Legal hunting for conservation of highly threatened species: The case of African rhinos

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Abstract

Legal hunting of highly threatened species – and especially the recreational practice of 'trophy hunting' – is controversial with selected ethical objections being increasingly voiced. Less attention has been paid to how hunting (even of threatened species) can be useful as a conservation tool, and likely outcomes if this was stopped. As case studies, we examine the regulated legal hunting in South Africa and Namibia of two African rhino species. Counter-intuitively, removing a small number of specific males can enhance population demography and genetic diversity, encourage range expansion, and generate meaningful socio-economic benefits to help fund effective conservation (facilitated by appropriate local institutional arrangements). Legal hunting of these species has been sustainable, as very small proportions of the populations of both species are hunted each year, and numbers of both today are higher in these countries than when controlled recreational hunting began. Terminating this management option and funding source could have negative consequences at a time when rhinos are being increasingly viewed as liabilities and COVID-19 has significantly impacted revenue generation for wildlife areas. Provided that there is appropriate governance and management, conservation of certain highly threatened species can be supported by cautiously selective and limited legal hunting.

Introduction

Cecil the lion's death invigorated challenges to both the social legitimacy and conservation contribution of legal recreational hunting of rare and threatened species ('t Sas-Rolfes, 2017). This prompted recent efforts to prevent hunting trophy imports to Western countries such as the United Kingdom and USA (Dickman *et al*., 2019). Aside from selected ethical arguments voiced against trophy hunting (Ghasemi, 2021) many argue that banning trophy hunting would enhance conservation of currently legally hunted species. However, for at least some species the inverse may be true (IUCN, 2016).

Moral critics of recreational hunting, and especially the taking of trophies, tend to raise objections by appealing to deontological or virtue ethical arguments, which are mostly concerned with the nature of, or motivations for, specified actions (Nelson *et al*., 2016). However, if biodiversity conservation is considered an ethical and policy imperative, then pragmatic consequentialist arguments, which are concerned with the outcomes of actions, are also relevant (Johnson *et al*., 2019). The European Union Court of Justice recently affirmed that, consistent with the precautionary principle, environmental policy and laws pertaining to hunting should be informed by good science (Epstein *et al*., 2019). This in turn, implies evaluating the consequences of policy measures in social-ecological context (Di Minin *et al.*, 2021).

We examine regulated hunting of Africa's two rhino species as a dual case study. After outlining its history, the ecological and socio-economic arguments for it, and evident impacts on rhino conservation, we conclude with some remarks on policy implications for conservation in general.

History of rhino hunting

Historically, Africa's free-ranging populations of white rhinos (*Ceratotherium simum*) and black rhinos (*Diceros bicornis*) were substantially reduced through expansive agricultural development and uncontrolled hunting for sport, meat, and rhino horn. Black rhinos were reduced to low numbers in South Africa and Namibia and by 1885 southern white rhinos (*C.s.simum*) had been reduced to a single population of only ~20-50 in what is today Hluhluwe-iMfolozi Park (HiP), South Africa. Numbers of both species have since recovered substantially in both countries, with legal hunting playing an enabling role.

White rhino

The southern white rhino population in HiP grew steadily under protection from 1885 and by 1961 numbers had reached such high levels that concerns about potential 'overgrazing' and accelerated bush encroachment led to the start of a bold white rhino translocation and reintroduction programme (Player, 2013). Hundreds of rhinos were moved to numerous public and private reserves within South Africa (including Kruger National Park), and to seven former African range states, and zoos and safari parks around the world. Early founder groups were often markedly female-biased, creating an excess male problem at source. The Natal Parks Board therefore sought to increase the number of areas willing to take more males by allowing private purchase of excess rhinos for a nominal fee and easing protective legislation to allow legal hunting in 1969. Recreational hunting of some of those animals under permit started when the total wild population was ~1,800.

Initially a regular low-cost supply of excess rhinos from State areas incentivized excessive hunting on some private reserves (Buys, 1987). This changed when market-driven live rhino auctions were introduced in 1989, and new South African legislation in 1991 further secured private ownership of rhinos. Increased live sale prices resulted, encouraging a focus on breeding, and white rhino numbers on private land in the country continued to grow (Emslie *et al*. 2019). Sales of excess animals provided an additional source of funding for state conservation, and promoted growth of the private conservation sector, with some of it driven by hunting tourism. Namibia also implemented legislative changes to enable private landowners to benefit from wildlife on their land, including white rhinos, while maintaining permitting controls over hunting. Zimbabwe has allowed a very small number (~6) of white rhino hunts on private land. The success of this model prompted the easing of international controls for South African white rhino hunting trophy exports in 1994.

From 2006, 'pseudo-hunting' of white rhino in South Africa to obtain horn for illegal sale into Asian markets temporarily became a problem (Milliken & Shaw, 2012). The implementation of control measures by South Africa in 2012 brought this abuse under control (Emslie *et al*., 2019). To date the number of white rhino hunted has not been subject to any quotas, but high prices have generally ensured that only a limited number are hunted each

year. The most recent South African white rhino biodiversity plan includes sustainable hunting as a key strategic component toward meeting its conservation target.

Black Rhino

Continental black rhino numbers declined sharply to around 2,360 in 1994/95. Subsequent protection and active use of translocations to enhance recovery saw numbers more than double across the continent, to an estimated 5,366-5,627 by 2017 (Emslie *et al*., 2019). South Africa and Namibia have both grown their black rhino numbers from a small base – from ~110 rhinos in South Africa in 1933, and ~300 in Namibia in 1970. Range and numbers have increased since, with both countries now conserving similar numbers, jointly comprising ~3,975 (70.6%) of Africa's black rhinos in 2018 – up from only about 2.9% (~741) in 1973.

