Aspects of the urban and rural ecology of Nile monitors (*Varanus niloticus*) in KwaZulu-Natal, South Africa

Euan Genevier

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ABSTRACT

Changing land use, which is taking place globally at an unprecedented rate, generally has a negative impact on biodiversity. However, some species persist in or even exploit these anthropogenically modified environments. Little is known about how reptiles are affected by urbanisation, especially in the global South. Nile monitors (*Varanus niloticus*) are Africa's largest lizard and are amongst the most exploited tetrapods on the continent, yet they can be found in highly urbanised and densely populated areas of KwaZulu-Natal, South Africa. They remain greatly understudied, with many gaps in even basic aspects of their ecology. They are projected to become threatened in parts of their range without successful management. This thesis focuses on two broad aspects of the ecology of Nile monitors in KwaZulu-Natal Province, South Africa. Namely, a) their distribution and persistence in a mosaic of land-use type and human population density in two focal areas and b) their population size and morphometrics in rural and urban areas. The aim was to fill important knowledge gaps in the ecology of this highly exploited, charismatic, and yet understudied species to make management recommendations and promote sustainable co-existence of people with Nile monitors in KwaZulu-Natal, South Africa.

The first aim was to determine if Nile monitors occur in urban environments in KwaZulu-Natal and what land use types they are associated with. Using 419 Nile monitor sightings, two hotspots were found in the region, one dominated by protected land and the other dominated by mosaics of anthropogenic land use. They were found throughout regardless of human density. However, their use of residential areas was greater in the region with less available natural land. Increasing human density was positively associated with the use of more wooded habitats, except for planted forests, which were avoided systematically. Their association with water was confirmed; however, increased human density also increased their mean distance to rivers, likely

reflecting their use of swimming pools, storm drains and other anthropogenic water features. Both managed and unmanaged green spaces (e.g., gardens, parks, golf courses) in urban mosaic landscapes are important features for the persistence of Nile monitors.

The second aim was to estimate the Nile monitor population size using a mark-resight method with baited camera traps on a gradient of land use in three different specific sites and also compare urban and rural morphometrics throughout KwaZulu-Natal. Nile monitor limbs, body width, tail base, heads, and body mass were larger in rural areas but only marginally. Furthermore, we found greater population size estimates with increased habitat modifications (golf course > farmland > nature reserve), but the average estimated snout-vent length decreased. Reduced size may result from intraspecific or interspecific competition and different predation dynamics in urban and rural ecosystems. More independent trigger events of potentially competing diurnal mesocarnivores were recorded in the nature reserve and farmland than on the golf course. Despite the larger population size estimate of monitors, there was evidence of poaching at the golf course. One individual was stolen from a trap, and four instances of deployed snare traps were recorded. The inclusion of dead individuals for morphometrics has shed light on the added threats caused by domestic dogs (*Canis lupus familiaris*), people and vehicles for urban monitor populations.

It is concluded that Nile monitors are successfully persisting in the mosaic urban landscapes of KwaZulu-Natal but are not as successful as some other urban varanids. The use of anthropogenic land-use, and thus conflict with people, will likely increase with continued land-use change. Future research should investigate Nile monitor interactions with people and pets as well as conflict mitigation strategies and both their urban and rural spatial ecology.

PREFACE

The data described in this thesis were collected in KwaZulu-Natal, Republic of South Africa from July 2022 to December 2023. Experimental work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, under the supervision of Prof Colleen T. Downs and Dr Cormac Price.

This thesis, submitted for the degree of Master of Science in the College of Agriculture, Engineering and Science, University of KwaZulu-Natal, School of Life Sciences, Pietermaritzburg campus, represents original work by the author and has not otherwise been submitted in any form for any degree or diploma to any University. Where use has been made of the work of others, it is duly acknowledged in the text.

…….

Euan EZ Genevier

January 2024

I certify that the above statement is correct, and as the candidate's supervisor, I have approved this thesis for submission.

……..

Prof Colleen T. Downs

Supervisor

January 2024

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DECLARATION 1 - PLAGIARISM

I, Euan Genevier, declare that

1. The research reported in this thesis, except where otherwise indicated, is my original research.
2. This thesis has not been submitted for any degree or examination at any other university.
3. This thesis does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other persons.
4. This thesis does not contain other persons' writing, unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:
    a. Their words have been re-written, but the general information attributed to them has been referenced
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DECLARATION 2 - PUBLICATIONS

DETAILS OF CONTRIBUTION TO PUBLICATIONS that form part and/or include research presented in this thesis.

PUBLICATION 1. Formatted for Landscape Ecology- provisionally accepted

Habitat use and distribution of Nile monitors (*Varanus niloticus*) in a mosaic of land use type and human population density in KwaZulu-Natal, South Africa

EEZ Genevier, C Price, NB Evans and CT Downs

Author contributions:

EEZG conceived paper with CP and CTD. CTD sought funding. EEZG and NBE collected data. EEZG analysed data, and wrote the paper. CTD and CP contributed valuable comments to the manuscript.

PUBLICATION 2. Formatted for the Journal of Zoology (London) - provisionally accepted

Population dynamics and morphometrics of Nile monitors (*Varanus niloticus*) on a gradient of urbanisation in KwaZulu-Natal, South Africa

EEZ Genevier, C Price, NB Evans and CT Downs

Author contributions:

EEZG conceived paper with CP and CTD. CTD sought funding. EEZG and NBE collected data. EEZG analysed data, and wrote the paper. CTD and CP contributed valuable comments to the manuscript.

Signed: ……

Euan EZ Genevier

January 2024

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Nile monitors are a particularly hard species to study as they are fast and skittish, usually disappear underwater within seconds of being seen, and can be difficult to manipulate. Therefore, most ecological studies use dead individuals caught by local fishermen or from museum specimens. A rather enjoyable quote from the leading expert on the species, Vivian de Buffrénil, is as follows: “When fighting for its life, a Nile monitor was a more dangerous adversary than a crocodile of a similar size” (1998). The success of this ambitious project would have been

impossible without Nick Evans's dedication, passion for research and tremendous skill and expertise in reptiles. Nick has caught more monitors than anyone in this study, has continuously promoted the project, and has advocated for the conservation of urban reptiles in Durban. I am incredibly grateful to him. The Durban aspects of this study have also greatly benefited from Dylan Leonard's passion and expertise. He has helped provide much-needed veterinary treatment to injured Nile monitors and two egg-bound females, as well as incubated wild Nile monitor eggs for us. The first wild Nile monitor eggs to be incubated and documented in South Africa since 1982.

Tala Collection Game Reserve proved to be an incredible site for this research. Not only because of the great monitor population and great scenery but also because of the tremendous help from Charl Frik. Charl has regularly checked traps for us, has continuously reported sightings, and has always been happy to help with data collection. His dedication to the project was very helpful, and such help was hard to come by.

Craig Cordier has assisted with many Nile monitors, either for data collection or moving large traps, by rescuing monitors in Pietermaritzburg and always giving sound advice when needed. Craig would go out of his way to help process Nile monitors, no matter where or what time of day. I am very grateful, as processing is not a one-man job. I would also like to thank every reptile remover who has contributed either by sending sightings or letting Nick or me collect an individual for processing and release. One that stands out in Pietermaritzburg is Dean Boswell. He has continuously sent sightings, let me process the Nile monitors he caught and even took measurements himself when I was abroad.

Finally, I would like to thank every intern and friend who has helped with fieldwork and anyone else who I may have missed.

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CHAPTER 1

INTRODUCTION

1.1 Background

Globally, with the continued increase in human populations, there is unprecedented changing anthropogenic land use. Generally, this impacts biodiversity negatively. However, some species persist and even exploit these changing environments (Downs et al., 2021; Singh et al., 2021). Little is known about how reptiles respond to anthropogenic changing land use, especially in Africa. The different selective pressures of urban, farmland or natural habitats can change animal behaviour, physiology, and morphology (Law et al., 2016; Putman and Tippie, 2020). The anthropogenic effect on behaviour can have ecological consequences (Uyeda et al., 2012; Wolfe et al., 2018; French et al., 2018).

Anthropogenic factors are having some devastating consequences on wildlife (Morner, 2002; Lowry et al., 2012; Taylor-Brown et al., 2019; McDonald et al., 2020). Human population growth and associated expansion in anthropogenic activities are the primary causes of an accelerated increase in extinction globally (Ceballos et al., 2015; Lowry et al., 2019). Sub-Saharan Africa is projected to experience the largest human population growth of all continents (Gu et al., 2021; Brum et al., 2022). Furthermore, much of Africa's urban development is fast and unplanned, making rapid unplanned urban development one of the major threats to sustainable development in Africa (Cobbinah et al., 2015; Okeke, 2021). Despite resulting in a general decline in biodiversity, many species across taxa successfully persist, adapt or even exploit urban environments both in South Africa and internationally (Lowry et al., 2012; Downs et al., 2021; Singh et al., 2021). There is great conservation value in understanding what factors facilitate the success of some species in cities and the failure of others (Lowry et al., 2013).

More than a fifth of all reptile species are threatened by extinction (Cox et al., 2022). There is a critical need for herpetofauna conservation and research gaps to be filled to make critical research-based management decisions. Most African reptiles are lacking in biological and ecological data (Dalhuijsen et al., 2014). Forty-five percent of families are threatened more than expected by chance, yet the vast knowledge gaps limit African reptile conservation (Tolley et al., 2016). Their main threat on the continent is land transformation for agriculture (Tolley et al., 2016, 2019). Varanid biology is poorly understood in general (Mendyk, 2018).

Nile monitors (*Varanus niloticus*) are Africa's largest lizard species and arguably the fourth largest globally, reaching up to 242 cm in total length in the wild (Haacke & Grove, 1995; Enge et al., 2004). Although Nile monitors are considered widespread in sub-Saharan Africa, they remain poorly studied throughout their range. They are opportunistic carnivorous scavengers and predators, capable of reaching large densities, which likely have a considerable impact on ecosystems (Western, 1974; Engeman et al., 2011; Savarie et al., 2011; Dalhuijsen et al., 2014). Local loss of monitors may increase the prevalence of carrion or cause an increase in competing scavengers (Cairncross et al., 2024). However, some become over abundant because of anthropogenic subsidies, resulting in conflict with people and pets (Bhattacharya & Kock, 2018; Rifie et al., 2023). Some research even suggests pest control is needed in parts of Indonesia and Thailand through harvesting for trade in urban areas, active persecution or translocation to sanctuaries (Wongtienchai et al., 2020; Yudha et al., 2022; Rifie et al., 2023). Abundance may also impact prey species, especially when there is an overwhelming preference for them, like the ringtailed possum (*Pseudocheirus peregrinus*) for lace monitors (*V. varius;* Jessop et al., 2010).

As a semi-aquatic species, Nile monitors generally use habitats near water, and thus, their performance and physiology depend on the quality of both aquatic and riparian habitats, much like *V. salvator* (Karunarathna et al., 2017). In India, the semi-aquatic *V. salvator* is even

praised in parts of its range for cleaning rivers of poultry carcasses and predating on locally feared snakes (Bhattacharya & Koch, 2018). Similar beliefs were recorded in Benin, where Nile monitors are considered sacred for their predation of snakes and rodents (Bio Oure et al., 2016). Adult Nile monitors also feed largely on carrion, which can help limit disease spread and clean freshwater systems (Dalhuijsen et al., 2014). They have been used as a natural tool to control Egyptian goose (*Alopochena egyptiaca*) populations on some golf courses in South Africa and contribute to rodent control in rural and urban environments (various pers. comm.). Several thousand tons of hazardous chemicals, such as pesticides that can no longer be used for their intended purpose, are stockpiled in Africa, often improperly and represent a hazard to human health and the environment (Berny et al., 2006). Nile monitors have been identified as a good sentinel species for these obsolete pesticide stockpiles in sub-Saharan wetlands (Berny et al., 2006; Ciliberti et al., 2006, 2011). They can, therefore, be an important tool for monitoring environmental contamination, the bioavailability of pollutants in food webs, and, thus, potential threats to local human populations and the environment (Ciliberti et al., 2006, 2011).

Although they are globally classified as least concern by the International Union for Conservation of Nature (IUCN) and nationally by the Atlas and Red List of Reptiles of South Africa, Lesotho and Swaziland, varanids are threatened by habitat destruction and heavy exploitation throughout their range for leather, bushmeat, the pet trade and traditional medicines (De Buffrénil & Castanet 2000; Bennett, 2002; De Buffrénil & Hémery, 2002; Bates et al., 2014; Wilms et al., 2021). The trade in Nile monitor skins has declined, even halving in parts of its range (De Buffrénil & Castanet 2000). However, it remains at a high level, and exploitation is still a significant concern because of the lack of detailed scientific data on the structure and dynamics of their populations (De Buffrénil & Castanet, 2000). According to CITES trade data, over 300,000 live Nile monitors were exported between 1975 and 2005 from

46 countries, making it the second most exported varanid globally and accounting for 23 % of the global monitor trade (Pernetta, 2009a,b).

In Sahelian regions of Africa, harvest estimates accounting for damaged skins and undocumented trade have reached 2 million in peak years (Dowell et al., 2015). This exploitation is so intensive that they are likely to go extinct in parts of their range unless it is successfully managed alongside climate-change-induced loss of suitable habitat (Ejigu & Tassie, 2020). Two decades ago, a study on Nile monitors in Sahelian Africa investigating the impact of exploitation found that mean adult snout-vent length, mean age, and longevity were reduced in most exploited populations, and there was a pattern of initial increased somatic growth then but steeply declining after 54 months (De Buffrénil & Hémery, 2002). Reduction in size class and density of large animals through selective harvesting is likely to lower the level of intraspecific competition as the diet of adult Nile monitors includes the same prey species as juveniles, thus increasing the food availability and growth rate of juveniles (De Buffrénil & Hémery, 2002). The extent of illegal hunting in South Africa is unknown. However, despite being protected by law, they are common in traditional medicine *muthi* markets (Whiting et al., 2011). A survey of the Faraday Market in Johannesburg found that 59% of traders had Nile monitor products for sale amongst many other regionally protected species (Whiting et al., 2011).

