? Journal for Nature Conservation **4**:147–156.
Mitchell, M. 1996. Acaricide resistance - Back to basics. Trop. Anim. Health Prod **28**:53-58.
Mongera, A., and C. M. Dooley. 2013. Of white tigers and solute carriers. Pigment Cell & Melanoma Research **26**:787-788.
Morgera, E. 2014. conceptualizing Benefit-sharing as the Pursuit of Equity in Addressing Global Environmental challenges.
Mosupi, I. 2016. Bank says no finance to captive lion trade. TimesLIVE. http://www.timeslive.co.za/thetimes/article1924712.ece.
Mungube, E. O., L. W. Wamae, G. A. Omondi, and G. Mwangi. 2015. Prevalence of multiple resistant Haemonchus and Ostertagia species in goats and cattle in Machakos, Eastern Kenya. Livestock Research for Rural Development **27**.
Musiker, D. G., T. France, and L. A. Hallenbeck. 1995. The public trust and Parens Patriae Doctrines: Protecting wildlife in uncertain political times. Pub. Land L. Rev., 16, p.87 **16**:87.
Myers, J. 1999. Investment Manias and Speculative Bubbles. Adapted from Profits Without Panic: Investment Psychology For Personal Wealth © Jonathan Myers 1999. http://www.psychonomics.com/research/a&s/specbubb.htm.
Myhr, A. I., and T. Traavik. 2002. The Precautionary Principle: Scientific uncertainty and omitted research in the context of GMO use and release. Journal of Agricultural and Environmental Ethics **15**.
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.
Naidoo, R., C. L. Weaver, R. W. Diggle, G. Matongo, G. Stuart-Hill, and C. Thouless. 2016. Complementary benefits of tourism and hunting to communal conservancies in Namibia. Conservation Biology **30**:628-638.
NAMC. 2006. Report on the investigation to identify problems for sustainable growth and development in South African wildlife ranching. National Agricultural Marketing Council, Pretoria.
NAPHA. 2015. Captive breeding puts hunting, conservation at risk: Shifeta. Media report by Namibia Professional Hunting Association. http://www.energy100fm.com/?q=content/captive-breeding-puts-hunting-conservation-risk-shifeta.
NAPHA. 2017. Press release from the Namibia Professional Hunting Association. November, 2017. http://www.napha-namibia.com/fileadmin/user_upload/NAPHA_Press_Release_24.11.2017.pdf
Needham, T., and L. C. Hoffman. 2013. The inheritance of coat colour and the implication of breeding for colour variants in game. Department of Animal Sciences, Faculty of AgriScience, University of Stellenbosch.
Nelson, F., P. Lindsey, and G. Balme. 2013. Trophy hunting and lion conservation: a question of governance? Oryx **47**:501-509.
Newman, D., and D. Pilson. 1997. Increased probability of extinction due to decreased genetic effective population size: experimental populations of Clarkia pulchella. Evolution:354-362.
Niehaus, C. 2014. Runaway Game Prices – An Economic Bubble With Major Conservation Risks? SA Hunter's Magazine.
Njobe, K., and N. King. 2016. A consolidated report of the national dialogue workshops on selective and intensive breeding of colour variants. Kay Ann Consulting.
Nogales, M., A. Martin, B. R. Tershy, C. J. Donlan, D. Veitch, N. Peurta, B. Wood, and J. A. Alonso. 2004. A review of feral cat eradication on islands. Conservation Biology **18**:310-319.

Nongwe, N. 2008. A systematic conservation assessment of habitat transformation and degradation in the Little Karoo, South Africa. Dissertation. University of Cape Town, Cape Town, South Africa.

Nowak, K. 2016. The End of 'Canned' Lion Hunting May Be in Sight. National Geographic, 11 March 2016. http://news.nationalgeographic.com/2016/03/140311-trophy-hunting-blood-lions-south-africa-conservation-captive-breeding/.

NSC. 2017. Reversal of policy on canned lion shooting. Letter to PHASA. November, 2017. NSC Facebook Page. https://www.facebook.com/search/str/Nordisk+Safari+Klub+canned+lion/keywords_pages?see_more_ref=eyJzaWQiOiIiLCJyZWYiOiJzZWVfbW9yZSJ9.

O'Connor, T. G. 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, T. G., P. Kuyler, K. P. Kirkman, and B. Corcoran. 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, T. G., J. R. Puttick, and M. T. Hoffman. 2014. Bush encroachment in southern Africa: changes and causes. African Journal of Range & Forage Science **31**:67-88.

O'Ryan, C., E. H. Harley, M. W. Bruford, M. Beaumont, R. K. Wayne, and M. I. Cherry. 1998. Microsatellite analysis of genetic diversity in fragmented South African buffalo populations. Animal Conservation **1**:85-94.

O'Regan, H. J., and A. C. Kitchener. 2005. The effects of captivity on the morphology of captive, domesticated and feral animals. Mammal Review **35**:215-230.

Ogada, D. L. 2014. The power of poison: pesticide poisoning of Africa's wildlife. Annals of the New York Academy of Sciences **1322**:1-20.

OPHAA. 2017a. Captive bred lion hunting; in support of PHASA and Stan Burger. Open letter from the Chair. www.sahunters.co.za., Operators and Professional Hunters Associations of Africa.

OPHAA. 2017b. Press release: PHASA suspended from OPHAA. November 2017. www.OPHAA.org.

Osborne, M. 2013. Is medicated grit the answer? www.williampowell.com/blog/shooting/is-medicated-grit-theanswer/ (accessed 15 September 2015).

.

Ostrowski, S., E. Bedin, D. M. Lenain, and A. H. Abuzinada. 1998. Ten years of Arabian oryx conservation breeding in Saudi Arabia - Achievements and regional perspectives. Oryx **32**:209-222.

OUTDOORHUB. 2015. HUNTING NEWS: Zimbabwe Park to Cull 200 Lions, Lack of Hunters. Article on the Outdoorhub. Accessed on 3 March 2016.

Packer, C., H. Brink, B. M. Kissui, H. Maliti, H. Kushnir, and T. Caro. 2011. Effects of Trophy Hunting on Lion and Leopard Populations in Tanzania. Conservation Biology **25**:142–153.

Packer, C., R. Holt, P. Hudson, K. D. Lafferty, and A. P. Dobson. 2003. Keeping the herds healthy and alert: implications of predator control for infectious disease. Ecology Letters **6**:797- 802.

Parrini, F., J. Koen, D. Dalton, and J. Eksteen. 2016. A conservation assessment of Hippotragus niger niger.*in* M. F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo, and H. T. Davies-Mostert, editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, Pretoria, South Africa.

Pato, F., L. Vázquez, N. Díez-Baños, C. López, R. Sánchez-Andrade, G. Fernández, P. Díez-Baños, R. Panadero, P. Díaz, and P. Morrondo. 2013. Gastrointestinal nematode infections in roe deer (Capreolus capreolus) from the NW of the Iberian Peninsula: Assessment of some risk factors. Veterinary Parasitology **196**:136-142.

Pekin, B. K., and B. C. Pijanowski. 2012. Global land use intensity and the endangerment status of mammal species. Diversity and Distributions.

Pertoldi, C., R. Bijlsma, and V. Loeschcke. 2007. Conservation genetics in a globally changing environment: present problems, paradoxes and future challenges. Biodiversity and Conservation **16**:4147-4163.

Pfitzer, S., and I. Colenbrander. 2005. The bushbuck.*in* J. D. P. Bothma and N. V. Rooyen, editors. Intensive wildlife production in southern Africa. Van Schaik, Pretoria.

Pfitzer, S., and H. Kohrs. 2005. The nyala. Pages 169-186 *in* J. d. P. Bothma and N. v. Rooyen, editors. Intensive wildlife production in southern Africa. Van Schaik, Pretoria.

PHASA. 2016. PHASA'S Stand on Captive-Bred Lions Hailed by International Hunting Community. Press release. PHASA website. Accessed March 2016.*in* P. H. A. o. S. Africa, editor.

PHAZ. 2017. Statement by the Professional Hunters Association of Zambia (PHAZ). November 2017. Lusaka, Zambia.

Phillips, L. 2015. Game rancher receives hefty fine from industry body for wildlife poisoning. Farmers Weekly, http://www.farmersweekly.co.za/news.aspx?id=79567&h=Gamerancherreceiveshefty finefromindustrybodyforwildlifepoisoning Accessed on 9 October 2015.

Pienaar, E. F., E. C. Rubino, M. Saayman, and P. van der Merwe. 2017. Attaining sustainable use on private game ranching lands in South Africa. Land Use Policy **65**:176-185.

Pienaar, R. C. 2013. The feeding ecology of extralimital Nyala (Tragelaphus angasii) in the Arid Moasic Thicket of the Southern Cape. Nelson Mandela Metropolitan University, Port Elizabeth.

Pieterson, D. W. 2013. Behavioural ecology and conservation biology of ground pangolins Smuthsia temminckii in the Kalahari desert. University of Pretoria, Pretoria.

Pieterson, D. W., A. E. Mckecknie, and R. Jansen. 2014. A review of the anthropogenic threats faced by Temmincks ground pangolin Smutsia temminckii in southern Africa. South African Journal of Wildlife Research **44**:167-178.

Piett, S., H. A. Hager, and C. Gerrard. 2015. Characteristics for evaluating the conservation value of species hybrids. Biodiversity and Conservation **24**:1931-1955.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics **52**:273-288.

Pitman, R. T., J. Fattebert, S. T. Williams, K. S. Williams, R. A. Hill, L. T. Hunter, R. Slotow, and G. A. Balme. 2016. The Conservation Costs of Game Ranching. Conservation Letters.

Pole, A., I. Gordon, L. Martyn, L. Gorman, and M. Macaskill. 2004. Prey selection by African wild dogs (Lycaon pictus) in southern Zimbabwe. Journal of Zoology **262**:207-215.

Polley, M. 2014. Canned lion hunting: The dark shadow of South Africa's wildlife tourism industry. Eturbonews, http://www.eturbonews.com/52794/canned-lion-hunting-dark-shadow-south-africa-s-wildlife-tourism-. Accessed 1 March 2016.

Potts, G. 1992. Conservation through wise-use hunting. Pages 718-726 Wildlife 2001: Populations. Springer.

Price, E. O. 1997. Behavioural genetics and the process of animal domestication. Pages 31-65 *in* T. Grandin, editor. Genetics and the Behaviour of Domestic Animals. Academic Press, New York.

Price, E. O. 1999. Behavioral development in animals undergoing domestication. Applied Animal Behaviour Science **65**:245-271.

Prince, H. H., P. B. Siegel, and G. W. Cornwell. 1970. Inheritance of egg production and juvenile growth in Mallards. Auk **87**:342-352.

Prins, H. H. T., and J. G. Grootenhuis. 2000. Introduction: The value of priceless wildlife. Pages 1-12 *in* H. H. T. Prins, J. G. Grootenhuis, and T. T. Dolan, editors. Wildlife conservation by sustainable use. Kluwer Academic Publishers, Boston/Dordrecht/London.