An excess male problem had also long been identified for black rhinos and confirmed by detailed population monitoring by the SADC Rhino Management Group (RMG) since 1989 (Adcock 2001). In response, in 2004, both South Africa and Namibia successfully applied for CITES quotas to export up to five black rhino trophies each per annum. Building on scientific recommendations (Leader-Williams *et al*., 2005) the SADC RMG, in consultation with stakeholders, developed a black rhino hunting permit application approval and scoring system for use in South Africa. This was adopted and became part of the country's current black rhino biodiversity management plan. Its criteria were designed to ensure that only applications to hunt specific black rhino that further demographic and/or genetic conservation of breeding populations are approved. From 2019 South Africa's black rhino export quota changed, to 0.5% of the total population (automatically adjusting the maximum quota up or down in response to changes in rhino numbers). In Namibia, the Ministry of the Environment and Tourism makes all the decisions relating to how many and which black rhino are to be hunted, pursuant of its conservation goals.

Biological conservation aspects

Regular translocations from established populations are undertaken to maintain their productivity, provide founders to expand range and numbers, and to adjust sex ratios.

Rhinos have on average a slightly (statistically significant) skewed average sex ratio at birth (53% males for black rhinos – Adcock, 2001). Chance variation around this mean results in

some populations having an even greater male bias. Rhino males are territorial and increased fighting occurs in black rhino populations when the number of potential males eligible to hold territories exceeds the dominant male carrying capacity determined by available food resources. Fighting, primarily by males, accounted for about 41% of known-cause natural black rhino deaths (N=640 – Adcock, 2019) with breeding females and calves comprising 70% of these losses (Adcock, 2019). Setting up new rhino populations is expensive and where possible a slightly skewed female biased founder group is desirable, but this can negatively impact on the sex ratio of the donor populations if not corrected for by removing additional males.

This excess male problem cannot be solved by simply moving excess males to other populations:

- Reserves with female-biased populations grow numbers faster and do not want more males.
- Introducing excess males into existing populations carries large risks for the males themselves but also the resident breeding stock (Brett 1998). Linklater *et al.* (2011) noted that restocking an area that already had black rhino had a higher mortality rate (13.4%) compared with an initial introduction (7.9%). Furthermore, although adult males accounted for a lower proportion of introduced animals, they accounted for a disproportionately higher percentage (21.9%) of the introduction-related deaths.
- There are limited opportunities to place excess males into male-only populations. In 2014 there were 11 such populations established in South Africa and one in Namibia; but these were generally in small areas not suitable for breeding herds. Inter-male fighting losses in male-only sites do occur, but are not heightened, due to lack of females (Adcock, 2019).

Middle-aged to old males (>25 years old) may be pushed out of their territories by younger dominant bulls into sub-optimal areas. Such animals are unlikely to breed again. Namibian authorities feared that leaving these marginalized animals in areas close to human settlements could lead to opportunistic poaching and stimulate further surges in illegal activity. Previous attempts to catch and reintroduce older black rhino bulls that were displaced from Etosha National Park were mostly unsuccessful. In most cases, the release of these older displaced

males back into existing rhino range led to fighting-related mortalities or the rhinos being displaced again. Limited conservation funding could be spent more effectively elsewhere.

Rhino population performance is density dependent (e.g., Okita Ouma *et al.,* 2008). Regular removals from established populations (including excess males) free up food reserves for other animals, especially breeding females. This both maintains productive densities and provides founder rhinos that can be invested in new areas with the potential for enhanced growth. Metapopulation growth rates increase with removals and creation of new breeding populations (Adcock, 2019). Compounding of increased metapopulation growth rates can result in many more rhinos, increasing a species' ability to withstand poaching. Managing for rapid population growth also minimizes loss of genetic heterozygosity through genetic drift (Brooks & Emslie, 1999). Accordingly, all official rhino plans/strategies recommend keeping established populations at productive densities through removals. The removal of some excess males in skewed populations through hunting can assist.

Managers need to limit inbreeding and maintain genetic diversity in populations – especially smaller ones. While some degree of inbreeding will be natural; if one or two dominant males have dominated the breeding for a significant period, their removal can enhance the genetic diversity and long-term viability of that population.

Karsten *et al.* (2011) found that this meta-population strategy appears to be delivering a genetically healthy population.

Socio-economic aspects

Rapid human population growth and associated economic pressures (especially prevalent in developing African countries) threaten wildlife through either unsustainable exploitation for subsistence and commercial purposes, or loss of wildlife habitats following land-use changes (IPBES, 2019). Rhinos are especially threatened by poaching to meet the persistent demand for their horns, leading to significant recurring financial obligations to cover essential security and management costs (Di Minin *et al*., 2015). Such costs vary by area. Items typically include infrastructure provision and maintenance (e.g., fencing), staff expenses (salaries, accommodation, etc.), vehicles/aircraft, equipment (weapons, monitoring, communications, etc.) and, in some instances, veterinary services and supplementary feeding.

The essential challenge for rhino conservators is to meet and contain these substantial costs, which have increased considerably with the increased poaching pressure over the last decade.

Contemporary African rhino conservators include government conservation agencies and, increasingly, private landowners (Emslie *et al*., 2019). In both South Africa and Namibia, white rhinos on private land are legally owned by the landowners; in South Africa some black rhinos are privately owned. The institutional arrangements in these countries include selective devolution of wildlife ownership and management authority, a model identified by Child (2019) as having performed best in terms of conserving large mammal populations both outside and within protected areas in Africa. Both countries also employ a conservation financing system that differs significantly from most other countries: being essentially decentralized, diversified, and supported by market mechanisms that channel direct monetary benefits from wildlife to relevant local levels rather than aggregating them centrally (where they are at greater risk of reallocation).