Most varanids, such as Nile monitors, are generalist opportunistic feeders (Traeholt, 1994; Guarino, 2001; Dalhuijsen et al., 2014). This facilitates their exploitation of anthropogenically modified landscapes such as farmlands and urban areas. Some varanid species, such as the ecologically similar *V. salvator* are known to take advantage of human food waste and significantly increase in abundance (Uyeda, 2009). The abundance of *V. salvator* in areas with anthropogenic food waste and wild areas of Tinjil Island, Indonesia, differed, with 4 individuals/km^2 in wild areas and 1400 individuals/km^2 in areas inhabited by

humans (Uyeda, 2009; Uyeda et al., 2013, 2015). This abundance is of potential concern in Malaysia as it clumps monitors near turtle nesting sites, which may result in a higher rate of predation on turtle eggs (Rusli et al., 2020; Lei et al., 2020). There is, therefore, a need for better waste management in this area.

Urban lace monitors (*V. varius*) were bigger and bolder than their natural habitat counterparts (Pettit et al., 2021). There have been no such morphometric comparisons of urban and rural Nile monitors. Because they prey on some small domestic animals, including chickens (*Gallus gallus domesticus*) (unpublished data), some landowners see them as vermin, which gives rise to human-wildlife conflict. Many myths and folklore also result in negative perceptions and sometimes persecution (unpublished data).

Multiple varanid species are successful urban and farmland exploiters (Traeholt, 1994, 1995, 1997; Uyeda et al., 2015). Anthropogenic activities alter some varanids' home range establishment, demographics and phenotypes (Jessop et al., 2012; Guerrero-Sanchez et al., 2021). Adapting to anthropogenic land can buffer the impacts of habitat loss and fragmentation on wild populations and generally provide access to abundant food. However, it comes with threats, such as a potential increase in harvesting, conflict, and previously unencountered risks, such as road collisions. It also increases the risk of zoonotic diseases and parasite spread (Mendoza-Roldan et al., 2020; Yudhana et al., 2021). Urban exploitation and tolerance to human proximity may be critical to varanid population maintenance as human population size and habitat loss increase (Uyeda, 2009).

Globally, urban lizards tend to be little researched (Brum et al., 2022). In the present study, I investigated Nile monitors' ecology, morphometrics, behaviour and habitat use to determine if they vary in urban and rural areas of KwaZulu-Natal, South Africa. The Nile monitor is an excellent model species for investigating varanid phenotypic plasticity, behaviour, and ecology with anthropogenic changing land use. I tested various predictions

related to this. Their phenotypic plasticity is already evident through their successful status in Florida; however, little is known about how they have adapted to urbanisation (Enge et al., 2004; Dowell et al., 2016; Briggs et al., 2022).

1.2 Problem statement

Nile monitors are the largest lizard in Africa, widespread and one of the most exploited tetrapods in Africa for bushmeat, leather, the pet trade and traditional medicines (Buffrénil & Hémery, 2002). They are projected to become threatened in parts of their range without successful management (Ejigu & Tassie, 2020). However, they are greatly understudied and much of their ecology remains unknown to science. There is a crucial need to fill knowledge gaps for successful research-based management.

Most varanids, such as *V. niloticus* are generalist opportunistic feeders which facilitates their urban exploitation (Traeholt, 1994; Guarino, 2001; Jessop et al., 2012; Dalhuijsen et al., 2014; Yu et al., 2021; Guerrero-Sanchez et al., 2022, 2023). Urban exploitation may be key to some varanid population persistence (Uyeda, 2009). The use of altered habitats and associated differences in both food abundance and quality can alter species behaviour, morphology, parasite load and population dynamics (Jessop et al., 2012; Guerrero-Sanchez et al., 2023). Understanding Nile monitors' urban and rural ecology in KwaZulu-Natal will produce knowledge of their basic ecology and plasticity in response to rapidly changing landscapes through the Anthropocene while giving much needed management recommendations (Ariefiandy et al., 2021).

As the Nile monitor faces significant risk in the form of over-exploitation and habitat loss, a better understanding of its ecology is needed. As documented above, there is a critical need for herpetofauna conservation and research gaps to be filled to make crucial research-based management decisions (Tolley et al., 2016, 2019). Varanid biology is poorly understood

in general (Mendyk, 2018). Most African reptiles lack biological and ecological data (Dalhuijsen et al., 2014). Little is known about the presence, abundance, behaviour, morphometrics and habitat use of Nile monitors in urban and farmland areas of South Africa and much more research is needed to make informed management decisions for this species.

1.3 Aims and objectives

The study aimed to determine if Nile monitors are successful exploiters of human-modified landscapes in KwaZulu-Natal Province, South Africa. I aimed to fill knowledge gaps in various aspects of their ecology, including spatial ecology and their use of urban mosaic landscapes, to make management recommendations. The objectives were:

Objective 1. To assess the habitat use and distribution of Nile monitors (*Varanus niloticus*) in a mosaic of land use types and human population densities in KwaZulu-Natal, South Africa

It was aimed to better understand the habitat use of Nile monitors throughout KwaZulu-Natal by comparing sighting records over a gradient of anthropogenic disturbance. I identified which factors affect their persistence in different land use types. Whether they modify their ecology, phenology, behaviour and habitat use in response to increasingly anthropogenically dense areas. It was determined if Nile monitors are affected by built-up cover and other land-cover types as well as their distance to water, roads and protected areas. The percentage cover of various habitat features around encounters of Nile monitors was mapped to infer their habitat use. The relative selection strength of land cover types, namely anthropogenic, barren, wooded, grassland and water, was calculated. This allowed us to compare preferences in land cover between sites. It was predicted that Nile monitors are associated with habitats close to water and successfully persist in mosaic urban landscapes.

Objective 2. To assess the population dynamics and morphometrics of Nile monitors along a gradient of urbanisation in KwaZulu-Natal, South Africa

Many lizards differ in morphology, mass and/or behaviour in response to urbanisation. Lace monitors (*V. varius*) in Australia are known to be more abundant, larger and bolder in anthropogenically modified habitats because of increased food availability (Pettit et al., 2021). However, high levels of human exploitation, pollution and prevalent risk of dogs in KwaZulu-Natal are likely to engender more complex results. I identified whether Nile monitors differ in morphology in urban areas and whether population dynamics differ across an urbanisation gradient. I also investigated human-Nile monitor interactions. It was hypothesised that Nile monitors' population dynamics, behaviour, and morphology vary in response to changing anthropogenic activities and human population density in KwaZulu-Natal. It was also predicted that despite persisting in urban areas, rural Nile monitor populations are greater with larger individuals than their urban counterparts. Consequently, Nile monitors response to changes in land use in terms of population size, and morphology were determined, and negative human/domestic animal – Nile monitor interactions were documented. Mark-resight methods were used for population estimates, and detailed morphological measurements were obtained by catching monitors using baited step plate traps and collaborating with reptile rescuers throughout KwaZulu-Natal. This allowed the determination of whether Nile monitors were successful exploiters or merely persisting in urban mosaic landscapes.

1.4 Structure of the thesis

The main body of this thesis was organised with the data chapters presented as draft manuscripts prepared for publication in peer-reviewed journals. The first chapter (Chapter 1) was the Introduction, which provides a literature review of the concepts covered in this study and the background literature. The next two chapters (Chapters 2 and 3) are empirical, each

covering a specific objective. Each chapter is formatted according to the journal it is intended to be submitted to. Because of this thesis format, a certain degree of repetition was unavoidable, especially in the methods section. However, this is deemed to be of little concern as this format allows the reader to read each chapter separately without losing the overall context of the thesis. Chapter 2 investigated the habitat use and distribution of Nile monitors (*V. niloticus*) in a mosaic of land use type and human population density in KwaZulu-Natal Province. Chapter 3 investigated the population dynamics and morphometrics of Nile monitors (*Varanus niloticus*) along a gradient of urbanisation in KwaZulu-Natal Province. The thesis ends with a summary chapter of synthesis, conclusive remarks, an emphasis on management recommendations, and proposed future research.

1.5 References

Ariefiandy, A., Purwandana, D., Azmi, M., Nasu, S.A., Mardani, J., Ciofi, C. and Jessop, T.S. 2021. Human activities associated with reduced Komodo dragon habitat use and range loss on Flores, Biodiversity and Conservation 30: 461–479. https://doi.org/10.1007/s10531-020-02100-8.

Auffenberg, W. 1988. Gray's Monitor Lizard. University of Florida Press, Florida.

Bates, M.F., Branch, W.R., Bauer, A.M., Burger, M., Marais, J., Alexander, G.J. and de Villiers, M.S. 2014. Atlas and Red List of Reptiles of South Africa, Lesotho and Swaziland. Animal Demography Unit (Cape Town) and the South African National Biodiversity Institute (Pretoria).

Bennett, D. 2002. Diet of juvenile *Varanus niloticus* (Sauria: Varanidae) on the Black Volta River in Ghana. Journal of Herpetology 36: 116–117. https://doi.org/10.1670/0022-1511(2002)036[0116:DOJVNS]2.0.CO;2

Berny, P.J., de Buffrénil, V. and Hémery, G. 2006. Use of the Nile monitor, *Varanus niloticus* L (Reptilia: Varanidae), as a bioindicator of organochlorine pollution in African Wetlands. Bulletin of Environmental Contamination and Toxicology 77: 359–366. https://doi.org/10.1007/s00128-006-1074-z.

Bhattacharya, S. and Koch, A. 2018. Effects of traditional beliefs leading to conservation of water monitor lizards (*Varanus salvator*) and threatened Marshlands in West Bengal, India. Herpetological Conservation and Biology 13: 408–414.

Bio Oure, R., Daouda, I.-H., Valentin, K. and Mensah, G. 2016. Inventaire, structure morpho-métrique et importance des varans sacrés de Kandi (Nord-Est Bénin). *International* Journal of Biological and Chemical Sciences 9: 2663. https://doi.org/10.4314/ijbcs.v9i6.12

Briggs-Gonzalez, V., Evans, P., Klovanish, C. and Mazzotti, F.J. 2022. A species bioprofile for the Asian Water Monitor (*Varanus salvator*). Southeastern Naturalist 21: 187–210. https://doi.org/10.1656/058.021.0302

Brum, P.H.R., Gonçalves, S.R.A., Strüssmann, C. and Teixido, A.L. 2022. A global assessment of research on urban ecology of reptiles: patterns, gaps and future directions. Animal Conservation 26: 1-13. https://doi.org/10.1111/acv.12799

Cairncross, R.J., Spencer, E.E., Meisuria, N., Crowther, M.S. and Newsome, T.M. 2024. Carrion use by a reptile is influenced by season, habitat and competition with an apex mammalian scavenger. Ecology and Evolution 14(8): e70211. https://doi.org/10.1002/ece3.70211.

Ceballos, G., Ehrlich, P.R., Barnosky, A.D., García, A., Pringle, R.M. and Palmer, T.M. 2015. Accelerated modern human–induced species losses: Entering the sixth mass extinction, Science Advances 1: e1400253. https://doi.org/10.1126/sciadv.1400253.

Ciliberti, A. 2006. The Nile monitor (*Varanus niloticus*), an indicator species for pollution in sub-Saharan wetlands. Bulletin of Environmental Contamination and Toxicology 77: 359–366. https://doi.org/10.1007/s00128-006-1074-z

Ciliberti, A. 2011. The Nile monitor (*Varanus niloticus*; Squamata: Varanidae) as a sentinel species for lead and cadmium contamination in sub-Saharan wetlands. Science of The Total Environment 409: 4735-4745

Cobbinah, P.B., Erdiaw-Kwasie, M.O. and Amoateng, P. 2015. Africa's urbanisation: Implications for sustainable development. Cities 47: 62–72. https://doi.org/10.1016/j.cities.2015.03.013.

Cox, N., Young, B.E., Bowles, P. et al. 2022. A global reptile assessment highlights shared conservation needs of tetrapods. Nature 605: 285–290. https://doi.org/10.1038/s41586-022-04664-7

Dalhuijsen, K., Alexander, G.J., Branch, W.R. 2014. A comparative analysis of the diets of *Varanus albigularis* and *Varanus niloticus* in South Africa. African Zoology 49: 83-93.

de Buffrénil, V., Castanet, J. 2000. Age estimation by skeletochronology in the Nile monitor (*Varanus niloticus*), a highly exploited species. Journal of Herpetology 34: 414–424. https://doi.org/10.2307/1565365

de Buffrénil, V., Hémery, G. 2002. Variation in longevity, growth, and morphology in exploited Nile monitors (*Varanus niloticus*) from Sahelian Africa. Journal of Herpetology 36: 419-426. https://doi.org/10.1670/0022-1511(2002)036[0419:VILGAM]2.0.CO;2

Dowell, S.A., de Buffrénil, V., Kolokotronis, S.-O. and Hekkala, E.R. 2015. Fine-scale genetic analysis of the exploited Nile monitor (*Varanus niloticus*) in Sahelian Africa. BMC Genetics 16: 32. https://doi.org/10.1186/s12863-015-0188-x.

Dowell, S.A., Wood, J.P., Campbell, T.S., Kolokotronis, S.-O. and Hekkala, E.R. 2016. Combining genetic and distributional approaches to sourcing introduced species: a case study on the Nile monitor (*Varanus niloticus*) in Florida. Royal Society Open Science 3: 150619. https://doi.org/10.1098/rsos.150619.

Downs, C.T., Alexander, J., Brown, M., et al. 2021. Modification of the third phase in the framework for vertebrate species persistence in connected urban environments based on a review of case studies from KwaZulu-Natal Province, South Africa. Ambio 50: 1866–1878. https://doi.org/10.1007/s13280-021-01501-5

Ejigu, D., Tassie, N. 2020. Present and future suitability of the Lake Tana Biosphere Reserve in Ethiopia for the Nile monitor (*Varanus niloticus*) using the MaxEnt model. Environmental Systems Research 9: 31. https://doi.org/10.1186/s40068-020-00197-y

Enge, K.M., Krysko, K.L. and Talley, B.L. 2004. Distribution and ecology of the introduced African rainbow lizard, *Agama agama africana* (Sauria: Agamidae), in Florida. Florida Scientist 67: 303–310. http://www.jstor.org/stable/24321176

Engeman, R., Jacobson, E., Avery, M. and Meshaka, W.E. 2011. The aggressive invasion of exotic reptiles in Florida with a focus on prominent species: A review. Current Zoology 57: 599–612. https://doi.org/10.1093/czoolo/57.5.599.

French, S.S., Webb, A.C., Hudson, S.B. and Virgin, E.E. 2018. Town and country reptiles: A review of reptilian responses to urbanization. Integrative and Comparative Biology 58(5): 948–966. https://doi.org/10.1093/icb/icy052.

