Realization of poaching effects on rhinoceroses in Kruger National Park, South Africa

Sam M. Ferreira*( )§, Cathy Greaver, Zoliswa Nhleko & Chenay Simms

Scientific Services, SANParks, Skukuza, 1350 South Africa

Received 1 August 2017. To authors for revision 5 September 2017. Accepted 19 October 2017

The persistence of black (*Diceros bicornis minor*) and white (*Ceratotherium simum simum*) rhinoceroses in the Kruger National Park (Kruger) is a key requirement for global rhinoceros conservation targets. Yet, poaching for rhinoceros horn poses a threat. In response, authorities are implementing an integrated response to curb the effect of poaching on rhinoceroses in Kruger. Nevertheless, researchers predicted both species would decline by 2016. The predictions were realized for southern white rhinoceroses, but it is uncertain whether the decline is real for south-central black rhinoceroses. Several evaluations are needed to elucidate uncertainties associated with detecting trends, the most important being to evaluate the effect of carcass detection rates on estimates of poaching rates. Nonetheless, poaching effects on rhinoceroses are disrupting conservation efforts to recover both southern white and south-central black rhinoceroses.

Keywords: population estimates, black rhino, white rhino, Kruger National Park.

INTRODUCTION

Wildlife trafficking remains one of the primary threats to biodiversity (Rosser & Manika, 2002). Several factors fuel poaching of high-value products derived from endangered species. Long histories of trading in such commodities, inelastic demand for products, high profit potential, weak law enforcement, unclear property rights and disincentives associated with human wildlife conflict all contribute to poaching onslaughts on species (Conrad, 2012).

The world had an estimated 29 324 rhinoceroses comprising five species made up of nine subspecies living in 14 countries by the end of 2015 (Emslie *et al.*, 2016). South Africa had 69.25%, India 9.93% and Namibia 9.44% and collectively the other 11 countries had 11.38%. On the African continent, South Africa was home to 79.23%, Namibia 10.80%, Kenya 4.38% and Zimbabwe 3.13% of Africa's rhinoceroses. The other seven countries contributed 2.46%. In Asia, India was home to 78.79% of Asia's rhinoceros population whilst Nepal contributed 17.45% and Indonesia 3.76%.

One of the world's key rhinoceros population localities is the Kruger National Park (Kruger) where south-central black (*Diceros bicornis minor*) and southern white (*Ceratotherium simum simum*) rhinoceroses occur. Kruger contributes ~22% to the south-central black rhinoceros numbers in South Africa, and ~49% to southern white rhinoceros numbers. During September 2015, between 313 and 453 south-central black and 8365 and 9337 southern white rhinoceroses were estimated for Kruger (Ferreira *et al.*, 2017). By the end of 2015, however, authorities detected 2936 rhinoceros carcasses attributed to poaching since 2011. The poaching of black and white rhinoceroses in Kruger reflects the symptoms of the drivers of poaching storms (Conrad, 2012).

Continued growth of south-central black rhinoceroses can be expected in the absence of poaching based on population data recorded during 2008 (Ferreira, Greaver & Knight, 2011). This did not materialize as poaching resulted in south-central black rhinoceros declines after 2008 (Ferreira *et al.*, 2015). Trends from 2008 to 2015, predict that between 244 and 431 south-central black rhinoceroses should be living in Kruger by 2016. Ferreira, Botha & Emmett (2012) in turn, predicted detectable declines for southern white rhinoceroses by 2016 given the dynamics noted at the end of 2010. Ferreira *et al.* (2015) recorded a possible decline between 2011 and 2015. Even so, these studies

*To whom correspondence should be addressed.
E-mail: sam.ferreira@sanparks.org

ISSN 2410-7220 [Print], ISSN 2410-8200 [Online] — DOI: https://doi.org/10.3957/056.048.013001

predict that 7645 to 9224 southern white rhinoceroses should be living in Kruger in 2016.