Two rhino conservation financing model variants exist in the two countries. South Africa's model is substantially market-oriented, whereas Namibia uses a hybrid model, treating the two species differently. The market-oriented model supplements rhino protection in state parks by enabling non-state landowners to benefit financially from activities such as photographic tourism, legal hunting, and live sales. This creates private incentives to protect and grow rhino populations on non-state land. State conservation agencies have also benefitted financially from live rhino sales to the private sector (Figure 1a). Figure 1b illustrates the mechanism for financing black rhino conservation in Namibia. Proceeds from black rhino trophy hunts are channeled into a state-administered Game Products Trust Fund and earmarked for contributions to specific rhino monitoring and management activities and support for community conservancies.

Critiques of the socio-economic effects of trophy hunting suggest that its contributions to country-level GDP are small relative to non-hunting wildlife tourism, and that benefits from hunting may be inequitably distributed, entrenching social inequality (Ghasemi, 2021). Whereas distributional concerns apply to all forms of wildlife tourism (hunting and non-hunting), and socio-economic transformation remains a pressing priority in many developing countries, the former claim is misleading. National GDP contributions are a poor indicator in terms of both broader socio-economic relevance and appropriate scale of analysis. GDP

metrics fail to consider essential ecosystems services and natural capital (Costanza *et al*., 1997). Nation states are an arbitrary level at which to make such assessments – more relevant are the global benefits of effective species conservation and ecosystem services provided by intact habitats, functionally populated with rhinos, and the more localized benefits that flow to specific rural landowners and communities, who are thereby incentivized to actively support conservation.

Arguments that contrast photographic with hunting tourism are furthermore misguided because these activities are mostly complementary rather than competing. Historically, hunting tourism has often acted as a pioneering developmental activity, providing the economic impetus to later establish photographic wildlife tourism operations. Hunting is often conducted in areas less suited to photographic tourism and which sometimes form buffer zones or corridors to supplement protected areas. The hunting fee and associated income generated per individual animal is substantial, such that a relatively small annual number of hunters (<100) can support proportionately far more rhinos and habitat (with far lower environmental impact) than countless non-hunting tourists (with a higher carbon footprint), who can repeatedly observe a small sample of habituated animals in a relatively confined area (Shumba *et al.,* 2021; Figure 2). A sensitivity analysis in Namibia demonstrates interdependence of the two activities and that the permanent loss of hunting tourism income would render most community conservancies economically unviable (Naidoo *et al.,* 2016).

Conservation impact of legal rhino hunting

By addressing the problems of excess males, high population densities, and inbreeding, limited targeted rhino hunting helps advance demographic and genetic conservation goals. The generation of additional revenue helps pay for and incentivize rhino conservation action.

While there have been no negative impacts following black rhino hunting, as discussed above hunting of white rhino was temporarily problematic on two occasions (over-hunting on private land in early years and later ‘pseudo-hunting’) although during those periods white rhino numbers still increased. Actions taken resolved both these issues. Figure 3a shows the growth of southern white rhino numbers since hunting started and how in relative terms the number hunted is very small. Figure 3b compares the number of hunts and proportions of populations hunted, showing the two temporarily problematic periods.

Up to and including 2018, a total of 45 black rhinos had been hunted in South Africa and 11 in Namibia; the Namibian black rhino hunts generated conservation revenues of >US$2 million. Total numbers of white rhinos hunted (until end 2018) are estimated at 2,537 (South Africa), 61 (Namibia) and 6 (Zimbabwe). In 2017 white rhino trophy fees alone generated ~US$6,7 million in South Africa.

Figures 4 and 5 show trends in numbers of southern white and black rhinos in the two major rhino hunting countries, South Africa and Namibia, relative to rhino numbers in other African range states, reflecting key policy change dates. Overall numbers of both species have increased since legal hunting restarted (with an 11-fold increase in white rhino numbers in South Africa and Namibia up to their peak in 2012), which has clearly been sustainable. The real threats to rhino populations are poaching for horn, causing escalated protection costs and reduced local incentives to conserve rhinos and their habitats – and not legal hunting (Emslie et al. 2019). The decline of white rhino numbers after 2012 is due to increased poaching (N>5,652), especially in the Kruger National Park (where hunting is not permitted), rather than legal hunting (N~400). Legal hunting is very selective whereas poachers kill valuable breeding females.

Conclusion

The African rhino case studies suggest that appropriately managed and regulated legal hunting (with trophy exports) can reinforce (rather than compromise) species and habitat conservation. This positive outcome is achieved through institutional arrangements that direct the flow of socio-economic benefits to locally relevant levels, thereby providing both 1) a source of finance for essential rhino security and management and 2) positive incentives for rural communities and private landowners to support conservation more generally. Similar results have been achieved for various other species in other contexts in southern Africa and elsewhere in the world (Cooney *et al*., 2017). Importantly, careful selection of animals to be hunted can ensure that small starting population sizes are not an impediment to the successful employment of this strategy. In the case of white rhinos, it helped enable their numbers and range to grow significantly.

Nowak *et al.* (2019) suggest that trophy hunting bans 'create opening for change'. In the case of African rhinos there is a high risk that such action now would result in negative socio-economic consequences at meaningful scales (Parker *et al*., 2019) with concomitant adverse outcomes for rhino conservation. As Africa struggles with declining sources of conservation funding in the wake of COVID-19 (Lindsey *et al*., 2020) policy makers must trade off such risks against the application of evolving ethical standards. Perhaps counter-intuitively, it is for relatively rare but actively managed species such as African rhinos that such complete hunting bans may carry the highest risk of an adverse conservation outcome.

Mindful of the increasing animosity towards trophy hunting, we suggest that regulation of hunting and trophy trade of threatened species should be evaluated on a case-by-case basis, given that there remain clear instances where legal hunting contributes positively toward achieving specified conservation goals.