Prugh, L. R., C. J. Stoner, C. W. Epps, W. T. Bean, W. J. Ripple, A. S. Laliberte, and J. S. Brashares. 2009. he Rise of the Mesopredator. BioScience **59**:779-791.

Quinn, M. A., and D. D. Walgenbach. 1990. Influence of grazing history on the community structure of grasshoppers of a mixed-grass prairie. Environmental Entomology **19**:1756-1766.

Reed, D. H., and R. Frankham. 2003. Correlation between fitness and genetic diversity. Conservation Biology **17**:230-237.

Reilly, B. K., E. A. Sutherland, and V. Harvey. 2003. The nature and extent of wildlife ranching in Gauteng province, South Africa. South African Journal of Wildlife Research **33**:141-144.

Reinert, H. K. 1991. Translocation as a conservation strategy for amphibians and reptiles: some comments, concerns, and observations. Herperologica **47**:357-363.

Reissmann, M., and A. Ludwig. 2013. Pleiotropic effects of coat colour-associated mutations in humans, mice and other mammals. Pages 576-586 *in* Seminars in cell & developmental biology. Elsevier.

Reusch, T. B., and T. E. Wood. 2007. Molecular ecology of global change. Molecular Ecology **16**:3973-3992.

Reyers, B., P. J. O'Farrell, R. M. Cowling, B. N. Egoh, D. C. L. Maitre, and J. H. J. Vlok. 2009. Ecosystem services, land-cover change, and stakeholders: finding a sustainable foothold for a semiarid biodiversity hotspot. Ecology and Society **14**:38.

Rhymer, J. M., and D. Simberloff. 1996. Extinction by hybridization and introgression. Annual Review of Ecology and Systematics **27**:83-109.

Ripple, W. J., and R. L. Beschta. 2006. Linking a cougar decline, trophic cascade, and catastrophic regime shift in Zion National Park. Biological Conservation **133**:397-408.

Ripple, W. J., J. A. Estes, R. L. Beschta, C. C. Wilmers, E. G. Ritchie, M. Hebblewhite, J. Berger, B. Elmhagen, M. Letnic, M. P. Nelson, O. J. Schmitz, D. W. Smith, A. D. Wallach, and A. J. Wirsing. 2014. Status and Ecological Effects of the World's Largest Carnivores. Science **343**:151-162.

Ripple, W. J., and E. J. Larsen. 2000. Historic aspen recruitment, elk, and wolves in northern Yellowstone National Park, USA. Biological Conservation **95**:361-370.

Roberts, J. A. 1968. Resistance of cattle to the tick Boophilus microplus (Canestrini). II. Stages of the parasite against which resistance is manifest. Journal of Parasitology **54**:667-673.

Roland Ward. 2017. Roland Ward Ltd. Webpage. https://rowlandward.com/.

Roth, H. H., and G. Merz. 1997. Wildlife resources: a global account of economic use. Springer, Berlin, Germany.

Roush, R. T. 1993. Occurrence, genetics and management of insecticide resistance. Parasitology Today **9**:174-179.

Rowe-Rowe, D. T. 1994. The ungulates of Natal. Natal Parks Board, Pietermaritzburg.

Ruggiero, L. F., R. S. Holthausen, B. G. Marcot, K. B. Aubry, J. W. Thomas, and E. C. Meslow. 1988. Ecological dependency: the concept and its implications for research and management.*in* Transactions of the Fifty-third North American Wildlife and Natural Resources Conference.

Ruggiero, L. F., M. K. Schwartz, K. B. Aubry, C. J. Krebs, A. Stanley, and S. W. Buskirk. 2000. Species Conservation and Natural Variation Among Populations. Pages 101-116 Ecology and conservation of lynx in the United States. University of Colorado Press, Boulder, USA.

Saccheri, I. J., P. M. Brakefield, and R. A. Nichols. 1996. Severe inbreeding depression and rapid fitness rebound in the butterfly Bicyclus anynana (Satyridae). Evolution:2000-2013.

SAHGCA. 2014. Policy Position: Intensive and Selective Breeding to Enhance or Alter Genetic Characteristics of Indigenous Game Species for Commercial Purposes. South African Hunters and Game Conservation Association, www.sahunters.co.za.

Sand, P. H. 2014. The Concept of Public Trusteeship in the Transboundary Governance of Biodiversity. Canadian Yearbook of International Law **41**:92.

SAPA. 2015. The True Meaning of Canned Hunting. http://www.sapredators.co.za/p25/faq/the-true-meaning-of-canned-hunting.html.*in* S. A. P. Association, editor.
SAPA. 2017a. Letter to the USA regarding the ban on the import of trophies from intensively-bred lions. http://www.sapredators.co.za/a14/general-articles/sapa-letter-to-secretary-zinke.html. South African Predator Association.
SAPA. 2017b. SAPA Norms and standards for hunting managed ranch lions. http://www.sapredators.co.za/p33/sapa/sapa-norms-and-standards-for-hunting-managed-ranch-lions.html. South African Predator Association.
Sasidharan, V., E. Sirakaya, and D. Kerstetter. 2002. Developing countries and tourism ecolabels. Tourism Management **23**:161-174.
Sax, D. F., and S. D. Gaines. 2003. Species diversity: from global decreases to local increases. Trends in Ecology & Evolution **18**:561-566.
Scaglione, F. E., C. Schroder, G. Degiorgi, O. Zeira, and E. Bollo. 2010. Cranial malformations in related white lions (Panthera leo krugeri). Veterinary Pathology Online **47**:1095.
Schepers, A., N. Matthews, and H. N. v. Niekerk. 2018. Exploring wildlife predation in the Limpopo Province of South Africa: A double-hurdle approach. African Journal of Ecology **56**:301-311.
Schlaepfer, M. A., C. Hoover, and C. K. J. Dodd. 2005. Challenges in evaluating the impact of the trade in amphibians and reptiles on wild populations. BioScience **55**:256-264.
Schlaepfer, M. A., D. F. Sax, and J. D. Olden. 2010. The Potential Conservation Value of Non-Native Species. Conservation Biology **25**:428-437.
Schmutz, S., and T. Berryere. 2007. Genes affecting coat colour and pattern in domestic dogs: a review. Animal genetics **38**:539-549.
Schoeman, A. 2016. Trofeediere laat biltongjagters ly. Netwerk24; http://www.netwerk24.com/Sake/Landbou/trofeediere-laat-biltongjagters-ly-20160529. Accessed November 2016.
Scholtz, R., G. A. Kiker, G. D. Duckworth, U. M. Scharler, H. G. Mwambi, and F. J. Venter. 2016. Different drivers create spatial vegetation cover and vertical structure in semi-arid African savannas. African Journal of Range & Forage Science **33**:91-100.
Schroder, B., and B. K. Reilly. 2013. A comparison between tick species collected in a controlled and control free area on a game ranch in South Africa. Journal of the South African Veterinary Association **84**:5.
Schroder, B., A. C. Uys, and B. K. Reilly. 2006. A survey of free-living ixodid ticks on a commercial game farm in the Thabazimbi District, Limpopo Province, South Africa. Journal of the South African Veterinary Association **77**:141-144.
Scott-Shaw, R., and C. D. Morris. 2008. Are there indicator species of veld condition with the non-graminoid component of mesic grasslands? 43rd Annual Grassland Society of South Africa Congress, Badplaas, South Africa.
Sewall-Wright, S. 1977. Evolution and the genetics of populations. Vol. 3: experimental results and evolutionary deductions. Chicago, IL: University of Chicago Press.
Shein, E. 1997. Quick riches. The Western Producer. http://www.producer.com/1997/12/ostrich-bubble-bursts-dont-expect-quick-riches/. Accessed February 2017.
Shillinger, J. E. 1938. Deer in relation to Fever Tick eradication in Florida. Trans. N. Am. Wild. Conf. **3**:882-885.
Shrader, A. M., I. Little, B. Coverdale, and P. T. 2016. A conservation assessment of Ourebia ourebi ourebi.*in* M. F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo, and H. T. Davies-Mostert, editors. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, Pretoria, South Africa.
Sih, A., A. Bell, and J. C. Johnson. 2004. Behavioral syndromes: an ecological and evolutionary overview. Trends in Ecology & Evolution **19**:372-378.

Sih, A., Kats, L.B., and E. F. Maurer. 2000. Does phylogenetic inertia explain the evolution of ineffective antipredator behaviour in a sunfish-salamander system? . Behaviour Ecology & Sociobiology **49**:48-56.

Skead, C. J. 2007. Historical incidence of the larger land mammals in the broader Eastern Cape. 2nd edition. Centre for African Conservation Ecology, Nelson Mandela Metropolitan University, Port Elizabeth.

Slabbert, A. 2013. Why do business heavyweights like Johann Rupert invest in buffalo? Investment Insights 9/10/2013. Moneyweb. www.moneyweb.co.za/moneyweb-investment-insights/can-game-farming-outperform-the-jse.

Slade, B., M. L. Parrott, A. Paproth, M. J. Magrath, G. R. Gillespie, and T. S. Jessop. 2014. Assortative mating among animals of captive and wild origin following experimental conservation releases. Biology Letters **10**.

Slotow, R. 2012. Fencing for purpose: A case study of elephants in South Africa. Pages 91-104 *in* M. J. S. M. W. Hayward, editor. Fencing for conservation. Restriction of evolutionary potential or a riposte to threatening processes. Springer, New York.

Smallidge, S. T., H. Halbritter, N. K. Ashcroft, and J. C. Boren. 2008. Review of livestock management practices to minimize livestock depredation by wolves: Applicability to the southwest. New Mexico State University, New Mexico.

Snyman, H. A. 1999. Soil erosion and conservation. Pages 355-380 *in* N. M. Tainton, editor. Veld Management in South Africa. University of Natal Press, Pietermaritzburg.

South African National Biodiversity Institute. 2011. National Biodiversity Assessment. Department of Environmental Affairs, Pretoria.

Spear, D., and S. L. Chown. 2009. The extent and impacts of ungulate translocations: South Africa in a global context. Biological Conservation **142**:353-363.

Spenceley, A., P. Relly, H. Keyser, P. Warmeant, M. McKenzie, A. Mataboge, P. Norton, S. Mahlangu, and J. Seif. 2002. Responsible Tourism Manual for South Africa, Department for Environmental Affairs and Tourism, July 2002. Responsible Tourism Manual for South Africa **2**.

Spoorex. 2017. Colour variants: The rise and fall. Spoorex News, https://www.spoorexnews.co.za/single-post/2017/07/24/Colour-Variants-The-Rise-and-Fall.

St John, F. F., A. M. Keane, G. Edwards-Jones, L. Jones, R. W. Yarnell, and J. P. G. Jones. 2011. Identifying indicators of illegal behaviour: carnivore killing in human-managed landscapes. Proc R Soc Lond Ser B **279**:804-812.