South-central black rhinoceroses require a 9% increase per annum and southern white rhinoceroses need 5% annual growth between 2016 and 2020 to reach Kruger's contribution to the South African population target (Ferreira *et al.*, 2017). These growth rates contrast against the predictions of previous studies. Here we report on the rhinoceros survey in Kruger during 2016 and evaluate the predictions from previous studies.

STUDY AREA

Kruger covers 19 485 km^2 in the low-lying savanna of South Africa. Annual rainfall exceeds 450 mm. Granite and gneiss deposits separated by Karoo sediment combine with wooded savanna comprising *Sclerocarya caffra* and *Senegalia nigrescens* on basalts and mixed *Combretum* spp. and *Senegalia* spp. on granites in southern Kruger. *Colophospermum* woodlands dominate the north to create 35 different landscape types across Kruger.

Management of Kruger benefits from 22 Sections that serve as management units where a section ranger, in charge of 15 to 25 field rangers, is responsible for the integrity of a section and conservation management. Conservation management largely focuses on implementing controlled fire regimes, the closure of boreholes and the destruction of dams with associated rehabilitation, clearing of alien plants and general law enforcement. A key priority in recent years is dedicated protection of rhinoceroses through implementing compulsory anti-poaching interventions, one of the key activities within the four pillared integrated rhinoceros management strategy of South Africa (Department of Environmental Affairs, 2014).

METHODS

We made use of two complimentary methods to estimate rhinoceros numbers. Most of Kruger's rhinoceroses occur south of the Olifants River where a sample-based block survey for 2016 was focused. To the sample-based block survey we added collated estimates of the minimum numbers of rhinoceros thought to be alive based on ranger reports for the sections north of the Olifants River.

The sampled-based block survey targeted eleven sections in southern Kruger using a helicopter (Bell Jet Ranger 206) to count south-central black and southern white rhinoceroses on 486 randomly placed blocks each 3 × 3 km in size (see Ferreira *et al.*, 2015 for more details). This design covered 48.8% of the area south of the Olifants River. Surveyors systematically completed transects comprising a 200 m observation strip on each side of the helicopter within each block, with flights 45 m above ground at a speed of 65 knots. The survey team comprised a pilot, a data recorder and two observers.

Jolly's estimator (Jolly, 1969) allowed landscape-specific and overall estimates after analyses accounted for bias. We used availability bias assessments conducted during 2013, derived from relationships between vegetation cover (Bucini *et al.*, 2011) and rhinoceros visibility (Ferreira *et al.*, 2015), and related vegetation cover to the average Enhanced Vegetation Index (EVI) on a block at the time. EVI is more responsive to canopy structural variations, including leaf area index, canopy type, plant physiognomy and canopy architecture (Huete *et al.*, 2002). We then used the EVI recorded for blocks during 2016 to estimate vegetation cover and then availability bias (Ferreira *et al.*, 2015) for each block during 2016. Observer bias came from estimates made during a previous black rhinoceros survey (Ferreira *et al.*, 2011). Detectability bias was minimal given that the size of the observation strips were narrower than those noted by previous studies (Kruger, Rielly & Whyte, 2008) when detectability was negligible.

We extracted population estimates for previous years from published literature (Ferreira *et al.*, 2011; Ferreira *et al.*, 2012; Ferreira *et al.*, 2015; Ferreira *et al.*, 2017). Where population estimate data were missing, we smoothed the trends by averaging between the last survey and the next available survey. We then used estimated three-year running means repeatedly drawing from the distribution of estimates defined by the 95% confidence interval of each estimate. This allowed us to obtain a smoothed curve represented as a swath of 95% confidence intervals for the entire time series since the introduction of southern white rhinoceroses and south-central black rhinoceroses during the 1960s (Penzhorn, 1971).

RESULTS

Estimated availability bias recorded on a specific survey block during 2013 (Ferreira *et al.*, 2015) associated positively with the average EVI on that same block at the time (vegetation cover = 0.019NDVI–22.67, r^2 = 0.32, t_{876} = 412.77, P <

0.01). This predicted that vegetation covered between 0% and 60.98% of a block across our survey area during 2016. Up to 26.4% and 17.1% of south-central black and southern white rhinoceroses were not available to be sampled due to vegetation cover following Ferreira *et al.* (2015). Collated observer bias was 3.8% of rhinoceroses missed by all observers (Ferreira *et al.*, 2011), while detectability bias was negligible (Kruger *et al.*, 2008).