Acknowledgements and data

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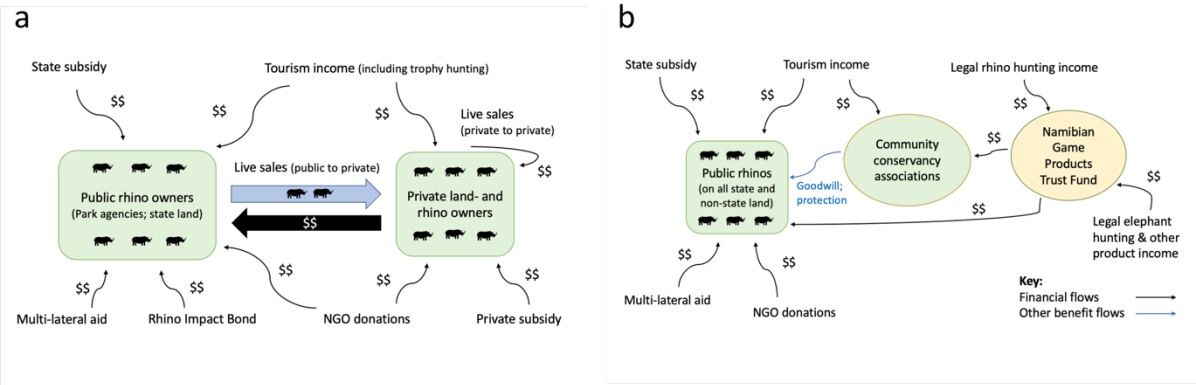
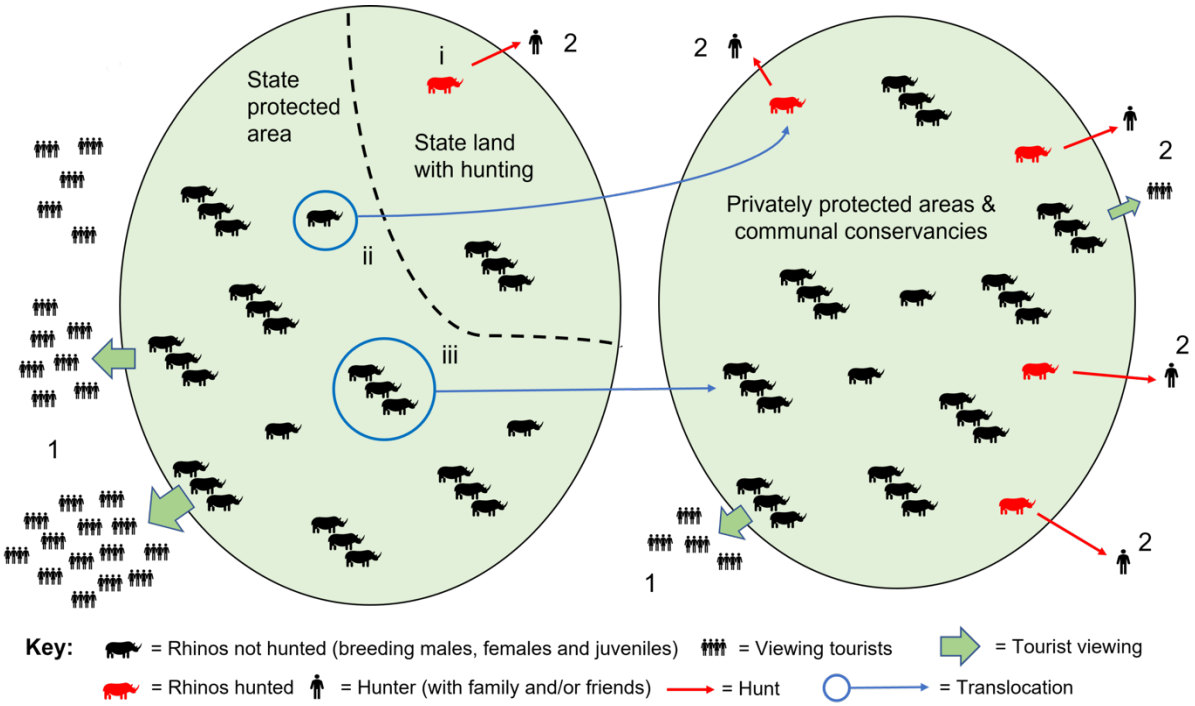
Article figures