Galdino, C., Horta, G. and Young, R. 2014. An update to a bead-tagging method for marking lizards. Herpetological Review 45: 587.

Green, B. and King, D. 1978. Home range and activity patterns of the sand goanna, *Varanus gouldii* (Reptilia: Varanidae). Wildlife Research 5: 417-424.

Gu, D., Andreev, K. and Dupre, M.E. 2021. Major trends in population growth around the World. China CDC Weekly 3: 604–613. https://doi.org/10.46234/ccdcw2021.160.

Guarino, F. 2001. Diet of a large carnivorous lizard, *Varanus varius*. Wildlife Research 28: 627-630. https://doi.org/10.1071/WR01001

Guerrero-Sanchez, S. 2019. Ecology and health of the Asian water monitor lizard (*Varanus salvator*) in the fragmented landscape of the Kinabatangan floodplain, Sabah, Malaysia. PhD Thesis, Cardiff University.

Guerrero-Sanchez, S., Frias, L., Saimin, S., Orozco-terWengel, P. and Goossens, B. 2023. The fast-food effect: costs of being a generalist in a human-dominated landscape. Conservation Physiology 11: coad055. https://doi.org/10.1093/conphys/coad055

Guerrero-Sanchez, S., Majewski, K., Orozco-terWengel, P., Saimin, S., Goossens, B. 2022. The effect of oil palm-dominated landscapes on the home range and distribution of a generalist species, the Asian water monitor. Ecology and Evolution 12: e8531. https://doi.org/10.1002/ece3.8531

Haacke, W.D., Groves, D. 1995. *Varanus niloticus niloticus* Nile Monitor Size. African Herp News 22: 45–46.

Jessop, T., Urlus, J., Lockwood, T. and Gillespie, G. 2010. Preying possum: Assessment of the diet of lace monitors (*Varanus varius*) from coastal forests in Southeastern Victoria. Biawak 4: 59–63.

Jessop, T.S., Smissen, P., Scheelings, F. and Dempster, T. 2012. Demographic and phenotypic effects of human mediated trophic subsidy on a large Australian lizard (*Varanus varius*): Meal ticket or last supper? PLoS One 7: e34069. https://doi.org/10.1371/journal.pone.0034069

Karunarathna, S., Surasinghe, T., Madawala, M., Somaweera, R., Amarasinghe, A.A.T., Karunarathna, S., Surasinghe, T., Madawala, M., Somaweera, R. and Amarasinghe, A.A.T. 2017. Ecological and behavioural traits of the Sri Lankan water monitor (*Varanus salvator*) in an urban landscape of Western Province, Sri Lanka. Marine and Freshwater Research 68: 2242–2252. https://doi.org/10.1071/MF17038.

Law, S.J., De Kort, S.R. and van Weerd, M. 2016. Morphology, activity area, and movement patterns of the frugivorous monitor lizard *Varanus bitatawa*. Herpetological Conservation and Biology 11: 467-475.

Lei, J., Booth, D.T., Rusli, M.U. and Zhang, Z. 2020. Spatial ecology of Asian water monitors adjacent to a sea turtle nesting beach. Zoological Science 38: 1-7. https://doi.org/10.2108/zs200071

Lowry, H., Lill, A. and Wong, B.B.M. 2013. Behavioural responses of wildlife to urban environments, Biological Reviews 88: 537–549. https://doi.org/10.1111/brv.12012.

McDonald, R.I., Mansur, A.V., Ascensão, F., Colbert, M., Crossman, K., Elmqvist, T., Gonzalez, A., Güneralp, B., Haase, D., Hamann, M., Hillel, O., Huang, K., Kahnt, B., Maddox, D., Pacheco, A., Pereira, H.M., Seto, K.C., Simkin, R., Walsh, B., Werner, A.S. and Ziter, C. 2020. Research gaps in knowledge of the impact of urban growth on biodiversity, Nature Sustainability 3: 16–24. https://doi.org/10.1038/s41893-019-0436-6.

Mendoza-Roldan, J.A., Modry, D. and Otranto, D. 2020. Zoonotic parasites of reptiles: A crawling threat. Trends in Parasitology 36: 677–687. https://doi.org/10.1016/j.pt.2020.04.014.

Mendyk, R. 2018. An annotated bibliography of captive reproduction in monitor lizards (Varanidae: *Varanus*). Part III. Soterosaurus. Biawak 12: 54–61.

Moss, J.B., Gerber, G., Laaser, T., Goetz, M., Oyog, T. and Welch, M. 2020. Conditional female strategies influence hatching success in a communally nesting iguana. Ecology and Evolution 10: 3424–3438. https://doi.org/10.1002/ece3.6139

Okeke, D.C. 2021. Prospects for sustainable urban development in Africa – (re)viewed from a planning perspective. International Planning Studies 26: 198–217. https://doi.org/10.1080/13563475.2020.1785278

Pernetta, A.P. 2009a. Redesigning the wildlife trade system. Science 324: 1389. https://doi.org/10.1126/science.324_1389a

Pernetta, A. 2009b. Monitoring the trade: using the CITES trade database to examine the dynamics of the trade in live monitor lizards (*Varanus* spp.). Biawak: Quarterly Journal of Varanid Biology and Husbandry 3: 37–45.

Pettit, L., Brown, G.P., Ward-Fear, G. and Shine, R. 2021. Anthropogenically modified habitats favor bigger and bolder lizards. Ecology and Evolution 11: 1586–1597. https://doi.org/10.1002/ece3.7124

Putman, B.J. and Tippie Z.A. 2020. Big city living: A global meta-analysis reveals positive impact of urbanization on body size in lizards. Frontiers in Ecology and Evolution 8: 580745. https://doi.org/10.3389/fevo.2020.580745

Rifaie, F., Yafi, K.M., Maireda, N.L. and Arida, E. 2023. Human-water monitor conflicts in Indonesia: Spatial patterns and mitigation alternatives. Journal of Tropical Ethnobiology 6: 1–15. https://doi.org/10.46359/jte.v6i1.160.

Rusli, M.U., Chen, G-N., Booth, D.T. and Lei, J. 2020. Diet preference and activity of Asian water monitor at Chagar Hutang turtle sanctuary. Journal of Sustainability Science and Management 15: 61–67.

Savarie, P., Engeman, R., Mauldin, R., Mathies, T. and Tope, K. 2011. Tools for managing invasions: acceptance of non-toxic baits by juvenile Nile monitor lizards and Burmese pythons under laboratory conditions. International Journal of Pest Management 57: 309–314. https://doi.org/10.1080/09670874.2011.598581.

Singh, N., Price, C. and Downs, C.T. 2021. Aspects of the ecology and behaviour of a potential urban exploiter, the southern tree agama, *Acanthocercus atricollis*. Urban Ecosystems 24: 905–914. https://doi.org/10.1007/s11252-020-01078-z

Stanner, M. and Mendelssohn, H. 1987. Sex ratio, population density and home range of the desert monitor (*Varanus griseus*) in the Southern Coastal Plain of Israel, Amphibia-Reptilia 8: 153-163. https://doi.org/10.1163/156853887X00414

Taylor-Brown, A., Booth, R., Gillett, A., Mealy, E., Ogbourne, S.M., Polkinghorne, A. and Conroy, G.C. 2019. The impact of human activities on Australian wildlife. PLoS One 14: e0206958. https://doi.org/10.1371/journal.pone.0206958.

Tolley, K., Alexander, G., Branch, W., Bowles, P. and Maritz, B. 2016. Conservation status and threats for African reptiles. Biological Conservation 204: 63-71. https://doi.org/10.1016/j.biocon.2016.04.006.

Tolley, K.A., Weeber, J., Maritz, B., Verburgt, L., Bates, M.F., Conradie, W., Hofmeyr, M.D., Turner, A.A., da Silva, J.M. and Alexander, G.J. 2019. No safe haven: Protection levels show imperilled South African reptiles not sufficiently safe-guarded despite low average extinction risk. Biological Conservation 233: 61–72. https://doi.org/10.1016/j.biocon.2019.02.006.

Traeholt, C. 1994. The food and feeding behaviour of water monitor, *Varanus salvator*, in Malaysia. Malayan Nature Journal 47: 331-343.

Traeholt, C. 1995. A radio-telemetric study of the thermoregulation of free living water monitor lizards, *Varanus s. salvator*. Journal of Comparative Physiology B 165: 125–131. https://doi.org/10.1007/BF00301476

Traeholt, C. 1997. Activity patterns of free-living water monitor lizards *Varanus salvator*. Malayan Nature Journal 50: 301-315.

Uyeda, L. 2009. Garbage appeal: relative abundance of water monitor lizards (*Varanus salvator*) correlates with presence of human food leftovers on Tinjil Island, Indonesia. Biawak 3: 9–17.

Uyeda, L., Iskandar, E., Kyes, R. and Wirsing, A.J. 2012. Proposed research on home ranges and resource use of the water monitor lizard, *Varanus salvator*. Forestry Chronicle 88: 542-546. https://doi.org/10.5558/tfc2012-103

Uyeda, L.T., Iskandar, E., Kyes, R.C. and Wirsing, A.J. 2015. Encounter rates, agonistic interactions, and social hierarchy among garbage feeding water monitor lizards (*Varanus salvator bivittatus*) on Tinjil Island, Indonesia. Herpetological Conservation and Biology 10: 753–764.

Uyeda, L.T., Iskandar, E., Wirsing, A. and Kyes, R. 2013. Nocturnal activity of *Varanus salvator* on Tinjil Island, Indonesia. Biawak 7:25–30.

Whiting, M., Williams, V., Hibbitts, T. 2011. Animals traded for traditional medicine at the Faraday Market in South Africa: Species diversity and conservation implications. Journal of Zoology 284:84–96. https://doi.org/10.1007/978-3-642-29026-8_19

Wolfe, A.K., Bateman, P.W. and Fleming, P.A. 2018. Does urbanization influence the diet of a large snake? Current Zoology 64(3): 311–318. https://doi.org/10.1093/cz/zox039.

Wongtienchai, P., Lapbenjakul, S., Jangtarwan, K., Areesirisuk, P., Mahaprom, R., Subpayakom, N., Singchat, W., Sillapaprayoon, S., Muangmai, N., Songchan, R., Baicharoen, S., Duengkae, P., Peyachoknagul, S. and Srikulnath, K. 2020. Genetic management of a water monitor lizard (*Varanus salvator macromaculatus*) population at Bang Kachao Peninsula as a consequence of urbanization with Varanus Farm Kamphaeng Saen as the first captive research establishment. Journal of Zoological Systematics and Evolutionary Research 59: 1–14. https://doi.org/10.1111/jzs.12436.

Yu, X., Zanudin, A.B.T.M., Rusli, M.U., Booth, D.T. and Lei, J. 2021 Diet reflects opportunistic feeding habit of the Asian water monitor (*Varanus salvator*). Animal Biology 72: 27–37.

Yudha, A., Kusrini, M. and Arida, E. 2022 Chasing for water monitors using dogs in West Java, Indonesia: a recreational hunting or pest control? Ethnobiology and Conservation 11: 04. https://doi.org/10.15451/ec2022-01-11.04-1-10

Yudhana, A., Praja, R.N. and Kartikasari, A.M. 2021. Sparganosis (*Spirometra* spp.) in Asian Water Monitor (*Varanus salvator*): A medical implications for veterinarians, breeders, and consumers. Veterinary World 14: 2482-2487. https://doi.org/10.14202/vetworld.2021.2482-2487

CHAPTER 2

Habitat use and distribution of Nile monitors (*Varanus niloticus*) in a mosaic of land use types and human population densities in KwaZulu-Natal, South Africa

Euan Genevier[1], Cormac Price[1,2], Nick Evans[1,3] and Colleen Downs[1*]

[1]*Centre for Functional Biodiversity, School of Life Sciences, P/Bag X01, Scottsville, Pietermaritzburg, 3209, South Africa.*

[2] *The HerpHealth lab. Unit for Environmental Sciences and Management, North-West University, Potchefstroom, South Africa 2531*

[3] *KZN Amphibian and Reptile Conservation.*

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*** Corresponding author:** Colleen T. Downs

Email: downs@ukzn.ac.za; ORCID: http://orcid.org/0000-0001-8334-1510

Other emails and ORCIDs:

Euan Genevier ORCID: https://orcid.org/0009-0006-4457-3195

Cormac Price Email: [redacted]; ORCID: https://orcid.org/0000-0002-9092-0796

Nick Evans Email: n[redacted]

Running header: Habitat use and distribution of Nile monitors

2.1 Abstract

Context Globally, with the continuous increase in human populations, unprecedented changes in land use are taking place. Little is known about how African reptiles respond to this rapid land use change.

Objectives We used sightings to establish if Nile monitors (*Varanus niloticus*) are persisting in a mosaic of human population densities and land use types in KwaZulu-Natal Province, South Africa.

Methods We collected sighting data throughout KwaZulu-Natal between April 2022 and July 2023, using three main methods, namely urban sightings obtained from reptile relocation groups and collaborations with reptile relocators; sightings obtained from personal encounters and citizen scientist observations using social media, posters, newspaper articles and interviews; and sightings from iNaturalist observations.

Results We obtained and used 419 sightings and then identified two hotspot areas in the province on which we chose to focus. Site A was dominated by natural habitat, and anthropogenic land use types dominated site B. We found a significant relative selection strength for most tested land covers over anthropogenic cover in site A, whereas most were insignificant in site B. Our results suggest that the use of anthropogenic land use types by Nile monitors in KwaZulu-Natal may be driven by habitat transformation and loss. Nile monitors were found persisting in urban mosaic landscapes and, when compared with available cover, were found in increasingly wooded (natural and urban vegetation) areas with increased human populations. Furthermore, they used small anthropogenic water bodies (e.g., swimming pools and storm drains), making them less reliant on rivers in urban environments. Their population demographics were similar in low and high human population density, but there appeared to be a skew towards smaller and younger individuals in medium human population density areas. Reproduction was occurring successfully in urban environments, but the detection of larger

monitors by potential wild and domestic predators and negative human interactions poses risks. Our study outlined the importance of managed and unmanaged green spaces (e.g., gardens, parks, golf courses) in urban mosaic landscapes for wildlife persistence, specifically Nile monitors, and supporting human well-being as well as reptile conservation. There is an overall need for more research on this highly exploited yet understudied reptilian urban adapter.