During 2016, 177 south-central black and 3031 southern white rhinoceroses were counted on sample blocks south of the Olifants River. Rangers reported five south-central black and between 49 and 67 southern white rhinoceroses north of the Olifants River.

Counts, biases and estimated numbers of southern white rhinoceroses north of the Olifants River translate to a total 7235 (95% CI: 6649–7830) white rhinoceroses living in Kruger during 2016. This was significantly lower than the 8875 (95% CI: 8365–9337) that occurred in Kruger during 2015 (Ferreira *et al.*, 2017) (Fig. 1). The 95% confidence interval of the 2016 estimate overlaps by only 7.1% with the predicted 95% confidence interval derived from previous trends (Ferreira *et al.*, 2017). Detectable declines were thus realized for southern white rhinoceroses during 2016.

The 2016 survey estimated 406 (95% CI: 349–465) south-central black rhinoceroses in Kruger with confidence intervals overlapping the 2015 estimate of 384 (95% CI: 313–453) (Ferreira *et al.*, 2017). Although black rhinoceros estimates are similar for 2015 and 2016, it is of concern that estimates are substantially lower than the 627 (95% CI: 588–666) estimated during 2008 (Ferreira *et al.*, 2011) (Fig. 1). Even so, the 95% confidence interval for 2016 overlapped by 43.2% with that of the confidence intervals of the predicted estimate.

DISCUSSION

Authorities implement several initiatives to protect rhinoceroses in Kruger (see Fig. 2 for a schematic

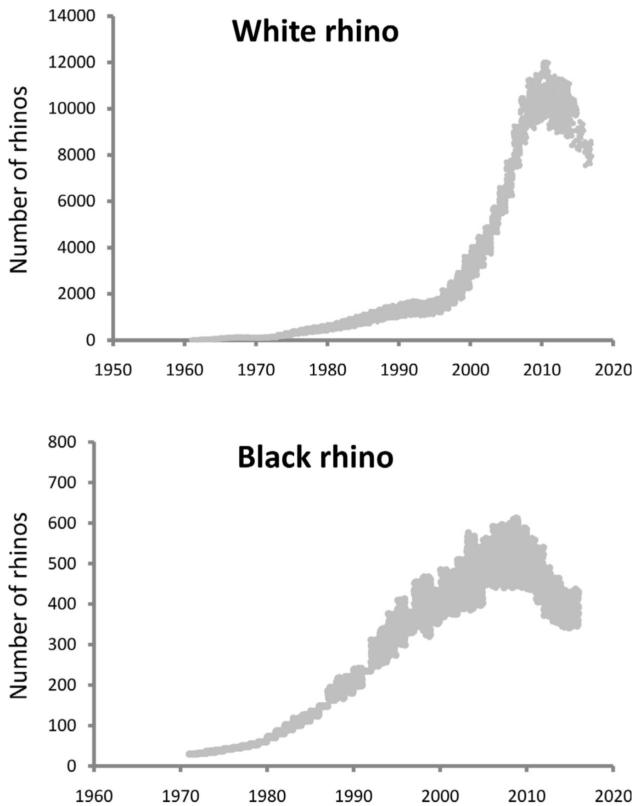

Fig. 1. White rhino and black rhino trends recorded for Kruger Narional Park since re-introductions. Figures extracted from previous published trends on black and white rhino. We provide a distribution of estimates within confidence intervals.