Figure 1



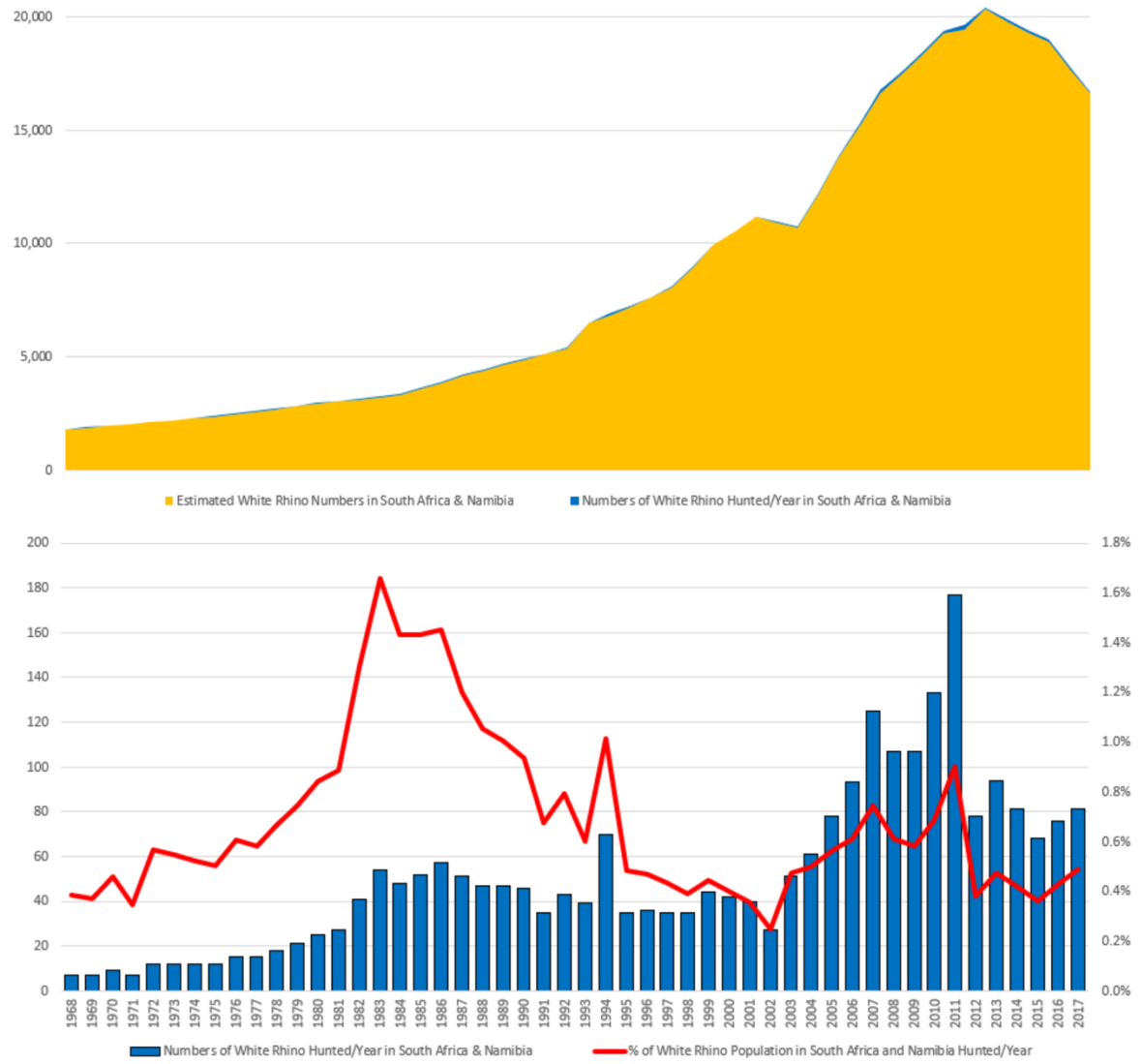
Financing models for rhino conservation: a) for South Africa and white rhinos in Namibia, and b) for black rhinos in Namibia

Figure 2



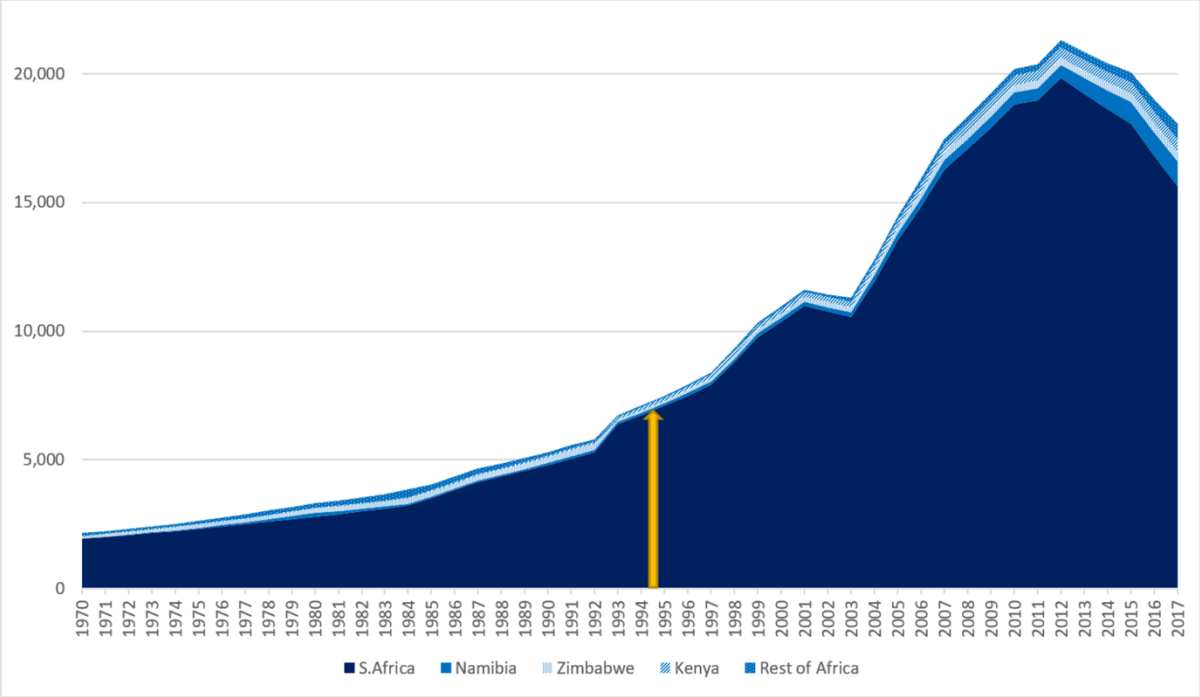
Rhino use on public and private lands, showing relative contributions of 1) numerous non-hunting tourist and 2) few legal hunting tourists. Rhinos move to private and hunting areas through i) natural dispersal, ii) translocation of excess males and iii) translocation of breeding herds. Hunts support additional range to that supported by non-hunting tourists alone and translocations provide financial support to reserves selling surplus rhinos.