Stahl, G. 1981. Genetic differentiation among natural populations of Atlantic salmon (*Salmo salar*) in northern Sweden. Ecological Bulletin (Stockholm) **34**:95-105.

Steyn, L. 2018. #ShockWildlifeTruths: Major SA hotel group continues to promote lion walks. Traveller24. Traveller24, http://www.traveller24.com/Explore/Green/shockwildlifetruths-major-sa-hotel-group-continues-to-promote-lion-walks-20180112.

Storfer, A. 1999. Gene flow and endangered species translocation: a topic revisited. Biological Conservation **87**:173-180.

Sutherland, D. R., A. S. Glen, and P. J. de Tores. 2010. Could controlling mammalian carnivores lead to mesopredator release of carnivorous reptiles? Proc R Soc Lond Ser B **278**:641-648.

Swanepoel, L. H., G. Balme, S. Williams, R. J. Power, A. Snyman, I. Gaigher, C. Senekal, Q. Martins, and M. F. Child. 2016a. A conservation assessment of Panthera pardus. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Swanepoel, L. H., M. J. Somers, W. van Hoven, M. Schiessmeier, C. Owen, A. Snyman, Q. Martins, C. Senekal, G. Camacho, W. Boshoff, and F. Dalerum. 2014. Survival rates and causes of mortality of leopards Panthera pardus in southern Africa. Oryx.

Swanepoel, M., A. J. Leslie, and L. C. Hoffman. 2016b. Farmers' perceptions of the Extra-limital Common Warthog in the Northern Cape and Free State Provinces, South Africa. Wildlife Society Bulletin **40**:112-121.

Swanepoel, P. A., P. R. Botha, C. C. du Preez, H. A. Snyman, and J. Labuschagne. 2015. Managing cultivated pastures for improving soil quality in South Africa: challenges and opportunities. African Journal of Range & Forage Science **32**:91-96.

Sykes, A. R., R. G. McFarlane, and A. S. Familton. 1992. Parasites, immunity and anthelmintic resistance. Pages 179-191 Progress in sheep and goat research,. CAB International, Wallingford, UK.

Symstad, A. J. 2000. A test of the effects of functional group richness and composition on grassland invasibility. Ecology **81**:99-109.

Tainton, N. M. 1972. The relative contribution of overstocking and selective grazing to the degeneration of tall grassveld in Natal. Pages 39-43 *in* Annual Congresses of the Grassland Society of Southern Africa.

Tainton, N. M., A. A.J., and J. E. Danckwerts. 1999. Principles of managing veld. Pages 169-193 *in* N. M. Tainton, editor. Veld Management in South Africa. University of Natal Press, Pietermaritzburg.

Taylor, A. 2016. The role of the wildlife industry in South Africa's green economy. Endangered Wildlife Trust, http://www.ewt.org.za.

Taylor, W. A., P. A. Lindsey, and H. Davies-Mostert. 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. The Endangered Wildlife Trust, Johannesburg.

Teletchea, F. 2015. Domestication and Genetics: What a Comparison Between Land and Aquatic Species Can Bring? Pages 389-401 Evolutionary Biology: Biodiversification from Genotype to Phenotype. Springer.

Teletchea, F. 2017. Wildlife Conservation: Is Domestication a Solution?*in* Global Exposition of Wildlife Management. InTech.

Temeyer, K., J. Pound, J. Miller, A. Chen, J. Pruett, F. Guerrero, R. Davey, P. Untalan, K. Lohmeyer, and A. Li. 2004. Organophosphate resistance in Mexican strains of Boophilus microplus: a major threat to the US cattle industry. SAAS Bull. Biochem. Biotechnol **17**:43-51.

Templeton, A. R., H. Hemmer, G. Mace, U. S. Seal, W. M. Shields, and D. S. Woodruff. 1986. Local adaptation, coadaptation, and population boundaries. Zoo Biology **5**:115-125.

Terborgh, J., K. Feeley, M. Silman, P. Nuñez, and B. Balukjian. 2006. Vegetation dynamics of predator-free land-bridge islands. Journal of Ecology **94**:253-263.

Tews, J., U. Brose, V. Grimm, K. Tielbörger, M. C. Wichmann, M. Schwager, and F. Jeltsch. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography **31**:79-92.

Thomas, S. 2013a. South Africa: Game Price Stampede. inancial Mail. Dec 2013. https://www.africahunting.com/threads/south-africa-game-price-stampede.12491/

Thomas, V. G. 2013b. 14. Creating marine protected area networks in Pacific North America for biodiversity conservation: Linking ecology to legislation. Biodiversity and Climate Change: Linkages at International, National and Local Levels:377.

Thorn, M., M. Green, F. Dalerum, P. W. Bateman, and D. M. Scott. 2012. What drives human-carnivore conflict in the North West Province of South Africa. Biological Conservation **150**:23-32.

Thorn, M., M. Green, D. Scott, and K. Marnewick. 2013. Characteristics and determinants of human-carnivore conflict in South African farmland. Biodiversity & Conservation **22**:1715-1730.

TREES. 2017. A marketing and spending analysis of trophy hunters 2015/2016 season., Tourism Research in Economic Environs and Society, North-West University, Potchefstroom, South Africa.

Trouwborst, A. 2006. Precautionary Rights and Duties of States. Nova et Vetera Iuris Gentium.

Trut, L., I. Oskina, and A. Kharlamova. 2009. Animal evolution during domestication: the domesticated fox as a model. Bioessays **31**:349-360.

Trut, L. N. 1999. Early Canid Domestication: The Farm-Fox Experiment: Foxes bred for tamability in a 40-year experiment exhibit remarkable transformations that suggest an

interplay between behavioral genetics and development. American Scientist **87**:160-169.
Turner, D. A. 2010. Comments concerning Ostrich Struthio camelus populations in Kenya. Scopus **30**:52-54.
Turvey, C., and D. Sparling. 2002. The Great Emu Bubble: A Retrospective Look at a New Industry Failure. Current Agriculture, Food & Resource Issues. Nr 3 / 2 0 0 2 / p . 4 9 - 6 . CAFRI Archives - Canadian Agricultural Economics Society. http://citeseerx.ist.psu.edu/viewdoc/download;jsessionid=3E2F72BB833759150F4491DFA6CD3B82?doi=10.1.1.529.7927&rep=rep1&type=pdf
UK. 2017. Trophy Hunting: UK and international policy. Briefing paper – House of Commons. Number 7908, 23 February 2017. www.parliament.uk/commons-library.
UN. 2012. The future we want, A/RES/66/288. United Nations, http://www.uncsd2012.org/content/documents/727The%20Future%20We%20Want%2019%20June%201230pm.pdf.
UTC. 2015. US Bans Imports of Lion Trophies: the Majestic Cats Protected by USFWS Endangered Species Act. MercoPress. http://en.mercopress.com/2015/12/30/us-bans-imports-of-lion-trophies-the-majestic-cats-protected-by-usfws-endangered-species-act. Accessed 4 April 2016.
Uys, R. G. 2006. Patterns of plant diversity and their management across South African rangelands. University of Cape Town, Cape Town.
Van der Merwe, P., M. Saayman, and W. Krugell. 2004. Factors that determine the price of game. Koedoe **47**:105-113.
Van Der Merwe, S. J. 2016. Official statement by African Lion Working Group (ALWG) on canned hunting and captive breeding. http://conservationaction.co.za/resources/reports/news-official-statement-by-african-lion-working-group-alwg-on-canned-hunting-and-captive-breeding/ Accessed 12 March 2016.
van der Merwe, V., C. Bissett, R. Groom, G. Mills, S. Durant, and K. Marnewick. 2016. A Conservation Assessment of Acinonyx jubatus. The Red List of Mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust,, Pretoria, South Africa.
Van Dyk, G., and R. Slotow. 2003. The effects of fences and lions on the ecology of African wild dogs reintroduced to Pilanesberg National Park, South Africa. African Zoology **38**:79-95.
van Ierland. 2010. Land use management on the border of Kruger National Park, the case of the Nghunghunyani buffer zone project.*in* K. Ferguson and J. Hanks, editors. Fencing Impacts: A review of the environmental, social and economic impacts of game and veterinary fencing in Africa with particular reference to the Great Limpopo and Kavango-Zambezi Transfrontier Conservation Areas. Mammal Research Institute, Pretoria.
Van Rooyen, C. 2017. Piet du Toit veiling lok byna R111milj. Landbou Nuus 14 Februarie, 2017. http://www.landbou.com/nuus/piet-du-toit-veiling-lok-byna-r111milj/.
van Rooyen, L. 2012. Colour variant game naturally profitable. Farmers Weekly. http://www.farmersweekly.co.za/article.aspx?id=24589&h=Colour-variant-game-naturally-profitable. Accessed 12 March 2016.
van Rooyen, N. 2002. Veld management in the savannas. Pages 571-617 *in* J. d. P. Bothma, editor. Game Ranch Management. Van Schaik, Pretoria.
van Wyk, A. M., D. L. Dalton, S. Hoban, M. W. Bruford, I. R. M. Russo, C. Birss, P. Grobler, B. Janse van Vuuren, and A. Kotzé. 2017. Quantitative evaulation of hybridization and the impact on biodiversity conservation. Ecology and Evolution **7**:320-330.
Van Wyk, J. A. 2001. Refugia - overlooked as perhaps the most potent factor concerning the development of anthelmintic resistance. Onderstepoort Journal of Veterinary Research **68**:55-67.
Volker, E. 2012. Ponzi Schemes in South Africa - A practical discussion. Auditing SA **2011/2012**:5-10.

von Brandis, R. G., and B. K. Reilly. 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.

Vuuren, B. J. v., I. Rushworth, and C. Montgelard. 2017. Phylogeography of oribi antelope in South Africa: evolutionary versus anthropogenic panmixia. African Zoology **52**:189–197.

Wallach, A. D., C. N. Johnson, E. G. Ritchie, and A. J. O'Neill. 2010. Predator control promotes invasive dominated ecological states. Ecology Letters **13**:1008-1018.

Walsberg, G. E. 1983. Coat color and solar heat gain in animals. BioScience **33**:88-91.

Waples, R. 1999. Dispelling some myths about hatcheries. Fisheries **24**:12-21.

Warren, C. D., J. M. Peek, G. L. Servheen, and P. Zager. 1996. Habitat use and movements of two ecotypes of translocated caribou in Idaho and British Columbia. Conservation Biology **10**:547-553.

WebFinance. 2017. Business Dictionary. http://www.businessdictionary.com/definition/sustainability.html.

Weilenmann, M., M. Gusset, D. R. Mills, T. Gabanapelo, and M. Schiess-Meier. 2010. Is translocation of stock-raiding leopards into a protected area with resident conspecifics an effective management tool? Wildlife Research **37**:702-707.