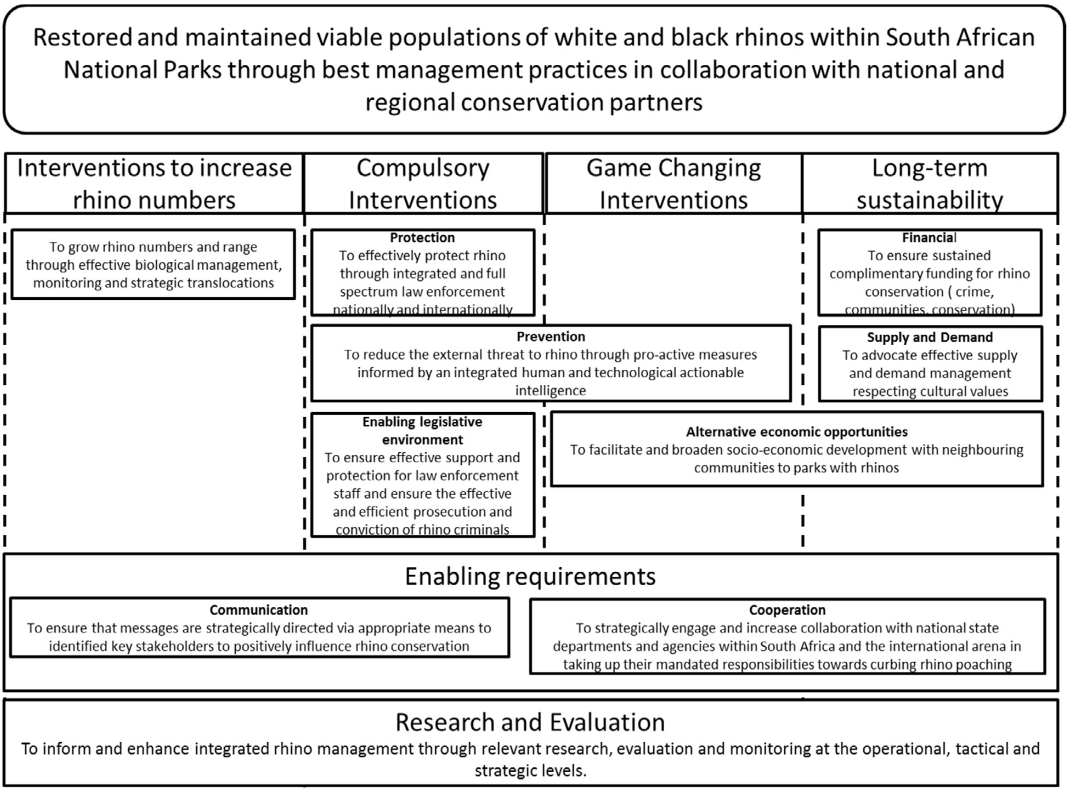

Fig. 2. Integrated management of rhinoceroses implemented by SANParks.

representation of these initiatives). Although a large focus is on compulsory anti-poaching (Department of Environmental Affairs, 2014), complimentary biological management remains a key intervention (Knight *et al.*, 2015). These interventions are largely the responsibility of conservation management authorities. The consequences for the two species of rhinoceroses that occur in the Kruger vary. Predictions for continued declines in south-central black rhinoceroses (Ferreira *et al.*, 2017) were not supported by our results. In contrast, predicted detectable declines for southern white rhinoceroses (Ferreira *et al.*, 2012) were realized by 2016.

Although the predictions for continued declines in the south-central black rhinoceros population in Kruger did not occur, several concerns caution interpretation of these results. Point estimates as well as their precision are particularly sensitive to sample-based approaches when populations are small, or densities are low (*e.g.* Barnes, 2002). Some estimates in a time series may thus be excessively high or low. For example, the high point estimate of 627 south-central black rhinoceroses in 2008 (Ferreira *et al.*, 2011) could be an anomaly. The decline in black rhinoceros numbers (Ferreira *et al.*, 2017) for instance, becomes non-significant if analysts exclude the estimate for 2009 (Ferreira *et al.*, 2011). Estimates of vital rates could help to address these uncertainties by checking what trends observed fecundity and survival schedules predict.

Continued declines in the southern white rhinoceros population did occur as predicted. Estimates for southern white rhinoceroses are unlikely to be biased by low density effects of small population estimates. The detected decline, however, could be sensitive to an outlier effect (Tsay, 1988) of the estimates during 2010 (Ferreira *et al.*, 2012) or 2016. Exclusion of the 2010 estimates, for instance, resulted in a non-significant annual decline from 2011 to 2015. Disregarding such outlier effects, the significantly lower estimate noted during 2016 compared to 2015 is of key concern and challenges the required contribution by Kruger to future southern white rhinoceros targets (Ferreira *et al.*, 2017).