Figure 3



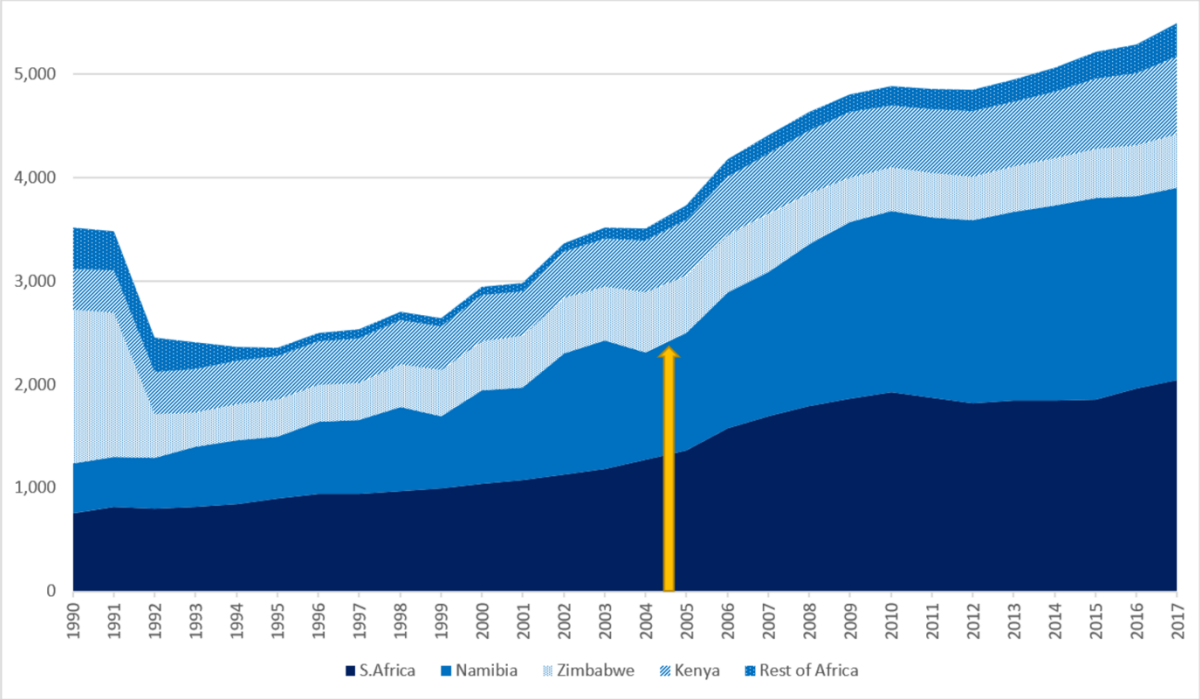
Above (a): Estimated numbers of live white rhinos in South Africa and Namibia (in yellow) and numbers legally hunted in these countries (blue).
Below (b): Absolute numbers of white rhino legally hunted (blue, left Y-axis) compared with relative % of population hunted (red, right Y-axis)

Figure 4



Total numbers of southern white rhinos by range state, following commencement of legal hunting in South Africa, with yellow arrow indicating date of eased restrictions on South African trophy exports

Figure 5



Total numbers of black rhinos by range state, before and after commencement of legal hunting, with yellow arrow indicating date of international approval of quotas for limited South African and Namibian hunts and trophy exports

Supplementary Material: Sources of data and information

Acknowledgements

Before detailing the sources of the data used in this paper, the authors would like to acknowledge and thank the many people in range states, and especially the official country representatives, who over the decades have contributed information on rhino numbers to the IUCN Species Survival Commission's African Rhino Specialist Group (AfRSG) and Southern African Development Community Rhino Management Group (SADC RMG). on the AfRSG. Over the last thirty years, this has allowed regular updating of rhino population data and facilitated the mandated reporting to the Convention on International Trade in Endangered Species of Fauna and Flora (CITES) Conferences of the Parties by IUCN, on trends in rhino numbers, hunting and poaching. The Rhino Resource Centre is a great source of many references with old historical estimates, and Kees Rookmaaker is thanked for his foresight and effort in creating and maintaining this invaluable source of rhino information.

The authors are especially grateful to Pierre du Preez, Piet Beytell, and Fillemon Iifo in Namibia, Thea Carroll and Mpho Tjiane in South Africa, and David Cumming in Zimbabwe for providing information on hunting in their countries over the years.

African Range states are also thanked for providing the AfRSG with regular poaching statistics.

Estimated Rhino Numbers

Estimated rhino numbers from 1973 used in this paper and shown in Figures 3, 4 and 5 were taken from a continental database of estimated rhino numbers per year by country and subspecies/genetic management cluster. This database was specifically set up by Richard Emslie as Coordinator of IUCN's Red List Authority on behalf of the IUCN Species Survival Commission's African Rhino Specialist Group (AfRSG) to collate historical numbers used as the basis for assessing population trends over the last 1, 2 or 3 generations as part of the latest IUCN African Rhino Red Listing assessment revisions (Emslie 2020).

Source of estimated numbers post 1992

Since 1992, data on African black and white rhino numbers have been collected and compiled by the AfRSG every one to three years (in 1992, 1993, 1995, 1997, 1999, 2001, 2003, 2005, 2007, 2010, 2012, 2015 and 2017 with complete black rhino estimates also available for 2018). Annual black rhino numbers for many major (but not all) rhino range states have also been collected and compiled by Keryn Adcock for the SADC Rhino Management Group through its regular status reporting (covering the period 1989-2018). These were used by AfRSG and IUCN's African Rhino Red List authority to improve continental estimates for years between AfRSG continental surveys. Eleven surveys of the status of white rhinos on private land in South Africa from 1987 to 2018 have also contributed to estimated numbers.

Regular updates of continental rhino numbers broken down by range state and subspecies/genetic management cluster are published in AfRSG Chair reports in the journal *Pachyderm*, and as part of

mandated joint IUCN/TRAFFIC rhino reports to CITES Conferences of the Parties (e.g., Emslie *et al*. 2019). Individual population breakdowns continue to be kept confidential at the request of some range states. To always use the best estimates possible, past estimates have sometimes been adjusted slightly up or down in the light of updated information or improved data analyses that has revised estimates.