Keywords: *Varanus*; Urban ecology; Habitat use; Citizen science; Anthropogenic disturbance

2.2 Introduction

Globally, with the continuous increase in human population, there is unprecedented changing anthropogenic land use (Ellis and Ramankutti 2008; Seto et al. 2012; Thaker et al. 2022). Human population growth is exponential in Africa, which is significantly increasing the need for resources, space, raw materials, employment, and other basic requirements (Brum et al. 2022). It is expected that 1.3 billion people will live in cities in Africa by 2050 (Cobbinah et al. 2015). Providing space and food for people is likely to result in continued rapid change in land use. Rapid unplanned urban development is a major threat to sustainable development in Africa (Cobbinah et al. 2015; Okeke 2021).

Urbanisation is an extreme form of land transformation which typically reduces ecological richness and biodiversity (Streicher et al. 2020; Downs et al. 2021). However, some species persist or even thrive in anthropogenic environments (Singh et al. 2021; Downs et al. 2021). The conservation of persisting wildlife is key to sustainable anthropogenic development. However, little is documented about the response of African wildlife to urbanisation and agriculture (Downs et al. 2021). There is a geographical and taxonomic bias in our knowledge of urban ecology in Africa, specifically African reptiles (Tolley et al. 2016; Brum et al. 2022; Woolley et al. 2023).

Habitat use is essential to understanding species and habitat conservation practices and priorities (de Gabriel Hernando et al. 2021). It can provide a baseline insight into their behaviour, habitat requirements, and ecological and behavioural plasticity to certain factors, especially in urban or rural mosaic landscapes (Reaney and Whiting 2003; Karunarathna et al. 2017; Downs et al. 2021). Green spaces (natural or managed) in urban areas can help offset urbanisation, maintain biodiversity and ecosystem services, as well as improve the well-being and enrich people (Markovchick-Nicholls et al. 2008; Downs et al. 2021; Hursh et al. 2023). However, these goals can be conflicting if a species is negatively associated with human activity (Markovchick-Nicholls et al. 2008). Populations adapted to anthropogenic land use are at risk of ecological traps, where species have adapted to conditions that may not remain favourable in the long run (Markovchick-Nicholls et al. 2008; Guerrero-Sanchez et al. 2022). There is also a potential risk of parasite transfer and the spread of zoonotic disease (Nebdiza-Roldan 2020; Mlangeni et al. 2024). A broad understanding of the habitat requirements for wildlife in urban areas is essential for sustainable urban planning and development (Hursh et al. 2023). Habitat use and distribution patterns can also help mitigate human-wildlife conflicts (Souza et al. 2018; Warrier et al. 2021).

Citizen scientists may help provide much-needed data with limited resources for understudied species or in urban mosaic landscapes (Josiah and Downs 2023). The inclusion of iNaturalist records broadens opportunistic datasets and can greatly increase sample sizes (Mo and Mo 2022; Forti and Szabo 2023). Citizen science programs are effective starting bases for exchange in sustainable city development between scientists, policymakers, and citizens (Kondratyeva et al. 2020). There are associated limitations in using platforms like iNaturalist, such as spatial and temporal sampling bias (Di Cecco et al. 2021; Geurts et al. 2023). However, these limitations can be partially mitigated by using such platforms as data supplementation to structured surveys, where the different sampling methods can be compared (Roberts et al. 2022;

Shin et al. 2022). These records should be analysed with caution regardless, but their value in biological research remains considerable (Roberts et al. 2022; Shin et al. 2022).

The Nile monitor (*Varanus niloticus*, Linnaeus 1758) is the longest lizard in Africa, reaching up to 242 cm in total length (Haacke and Grove 1995; Enge et al. 2004). It is listed as Least Concern both globally and regionally in southern Africa (Bates et al. 2014; Wilms 2021). However, it is among the most heavily exploited tetrapods in Africa (de Buffrénil and Rimblot-Baly 1999; de Buffrénil and Castanet 2000; de Buffrénil and Hémery 2002; Dowell et al. 2015). It is also projected to become threatened in parts of its range through extensive climate-change-induced loss in habitat suitability (Ejigu and Tassie 2020). The extent of exploitation in South Africa is unknown as most research on the topic comes from Sahelian Africa (e.g., de Buffrénil and Rimblot-Baly 1999; de Buffrénil and Castanet 2000; de Buffrénil *et al.* 2002; Dowell et al. 2015). The Nile monitor is protected by law in South Africa, and few live animals have been legally exported from the region between 1975 and 2005, according to CITES data (Pernetta 2009). However, local exploitation occurs for traditional medicine and bushmeat, with various body parts, skins and oils available in traditional medicine markets (Alexander 1990; Whiting et al. 2011). In a survey of traded animals at the Faraday muthi market in Johannesburg, 84.4% of traders disclosed having monitor lizard (*V. albigularis albigularis* and *V. niloticus*) products. More specifically, 59% of traders disclosed having Nile monitor products for sale (Whiting et al. 2011). Therefore, despite little evidence of international trade from South Africa, the local exploitation of varanids is evident despite their protected status.

Home range size is unknown for Nile monitors. Other varanid species home range estimates vary from 0.1 ha to 3391 ha and differ significantly between species, sex, habitat type and geographical location (Table 2.1; Weavers 1993; Thompson 1994; Guarino 2002; Ciofi et al. 2007; Guerrero-Sanchez et al. 2021, 2023). A broad positive correlation exists between animal size and its estimated home range (Ofstad et al. 2016; Ariano-Sánchez et al. 2020). The

relationship between body mass and home range size in varanids is linear (Guarino 2002). Being the largest varanid in Africa, Nile monitors likely have a relatively large home range size. However, there have been no detailed studies on Nile monitor habitat use or home range in Africa.

Here, we investigated three main components of Nile monitors' occurrence in the anthropogenically modified mosaic landscapes of KwaZulu-Natal Province, South Africa. Firstly, we investigated if their distribution and persistence are affected by human population densities. Secondly, we assessed if their habitat use differed with increased human density. Thirdly, we evaluated if body size estimates differed with increased human population densities. We predicted that Nile monitors persist in both modified mosaic and unmodified landscapes, are strongly associated with waterbodies, use artificial structures and artificial waterbodies, and urban vegetation in urban areas, and that estimated sizes of individuals decrease with increased human population density.

Table 2.1: Varanid home range or activity area estimates from previous studies with their associated tracking method, land-use type, region, sample size and reference. (Note: F = female; M = male; J = juvenile, HR = home range. * Values based on means).

Species	Min HR estimate (ha)	Max HR estimate (ha)	Method	Land-use	Region	Sample size	Reference
V. rosenbergi	7.8	7.8	Radio telemetry	n/a	Southern Australia	n/a	Green and King 1978
V. gouldii	19.44	24.83	Radio telemetry	n/a	Southern Australia	n/a	Green and King 1978
V. griseus	13.7 (F) ; 83.5 (M)	60.7(F) ; 115.9 (M)	Radio telemetry	Urban	Israel	9 (4 M; 5 F)	Stanner and Mendelssohn 1987
V. olivaceus	1.48	1.48	Radio telemetry	n/a	Philippines	12	Auffenberg 1988
V. bengalensis	4.4 (F)	5.3 (M)	Radio telemetry	n/a	Southern Pakistan	n/a	Auffenberg et al. 1991
V. varius	13 (M)	127 (M)	Radio telemetry	n/a	Southeastern Australia	12	Weavers 1993
V. gouldii	8	91	Radio telemetry	Urban	Western Australia	10	Thompson 1994
V. tritiris	3.7 (F)* ; 71.4 (M)*	9.3 (F)* ; 40.4 (M)*	Radio telemetry	Reserve	Southern Australia	11 (5 M; 6 F)	Thompson et al. 1999
V. albigularis	15.9 (M) ; 5.5 (F) ; 12 (J)	20.7 (M) ; 6.7 (F)	Radio telemetry	Reserve	Northern Namibia	31	Phillips 1995
V. griseus	7.5	22.8	Radio telemetry	Reserve	Egypt	4 (2 M; 2 F)	Ibrahim 2002
V. glauerti	1.25	7.36	Radio telemetry	Reserve	Northern Australia	6 (5 M ; 1 F)	Sweet 1999
V. glebopalma	3.5	7.76	Radio telemetry	Reserve	Northern Australia	9 (5 M ; 4 F)	Sweet 1999

V. varius	4 (J) ; 10.5 (M)	387 (M)	Radio telemetry	Reserve	Southeastern Australia	23 (2 F ; 19 M ; 2 J)	Guarino 2002
V. komodoensis	170	530	Radio telemetry	Reserve	Komodo and Flores	5 (3 F; 2 M)	Ciofi et al. 2007
V. komodoensis	2	3391	GPS and radio	Reserve	Northeastern Komodo	20 (12 M ; 8 F)	Purwandana et al. 2021
V. indicus	0.9 (F)*	4.7 (M)*	Radio telemetry	Reserve	Northern Australia	11 (6 M ; 5 F)	Smith and Griffiths 2009
V. mertensi	6.3 (F)*	13.6 (M)*	Radio telemetry	Reserve	Northern Australia	13 (8 M ; 5 F)	Smith and Griffiths 2009
V. salvator	0.1	138.9	GPS	Plantation and Forest	Northern Borneo	14	Guerrero-Sanchez et al. 2022
V. varius	0.5	146.7	GPS	Reserve	Southeastern Australia	16 (11 M ; 5 F)	Lei and Booth 2018
V. varius	2.12 (M)	194.86 (M)	GPS	Reserve	Southeastern Australia	9 (M)	Pascoe et al. 2019

2.3 Methods

2.3.1 Study area

The province of KwaZulu-Natal is located on the east coast of South Africa and borders Lesotho, Eswatini, and Mozambique internationally, and the provinces of the Free State, Eastern Cape and Mpumalanga nationally, as well as a coastline of some 600 km along the Indian Ocean (Ndlovu and Demlie 2020, Fig. 2.1b). Its climate is heavily influenced by the Indian Ocean, with high temperatures and rainfall in the austral summer that often rise above 25°C and can exceed 40°C during the day, and cool, dry winters that average ~ 20°C (Fairbanks and Benn 2000; Ndlovu and Demlie 2020). The mean annual rainfall in KwaZulu-Natal ranges from 650 mm in eastern grasslands to 1400 mm in eastern Coastal Bushveld (Kruger and Nxumalo 2017; Ndlovu et al. 2021; Mugiyo et al. 2022). This influence of relatively mild winters and high humidity creates favourable conditions for relatively large reptiles in KwaZulu-Natal, eastern Limpopo and eastern Mpumalanga (Alexander 2007; Bates et al. 2014).

We focussed on the two areas in KwaZulu-Natal where most of our sightings occurred (92%; Fig. 2.1b). The first, a more northern focal area (site A, Fig. 2.1b) was 4900 km^2 in the uMkhanyakude and Zululand District Municipalities, between Ulundi (28°18'21.2"S 31°26'12.2"E), St Lucia (28°23'03.8"S 32°25'29.3"E) and Jozini (27°26'42.6"S 32°03'14.7"E). This area is predominantly Zululand Lowveld vegetation and settlements. It includes protected areas, for example, the biologically important wetlands of iSimangaliso Wetland Park and Hluhluwe-iMfolozi Park (Rowe-Rowe and Taylor 1996; Fairbanks and Benn 2000). The main agricultural production in the area includes livestock (meat and dairy production) and subsistence farming in the arid inland region, and sugar cane (*Saccharum officinarum*) nearer the coast (Waldner et al. 2017). Altitude in this area ranges from 0 to 787 m a. s. l.

The second focal area (site B, Fig. 2.1b) was 8350 km^2 in more southern KwaZulu-Natal, in the eThekwini, Ugu, iLembe and uMgungundlovu District Municipalities, situated between KwaDukuza (29°22'13.0"S 31°21'06.9"E), Nottingham Road (29°20'57.4"S 29°59'18.0"E), and Mtwalume (30°29'23.1"S 30°37'46.9"E). It has inland Sub-Escarpment Savannah and Indian Ocean Coastal Forest biomes, with two major cities, Pietermaritzburg and Durban (McPherson et al. 2016). The dominant agricultural production includes livestock farming (meat and dairy production), sugar cane and exotic timber plantations *Acacia mearnsii*, *Pinus patula* or *Eucalyptus* spp. (Ngcobo et al. 2018). Altitude in site B ranged from 0 to 1540 m a. s. l.