Weiss, E. B. 1992. Environment and trade as partners in sustainable development: a commentary. The American Journal of International Law **86**:728-735.

Wightman, A., and R. E. Tingay. 2015. The intensification of Grouse Moore management in Scotland. Commissioned and published by the Legue Against Cruel Sports.

Wilkins, A. S., R. W. Wrangham, and W. T. Fitch. 2014. The "Domestication Syndrome" in Mammals: A Unified Explanation Based on Neural Crest Cell Behavior and Genetics. Genetics **197**:795–808.

Williams, V. L., D. J. Newton, A. J. Loveridge, and D. W. Macdonald. 2015. Bones of Contention: An Assessment of the South African Trade in African Lion *Panthera leo* Bones and Other Body Parts., TRAFFIC, Cambridge, UK &WildCru, Oxford, UK.

Winnercom. 2014. Tuskers. [online video]. Friday, 3 January 2015. http://videos.sorensonmedia.com/Winnercom/TUSKERS/7976ab2499effx4138p 93c5Pee7c3dfb2023. [Accessed 22 November 2016].

Wirminghaus, J. O., C. T. Downs, C. T. Symes, and M. R. Perrin. 1999. Conservation of the Cape parrot in southern Africa. South African Journal of Wildlife Research **29**:118-129.

Wood, M. C. 2013. Nature's trust: Environmental law for a new ecological age. Cambridge University Press.

Woodroffe, R., S. Hedges, and S. M. Durant. 2014. To fence or not to fence. Science **344**:46-48.

Woodroffe, R., and C. Sillero-Zubiri. 2012. Lycaon pictus. The IUCN Red List of Threatened Species 2012: e.T12436A16711116.

World Health Organization. 1965. Technical Report Series.

Writer, S. 2015. Breeder aims to be a big game changer in 'white' industry. Business Day. http://www.bdlive.co.za/national/science/2015/09/04/breederaimstobeabiggamechangerinwhiteindustry?service=print. Accessed 9 April 2016.

WRSA. 2016. WRSA setting the benchmark for responsible and sustainable wildlife ranching. Press release. September 2016. http://www.wrsa.co.za/wrsa-setting-the-benchmark-for-responsible-and-sustainable-wildlife-ranching/.

WSF. 2017. WSF Severs Ties with PHASA over Captive Bred Lion Hunting Policy Reversal WSF Facebook Page. November, 2017. https://www.facebook.com/pg/WildSheepFoundation/posts/?ref=page_internal.

Young, R. 2003. Environmental Enrichment for Captive Animals. Blackwell Science, Oxford.

Zeder, M. A. 2015. Core questions in domestication research. Proceedings of the National Academy of Sciences **112**:3191-3198.

ZPHGA. 2017. Open letter to the President, EXCO and Members of PHASA. ZPHGA, Harare, Zimbabwe.

APPENDIX 1

Table 6: Hierarchical ranking of impacts from highest to lowest risk with the highest risk at an ecosystem's level, followed by the quality of evidence available, and the probability of occurrence within the wildlife sector.

Issue	Impact	Ecosystem level impact	Evidence	Prob of occurrence in industry	Impact on Threatened species
6	Fragmentation of the landscape through impermeable fencing restricts movement of free - ranging species and reduces habitat quality and quantity	1	1	1	3
10	Off-label use of pesticides and unlawful use of hazardous substances cause mortality of indigenous species resulting in changes in ecosystem functioning and increased threats to the conservation of threatened species	1	1	2	2
6	Concentration of species in small areas with impermeable fences for intensive breeding purposes results in habitat degradation within such areas	2	1	1	4
7	The killing of predators and other conflict species may result in a reduction in population numbers which in turn may lead to a change in the conservation status of the species and thereby furthering the extinction risk of the species	2	1	1	2
7	The disruption of social structures of species targeted for removal may exacerbate the conflict potential as a result of the constant removals of individuals. This in turn could result in a decline in the survival rate of the affected population. This constant removal of dispersers may also create a loss or disruption of dispersal opportunities, thereby increasing local extinction risk	2	1	1	2
7	The removal of predators will at a certain scale disrupt predation as a natural process in the broader landscape/environment thereby affecting ecosystem functioning	2	2	1	5
5	Out of range introductions into habitats which are suitable could potentially lead to species becoming established within the landscape (invasive) and thereby impacting upon both the habitat and native species	3	2	2	4
8	The development of microbial, helminth and ectoparasite resistance to stock remedies and veterinary medicines resulting in microbes, helminths and ectoparasites that may start infesting free-roaming game and livestock on a large scale with conservation and economic consequences	3	1	2	2
5	The removal of wild specimens of naturally rare species or species with currently small population sizes, in South Africa or other African countries where sourcing is often cheaper, can lead to population declines resulting in a lower overall conservation status and a higher extinction risk for these species	4	2	2	2
4	Expression of deleterious attributes that may lead to physical, behavioural and lethal outcomes	5	1	1	1
4	Loss of genetic diversity resulting in decreased fitness and reduced adaptive potential.	5	1	1	1
4	The introgression of non-native genes into wild populations from deliberate hybridization/cross breeding may result in outbreeding depression with animals that may be less adapted to current environmental conditions as well as the loss of genetic diversity at a species level	5	2	2	2
4	Physiological stress as a result of maladaptation of animals to their current environment	5	3	3	3
4	Domestication of wild species resulting in a loss of their natural ability to adapt to wild conditions	5	1	2	2
5	Changes in natural genetic composition, evolutionary trajectory and adaptive potential of wild populations through the introgression of captive population genetics wherein genetic changes in the captive population may lead to an altering genetic composition and/or evolutionary trajectory and/or adaptive potential of wild populations through deliberate and accidental introductions	5	2	2	1
8	Disruption of the process of natural selection in terms of host-parasite evolution with resulting loss of disease and parasite resistance within the game population	5	2	2	2
9	Venison from intensively bred game may be contaminated by antimicrobials, ectoparasiticides, anthelmintics and/or anti-inflammatory agents, thereby posing a health risk to humans; intensively produced venison may not be as healthy as wild venison, potentially damaging the brand image and value of extensively produced venison	5	2	2	5

1 = high agreement based on significant evidence or ≥80% probability of occurrence or impact; 2 = high agreement base on limited evidence or 60-79% probability; 3 = low agreement, albeit with significant evidence or 40-59% probability; 4 = low agreement based on limited evidence or 20-39% probability; 5 = <20% probability.

APPENDIX II

Figure 17: Milestones in the campaign against hunting of captive-bred lions as reported on the Facebook Webpage for Blood Lions (Blood lions 2016).

- Over 300 000 You Tube views of the movie trailer to date.
- 37 000+ followers with a weekly reach of over 60 000 people.
- The FB campaign has reached over 11 million people in 12 months
- The Twitter campaign has reached nearly 4 million people in 12 months
- Five international tweet storms with millions involved worldwide
- International media coverage has generated publicity for the campaign worth over R21 600 470
- Worldwide distribution by PBS International, with screenings by Discovery channel, Animal Planet and MSNBC in 185 countries and territories
- Critical parliamentary screenings in Australia, Botswana, European Parliament, Brussels and Finland. Italy and Spain amongst others are still to come.
- Australia, France and Netherlands have banned the importation of lion body parts.
- US Fish and Wildlife Service have placed tighter restrictions on the import of lion "trophies"
- 42 major airlines no longer carry lion hunting trophies
- National Geographic called their Blood Lions feature one of the 12 most powerful stories bringing awareness to conservation, poaching and wildlife trafficking over the last decade
- Screening, key-note address, workshops at ITB Berlin, world's leading trade travel show.
- The film screened with Q&A at World Travel Market in South Africa, The Conservation Lab and Indaba
- The wider trophy hunting community has certainly taken notice of Blood Lions with engagement taking place at various levels in different countries. In particular, we welcome the outcome of the recent PHASA AGM (http://phasa.co.za/…/682-position-paper-on-captive-bred-lio…hunting.html) where the majority of members voted against captive breeding and canned hunting
- African Lion Working Group publishes statement on "Captive-bred Lion Hunting and Associated Activities"
- World Youth Student Travel Conference screenings (September 2015) motivated tourism industry to set up closer scrutiny of lion petting and breeding facilities with result that many volunteer and other interaction programmes have been removed from itineraries and websites.
- Fair Trade Tourism revised strict criteria regarding animal interactive Voluntourism.
- Partnership with Global Nature Fund to support a Blood Lions Campaign in Germany, as well as Campaign Against Canned Lion Hunting international and South Africa.
- Blood Lions, with Wildlife ACT - Volunteer in Africa and members of the tourism industry initiated and launched a 'Born to Live Wild' Pledge which has been endorsed by some of the most influential tourism industry leaders worldwide.
- Tourvest, one of Africans leading travel operators, embraces Born to Live Wild tourism pledge with internal pledge across group of companies.
- Blood Lions and Empowers Africa raised funds in the US to support an additional inspector on the NSPCA Wildlife Unit in South Africa.

Table 7: Summary of position statements of prominent local, African and international hunting organisations on hunting of intensive and selectively breeding of game.

Organisation	Description	Summary of Statement Hunting / Conservation	Reference
Local organisations			
Confederation of Hunters Associations of South Africa (CHASA)	It is a federation of over 25 hunting, hunting-related and shooting affiliates across South Africa.	• CHASA recognises that the Wildlife Industry, whilst underpinned by fair chase hunting, of necessity includes activities to harvest game or reduce damage causing animals or otherwise manage offtake. The differences between these various legitimate, essential activities is a question of semantics amongst hunters. We are bound to stand in defence and further the interests of those who participate across all these various activities. • CHASA aims to preserve and cultivate the traditions and lore of hunting. Notwithstanding the statement in the preceding paragraph, CHASA encourages all hunters to seek personal fulfilment in their quarry taken in a manner as close as is reasonably possible to that described in our Fair Chase Policy whenever the intent of the harvest is for the celebration of a trophy or entering into our SA Record Book. To this end, any animal taken from a source and/or in a manner which is not close to this standard, should be taken for personal or consumptive use and be seen as a harvest or management hunt only. • CHASA is opposed to the deliberate breeding of hybrids and discourages its members, and hunters in general, to seek to hunt, and thus create a demand for such animals. • CHASA condemns the irresponsible practice of “put & take hunting” where animals are hunted so soon after translocation that they are not habituated to their new territory. • CHASA will condemn any breeding practice where proper scientific evidence indicates that it could be harmful to existing wildlife meta-populations and/or biodiversity.	CHASA, 2016
		• Recognises that there are members who do hunt captive-bred lions. • Ratifies the SAPA Norms & Standards for hunting captive-bred lions. • Strongly endorse the stated ambitions of SAPA relating to their self-governance and oversight role. • Urges members who are desirous of hunting captive-bred lions to ensure that their hunt is conducted in accordance with SAPA Norms & Standards, and preferably on a farm accredited by SAPA. • Supports the policy position of PHASA on captive-bred lion shooting.	CHASA, 2017a. CHASA Position Statement on the Hunting of Captive-bred Lions. CHASA, Uitenhage, South Africa.