Source of estimated numbers 1980-1991

Before the formation of the AfRSG, its predecessor, the IUCN SSC African Elephant and Rhino Specialist Group, compiled continental estimates for each country for 1980, 1984 and 1987 (Western & Vigne, 1985; Cumming *et al*., 1990).

Source of estimated numbers pre-1980

Klingel's (1979) survey of African rhinoceroses for IUCN also provided estimates for many populations across Africa in the late 1970s. Numerous other data sources, listed in the IUCN African Rhino Red List Assessment supplementary document (Emslie, 2020), were also used to derive the best possible estimates for the period 1973-1979 (and also before this). Brooks & Emslie (1999) also provided some historical numbers.

Interpolation of numbers between years

Population estimates were not available for every country every year. Estimates for missing years were interpolated by applying constant annual growth/decline rates that match up with available starting and end period estimates[1].

Uncertainty around point estimates of numbers

Population measurement error is often a large source of uncertainty for many species, but this is less of a problem for African rhinos, with many populations being monitored using individual identification-based methods. Very large rhino populations in larger protected areas and some other smaller, less-well-monitored sites, plus incomplete reporting on some white rhino populations on private land in South Africa, are the main sources of uncertainty in rhino numbers.

Several unrounded point estimates are given in the paper, prefaced by ~. These must be interpreted as having a degree of uncertainty around them. For example, as was reported to CITES CoP18, Emslie *et al.'*s (2019) bootstrapped estimates of 90% confidence levels around the 2017 continental point estimate of 18,067 white rhino were 17,212 to 18,915; and were 5366 to 5627 around the black rhino estimate of 5495. [2] Importantly, trends in estimated numbers of both species over time shown in Figures 3, 4 and 5 are of a much greater magnitude that the degree of uncertainty around individual point estimates for any given year (as indicated by above bootstrapped confidence levels).

Source of further information

[1] For example, supposing one only had estimates for a population of 211 in year 0 and 249 in year 3 (i.e., there were no estimates for years 1 and 2). This represents a total increase over the three years of 18.00948%. Raising 1.1800948 to the power of (1/number of years) gives 1.0567 or an average annual compounded increase of 5.67%. Applying this annual growth rate gives rounded interpolated estimates for year 1 of 223, and 236 for year 2. Applying this annual growth rate for a third year returns 249 – the same as the recorded population estimate for year 3.

[2] The bootstrapping was based on actual calculated confidence levels or best estimates of likely estimate precision for each population. What this means statistically, is that if we were to repeatedly estimate rhino numbers many times, then on average in about 9 out of 10 cases (assuming estimation is unbiased) the true number of rhinos would be expected to fall within our estimated 90% confidence levels.

For further information on numbers and data sources used in African Rhino Red Listing (which are also used in this paper) see section titled "Population estimates used for assessing changes over one, two and three generations" in *Supplementary Document for most recent IUCN African Rhino Red List Assessments:* https://www.iucnredlist.org/species/pdf/152728945/attachment

Figure I below is reprinted from the African Rhino Red Listing Supplementary document (Emslie 2020) and shows the estimated continental trends in rhino numbers since 1973 by subspecies/genetic management cluster.

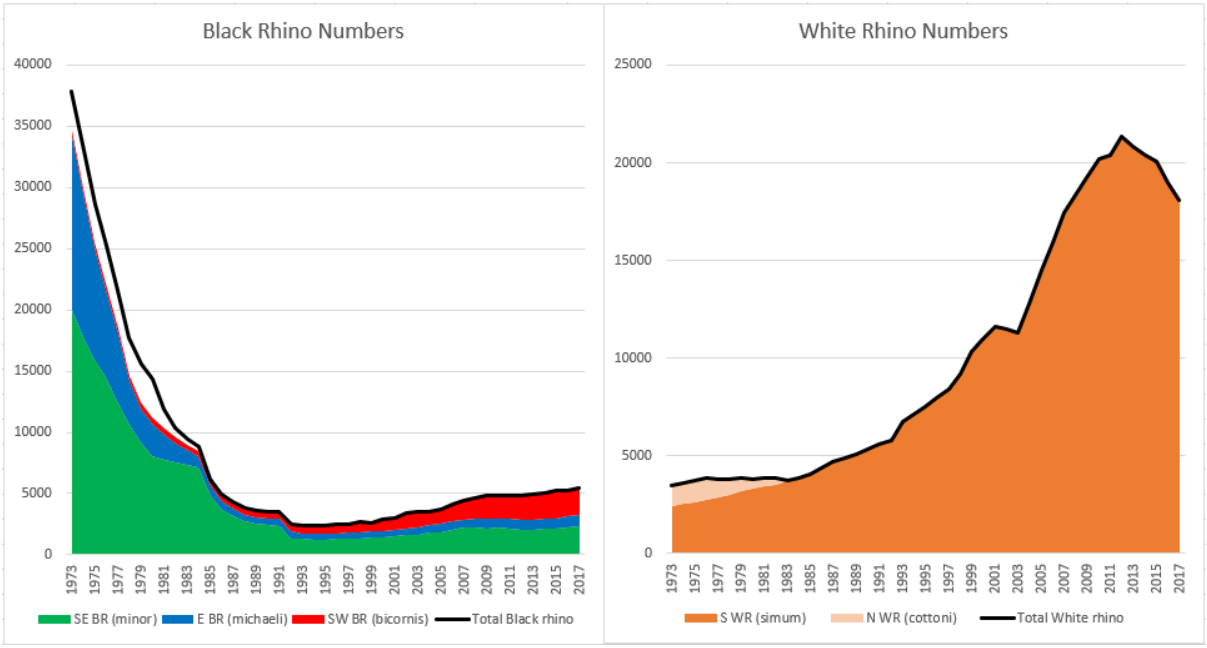


Figure I: Graph of estimated black and white rhino numbers by subspecies/genetic management cluster from the African rhino historical numbers database developed used to assess changes in numbers over three generations (44 years) under Criteria A2 and A4.