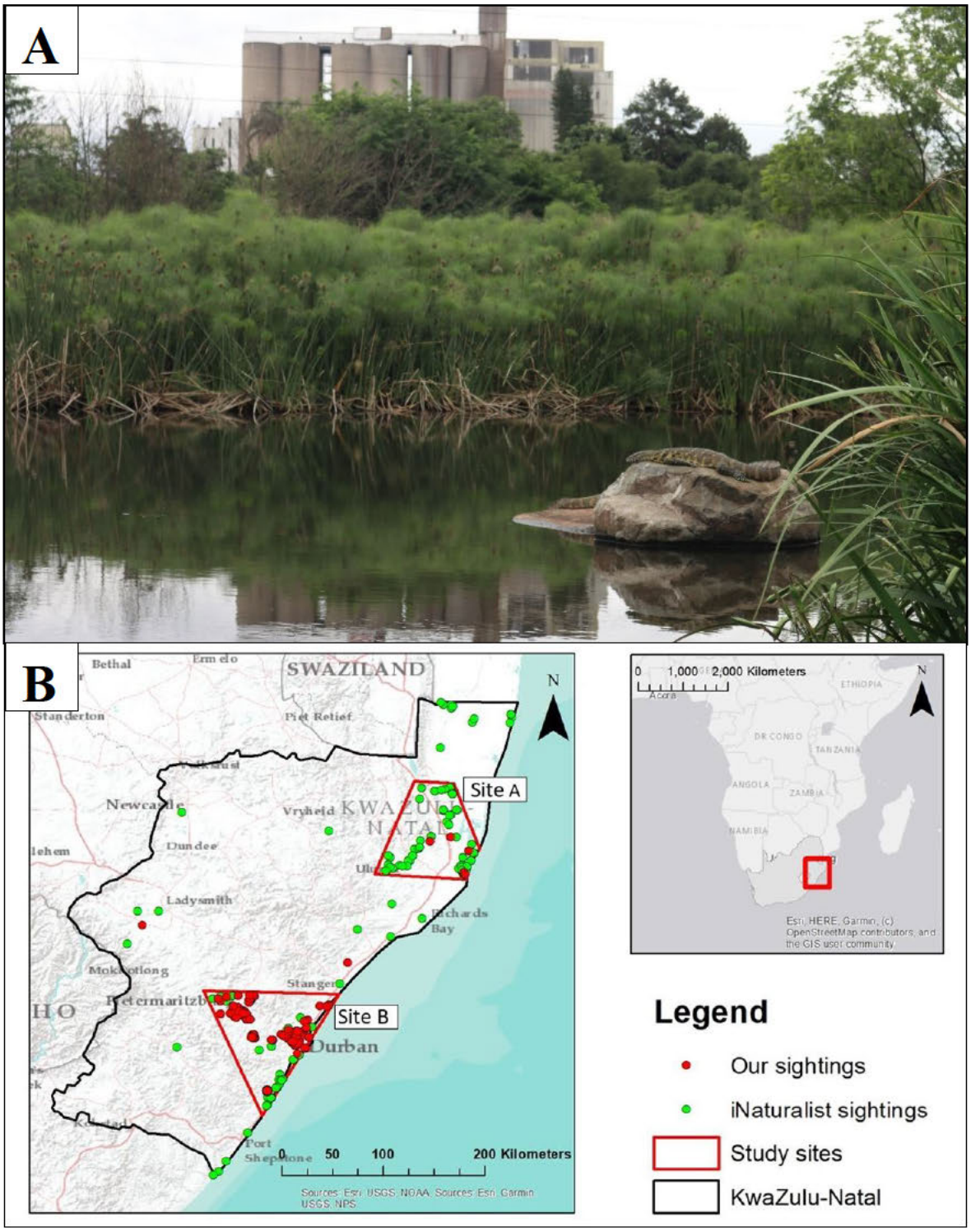

Figure 2.1: A) Photograph of two Nile monitors basking at an urban pond in Pietermaritzburg (E Genevier©), and B) locations of sightings of Nile monitors throughout KwaZulu-Natal, South Africa and our focal areas (sites A and B).

2.3.2 Nile monitor sighting records

We collected Nile monitor sightings throughout KwaZulu-Natal Province (Fig. 2.1b) between April 2022 and July 2023. We used three methods for data collection to maximise effort in acquiring confirmed Nile monitor locations: 1) urban sightings were obtained from several consenting reptile relocation groups and through collaboration with reptile relocators; 2) further sightings were obtained from personal encounters and citizen scientist observations using social media, posters, newspaper articles and interviews; and 3) these sightings were supplemented with iNaturalist observations. These sightings included ones where conflict was observed or reported between humans and domestic animals with Nile monitors. We ensured each sighting was reliable, and it was only included if it was confirmed with photographic evidence by the author and our geographical coordinates were confirmed by the observer. Verifying citizen scientist identification was important as they are commonly confused with Nile crocodiles (*Crocodylus niloticus*) and locally sympatric rock monitors (*Varanus albigularis albigularis*; Dalhuijsen et al. 2014; pers. obs.). Personal sightings were recorded at ≤ 10 m accuracy. Sightings from reptile removal groups were either recorded as a house address or geographical location using a global positioning system (GPS) at ≤ 10 m accuracy. iNaturalist sightings were only included if they were classified as Research Grade, included a photograph and had a public location with 500 m accuracy or less. To be classified as "Research Grade", observations in iNaturalist need to include a date and a location and have a minimum of two identifications, with the majority agreeing with the taxonomic classification (Durso et al. 2021). The authors confirmed every iNaturalist sighting. Tracks and scat observations were not included. When possible, we estimated the general size range category of individuals either in the field or with photographs. The size classes used were hatchling (estimated under 40 cm total length), sub-adult (>40 cm and <100 cm total length), adult (>100 cm and <150 cm total length) and mature adult (>150 cm total length). Size class estimates are often used as they can

give valuable insight into demographics without requiring the capture of individuals (Price et al. 2022a). We validated our field estimates by assessing the size class of individuals that had been previously captured, measured, and externally marked in a separate study (Genevier et al. in prep., Chapter 3). We did not sex individuals.

2.3.3 Statistical and landscape analyses

Randomised locations (hereafter "non-sightings") were generated using ESRI ArcMap (version 10.8.2, Redlands, California). These were used as a control group to establish habitat features that were positively and negatively associated with Nile monitor encounters. To assess "availability" adequately, we generated ten random points for every sighting in each site. We mapped a 500 m radius around each sighting and non-sighting location as a 78.54 ha land buffer area. Typically, such buffer areas fall within a known home range size for a species (McPherson et al. 2016; Mowry et al. 2021). With the present lack of studies on the Nile monitor's spatial ecology, we had to assume that the use of a 78.54 ha buffer area appropriately represents a fraction of their area used. This assumption was based on mark-resight data and the home range literature on other species of similar size, and was later validated by using point land-cover (Table 2.1; unpublished data). Moreover, the nature of our sightings does not only reflect actual animal habitat use in their present home range but also reflects encounters during a change or expansion of the present individual home range. A likely scenario for our domestic conflict records is that individual monitors were moving from one area to a new one rather than being in the core area of their home range (EG unpublished data). Therefore, our analysis used a relatively small buffer area to mitigate this bias.

All landscape data were measured using ArcGIS 10.7 (ESRI Inc. 2010). Further parameters were collected from physical visits (when possible), aerial imagery and eThekwini Municipality's corporate global information system (GIS) website. Recorded parameters were

quantified using percentage cover (% c) in the buffer area or the nearest distance to the sighting and non-sighting point. We calculated the % c of 24 land-cover types (Table 2.2). Each sighting was then classified as either low, medium, or high human population density point in R studio (Packages "raster", "sf" and "dplyr"; Version 4.0.2; R Core Team 2023) using 2020 data (CIESIN 2020). Human population density was based on a 1 km radius of each sighting and non-sighting point. We calculated the nearest distance to rivers, roads and terrestrial protected areas. Additionally, the presence and absence of pools and domestic animals were recorded when possible; however, this was done for sighting points only.

Table 2.2: The different resolutions of land cover types in the present study.

Low resolution	**Medium resolution**	**High resolution**
Forested land	Natural cover	Natural wooded land
		Shrubs
		Urban vegetation
	Plantation	Planted forest
Grass	Grassland	Natural grassland
Water	Waterbodies	Artificial waterbodies
		Natural waterbodies
Wetlands	Wetlands	Herbaceous wetlands
		Woody wetlands
Barren	Barren land	Consolidated
		Unconsolidated
Crop	Cultivated	Permanent crops
		Temporary crops
		Fallow lands and old fields
Anthro	Built-up	Residential
		Village
		Smallholdings
		Commercial

	Industrial
	Transport
Mines and quarries	Surface infrastructure
	Extraction sites
	Waste and resource dumps

We then used the lowest-resolution land use type at each point with logistic regressions with binomial family error to calculate resource selection functions (RSFs), identifying the relative probability of Nile monitor sightings and the landscape drivers in the defined sites, using observed and weighted randomly sampled locations (Table 2.2; Johnson et al. 2009; Steen et al. 2015). The models were built using fixed effects, without random effects, as no hierarchical structure was specified. The independent variables used were land cover types consisting of five variables (barren, anthropogenic, cover, grassland, water and wetland), human population index and three distance to features variables (rivers, protected areas and roads). We used a Variance Inflation Factor (VIF) test to assess multicollinearity across all predictors in the full model, as our data contained categorical and continuous variables using the car package (version 3.1-3; R Core Team 2023) in RStudio (de Jongh et al. 2015). We used the Akaike's information criterion (AIC) approach to select the best-fit candidate model. We calculated the relative importance of each variable by summing the Akaike weights (wi) of each model said variable occurred in (Pillay et al. 2024).

2.4 Results

We obtained a total of 419 usable Nile monitor sightings, of which 28.9% (n = 121) were from iNaturalist (Fig. 2.2). There were 19.3% (n = 81) of Nile monitor sightings in site A and 80.7% (n = 338) in site B. Most sightings (54%) were from urban mosaic landscapes, from Durban

and Pietermaritzburg reptile removal groups. The majority of iNaturalist sightings (53%) included were from protected areas in site A.

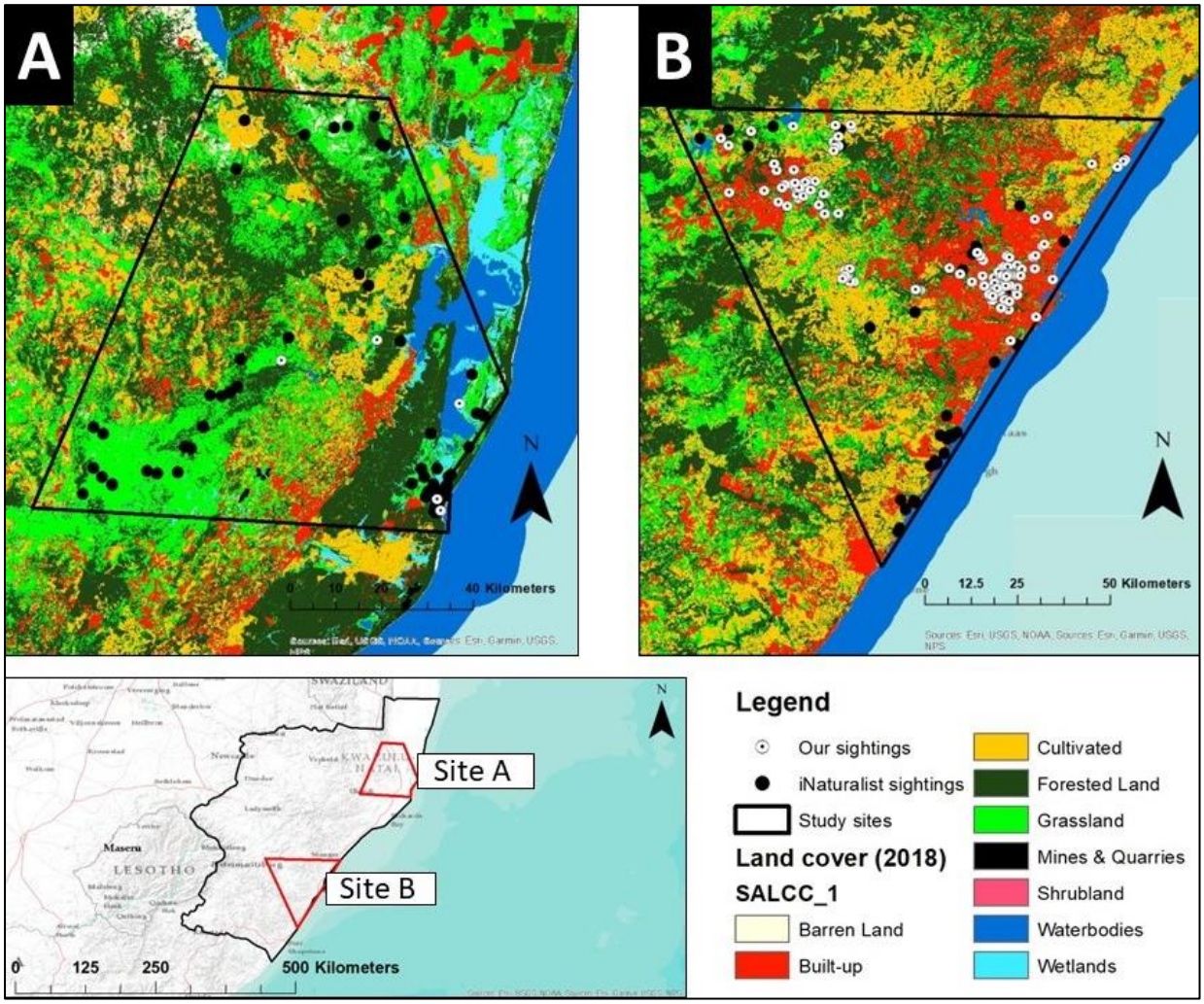

Figure 2.2: Geographical localities of Nile monitor sightings in sites A and B with land-cover types of each shown in KwaZulu-Natal, South Africa, in the present study.

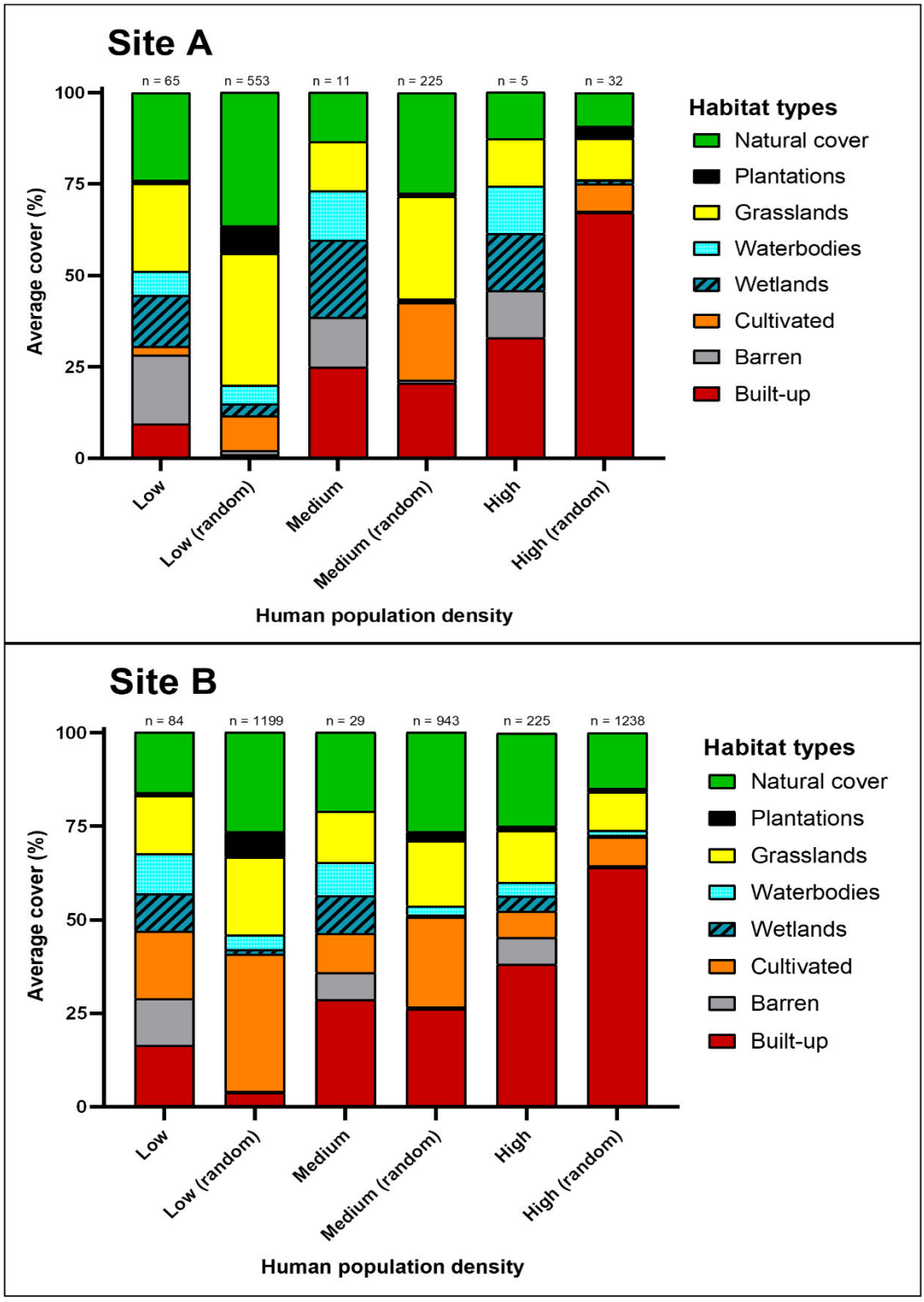

Figure 2.3: Average cover in 500 m of each Nile monitor sighting and non-sighting (random) locations on a gradient of human population density in sites A and B and number of points per category (n).