			CHASA, 2017b. Letter to DSC Executive and Hunters. https://www.facebook.com/search/top/?q=CHASA
Professional Hunters Association of South Africa (PHASA)	The association in SA with the core business of serving the professional hunting industry. It has approximately 1200 members.	PHASA rejects: • Hunting of canned or captive-bred lions. • The hunting of animals in any area other than an "extensive wildlife system" as defined in the Threatened and Protected Species (TOPS) Regulations issued i.t.o. Act 10 of 2004. • Any notion or claim that colour variants are bred to satisfy a significant demand in the trophy hunting market. • Any notion or claim that breeding practice aimed at increasing horn size is necessary because trophy hunting depleted the gene pool. • Any notion or claim that the breeding of animals with abnormally large horn length lengths is driven by a significant demand in the trophy hunting market. • Highly controversial practices such as artificial insemination, cloning, genetic manipulation and any procedure that produces artificial colour variants. • The inclusion of any further colour variants in trophy hunting record books. • Any form of "catalogue marketing" of individual wild animals or groups of wild animals for hunting purposes.	PHASA, 2015; PHASA, 2016a
		• PHASA vehemently rejects all forms of canned or illegal hunting. • Does not condone all forms of captive-bred lion hunting. • At the 40th Annual General Meeting (AGM) in 2017 voted in favour of the following resolution: "PHASA accepts the responsible hunting of ranched lions on SAPA accredited hunting ranches within the relevant legal framework and/or according to recommendations of the applicable hunting association, such as SCI's fair chase standards." • Committed to upholding fair chase and ethical conduct of members partaking in such hunts as contained in the National Environmental Management: Biodiversity Act, 2004 (ACT 10 of 2004): Threatened or Protected Species (Tops) Regulations "Codes of Ethical Conduct and Good Practice"	PHASA, 2017a. PHASA - adopts new constitution and resolution at 2017 AGM. https://www.phasa.co.za/what-is-in-the-news/phasa-press-release.html PHASA, 2017b. PHASA facts vs fiction. https://www.phasa.co.za/what-is-in-the-news/phasa-press-release.html
South African Hunters &	Established in 1949, it is the biggest hunting and conservation association	• Opposes artificial and unnatural manipulation of wildlife to enhance or alter species' genetic and phenotypic characteristics (e.g. coat colour, body size	SA Hunters, 2014

Game Conservation Association (SA Hunters)	in SA and Africa with more than 40 000 members. It represents consumptive hunters, approximately 1000 farmers/landowners, sport shooters and gun owners.	or horn size) in particular through intentional cross-breeding of species, subspecies or evolutionary significant local phenotypes and or the use of domestic livestock breeding methods such as, but not limited to, line breeding, germplasm and semen production or trading, artificial insemination, embryo transfer, castration, growth hormone treatments, controlled or unnatural breeding programs and cloning. • Opposes the intentional breeding of indigenous wild animals in intensive- or highly altered semi-intensive production systems for purely commercial purposes. • Encourages Government to institute adequate control mechanisms for the regulation of commercial breeding and production operations with indigenous wild animals. • Urges all SAHGCA members to abstain from trading in and hunting animals so manipulated as contemplated. • Is committed to further develop and promote the principles, criteria, indicators and incentives for responsible wildlife utilisation, including hunting, as well as extensive wildlife ranching based on sound conservation principles.	
South African Movement for the Promotion of Ethical Outfitters (SAMPEO)	A group of nine experienced professional hunters and outfitters in SA that distance themselves from hunting of lions bred in controlled environments.	• Condemn the immoral practice of canned/captive-bred lion shooting, where lions are bred for the sole purpose of being killed by paying clients and play no meaningful contribution to wildlife conservation, financial or otherwise that aids the species the African Lion (*Panthera leo*) in its natural state. • See no meaningful distinction between the terms “canned” or “captive-bred” lion. • The activities of a few have severely tarnished the reputation of our industry. They have caused major harm to those of us who are committed to acceptable hunting practices that enhance the already significant conservation efforts that have been and are made by hunting in South Africa.	SAMPEO, 2015
African organisations			
Federation of Namibian Tourism Associations in Namibia (FENATA)	This federation represents the different tourism products in Namibia, including amongst others, accommodation facilities, the tour operators, professional hunters, community-based tourism enterprises, tourism products within Communal Conservancies, travel agents, tour guides, protected desert areas and businesses selling commodities to tourists.	• Request MET to ban the import and export of all gene-manipulated wild game species into or out of Namibia, as well as all game trophies bred for colour variation or game animals which are used for artificial breeding of outsized trophies.	FENATA, 2016
Namibia Professional Hunting Association	The association represents	• We condemn the artificial breeding of wild animals for the hunting industry.	NAPHA, 2016

(NAPHA)	professional hunters in Namibia. It has over 400 members.	• We are particularly concerned about the increasing selective line breeding of wild animals to produce colour variants or outsized horn growth. • We consider these practices detrimental to all conservation orientated wildlife management practices.	
		• States there is a distinct and profound difference between the definitions of the concepts of "legal" and "ethical" and that, just because something might be legal (or not yet deemed to be illegal), that it is therefore ethical. • Rejects the definition of the term "ethical" as meaning "all types of hunting permissible by law", as it is seen to fly in the face of the Code of Ethical Sport Hunting Conduct for Africa. • See hunting of captive-bred lions in direct contravention of what is considered fair chase and ethical hunting. Therefore, it cannot be called hunting. • It is the view that canned and captive shooting are rejected by all ethical hunters who believe that there is small difference between the two. • States that PHASA and hunting captive-bred lions place all the hard work undertaken by various institutions in support of sustainable hunting as a tool of conservation, in jeopardy and that supporting it would be detrimental to the entire hunting industry worldwide. • Condemn the decision by PHASA to support captive-bred hunting in the strongest possible terms and distances itself from this decision which has severely tarnished the reputation of the entire African hunting industry. •	NAPHA, 2017 Press release from the Namibia Professional Hunting Association. November, 2017. http://www.napha-namibia.com/fileadmin/user_upload/NAPHA_Press_Release_24.11.2017.pdf
Namibian Ministry for Environment and Tourism	Government department responsible for hunting, conservation and tourism.	• Hunting outfitters and professional hunters who put wildlife that is manipulated and bred intensively in captivity up for sale are putting hunting and conservation at risk. • Captive breeding mostly for financial purposes has its downside, such as behavioural problems in animals that are eventually released as they are unable to hunt or forage, and loss of habitat, amongst others. • Will not be allowed to get out of control in Namibia as they threaten to destroy what the Namibian hunting and conservation community has worked hard to establish over the past 60 years. • No one who cares for the conservation of wildlife and wildlife habitats and all they have to offer should allow this to happen so that a few greedy people can make a short-term profit, which benefits only them at such a high cost to the country. • Anything which damages or abuses hunting will have a negative effect on conservation in Namibia.	NAMPA, 2015
Outfitters and Professional Hunters Associations of Africa (OPHAA)	An international association that represents nationally recognized African hunting associations. Its membership include approximately 11	• Promote legal and ethical fair-chase sustainable hunting in Africa. • We condemn the artificial breeding of wild animals for the hunting industry.	Boretsky, 2015

	hunting organisations from 9 African countries (Botswana, Ethiopia, Mozambique, Namibia, South Africa, Tanzania, Uganda, Zambia and Zimbabwe).		
		• Suspended PHASA from OPHAA after PHASA's policy change to conditionally support captive-bred lion hunting at the end of 2017. • States that captive-bred lion hunting brings the entire hunting industry in every African state where hunting is permitted, in ill repute. • It disregards the fundamental fair-chase principle and jeopardise conservation efforts and livelihoods generated by well-managed and ethical hunting operations.	OPHAA, 2017b Press release: PHASA suspended from OPHAA. November 2017. ww.OPHAA.org
Safari Operators Association of Zimbabwe (SOAZ) Zimbabwe Professional Hunters & Guides Association (ZPHGA) Zimbabwe Tour Operators Association (ZTOA)	All members of the Zimbabwean Wildlife Sector	• Commitment in promoting and encouraging the legal and ethical fair-chase sustainable use of wildlife resources for the benefit of wildlife, communities and the tourism industry. • View captive-bred lion hunting as abhorrent and unethical.	SOAZ, 2017. Statement by the Safari Operators Association of Zimbabwe; the Zimbabwe Professional Hunters & Guides Association and the Zimbabwe Tour Operators Association. November, 2017. SOAZ, Harare, Zimbabwe.
Zimbabwe Professional Hunters & Guides Association (ZPHGA)	Represent professional hunters & guides	• Acknowledges that the wildlife management model in South Africa is vastly different to its neighbours' and is based on wildlife ownership by the landowners and a game ranching model. • Finds that taking a decision to support captive-bred shooting only based on laws and regulations and reported economics of the practice and not considering the will of the (hunting) world, as puzzling. • With the experience from Cecil, experienced the implications of world perception and influence and realised how actions have ramifications stretching from Zimbabwe to Alaska. • The practice of captive-bred hunting can no longer be tolerated by fellow African professional hunting organisations and the world and will never be perceived as fair chase. • The impression that captive-bred hunting will be accepted by fellow professional hunting organisations and the general public is wrong, without question.	ZPHGA, 2017. Open letter to the President, EXCO and Members of PHASA. ZPHGA, Harare, Zimbabwe.