How real is the detected decline? Several factors could influence the low estimates of southern white rhinoceroses that we noted. Three of these associate with environmental conditions. Kruger experienced a drought during 2016 (Smalman, 2016). Southern white rhinoceros calf survival decreases when environmental conditions worsen in Kruger (Ferreira *et al.*, 2015). In addition, responses of large mammals to constrained resources, either through environmental variation or density effects, predicts first a reduction in calf and juvenile survival, then decreased reproduction and ultimately also decrease in survival of adults (Eberhardt, 2002). A severe drought may thus result in increased mortalities, specifically for a grazer like the southern white rhinoceros. Note that Kruger management recorded 127 natural mortalities of southern white rhinoceroses between the 2015 and 2016 surveys compared to the 83 noted between the 2014 and 2015 surveys. South-central black rhinoceroses, a browser and thus less sensitive to drought impacts, had 11 natural mortalities between the 2015 to 2016 surveys compared to 14 between 2014 and 2016 (SANParks, unpubl. data). The effects of drought-induced natural mortalities on population estimates thus requires consideration when interpreting our results.

A second biological effect associates with how southern white rhinoceroses respond to spatial variability in resources. Availability of water and food play a key role in how southern white rhinoceroses use landscapes (Owen-Smith, 1988). The spatial patterns of EVI during the 2016 drought across Kruger highlighted some areas that are impacted more than others (SANParks, unpubl. data). We expect that southern white rhinoceroses will choose landscapes with improved grass resources rather than other landscapes. In addition, Kruger substantially reduced the distribution of water during droughts as a result of the closure of boreholes and removal of constructed dams over an extended period (Smit, 2013). Adjacent areas to the west of Kruger, however, retained high densities of boreholes and more uniform distributions of water (Smit, 2013). Southern white rhinoceros habitat selection during the drought thus predicts individual movements out of Kruger into adjacent private reserves that could reduce estimates. Thus the effects of local movement also require consideration when interpreting future trends.

A further consequence of individual southern white rhinoceros responses to resource availability is drought-induced clumping at resource hotspots. We thus expect that southern white rhinoceroses are more clumped in their distribution as they congregate around key water and food resources during droughts. For instance, during 2015, survey blocks with rhinoceroses had 10 individuals on average. Survey blocks with rhinoceroses had 13 individuals on average during 2016 and there were many more blocks without rhinoceroses. This suggest that clumping occured and could impact estimates as well as their precision (Ferreira & van Aarde, 2009).

Biological management, a key component of conserving both rhinoceros populations in Kruger (Knight *et al.*, 2013; Knight *et al.*, 2015), could complicate how rhinoceroses respond biologically to environmental conditions. In fact, that is the primary purpose of biological management – maximize population growth to help achieve rhinoceros conservation targets. SANParks has removed 1402 southern white rhinoceroses from Kruger since 1990 (SANParks, unpubl. data) and has contributed to the establishment of populations on various properties. For instance, between 5563 and 6110 southern white rhinoceroses resided on private and provincial reserves as additional rhinoceros conservation areas by the end of 2015 (Emslie *et al.*, 2016). Removals, however, may limit populations when it forces the source population into an ecological trap (Schlaepfer, Runge & Sharman, 2002). In such a case, the total number of births cannot offset the total number of individuals removed at a local scale. During 2015 and 2016, authorities removed 217 southern white rhinoceroses from focal areas north of the Sabie River. Even so, removed individuals are not a loss to the overall rhinoceros targets of South Africa (Knight *et al.*, 2013, Knight *et al.*, 2015). A key requirement is thus to evaluate how removed individuals perform in smaller rhinoceros strongholds where poaching risks are lower and conditions are favourable for breeding. This will allow estimating offsets of improved breeding in strongholds against poaching losses in Kruger.