Habitat types of Nile monitors varied between sites, with site A having 11 of the 24 high-resolution land cover types and site B having 15 (Table 2.2, Fig. 2.2, Fig. 2.3). All five high human density sightings in site A came from the outskirts of St Lucia Village, which is small with some residential and commercial areas and falls in a protected area, where monitor populations can be functionally connected to iSimangaliso Wetland Park through the estuary. No sightings were recorded in the dense residential parts or agricultural land. Site A is predominantly natural grassland and natural wooded land in low human density areas (37% and 49%, respectively) and dominated by residential land in high human density (53%; Fig. 2.2, Fig. 2.3). Site B had more anthropogenic land use, and nature reserves are generally smaller and surrounded by either agricultural land or urban environments (Fig. 2.2, Fig. 2.3). Sixty-five per cent (n = 103) of sightings in Durban had swimming pools on the property or neighbouring property. Areas used by Nile monitors had more water cover, no planted forests and more barren land throughout. Despite showing a strong avoidance of built-up land in highly populated areas of site A, there was less of an avoidance in highly populated areas of site B. Although exotic tree plantations were nearly absent from buffer areas around true locations, natural cover remained consistent throughout, with a slightly increased cover in medium and high human populations of site B (Fig. 2.2, Fig. 2.3).

The relative selection strength of land use types over anthropogenic land use differed largely between sites (Fig. 2.4) for Nile monitors. There was a highly significant association with water ($P < 0.001$), cover and wetlands ($P = 0.002$ and $P = 0.001$, respectively) and a significant association with barren ($P = 0.021$) over anthropogenic land use in site A (Fig. 2.4, Fig. 2.5, Table 2.3). The best model fits for the relative selection strength of Nile monitors in Site A, based on ≤ 2 delta AICc, were land cover (wi = 1), river distance (wi = 1), protected area distance (wi = 1), road distance (wi = 1) and pop density (wi = 0.97; Table 2.3). Coefficient

estimates of the top model suggest an increase in Nile monitor selection strength in site A close to rivers ($\beta = -0.001 \pm 0.000$ SE, $p < 0.001$), protected areas ($\beta = -0.001 \pm 0.000$ SE, $p < 0.001$), roads ($\beta = -0.005 \pm 0.001$ SE, $p < 0.001$), and increased human population density ($\beta = 0.001 \pm 0.000$ SE, $p < 0.001$). Nile monitor selection strength was also positively associated with waterbodies ($\beta = 4.202 \pm 0.703$ SE, $p < 0.001$), wetlands ($\beta = 2.502 \pm 0.779$ SE, $p = 0.001$), barren land ($\beta = 1.926 \pm 0.834$ SE, $p = 0.021$) and grassland ($\beta = 1.079 \pm 0.633$ SE, $p = 0.088$).

The best two model fits for the relative selection strength of Nile monitors in Site B, based on ≤ 2 delta AICc, were river distance (wi = 1), protected area distance (wi = 1), road distance (wi = 1), pop density (wi = 1) and land cover (wi = 0.48; Table 2.4). Coefficient estimates of the top two models suggest an increased Nile monitor selection strength in site B close to rivers ($\beta = -0.00032 \pm 0.00005$ SE, $p < 0.001$), protected areas ($\beta = -0.00011 \pm 0.00001$ SE, $p < 0.001$), roads ($\beta = -0.00644 \pm 0.00070$ SE, $p < 0.001$), less human population density ($\beta = -0.000037 \pm 0.000011$ SE, $p < 0.001$), with the most important land cover type being grassland ($\beta = 0.434 \pm 0.166$ SE, $p = 0.009$), while the effect of other cover types, including barren land, water, and wetlands, were not significant.

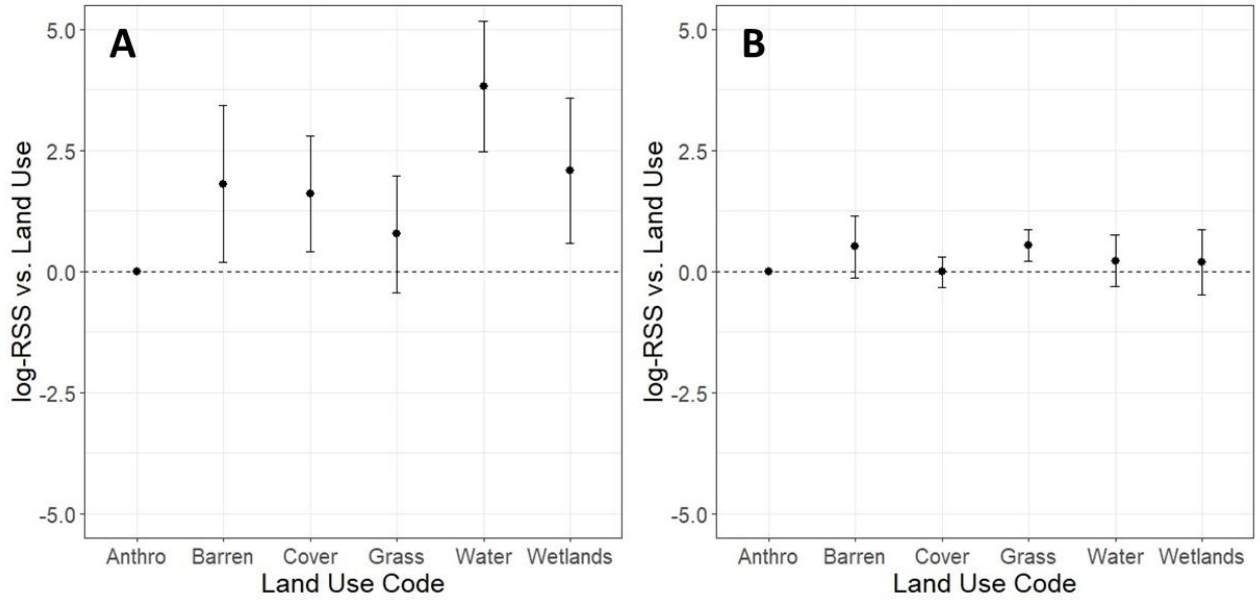

Figure 2.4: Relative selection strength (Log-RSS) by Nile monitors for various land use types compared with anthropogenic land use in site A (left) and site B (right). (Note: Whiskers show 95% confidence intervals).

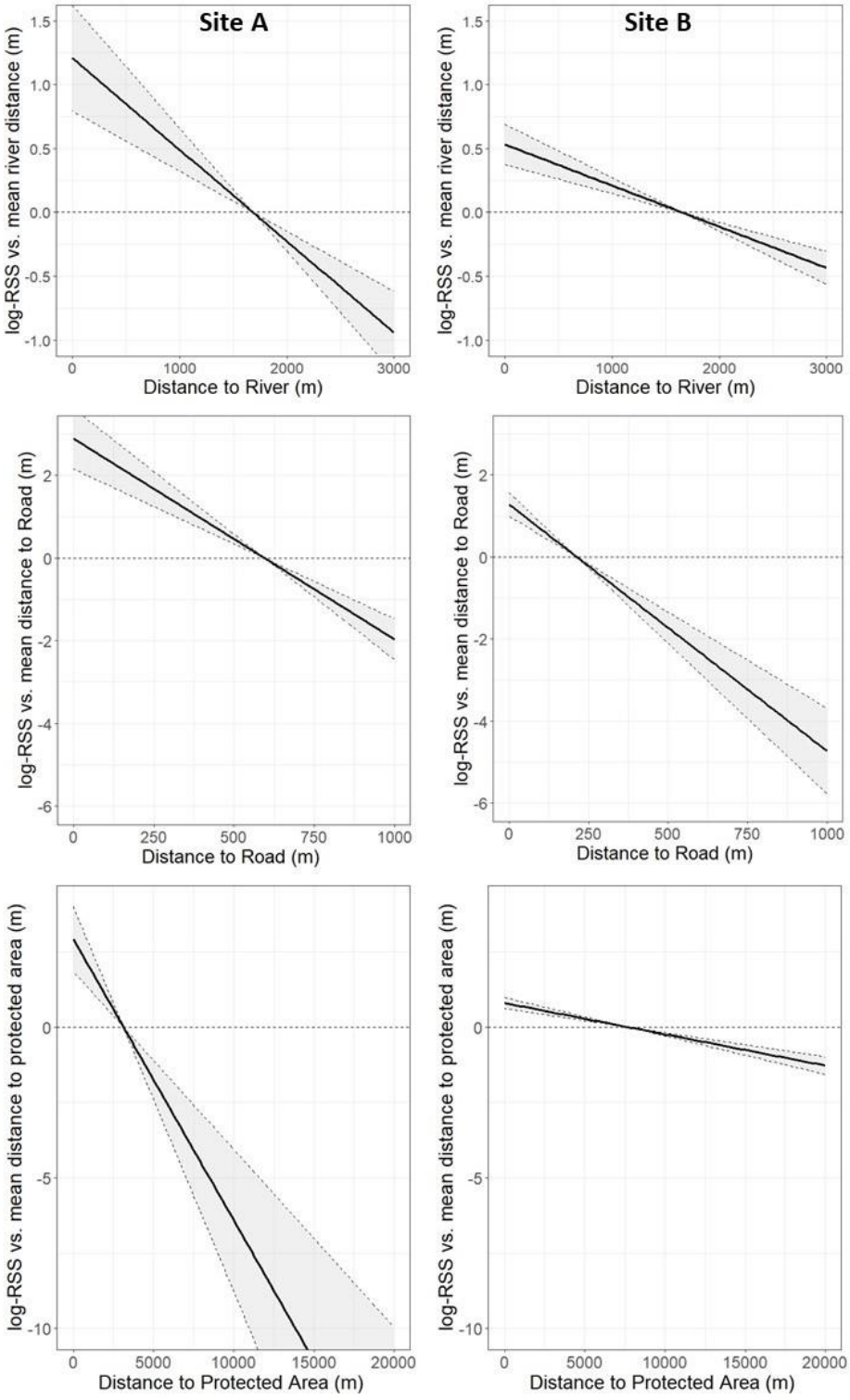

Figure 2.5: Nile monitor sightings relative selection strength (Log-RSS) to distance to rivers, roads and protected areas in site A (left) and site B (right).

Table 2.3: The top generalised linear models showing the effects of relative selection strength of landscape features for Nile monitors (*Varanus niloticus*) in site A of this study.

Model	AICc	LogLik	delta_AICc	Weight
Land cover + River + Protected + Road + Pop	1642.20	-810.95	0.00	0.97
Land cover + River + Protected + Road	1649.49	-815.62	7.29	0.03
River + Protected + Road + Pop	1687.92	-838.91	45.71	0.00
River + Protected + Road	1690.80	-841.37	48.60	0.00
Land cover + Protected + Road + Pop	1693.49	-837.62	51.29	0.00
Land cover + Protected + Road	1695.20	-839.50	53.00	0.00
Protected + Road	1726.09	-860.02	83.88	0.00
Protected + Road + Pop	1726.95	-859.44	84.75	0.00
Land cover + River + Road + Pop	1779.04	-880.39	136.83	0.00
Land cover + River + Road	1784.69	-884.24	142.49	0.00
River + Protected + Pop	1791.43	-891.68	149.22	0.00
Land cover + River + Protected + Pop	1792.54	-887.14	150.33	0.00
River + Protected	1803.21	-898.58	161.01	0.00
Land cover + River + Protected	1806.74	-895.27	164.54	0.00
River + Road + Pop	1812.45	-902.19	170.25	0.00
Land cover + Road + Pop	1819.01	-901.40	176.81	0.00
Land cover + Road	1823.17	-904.50	180.96	0.00
Protected + Pop	1827.08	-910.52	184.88	0.00
Land cover + Protected + Pop	1827.35	-905.57	185.15	0.00
River + Road	1828.41	-911.18	186.21	0.00
Protected	1832.93	-914.45	190.73	0.00
Land cover + Protected	1835.21	-910.52	193.00	0.00
Road + Pop	1847.16	-920.56	204.95	0.00
Road	1862.15	-929.06	219.95	0.00
Land cover + River	1872.84	-929.34	230.64	0.00
Land cover + River + Pop	1874.43	-929.11	232.23	0.00
River	1886.15	-941.06	243.95	0.00
River + Pop	1886.72	-940.33	244.51	0.00
Land cover	1902.07	-944.97	259.87	0.00
Land cover + Pop	1903.86	-944.85	261.66	0.00
Null model	1916.82	-957.40	274.62	0.00
Pop	1917.39	-956.68	275.19	0.00

Table 2.4: The top generalised linear models showing the effects of relative selection strength of landscape features for Nile monitors (*Varanus niloticus*) in site B of this study.