		• Standards and certification of captive breeding for hunting facilities, is simply a way of trying to justify the practice and hoodwinking the naive into believing that the practice can be considered fair chase, sustainable and a conservation tool, even though these regulations do not comply with current SCI recommendations or are not in line with fair chase in the rest of Africa. • Lion Production is not Lion Ranching and cannot support fair chase hunting. • Challenges to overcome changes in global wildlife management policies while maintaining and protecting professional fair chase hunting as the foundation for protecting marginal and isolated ecosystems and wildlife and branding responsible hunting as conservation tool is undermined by captive-bred shooting. • With its decision to support captive-bred lion shooting, PHASA has inadvertently divided and alienated itself from the professional hunting fraternity. • No longer recognise PHASA as a professional hunters association.	
Professional Hunters Association of Zambia (PHAZ)		• No longer recognise PHASA as a professional hunters association and distance themselves from PHASA due to its direct contradiction in firstly 'professional' and 'ethical' hunting and secondly in the public approval of captive-bred lion shooting. • The very perception of captive-bred lion shooting is hugely detrimental to endeavours to promote 'fair chase', 'ethical' and 'professional' hunting as a management tool in conservation. • Any member of PHAZ found to be involved with captive-bred lion shooting in any way, such as, promoting/booking/selling/guiding and observing' will cease to be a member of PHAZ due the direct conflict. • Committed to promoting and encouraging the legal and ethical fair chase sustainable use of wildlife resources for the benefit of wildlife, communities and the tourism industry in all member countries of OPHAA.	PHAZ, 2017. Statement by the Professional Hunters Association of Zambia (PHAZ). November 2017. Lusaka, Zambia.
International organisations			
Boone & Crockett Club	The most influential and prestigious hunting and conservation body in North America, founded originally by President Theodore Roosevelt. It has only 100 full members of which almost all are wealthy, influential, opinion-makers. While the organisation is focussed on North American wildlife and habitat it has joined the CIC in Europe to influence and affect hunting and conservation on a broader basis.	• The Club will speak out when necessary to defend hunting and its value to conservation. This includes pointing out activities that undermine the public support of hunting. • The practices of deer breeding and shooting operations should not be accorded the same level of public acceptance as the ethical hunting of wild, free-ranging game that is the foundation of the North American Model of Wildlife Conservation and forms the tradition of the Club and the majority of hunters. • The benefits that hunting brings to conservation, wildlife management, wildlife health, and land stewardship, and the opportunity for future generations to freely hunt wild species is worth much more than an industry seeking short-term profits.	B&CC, 2015

		• Selective breeding and artificially growing deer and elk with unnaturally large antlers to be sold and then shot in a put-and-take situation is not representative of traditional hunting, and these practices should be discouraged. • The captive-cervid industry is ignoring the fact that society rightfully expects hunting to be conducted ethically. • If hunting is perceived as less than fair (i.e., less than desirable, reputable, and legitimate) our society may no longer tolerate hunting in any form.	
		• Oppose canned "hunting" as a violation of the principles of fair chase and an affront to the time-honoured traditions of hunting. • Just because captive-bred lion shooting is "legitimate" does not make it ethical – it flies in the face of the ethical standards sportsmen have carried with them for more than 100 years. • Canned shoots should be of great concern to all sportsmen and sportswomen, not only as a matter of doing right by the game we hunt, but because those who do not hunt confuse the activity with ethical fair chase hunting—a gross misconception that undermines public support for hunting. • Breeding lions or any wild animal to be shot in a bogus situation is not hunting, not good for the future of hunting, should not be passed off as hunting, and people should not confuse it with hunting. • Applaud the action of organizations and companies that have chosen to say "no more" to African captive-bred lion shooting.	B&CC, 2017. Press release on canned shoots. November, 2017. Boone & Crockett Club Webpage. http://mailchi.mp/boone-crockett/save-the-date-to-attend-a-boone-and-crockett-club-reception-175225?e=6e34656bff
International Council for Game & Wildlife Conservation (CIC)	CIC represents 26 USA State Members, a wide range of organisations engaged in hunting and conservation, as well as individuals such as private members and scientific experts from 86 countries around the world	• Expresses its full commitment to further develop and promote principles, criteria and indicators for sustainable fair chase hunting. • Excludes all trophies of manipulated animals from being scored with the copyrighted CIC Trophy Evaluation Methods. • Urges all CIC members to abstain from hunting manipulated animals. • Opposes artificial and unnatural manipulations of wildlife including the enhancement or alteration of a (e.g. pelage colour, body size, horn or antler size) in particular through: intentional cross-breeding of species, subspecies or evolutionary significant local phenotypes; and the use of domestic livestock breeding methods like flow cytometry or genetic testing, germ plasm and semen production or trading, artificial insemination, embryo transfer, castration, growth hormone treatments, controlled or unnatural breeding programmes, cloning.	CIC, 2011
Dallas Safari Club (DSC)	Established in 1982, situated in Dallas USA with in excess of 6000 members around the world. Has given grants totalling more than $5 million to directly support its mission statement of conservation, education and protecting hunters' rights. Host one of	• DSC has a responsibility to support and encourage ethical hunting practices, even where ethical practices do not align with what is legally permitted. • The practice of captive-bred lion hunting is not a practice that is in keeping with its values of ethical and fair chase hunting. • DSC does not support the practice of captive-bred lion hunting.	DSC, 2018. Dallas Safari Club Position on Captive-bred Lion Hunting. http://dscnewscenter.org/2018/01/dsc-

	the most prominent hunting show in America and the world.		position-on-captive-bred-lion-hunting/
Hohe Jagd & Fischerei" Fair	One of the prominent hunting shows in Europe.	• Dissociates itself from the promotion of shooting farmed game animals and lions bred in captivity. • Strives to promote ecologically sustainable and ethically acceptable hunting practices. • The shooting of lions bred in captivity, and of genetically manipulated African game animals – in enclosed areas – bears no relation to the purposes and principles of hunting, and severely damages the public's conception of hunting and hunters. • This is also the case with regards to the shooting of artificially bred colour variants and mutations of game animals that cannot be found out in the wild.	HJFF, 2016
International Professional Hunters' Association (IPHA)	Established in 1969 with over 400 members coming from some 30 countries around the world.	• Strongly oppose hunting captive-bred, or ranched lions and the Professional Hunters' Association of South Africa's recent decision to condone the practice. • Finds no compelling evidence that the breeding and raising of lions in captivity for the ultimate purpose of being shot within fenced areas of any size promotes conservation of species or habitats. • Finds no conservation value in hunting of captive-bred lions under any circumstances. • Makes no distinction between captive-bred lions and so-called "ranched" lions that are bred in captivity and released onto hunting ranches, whether or not these practices meet the accreditation standards of PHASA and/or the association of predator breeders in South Africa (SAPA). • Views the practice of shooting captive-bred lions as detrimental to the reputation of the entire hunting industry at a key time when the ethics and conservation value of legal and ethical hunting faces increasing public scrutiny and challenges. • Will immediately review/revoke the membership of any person determined to be participating in the practice of hunting captive-bred or ranched lions. • Acknowledge the conservation contribution of "wild-managed" lion populations that are free-ranging, self-sustaining predators on vast fenced reserves in South Africa whose management may include carefully controlled and sustainable quotas for fair-chase hunting.	IPHA, 2017. IPHA'S statement against hunting of captive-bred &/or ranched lions. IPHA Facebook page. https://www.facebook.com/Internationalprohunters/?hc_ref=ARQjhgQ_RYoCgJj4VX9RXiqwWENBpleAx7s89HO7jgSkLLCOwM6UzoEvKNdHtx3cuLc&fref=nf
"Jagd & Hund" Exhibition	Europe's biggest hunting exhibition.	• Strictly forbid the selling or advertising of any type of killing captive breed lions or artificial breed game at the "Jagd & Hund" show. • Exhibitors who would not follow the ban were advised that their booth would be closed - and they would lose the chance to return to the exhibition - forever. • With this clear position the Dortmund exhibition sent a sign to the world that hunters all over the world would not close their eyes in fact of the pervert	J&H, 2016

		breeding industry in South Africa - and will not agree to call such practices "hunting" and those who take part of it aren't "hunters" any longer.	
Nordic Safari Club (NSC)	One of Europe's largest hunting associations and second largest hunting market to SA.	• Will boycott canned lion hunting in South Africa. • Boycott promotions of canned lion hunts at its trade shows. • Are against shooting lions that were fed by humans, as this is not hunting, but killing a half-tame animal for profit. • Scandinavian hunters not interested in South African hunts. All hunts not just lions. • Members may not import lion trophies from South Africa. • Nordic hunters will uphold the hunting ethics that the South African hunting industry and government had dropped.	Tempelhof, 2014
		• It is important to protect the image of South African hunters against breaching of ethical principles particularly relating to canned lion shooting, breeding of artificial colour variants and genetic mutations. • Removed all South African lion trophies from their record books. • Banned all advertisements from operators offering canned lions in their magazine or any editorial material relating to the practice. • Asked members to refrain from buying hunts or doing any business with outfitters offering canned lion shooting. • Warned that the associated bad publicity cannot be afforded in a time where many proposals restricting trophy import in the EU are in the pipeline.	NSC, 2017. Reversal of policy on canned lion shooting. Letter to PHASA. November, 2017. NSC Facebook Page. https://www.facebook.com/search/str/Nordisk+Safari+Klub+canned+lion/keywords_pages?see_more_ref=eyJzaWQiOiIiLCJyZWYiOiJzZWVfbW9yZSJ9
Rowland Ward	Rowland Ward has been a world-renowned brand in the sporting and outdoor market since 1870. It houses the "Records of Big Game series", one of the two world famous recognised international trophy record books.	• Hunting within game-proof fences is acceptable if it promotes the general well-being and conservation of habitat and the species enclosed. Enclosures, however, must promote self-sustaining, breeding populations that can feed themselves from naturally occurring vegetation and prey without continual supplemental feeding by humans. • They shall provide enough acreage and vegetation that animals can easily hide from humans and predators alike, and they must offer a hunting scenario whereby the outcome of obtaining a certain animal is by no means guaranteed. • Animals that are released solely for hunting purposes shortly thereafter will not be accepted for entry into the record book. • Any animal shot in an enclosure that lacks adequate food and acreage is not eligible for entry into the record book. • Colour variations of species in certain animal populations and particular regions have been naturally occurring probably since the dawn of time. Rowland Ward Ltd., in fact, has several categories that are, by and large, based on naturally occurring coloration-only differences, such as the Angola impala. However, Rowland Ward Ltd. will not accept animals that are	Rowland Ward, 2017

		specifically bred with the goal being to establish a separate colour-based category for trophy hunting. Rowland Ward Ltd. will not create categories for such animals. • No hybrid animals will be accepted unless such animals have a natural hybridization zone in a completely free range, such as the Armenian mouflon and the Transcaspian urial.	
Safari Club International (SCI)	With 55 000 members, it is the most influential and wealthiest hunting organisation in North America with a focus on Africa. It is the home for Americans who hunt internationally. It has a lobbying force in Washington, D.C. and senior politicians like the Bush family and senior military people like Schwartzkopf have spoken at its annual convention. It generates in excess of $1 million for conservation projects predominantly in Africa.	The SCI Record Book Committee will review and add new big game animal species and sub-species as entries to the Record Book and World Hunting Award programme given the following requirements: • All new SCI Record Book entries will use the best available science regarding the taxonomic status of an animal; • The SCI Record Book entries will add new species to the record book based on scientific evidence that the entry represents a valid taxonomic species or grouping of related sub-species and not simply a hybrid, a colour variant, or genetic mutation of an existing species; • The SCI Record Book committee does not support procedures or practices with wildlife that produce non-typical colour variants, horns, antlers, or body size; • The SCI Record Book committee discourages breeding practices that genetically manipulate wildlife species to alter appearance or size, including assisted reproductive technologies that include genetic manipulation and wildlife cloning. • The SCI has stated that colour variant springbok records would remain in the record book based on the 'grandfathering' principle.	Boretsky, 2015
Spiral Horn Antelope Club	It is a ten-year old specialist hunting club for those interested in the 30 species and subspecies of spiral horn (tragelaphine) antelopes. It has approximately 400 members.	• The intensive breeding and domestication of wildlife to produce animals with exaggerated horn lengths and unnatural colour variations is, along with canned hunt killing, causing overseas hunters to avoid South Africa. This, in turn, is having a seriously adverse effect on hunting and, consequently, on conservation in this country.	DEA, 2016b
Wild Sheep Foundation (WSF)	A North American hunting organisation focussed on enhancing wild sheep populations, promoting professional wildlife management, educating the public and youth on sustainable use and the conservation benefits of hunting while promoting the interests of the hunter and all stakeholders.	• On captive-bred lion hunting: *"Just because they have the right – does not make it right"*	WSF, 2016
		• Recognises the South African wildlife conservation model, based on privatisation of wildlife and regulated hunting, has played an integral and critical role in the country's incredible conservation success.	WSF, 2017. WSF Severs Ties with PHASA over Captive-bred Lion Hunting