Evaluating breeding offsets through biological management and moving some individuals to strongholds is particularly important since southern white rhinoceros recruitment in Kruger may not offset the recorded poaching. Rangers achieved a reduction in realized poaching rates based on detected carcasses. Realized poaching rate was 9.3% for southern white and 12.3% for south-central black rhinoceroses between the

2014 and 2015 surveys – between the 2015 to 2016 surveys, the rates were 7.3% and 11.2% respectively (SANParks, unpubl. data). Authorities, however, need to reduce realized poaching rates to <6.0% to have recruitment in Kruger offset poaching (Ferreira *et al*., 2017).

Our evaluation of the potential mechanisms that contribute to the substantial reduction in southern white rhinoceros estimates in Kruger assumed high confidence in carcass detection rates. SANParks aim to detect >75% of rhinoceros carcasses in Kruger within three days of death of an individual. Detection ranged from 52 to 71% within 3 days, with more than 88% of carcasses detected within two weeks (Ferreira *et al*., 2012). A key challenge is the uncertainty in how many carcasses rangers never detect. It is thus of key importance to evaluate carcass detections and adjust estimates of fatalities (*e.g.* Huso, 2011) as poaching rates could be significantly higher.

CONCLUSION

Our evaluation highlights that poaching has a negative effect on at least one of the rhinoceros species occurring in Kruger. Authorities require continued implementation of the various initiatives. This is particularly important given the additional challenges imposed by drought on rhinoceros persistence in the face of the excellent achievements of rangers that reduced realized poaching rates. Importantly, SANParks' biological management initiatives need to be adaptive and the consequences of removals for source populations should be weighed against the gains made at targeted rhino strongholds.

ACKNOWLEDGEMENTS

We are grateful to John Bassi the pilot and all the observers Adolf Manganyi, Pauli Viljoen, Marius Renke, Marius Snyders, Ben Wigley and Irwing Knight. SANParks provided financial and logistical support. Sam Ferreira completed the conceptual approach, analyses and writing. Cathy Greaver oversaw the logistics of field data collections and contributed to the writing. Zoe Nhleko completed the data capture and quality checking after each day of survey. Chenay Simms assisted with spatial planning, quality checking of data and contributed to the writing.

[§]ORCID iD

S.M. Ferreira orcid.org/0000-0001-6924-1150

REFERENCES

Barnes, R.F.W. (2002). The problem of precision and trend detection posed by small elephant populations in West Africa. *African Journal of Ecology,* 40, 179–185.

Bucini, G., Hanan, N.P., Boone, R.B., Smit, I.P.J., Saatchi, S., Lefsky, M.A. & Asner, G.P. (2010). Woody fractional cover in Kruger National Park, South Africa: remote-sensing based maps and ecological insights. In M.J. Hill & N.P. Hanan (eds.), *Ecosystem function in savannas: measurement and modeling at landscape to global scales,* (pp. 219–237). Boca Raton, U.S.A.: CRC/Taylor and Francis.

Conrad, K. (2012). Trade bans: a perfect storm for poaching? *Tropical Conservation Science*, 5, 245–254.

Department of Environmental Affairs, 2014, *Minister Edna Molewa leads implementation of integrated strategic management of rhinoceros in South Africa*. Retrieved from https://www.environment.gov.za/mediarelease/molewaintegratedstrategicmanagement rhinoceros on 23 February 2017.

Eberhardt, L.L. (2002). A paradigm for population analysis of long-lived vertebrates. *Ecology,* 83, 2841–2854. https://doi.org/10.1890/0012-9658(2002)083[2841:APFPAO]2.0.CO;2.

Emslie, R.H., Milliken, T., Talukdar. T., Ellis, S., Adcock, K. & Knight, M.H. (2016). African and Asian rhinoceroses – status, conservation and trade. CoP17 Doc 16, Annex 5. Retrieved from https://cites.org/sites/default/files/eng/cop/17/WorkingDocs/E-CoP17-68-A5.pdf on 20 February 2017.