Model	AICc	LogLik	delta_AICc	Weight
River + Protected + Road + Pop	7662.00	-3825.99	0.00	0.52
Land cover + River + Protected + Road + Pop	7662.18	-3821.05	0.18	0.48
Land cover + River + Protected + Road	7673.33	-3827.63	11.32	0.00
River + Protected + Road	7675.70	-3833.84	13.69	0.00
Land cover + Protected + Road + Pop	7712.23	-3847.09	50.23	0.00
Protected + Road + Pop	7712.61	-3852.30	50.61	0.00
Land cover + Protected + Road	7723.35	-3853.65	61.34	0.00
Protected + Road	7725.56	-3859.77	63.55	0.00
River + Road + Pop	7752.66	-3872.32	90.66	0.00
Land cover + River + Road + Pop	7754.37	-3868.15	92.36	0.00
Land cover + River + Road	7757.16	-3870.56	95.16	0.00
River + Road	7757.25	-3875.62	95.25	0.00
Road + Pop	7793.51	-3893.75	131.50	0.00
Land cover + Road + Pop	7794.55	-3889.25	132.55	0.00
Land cover + Road	7797.52	-3891.74	135.51	0.00
Road	7797.95	-3896.97	135.94	0.00
Land cover + River + Protected	7846.81	-3915.38	184.81	0.00
Land cover + River + Protected + Pop	7848.75	-3915.35	186.75	0.00
River + Protected	7861.95	-3927.97	199.94	0.00
River + Protected + Pop	7863.49	-3927.74	201.48	0.00
Land cover + Protected	7893.68	-3939.82	231.67	0.00
Land cover + Protected + Pop	7895.59	-3939.77	233.59	0.00
Protected	7905.56	-3950.78	243.56	0.00
Protected + Pop	7907.22	-3950.61	245.22	0.00
Land cover + River	7945.37	-3965.67	283.37	0.00
Land cover + River + Pop	7945.71	-3964.83	283.70	0.00
River + Pop	7954.48	-3974.23	292.47	0.00
River	7956.58	-3976.29	294.58	0.00
Land cover	7983.19	-3985.58	321.19	0.00
Land cover + Pop	7983.79	-3984.87	321.78	0.00
Pop	7990.60	-3993.30	328.60	0.00
Null model	7992.18	-3995.09	330.18	0.00

On average, true sighting points of Nile monitors were consistently closer to roads, protected areas and rivers than random points, regardless of human population density (Fig. 2.5, Fig. 2.6). Nile monitors were seen further away from rivers with increased human population density (Fig. 2.5, Fig. 2.6). The distance to protected areas was similar in low- and high-density areas but greater in medium human density. Nile monitors were seen 63% closer to protected areas than random in low-density areas, 46% in medium and 49% in high. Distance to roads greatly decreased with higher human density levels in both random and true sighting locations (Fig. 2.5, Fig. 2.6).

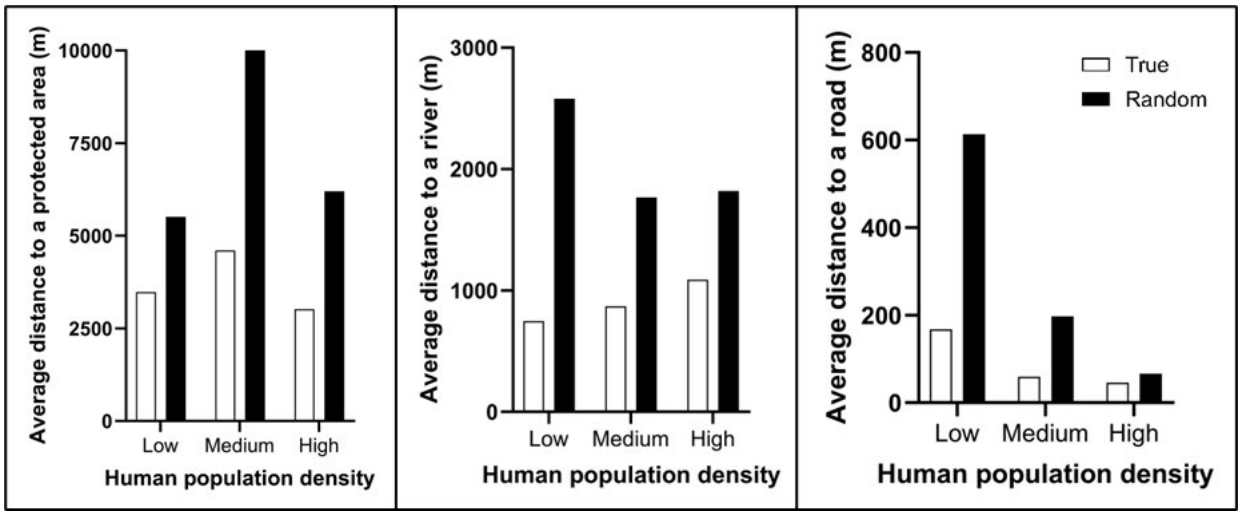

Figure 2.6: Average distance to protected areas, rivers and roads for both sightings and non-sighting locations of Nile monitors in our study areas (sites A and B combined). (White = true sightings; Black = random locations).

We could estimate the size class of 90% (n = 376) of Nile monitor individuals sighted. We found a total of 23 hatchlings, 113 sub-adults, 194 adults and 46 mature adults. There was little difference in the proportion of size classes between low and high human population density areas (Fig. 2.7). However, a greater proportion of juveniles and subadults with a low

proportion of adults were observed in medium human dense areas (Fig. 2.7). Adults were the predominant size class observed in low- and high-human density areas, with sub-adults dominating the medium-density areas.

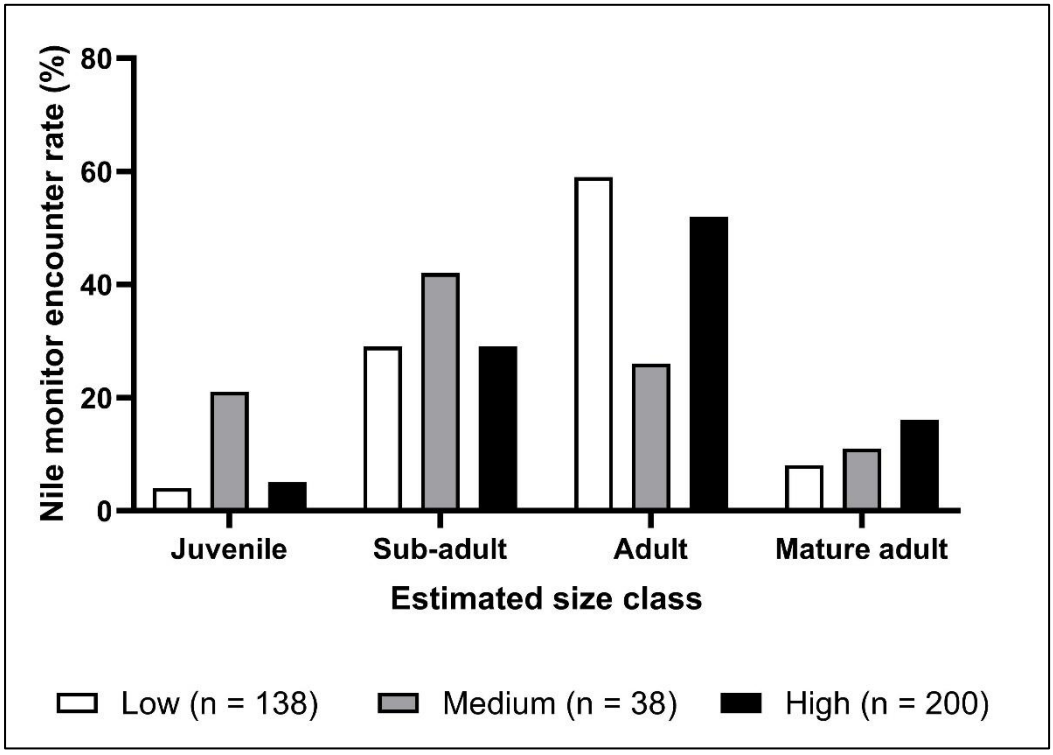

Figure 2.7: Nile monitor encounter rate of each estimated size class across a gradient of low to high human population density areas. (Note: Juveniles were estimated to be less than 40 cm in total length; Sub-adults were estimated to be between 40 cm and 100 cm; Adults were between 100 cm and 150 cm; Large adults were estimated to be larger than 150 cm).

2.5 Discussion

Our study showed that Nile monitors use a wide variety of habitat types across the landscape and can adapt to various levels of anthropogenic disturbance as found in other South African species across taxa (Ramesh & Downs, 2014; Thatcher et al. 2020; Maphalala et al., 2020; Singh et al., 2021; Downs et al. 2021). Many urban exploiting species alter their behaviour in

response to urbanisation (Downs et al. 2021). The reduced reliance on river systems in urban areas suggests behaviour plasticity in the Nile monitor's adaptation to urban environments. The lack of residential sightings in site A can be interpreted in three ways: Firstly, when lots of natural habitats with low human density are available, Nile monitors do not exploit urban centres and will remain in natural habitats or urban outskirts. Secondly, these residential areas are simply unsuitable for monitors. Lastly, our lack of sightings from reptile relocation groups and large reliance on iNaturalist observations for the area resulted in sample bias against residential sightings. However, residential sightings from iNaturalist did occur in site B. There are also reports of citizen science observation being spatially aggregated around tourist locations (Auguste et al. 2020; Durso et al. 2021). Yet, our human-dense sightings of the area come from St Lucia, an important tourist destination (Forbes et al. 2020). Therefore, if such spatial bias is expected, more residential sightings would likely have been reported if they had occurred.

We found that Nile monitors persisted in urban mosaic landscapes and occurred in proximity to humans in KwaZulu-Natal. Urban use has been documented in multiple varanids, such as the ecologically similar Asian water monitor (*Varanus salvator*; Uyeda 2009; Karunarathna et al. 2017; Wongtienchai et al. 2020; Briggs-Gonzalez et al. 2022; Trivalairat and Srikosamatara 2023). Asian water monitors are overall less abundant in oil plantations than neighbouring forests in Northern Borneo, despite having abundant food (Guerrero-Sanchez et al. 2021; 2022). This is potentially because of limited refugia intensifying intra-specific competition amongst other aspects which overall reduced habitat suitability (Guerrero-Sanchez et al. 2021). Our results suggest Nile monitors are less abundant in exotic tree plantations (mainly pine *Pinus* spp. and eucalyptus *Eucalyptus* spp. monocultures for timber), yet rodents can be relatively abundant in South African timber plantations (Ferguson et al. 2003). This corroborates that the limited availability of refugia in plantations may result in less suitable

habitats despite the abundance of food. However, more research is needed on the perceived avoidance of plantations and associated prey by Nile monitors where adjacent suitable habitats are present.

The proximity of Nile monitors to humans outlines concerns for potential conflicts with people and domestic animals. This may explain the preference for closed habitats (natural wooded land and urban vegetation) in human-populated areas, which offer more refugia (hiding and escape opportunities). It has been suggested that urban exploitation may be a key to the population persistence of urban exploiting varanids (Uyeda 2009). The evident persistence of Nile monitors in high human population density areas, paired with a seemingly healthy population demographic, further suggests that urban use may be an important aspect of their population maintenance in an increasing land use change scenario.

We also outline the importance of managed and unmanaged green spaces in mosaic urban landscapes for the persistence of wildlife, particularly Nile monitors, as documented for other species in KwaZulu-Natal (Downs et al. 2021), based on their occurrence in the present study. Nile monitors use of small anthropogenic waterbodies, such as swimming pools and storm drains, based on their occurrence in urban mosaic landscapes in the present study, suggests a certain level of behavioural plasticity and adaptation to highly anthropogenic ecosystems.

Nile monitors are projected to become threatened in parts of their range through climate change-induced loss of suitable habitat (Ejigu and Tassie 2020). Under several future climate scenarios, Ejigu and Tassie (2020) modelled the present and future habitat suitability for Nile monitors at Lake Tana Biosphere Reserve, Ethiopia. Their best-case-scenario projection predicted a reduction in habitat suitability in areas of over 54% by 2050 and over 90% by 2070. They argued that although they are presently listed as Least Concern, they may become threatened or endangered without effective conservation through loss of suitable habitat.

Wetlands in parts of South Africa have declined in distribution and size through agriculture and urban development and are expected to decline further through rising water scarcity and increased occurrence and intensity of droughts (Grundling et al. 2013; Price et al. 2022b; Thamaga et al. 2022). Reduced wetland connectivity through drought and anthropogenic land use change may also affect population demographics in some semi-aquatic species (Price et al. 2021). Perhaps the loss of wetland habitats will cause population declines in Nile monitors in their affected range (Ejigu and Tassie 2020). Our results suggest that Nile monitors can use small artificial waterbodies regardless of associated foraging opportunities. Many nature reserves in South Africa presently use artificial waterholes to mitigate water scarcity or unpredictability (Sutherland et al. 2018). Future research should investigate the use of such structures by Nile monitors and their feasibility as a management tool in increasingly arid parts of their range. More research is needed on the potential impacts of treated pools and chemical runoff associated with cities such as Durban and increased flooding events, which have likely doubled in frequency in Durban in the last century (Lebepe et al. 2022; Grab and Nash 2023). This study serves as a baseline understanding of the urban use of Nile monitors and a call for research in this highly exploited yet understudied species (Whiting et al. 2011).

Although monitors, like many other reptiles and ectotherms, sometimes use road surfaces for basking or foraging, the proximity to roads of our sightings likely reflects a limitation in the use of non-structure opportunistic data (i.e. reptile removal groups and iNaturalist; pers. obs.; Colino-Rabanal and Lizana 2012; Azmi et al. 2021). Many of our sightings from site A are from fenced reserves, where visitors are generally prohibited from exiting vehicles outside designated areas. Furthermore, sightings from reptile removal groups are predominantly from people's homes. There was, therefore, an overall expected bias towards road proximity.