		• Condemns the practice of breeding and captive-rearing of predators which cannot sustain themselves naturally and then releasing them for the sole purpose of shooting them under restrictive conditions. Recognizes that while legal by South African law, many within South Africa's hunting and conservation community condemn such practices. • Opposes captive-bred hunting as the practice has not been scientifically proven to enhance free-ranging populations or otherwise provide conservation benefits to wild lions and is contrary to the principles of fair chase hunting. • Severed affiliation and sponsorship support with PHASA's after their reversal of its 2015 policy condemning the practice of hunting captive-bred lions under controlled conditions. • WSF will continue to support outfitters and professional hunters in South Africa who are committed to conservation through ethical hunting and the sustainable use of the country's incredible wildlife resource.	Policy Reversal WSF Facebook Page. November, 2017. https://www.facebook.com/pg/WildSheepFoundation/posts/?ref=page_internal

Table 8: Summary of position statements of organisations representing game farming/breeding/management and prominent conservation organisations that have been known to participate in the debate on intensive and selective breeding of game for pure commercial purposes.

Organisation	Type of Organisation	Summary of Statement Hunting / Conservation	Reference
Local			
Association of Zoos and Aquariums (AZA)	The association is dedicated to the advancement of zoos and aquariums in the areas of conservation, education, science, and recreation. AZA represents more than 230 institutions in the United States and overseas, which collectively draw more than 183 million visitors every year.	• Intentional breeding to achieve rare colour-morphs may seriously compromise the welfare of individual animals and such breeding practices are also problematic from a population management and conservation perspective.	AZA, 2011
Endangered Wildlife Trust (EWT)	A non-governmental, not-for-profit conservation organisation, dedicated to conserving threatened species and ecosystems in southern Africa to the benefit of all people. Member of the (IUCN).	• Selective and intensive breeding of colour variant animals does not directly contribute to biodiversity conservation, and does not allow for natural evolutionary processes to take place.	DEA, 2016b
Game Rangers Association of Africa (GRAA)	An association that represents more than 1500 game rangers across Africa, from approximately 20 countries. It is a member of the International Ranger Federation (IRF)	• Is against the manipulation of wild animals using hormones, artificial feeding and other selective breeding techniques to obtain animals with superior physical proportions and un-natural colour variations.	GRAA, 2016
National Association of Conservancies/Steward-ship of SA (NACSSA)	An association of environmentally conscious land-owners and land-users that choose to cooperatively manage their natural resources in an environmentally sustainable manner without necessarily changing the land-use of their properties. NACSSA represents ±750 conservancies in South Africa that manage about 3 million hectares of land.	• Opposes the selection of aberrant forms of wildlife for breeding purposes (e.g. colour variants). • Urges government to regulate against the breeding and distribution of genetically manipulated game (e.g. colour variants).	NACSSA, 2015

NSPCA	The SPCAs are governed by the SPCA Act 169 of 1993 which is administered by the NSPCA, constituting us as a statutory body. Over 90% of all animal welfare investigations and prosecutions in SA is led by the NSPCA.	• In the interest of human safety, animal welfare and biodiversity, we appeal to our government to ban the intensive and selective breeding of wild animals in South Africa. • This type of breeding or management of wildlife has absolutely no benefit to the individual animal, the species, biodiversity or conservation as a whole. • Due to the high financial value of these colour morph antelope farmers take extreme measures to protect them from their natural predators, including lethal control methods.	NSPCA, 2015
South African Predator Association (SAPA)	SAPA represent lion breeders in SA and coordinate and promote the interests of its members with the view of establishing and maintaining a healthy and profitable predator breeding and hunting sector	• Support the breeding and hunting of captive-bred lions according to specific norms and standards; • Differentiates between "canned hunting" that is not supported and the hunting of captive-bred lions (supported) • Provides standards for: – conditions deliberately aimed at preventing human imprinting for lions to be hunted; – size of the hunting area; – release period prior to the hunt; – hunting methods; and – misrepresentation of facts to hunting clients (hunters).	SAPA, 2017a-b
Wildlife Ranching South Africa (WRSA)	A national association representing land-owners with an interest in game, game ranchers and breeders, professional hunters, hunting outfitters, taxidermists, game reserves and mixed farmers.	WRSA supports the breeding of colour variants. • Prohibits its members from undesirable breeding practices such as: – cross-breeding; – breeding animals with genetically detrimental conditions, such as albinism and dwarfism; – genetically manipulating species; – using artificial reproductive technologies such as artificial insemination; – embryo transfers and cloning – except where these can assist in the preservation of threatened species and with the explicit approval from the Department of Environmental Affairs.	WRSA, 2016
International			
African Lion Working Group (AWG)	Expert group for the promotion of comprehensive, scientifically based conservation strategies for all free roaming lion populations in Africa.	• Captive-bred lion hunting does not provide any demonstrated positive benefit to wild lion conservation efforts and therefore cannot claim to be conservation.	Van der Merwe, 2016
International Union for Conservation of Nature (IUCN)	Biggest conservation organisation in the world with 1 300 members from 170 countries and support of >11 000 scientists.	• Acknowledge that sustainable, legal and ethical hunting generates income and supports human livelihoods in areas where other farming practices are less viable. • Request:	IUCN, 2016b; IUCN, 2016c

		– termination of the practice of breeding lions in captivity for the purpose of 'canned shooting' through a structured, time-bound process; – restriction of captive breeding of lions to registered zoos or registered facilities whose documented mandate is as a recognised, registered conservation project; – development of norms and standards for the management of captive-bred lions in South Africa that address welfare, biodiversity and utilisation aspects; and – prohibition of the hunting of captive-bred lions under any conditions. • Concerned that large-scale intensive and selective breeding may have direct and indirect detrimental consequences for biodiversity that will reduce the ability of eco-tourism and hunting to contribute sustainably to the economy and human well-being. • Recommend: – adoption of a risk-averse strategy in permitting establishment or expansion of this practice; – prohibition of intentional hybridization of large wild mammals across species, subspecies or other recognised evolutionary boundaries; – prohibit release of selectively bred animals into the wild until the risks are understood and can be managed; – development and implementation of norms and standards for husbandry practices of intensively bred species; – establish monitoring systems to document the extent and impact of these activities, and support research to provide more information to anticipate and manage risks; and – develop and implement certification systems for wildlife operations to ensure transparency so that end-users know the origin of the animals they are using and/or buying.	
IUCN Antelope Specialist Group (IUCN ASG)	A specialist group of 73 volunteer members, representing 27 countries. Members include field biologists, academics, wildlife managers, captive breeders, government officials, NGO staff, and others from diverse and inter-related fields.	• Direct threat to biodiversity by risking the survival of indigenous taxa. • Distortion of natural processes of evolution. • Weakened resilience or reduced adaptive capacity to environmental changes. • Reduced reproductive fitness. • Opposes all IGM of antelopes for commercial or amenity purposes, with particular reference to: (i) hybridization of different species (ii) crossing of different subspecies (iii) selective inbreeding of a population	IUCN SSG ASG, 2015

South African National Biodiversity Institute (SANBI)	SANBI receives its mandate from NEMBA and advises the Minister of DEA on matters of Biodiversity Policy.	• The breeding of genetically inferior recessive colour morphs does not further the conservation of South Africa's wild biodiversity and therefore cannot be supported. • The Scientific Authority currently views this as a low risk threat to the species that are likely to be affected and therefore does not recommend that it be legislated against, but the situation needs to be monitored. • Should be discouraged or dis-incentivised.	SANBI, 2010

APPENDIX III

SUMMARY OF CASE STUDIES THAT HIGHLIGHT SOME OF THE SOCIAL, ENVIRONMENTAL AND ECONOMIC RISKS ASSOCIATED WITH PRACTICES WHERE WILDLIFE WAS BRED INTENSIVELY FOR PURE COMMERCIAL PURPOSES WITHOUT PROPER CONSIDERATION OF ENVIRONMENTAL AND SUSTAINABILITY PRINCIPLES.

Lion breeding as a case study

The shooting of lions from intensively-bred camps first attracted international attention after the 1997 broadcast of the so called "Cook report," a British television report which showed shocking footage of lions being shot near the Kruger National Park (SAPA 2017b). The captive breeding and subsequent shooting of lions in South Africa increased dramatically since then. In 2015, the lion population in South Africa was about 9 100, of which approximately 68% were in intensive breeding facilities and 32% free-ranging in protected areas (Williams et al. 2015). It is estimated that there are currently between 5 915 and 8 000 lions in 294 facilities (TREES 2017).

The growth and economic contribution of the intensive lion breeding sector and the economic contribution of shooting animals from these facilities changed dramatically in recent years. Statistics on trophy hunting from the Department of Environmental Affairs indicate that the annual number of international hunters visiting South Africa has seen a dramatic decrease of 28% from 9 138 in 2011 to 6 539 in 2016 (DEA 2011, 2016). At an average spending of approximately R262 000 by international hunters per trip (TREES 2017), the country has lost almost R288 million in direct income from trophy hunting between 2014 and 2016.

Table 9: Trends from the top ten income generators of trophy hunted game species (DEA 2016).

TOP 10 INCOME GENERATORS (Mil. Rand)				
	2014	**2015**	**2016**	**% Change 2014-2016**
Lion	195	181	111	-43
Buffalo	127	145	220	73
Kudu	78	104	110	40
White rhino	72	76	83	14
Sable antelope	57	73	117	106
Gemsbok - Oryx	39	51	49	27
Nyala	45	46	76	71
Burchell zebra	39	45	51	29
Waterbuck	36	40	51	39
Blue wildebeest	36	39	50	39

Although the income generated from the top ten trophy species increased since 2014, some with as much as (106%), the income from lion hunting has decreased with 43% between 2014 and 2016 and the number of lions hunted with 61% ((DEA 2016), Table 8). The majority of lions hunted in South Africa (>98%) are from captive-bred lion breeding facilities. Research by North-West University indicates that 56% of hunters are not informed that they are hunting

captive-bred lions (TREES 2017). Lion hunting contributed approximately 18% of the income generated from trophy hunting in 2014, but this dropped to 8% in 2016, that amounted to approximately R110 million (DEA 2016). This is a drop of approximately R84 million in income generated since 2014 (DEA 2014, 2016).