Ferreira, S.M., Bisset, C., Cowell, C., Gaylard, A., Greaver, C., Hayes, J., Hofmeyr, M., Moolman-van der Vyver, L. & Zimmerman, D. (2017). The status of rhinoceroses in South African National Parks. *Koedoe*, 59, a1392. https://doi.org/10.4102/koedoe.v59i1.1392

Ferreira, S.M., Botha, J.M. & Emmett, M.C. (2012). Anthropogenic influences on conservation values of white rhinoceros. *PLOS ONE*, 7, 1–14. https://doi.org/10.1371/journal.pone.0045989

Ferreira, S.M., Greaver, C.C. & Knight, M.H. (2011). Assessing the population performance of the black rhinoceros in Kruger National Park. *South African Journal of Wildlife Research*, 41, 192–204. https://doi.org/10.3957/056.041.0206

Ferreira, S.M., Greaver, C., Knight, G.A., Knight, M.H., Smit, I.P.J. & Pienaar, D. (2015). Disruption of rhino demography by poachers may lead to population declines in Kruger National Park, South Africa. *PLOS ONE*, 10, 1–18. https://doi.org/10.1371/journal.pone.0127783

Ferreira, S.M. & van Aarde, R.J. (2009). Aerial survey intensity as a determinant of estimates of African elephant population sizes and trends. *South African Journal of Wildlife Research*, 39,181–191. https://doi.org/10.3957/056.039.0205

Huete, A., Didan, K., Miura, T., Rodriguez, E.P., Gao, X. & Ferreira, L.G. (2002). Overview of the radiometric and biophysical performance of the MODIS vegetation indices. *Remote Sensing of Environment*, 83, 195–213. https://doi:10.1016/S0034-4257(02)00096-2

Huso, M.M. (2011). An estimator of wildlife fatality from observed carcasses. *Environmetrics*, 22, 318–329.

Jolly, G.M. (1969). Sampling methods for aerial censuses of wildlife populations. *East African Agricultural and Forestry Journal*, 34, 46–49.
https://doi.org/10.1080/00128325.1969.11662347

Knight, M.H., Balfour, D. & Emslie, R.H. (2013). Biodiversity management plan for the black rhinoceros (*Diceros bicornis*) in South Africa 2011–2020. *Government Gazette* (*South Africa*) 36096: 5–76.

Knight, M.H., Emslie, R.H., Smart, R. & Balfour, D. 2015. Biodiversity management plan for the white rhinoceros (*Ceratotherium simum*) in South Africa 2015–2020. Pretoria, South Africa: Department of Environmental Affairs.

Kruger, J.M., Reilly, B.K. & Whyte, I.J. (2008). Application of distance sampling to estimate population densities of large herbivores in Kruger National Park. *Wildlife Research* 35, 371–376.
https://doi.org/10.1071/WR07084

Owen-Smith, R.N. (1988). *Megaherbivores: the influence of very large body size on ecology*. Cambridge, U.K.: Cambridge University Press.

Penzhorn, B.L. (1971). A summary of the re-introduction of ungulates into South African National Parks (to 31 December 1970). *Koedoe,* 14, 145–159.

Rosser A.M & Manika S.A. (2002). Overexploitation and species extinctions. *Conservation Biology*, 16, 584–586.

Schlaepfer, M.A., Runge, M.C. & Sherman, P.W. (2002). Ecological and evolutionary traps. *Trends in Ecology & Evolution*, 17, 474–480.
https://doi.org/10.1016/S0169-5347(02)02580-6

Smalman, N. (2016). Devastating effects of drought becomes clear Kruger.
Retrieved from http://lowvelder.co.za/313600/devastating-effects-of-drought-becomes-clear-kruger on 5 July 2017.

Smit, I.P.J. (2013). Systems approach towards surface water distribution in Kruger National Park, South Africa. *Pachyderm*, 53, 91–98.
https://doi.org/10.4102/koedoe. v55i1.1107

Tsay, R.S. (1988). Outliers, level shifts, and variance changes in time series. *Journal of Forecasting*, 7, 1–20.
https://doi.org/10.1002/for.3980070102

Responsible Editor: A.M. Shrader