References:

Brooks, M., Emslie, R., 1999. African rhino: status survey and conservation action plan. IUCN, Gland, Switzerland.

Cumming, D.H.M, Du Toit, R.F. and Stuart S.N. 1990. *African Elephants and Rhinos – Status Survey and Conservation Action Plan*. IUCN SSC African Elephant and Rhino Specialist Group. IUCN, Gland, Switzerland.

Emslie, R., 2020. Red List, supplementary information: African Rhinos (*Ceratotherium simum* and *Diceros bicornis*, including the subspecies). In: IUCN 2020. IUCN Red List of Threatened Species. <www.iucnredlist.org>: pp. 1-54.

Klingel, H. 1979. Survey of African Rhinoceroses**.** Report to IUCN SSC African Elephant and Rhino Specialist Group**.**

Western, D. and Vigne, L. 1985**.** The Status of Rhinos in Africa. *Pachyderm* 4: 5-6.

Legal hunting information

A database of the numbers of rhino hunted has been built up over time by the authors and has been updated in recent years with data provided by South African and Namibian authorities. Data has also been reported to the Parties to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in joint IUCN/TRAFFIC rhino reports. These reports form part of the CITES Secretariat's reporting on rhinos at CITES Conferences of the Parties (CoP's) – see, for example, Emslie et al., (2019).

Much of the early compilation of South African hunting numbers came from an earlier analysis by Adcock and Emslie (1994) published in the proceedings of a symposium, "Rhinos as Game Ranch Animals". Allocation of estimated total white rhino numbers hunted in the early years (only given as a total for 1968-1987 in Adcock & Emslie 1994) were based on a survey of white rhino on private land by Daan Buys (1987). Estimates have since been improved for this paper by estimating numbers hunted broken down by year based largely on the raw data in an Appendix of Buys (1987).

As part of data collection for an international study of Cost:Benefits of different rhino conservation approaches in Africa and Asia in the early 1990s, Richard Emslie interviewed a number of hunting outfitters, analysed South African CITES export permits over a limited period, and obtained a full record of the number of white rhino hunted per year in the former semi-autonomous territory of Bophutatswana from 1982-1994. He used this information to derive best estimates of numbers hunted for 1987-94.

Data on numbers hunted between 1994 and 2004 were provided to the authors by the former South African Department of Environmental Affairs, currently the Department of Fisheries, Forestry, and the Environment (DFFE). SA data from 2004 is as reported in Emslie *et al*. (2019) and this was based on information provided by the same Department.

Data on rhino trophy hunting fees was similarly provided to the authors by DFFE.

Historical Namibian hunting data (numbers hunted, and black rhino trophy fees) was provided to the authors by the current Namibian Ministry for Environment, Forestry, and Tourism.

Thus, for both South Africa and Namibia, the authors were able to compile a good record of estimated numbers of both species hunted per year since legal hunting restarted.

Zimbabwean hunting data was obtained by analysing trophy export permits recorded in the CITES trade database. This indicated that six white rhinos have been legally hunted in Zimbabwe. David Cumming also confirmed that some limited white rhino hunts had taken place in the country. As so few white rhinos and no black rhinos have been hunted in Zimbabwe, this study has focused on the two countries that hunted larger numbers of both species of rhino (South Africa and Namibia). Figures 4 and 5 however do present trends in estimated rhino numbers over time in Zimbabwe.

For many years the majority of African rhino have been conserved by four countries – South Africa, Namibia, Zimbabwe, and Kenya. For this reason, Figures 4 and 5 provide a breakdown of numbers for these four countries, with the remainder summed and included as Rest of Africa totals.

References:

Adcock, K., Emslie, R., 1994. The role of trophy hunting in white rhino conservation, with special reference to Bop Parks, in: Penzhorn, B.L., Kriek, N.P.J. (Eds.), Proceedings of a Symposium on "Rhinos as Game Ranch Animals", Onderstepoort, 9 & 10 September 1994. Presented at Rhinos as Game Ranch Animals, South African Veterinary Association, Onderstepoort, South Africa, pp. 35–41. ISBN 1-875088-17-2

Buys D. (1987). A summary of the introduction of white rhino onto private land in the Republic of South Africa: Report for Rhino and Elephant Foundation. http://www.rhinoresourcecenter.com/pdf_files/129/1291956323.pdf

Emslie, R., (2020). Red List, supplementary information: African Rhinos (*Ceratotherium simum* and *Diceros bicornis*, including the subspecies). In: IUCN 2020. IUCN Red List of Threatened Species. <www.iucnredlist.org>: pp. 1-54. https://www.iucnredlist.org/species/pdf/152728945/attachment

Emslie, R. H., Milliken, T., Talukdar, B., Adcock, K., & Knight, M. H. (2019). *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 (Rev. CoP17), CoP18 Doc. 83.1 Annex 2.* CITES Secretariat. https://cites.org/sites/default/files/eng/cop/18/doc/E-CoP18-083-01.pdf

Poaching data

Since 2006 the AfRSG has regularly compiled and updated reported poaching data based on information kindly supplied by range states. Poaching updates are also published in AfRSG Chair reports in the journal *Pachyderm* and in joint IUCN/TRAFFIC rhino reports to CITES CoP's (e.g., Emslie *et al*., 2019).

Reported poaching figures represent minimum numbers, because in some populations (especially in very large areas with lower field ranger densities) as many as 20% of carcasses may