The downward trend started prior to the trade ban on the import of lion trophies from captive-bred lion populations by the United States in late 2016. As discussed in the body of this report and confirmed by media reports, opinions from prominent role players and the captive lion breeding sector, these declines are linked to reputational damage associated with shooting of captive-bred lions for trophies. As the USA represents approximately 60% of the trophy market of South Africa (DEA 2014), the USA trade ban resulted in significant reduction in cash flows of the intensive lion breeding facilities and affected trophy hunters. It has been described as "devastating" to the sector (SAPA 2017a). A preliminary assessment by SAPA after a period of nine months, indicated that there was a loss of at least 320 lion hunts, which equates to a direct loss of income of approximately R78 million (Van de Vyver, pers. comm. 2016)[14]. With a drop in international demand, lion breeders started offering cheap lion hunting packages to locals (Lombaard 2016). Although hunting only contributes 24% of the income of breeding facilities, it enables multiple use strategies. Secondary income sources from products such as skins and bones contributes 9% of the income of breeding facilities (TREES 2017). Under the current proposed CITES export quota for 800 lion bone carcasses, this cannot replace income generated from the shooting of intensively bred lions as this quota is what have been traded previously as a secondary income stream from hunting. At the current (2017) price of R25 000 per carcass, it may not be economically viable to breed lions exclusively for the bones (Williams et al. 2015). The future implications of a regulated lion skeleton export quota on the viability of breeding facilities is unknown as it may affect the current price, supply and demand, which may in turn have an impact on the feasibility of poaching wild lions for their bones. This highlights the importance of considering risks for other cumulative impacts resulting from reputational damage.

At an average cost of R21 000 to feed one lion for a year, it will cost approximately R168 million to feed an estimated 8 000 lions in the country per annum, with average operational costs of approximately R50 000 per facility (TREES 2017). With a life expectancy of between 15 to 20 years for a lion, the financial burden to breeders is very high. With the viability of breeding facilities under pressure, concerns are being raised as to the emerging animal welfare risk. Increased incidents of neglect and euthanasia have already been reported (Confidential sources from provincial conservation agencies; (Africa Geographic 2016)).

Other indirect and cumulative impacts include increased administrative cost to government and the wildlife sector in dealing with the impacts of new trade restrictions as well as the indirect societal costs associated with redirecting limited public resources away from growing other sub-sectors of the wildlife economy, conservation and brand building of South Africa as responsible wildlife-based tourism destination. Other cumulative impacts include lion breeding facilities closing down with approximately 660 people that have already lost their jobs (Van de Vyver, pers. comm. 2016).

It is clear from the lion case study that reputational damage that can result from negative stakeholder perceptions about perceived irresponsible hunting practices and the shooting of intensively-bred lions, poses economic risks to both the hunting and lion breeding sectors, with concomitant social, economic and conservation risks for the broader wildlife industry. This case study highlights the importance of following an integrated approach in considering risks associated with intensive and selective breeding of game. It is evident that reputational risk and its impact on demand was not anticipated by lion breeders, even though it is one of the

[14] Van de Vyver, C. (2016). E-mail to SA Hunters on: Statistics on lion breeding facilities.12/5/2016.

biggest risk for any business (Humphries 2003, Ernst and Young 2016). This is in line with the theory on reputational management and business sustainability principles that negative perceptions about one sub-sector of the wildlife industry can affect the sustainability of another sub-sector in the same value chain.

Intensive breeding of emu

In evaluating the emu breeding industries collapse in the mid 1990's in Canada, Turvey and Sparling (2002) found that it is critical for industry participants to understand end-market requirements and the production system implications before developing a new industry. More than three hundred Canadian farmers switched from conventional farming to emu breeding, believing that the beneficial health properties of emu meat and oil make for a good investment opportunity. Rapid expansion to supply in the demand for new breeding stock fuelled high prices. In the absence of a viable consumer market that bought into the value propositions offered by emu breeders, the oversupply resulted in plummeting prices. The end result was a collapse in the emu breeding industry (Turvey and Sparling 2002).

The emu breeding case study highlights the risks associated with driving supply in breeding stock without proper consideration of a consumer market, similar to what is happening in the intensive and selective breeding of colour variants in South Africa.

Intensive and selective breeding of colour variants

In 2013, it was reported in the financial media that intensive and selective game breeding produced annual return on capital employed as high as 130% for black impala. It was said to outperform the JSE All Share Index (ALSI) of 26.68% and investment in property (typically 10%) (Slabbert 2013).

It has been reported that 43% of game ranchers are involved in some form or another of intensive and selective breeding (Taylor et al. 2015). The proliferation of game breeding facilities were said to be fuelled by exceptionally high prices for breeding stock and the promise of trophy hunters that will flock to South Africa to hunt the grand slam of colour variants (van Rooyen 2012, Writer 2015). Other reasons that were mentioned that fuelled prices included initiatives for possible listing of rare game species on the Johannesburg Stock Exchange; export of species and re-introduction in other African countries; the attraction of corporate investors who got higher rates of return than in other assets; and WRSA's model for land reform putting intensive and selective breeding forward as a highly profitable option which could be pursued on small pieces of land (Cloete 2015).

At the end of 2014, SA Hunters, by far the biggest hunting and conservation association in South Africa, openly distanced itself from hunting of game that have been intensively- or selectively-bred for the purposes of hunting, stating that it is not only against the spirit of responsible hunting, but it also has negative conservation implications and hold significant risks for the wildlife sector as a whole (SAHGCA 2014). Several local and international hunting and conservation organisations, including the IUCN, followed suit raising concern about the reputational risk associated with shooting game from intensively- and selectively-bred operations, as well as potential conservation and economic concerns. See Table 7 and 8 in Appendix 2 for the position statements of hunting and conservation organisations about intensive and selective breeding.

It became evident that there was not a significant demand for trophies of colour variants as predicted. As far as local demand by hunters for colour variants is concerned, a recent study by North West University revealed that the biggest majority (81%) of local meat hunters in the survey have not hunted colour variants (TREES 2017). Those that have shot colour variants, indicated it was for meat purposes and because animals were available at a good price, not for trophies (TREES 2017). According to a number of websites, trophy hunting packages for colour variants have been sold, but no reliable data could be sourced indicating the extent of

demand for trophies of colour variants. However, PHASA (2016) reported that they did not see any substantive demand for trophies of colour variants, although a small number of trophies for naturally occurring colour variants have previously been recorded in the Roland Ward Trophy Book (Roland Ward 2017).

Since the second half on 2016, economists and reporters started referring to an oversupply of colour variants and significant decreases in prices in South Africa (Botha 2016, Cloete 2016, Schoeman 2016, Van Rooyen 2017). Cloete, a prominent agricultural economist that regularly comment on economic trends for the wildlife ranching sector, stated that improved economic and climatic conditions are not likely to turnaround the decrease in prices as it is likely to be offset by a further growth in supply (Cloete 2016).

By 2017, Cloete indicated that it was the first time in the history of this young industry that the breeding segment of the wildlife industry is facing uncharted territory in that "price pressure is being instigated from both the demand and supply side" (Cloete 2017). A similar trend was predicted for higher-value species, especially female animals. Lower profitability, larger breeding stocks and an uncertain market/investment sentiment were given as reasons that contributed towards both demand and supply pressure (Cloete 2017). By 2018, prices for black impala ewes dropped from R610 000 in 2014 to R7 500 in 2018 (African Wildlife Auctions 2018) and some game breeders have opened the gates of their intensive breeding camps, offering hunting packages where black impala and those with typical colouring are offered at the same price.

The tulip and ostrich bubbles

At the end of the 16th century, prices in rare colour variants of tulips escalated to levels far above the intrinsic value of the flowers (Beattie 2017). Word spread that unimaginable profits could be made and sellers travelled round the country selling a variety of tulip investments to speculators that were trading their land, life savings and farms to get more tulip bulbs (Myers 1999). At auctions the prices of bulbs were bid up to exuberant heights as speculators purchased bulbs at higher and higher prices, intending to re-sell them for a profit (Myers 1999). All items and commodities have an intrinsic worth, but all that was supporting the tulip market price was a belief by speculators that new colour varieties of bulbs would continue to be worth more than it was the previous day, following what is known as the greater-fool theory of investing (Economist 2012). When tulip traders could no longer find new investors willing to pay increasingly inflated prices for their bulbs, prices plummeted (Mackay 1841, Garber 1990, Myers 1999). The crash in the tulip market had huge socio-economic implications (Mackay 1841, Garber 1990, Myers 1999).

Similarly, when the ostrich bubble in the United States burst in the 1980's, people lost their lifetime earnings and it lead to animal welfare situations, with some breeders setting their birds free. A few breeders kept their birds and continued to work hard to develop processing and other markets. However, the latter was not easy as the prices of the leather were exuberant and few members of the public have tried ostrich meat (Shein 1997).

These two case studies highlight the risks associated with escalating prices of products by creating hype in the market to levels far above the intrinsic value of the asset, and has been compared with trends in the intensive and selective game breeding industry in South Africa by some, including investment analyst Chris Niehaus[15] (Niehaus 2014). Trading prices that are not related to the intrinsic or asset value of a product should raise concern (Graham 2003), especially if much higher than four times the asset value. In the last couple of years the prices for colour variants and those bred for exceptional horn lengths have skyrocketed with growth

[15] Chris Niehaus - former Deputy Chairman of the Johannesburg Stock Exchange and ex-CEO of HSBC Investment Services (Africa) and HSBC Securities (Asia).

in auction prices as high as 446% and 449% per year for some colour variants (Bezuidenhout 2012, Thomas 2013a).

The lessons learned from these case study

- That game breeders that purposely breed game intensively and selectively have to recognise the importance of understanding market drivers and end-market or consumer market requirements, including consumer perceptions/preference as well as broader environmental and market trends and how that can change demand (there is a worldwide trend toward greater responsibility and sustainability);
- Production system implications, e.g. how populations are managed for the hunting market, are critical as consumers and the public are increasingly concerned with how products are developed/produced in the light of threats to the environment such as climate change, species extinction, land degradation, over and irresponsible use of natural resources (Dodds et al. 2010, UN 2012);
- Speculation linked to an extraordinary 'investment interest' in a market can lead to 'n escalation of prices as new investors buy into this market. This can lead to inflated prices and oversupply, especially in cases of poorly developed consumer markets, as is the case for colour variants. The common outcome of such is massive profit for the originators and early investors and significant financial loss for the remainder (Krige 2012, Volker 2012); and
- Strategic environmental assessments in line with NEMA, that consider the full lifecycle of a new development/sector within the wildlife economy ensure identification of current and potential future social, environmental and economic risks that can assist in pro-actively developing the necessary mitigation measures to either avoid or ameliorate impacts.

APPENDIX I

POTENTIAL RESEARCH PRIORITIES HIGHLIGHTED THROUGH THE PROCESS