Further research is needed on morphological response to anthropogenic stressors. We cannot infer whether the observed differences in size class occurrences are a direct consequence of size-selected persecution or phenotypic plasticity in an urban environment. We suggest that further research be conducted on their morphology and people's perceptions, with an emphasis on medium human dense areas because of our observed smaller size classes. It does, however, suggest the Nile monitors are finding suitable breeding sites in the mosaic urban matrix. Previous breeding records focus on their nesting behaviour in termite mounds (Cowles 1930; Branch and Erasmus 1982). During this study, we found urban Nile monitor nests in private residential gardens (e.g., in a compost heap, a failed attempt to lay in gaps in a garden retaining wall and uncovered eggs under a concrete deck, unpublished data). Additionally, a large individual was observed excavating a termite mound in a Durban urban green space by a member of the public (pers. comm.). Based on photographs, the individual's size, activity and the time of year (June 2022), we believe this was a nesting attempt. The mound was visited 12 days later, but no eggs were found. Whether the attempt failed or the nest was raided remains unknown.

The use of a 78.54 ha buffer area in this study was a limitation as it may not reflect actual habitat use. The lack of knowledge on Nile monitor home range makes it difficult to assess this bias and may have resulted in under or over-exaggerated associations. Many species of monitors increase home range size with body mass, and it can vary greatly by individual, geographical location, habitat type, sex, diet, weather, and season (Thompson et al. 1999; Guarino 2002; Purwandana et al. 2021; Guerrero-Sanchez et al. 2022, 2023). Furthermore, many of our first-time urban encounters and conflicts were likely a result of a change or expansion of an individual monitor's activity area rather than an encounter in the core area of their home range. However, the use of point low-resolution land use type for the resource selection functions showed similar results of strong association with water and a difference in

tolerance to humans and anthropogenic land use. This suggests that our chosen buffer size was accurate enough to show habitat associations of Nile monitors. This potential bias could be mitigated by a detailed study of their spatial ecology in which core home range characteristics could be assessed. We successfully detected habitat characteristics associated with Nile monitor encounters on a gradient of human population density, further enabling our ability to make research-based management decisions for urban and rural Nile monitors in KwaZulu-Natal Province.

2.6 Conclusions

This study found that Nile monitors use natural, undisturbed habitats over anthropogenic land-use types when it is abundant. However, they will use anthropogenic land-use types with no significant avoidance when the area is prominently modified at a landscape scale. Therefore, we suggest that the urban use of Nile monitors in KwaZulu-Natal is partly driven by habitat transformation and loss. With continued land use change, urbanisation and climate change-induced loss of wetlands, the use of anthropogenic habitats by Nile monitors and conflicts with people will potentially increase. In urban landscapes, Nile monitors use small anthropogenic water bodies such as treated and untreated swimming pools and storm drains, showing behavioural adaptation and reduced reliance on river systems. This also outlines the importance of both natural and managed green spaces in urban mosaic landscapes of KwaZulu-Natal for the persistence of Nile monitors. There is a need for more research on this highly exploited yet understudied species. A better understanding of their spatial and temporal activity in urban and natural habitats and how they navigate complex urban centres would benefit future conservation efforts. Furthermore, the management of urban Nile monitors would benefit from a better understanding of people's perceptions towards them, their associated myths and folklore, as well as their illegal exploitation and persecution in South Africa.

2.7 Acknowledgements

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2.8 Data availability statement

The data used in this study belong to the University of KwaZulu-Natal and are available from the authors on reasonable request.

2.9 Declaration of competing interest

The authors declare they have no financial interests/personal relationships which may be considered as potential competing interests.

2.10 References

Alexander GJ (1990) Reptiles and amphibians of Durban. Durban Museum Novitates 15(1):1–41. https://doi.org/10.10520/AJA0012723X_1939.

Alexander GJ (2007) Thermal biology of the Southern African Python (*Python natalensis*): Does temperature limit its distribution. In: Henderson RW, Powell R (Eds), Biology of the boas and pythons. Eagle Mountain Publishing, Eagle Mountain, Utah. pp 50–75.

Ariano-Sánchez D, Mortensen RM, Reinhardt S, Rosell F (2020) Escaping drought: Seasonality effects on home range, movement patterns and habitat selection of the Guatemalan Beaded Lizard. Global Ecology and Conservation 23:e01178. https://doi.org/10.1016/j.gecco.2020.e01178

Auffenberg W (1988) Gray's Monitor Lizard. University of Florida Press, Florida.

Auffenberg W, Arian QN, Kurshid N (1991) Preferred habitat, home range and movement patterns of Varanus bengalensis in southern Pakistan. Mertensiella 2:7-28.

August TA, Pescott OL, Joly A, Bonnet P (2020) AI Naturalists might hold the key to unlocking biodiversity data in Social Media Imagery. Patterns 1(7):100116. https://doi.org/10.1016/j.patter.2020.100116

Auguste RJ (2020) Using citizen science to rapidly determine the distribution of exploited Green Iguanas (*Iguana iguana*) across urban areas in Trinidad and Tobago. Reptiles and Amphibians 27(3):419–421. https://doi.org/10.17161/randa.v27i3.14859

Azmi M, Ardiantiono A, Nasu S, Kasim M, Ariefiandy A, Purwandana D, Ciofi C, Jessop T (2021) Incidences of road kills and injuries of Komodo dragons along the north coast of Flores Island, Indonesia. Herpetological Conservation and Biology 16:11–16.

Bates MF, Branch WR, Bauer AM, Burger M, Marais J, Alexander GJ, de Villiers MS (2014) Atlas and Red List of Reptiles of South Africa, Lesotho and Swaziland. Animal Demography Unit (Cape Town) and the South African National Biodiversity Institute, Pretoria.

Branch WR, Erasmus H (1982) Notes on reproduction in South African water monitors, *Varanus niloticus* (Sauria: Varanidae). Journal of the Herpetological Association of Africa 28(1):4–9. https://doi.org/10.1080/04416651.1982.9650104

Briggs-Gonzalez V, Evans P, Klovanish C, Mazzotti FJ (2022) A species bioprofile for the Asian water monitor (*Varanus salvator*). Southeastern Naturalist 21(3):187–210. https://doi.org/10.1656/058.021.0302

Brum PHR, Gonçalves SRA, Strüssmann C, Teixido AL (2022) A global assessment of research on urban ecology of reptiles: patterns, gaps and future directions. Animal Conservation 26:1–13. https://doi.org/10.1111/acv.12799

Ciliberti A, Berny P, Vey D, de Buffrénil V. (2012) Assessing environmental contamination around obsolete pesticide stockpiles in West Africa: Using the Nile monitor (*Varanus niloticus*) as a sentinel species. Environmental Toxicology and Chemistry / SETAC, 31:387–94. https://doi.org/10.1002/etc.731

Ciofi C, Puswati J, Winana D, de Boer M, Chelazzi G, Sastrawan P (2007) Preliminary analysis of home range structure in the Komodo monitor, *Varanus komodoensis*. Copeia 2007:462–470. https://doi.org/10.1643/0045-8511(2007)7[462:PAOHRS]2.0.CO;2

Cobbinah PB, Erdiaw-Kwasie MO, Amoateng P (2015) Africa's urbanisation: Implications for sustainable development. Cities 47:62–72. https://doi.org/10.1016/j.cities.2015.03.013

Cockburn J, Rouget M, Slotow R, Roberts D, Boon R, Douwes E, O'Donoghue SH, Downs C, Mukherjee S, Musakwa W, Mutanga O, Mwabvu T, Odindi J, Odindo A, Procheş Ş, Ramdhani S, Ray-Mukherjee J, Sershen N, Schoeman C, Willows-Munro S (2016) How to build science-action partnerships for local land-use planning and management: Lessons from Durban, South Africa. Ecology and Society 21:28. https://doi.org/10.5751/ES-08109-210128

Colino-Rabanal V, Lizana M (2012) Herpetofauna and roads: A review. Basic and Applied Herpetology 26:5–31. https://doi.org/10.11160/bah.12008

Cowles RB (1930) The life history of *Varanus niloticus* (Lin.) as observed in Natal, South Africa. Journal of Entomology and Zoology 22(1):1-31.

Dalhuijsen K, Branch WR, Alexander GJ (2014) A comparative analysis of the diets of *Varanus albigularis* and *Varanus niloticus* in South Africa. African Zoology 49:83–93. https://doi.org/10.1080/15627020.2014.11407621

de Buffrénil V, Castanet J (2000) Age estimation by skeletochronology in the Nile monitor (*Varanus niloticus*), a highly exploited species. Journal of Herpetology 34:414. https://doi.org/10.2307/1565365

de Buffrénil V, Hémery G (2002) Variation in longevity, growth, and morphology in exploited Nile monitors (*Varanus niloticus*) from Sahelian Africa. Journal of Herpetology 36: 419. https://doi.org/10.2307/1566186

de Buffrénil V, Rimblot-Baly F (1999) Female reproductive output in exploited Nile monitor lizard (*Varanus niloticus* L.) populations in Sahelian Africa. Canadian Journal of Zoology 77:1530–1539. https://doi.org/10.1139/z99-130

de Gabriel Hernando M, Karamanlidis A, Grivas K, Krambokoukis L, Papakostas G, Beecham J. (2021) Habitat use and selection patterns inform habitat conservation priorities of an endangered large carnivore in southern Europe. Endangered Species Research 44:203-215. https://doi.org/10.3354/esr01105

de Jongh P, de Jongh E, Pienaar M, Gordon-Grant H, Oberholzer M, Santana L (2015) The impact of pre-selected variance inflation factor thresholds on the stability and predictive power of logistic regression models in credit scoring, ORiON 31(1):17–37. https://doi.org/10.5784/31-1-162.

de Souza JC, da Silva RM, Gonçalves MPR, Jardim RJD, Markwith SH (2018) Habitat use, ranching, and human-wildlife conflict within a fragmented landscape in the Pantanal, Brazil. Biological Conservation 217:349–357. https://doi.org/10.1016/j.biocon.2017.11.019

Di Cecco GJ, Barve V, Belitz MW, Stucky BJ, Guralnick RP, Hurlbert AH (2021) Observing the observers: How participants contribute data to iNaturalist and implications for biodiversity science. BioScience 71(11):1179–1188. https://doi.org/10.1093/biosci/biab093

Dowell SA, de Buffrénil V, Kolokotronis S-O, Hekkala ER (2015) Fine-scale genetic analysis of the exploited Nile monitor (*Varanus niloticus*) in Sahelian Africa. BMC Genetics 16:32. https://doi.org/10.1186/s12863-015-0188-x

Downs C, Alexander J, Brown M, Chibesa M, Ehlers Smith Y, Gumede T, Hart L, Josiah K, Kalle R, Maphalala M, Maseko M, McPherson S, Ngcobo P, Patterson L, Pillay K, Price C, Raji I, Ramesh T, Schmidt W, Ehlers Smith D (2021) Modification of the third phase in the framework for vertebrate species persistence in urban mosaic environments. Ambio 50:1866–1878. https://doi.org/10.1007/s13280-021-01501-5

Durso AM, Ruiz de Castañeda R, Montalcini C, Mondardini MR, Fernandez-Marques JL, Grey F, Müller MM, Uetz P, Marshall BM, Gray RJ, Smith CE, Becker D, Pingleton M, Louies J, Abegg AD, Akuboy J, Alcoba G, Daltry JC, Entiauspe-Neto OM, Freed P, de Freitas MA, Glaudas X, Huang S, Huang T, Kalki Y, Kojima Y, Laudisoit A, Limbu KP, Martínez-Fonseca JG, Mebert K, Rödel M-O, Ruane S, Ruedi M, Schmitz A, Tatum SA, Tillack F, Visvanathan A, Wüster W, Bolon I. (2021) Citizen science and online data: Opportunities and challenges for snake ecology and action against snakebite. Toxicon X:9–10, 100071. https://doi.org/10.1016/j.toxcx.2021.100071

Ejigu D, Tassie N (2020) Present and future suitability of the Lake Tana Biosphere Reserve in Ethiopia for the Nile monitor (*Varanus niloticus*) using the MaxEnt model. Environmental Systems Research 9:1–15. https://doi.org/10.1186/s40068-020-00197-y

Ellis EC, Ramankutty N (2008) Putting people in the map: anthropogenic biomes of the world. Frontiers in Ecology and the Environment 6:439–447. https://doi.org/10.1890/070062

Enge K, Krysko K, Hankins K, Campbell T, King A (2004) Status of the Nile Monitor (*Varanus niloticus*) in Southwestern Florida. Southeastern Naturalist 3:571–582. https://doi.org/10.1656/1528-7092(2004)003[0571:SOTNMV]2.0.CO;2

Fairbanks DHK, Benn GA (2000) Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. Landscape and Urban Planning 50:237-257. https://doi.org/10.1016/S0169-2046(00)00068-2

Ferguson JWH, van Jaarsveld AS, Johnson R, Bredenkamp GJ, Foord SH, Britz M (2003) Rodent-induced damage to pine plantations: a South African case study. Agriculture, Ecosystems and Environment 95(1):379–386. https://doi.org/10.1016/S0167-8809(02)00164-0

Forbes N, Forbes A, James B (2020) Restoration of Lake St Lucia, the largest estuary in South Africa: historical perceptions, exploitation, management and recent policies. African Journal of Aquatic Science 45(1–2):183–197. https://doi.org/10.2989/16085914.2020.1719816

Forti L, Szabo J (2023) The iNaturalist platform as a source of data to study amphibians in Brazil. Anais da Academia Brasileira de Ciências 95:20220828. https://doi.org/10.1590/0001-3765202320220828

Genevier EEZ (2024) Aspects of the urban and rural ecology on Nile monitors (*Varanus niloticus*) in KwaZulu-Natal, South Africa. Thesis, University of KwaZulu-Natal.

Geurts E, Reynolds J, Starzomski B (2023) Turning observations into biodiversity data: Broadscale spatial biases in community science. Ecosphere 14. https://doi.org/10.1002/ecs2.4582

Grab S, Nash D (2023) A new flood chronology for KwaZulu-Natal (1836–2022): the April 2022 Durban floods in historical context. South African Geographical Journal 1–22. https://doi.org/10.1080/03736245.2023.2193758

Green B, King D (1978) Home range and activity patterns of the sand goanna, *Varanus gouldii* (Reptilia: Varanidae). Wildlife Research 5:417-424. https://doi.org/10.1071/WR9780417

Grundling AT, Van den Berg EC, Price JS (2013) Assessing the distribution of wetlands over wet and dry periods and land-use change on the Maputaland Coastal Plain, north-eastern KwaZulu-Natal, South Africa. South African Journal of Geomatics 2(2):120–138.

Guarino F (2002) Spatial ecology of a large carnivorous lizard, *Varanus varius* (Squamata: Varanidae). Journal of Zoology 258:449-457. . https://doi.org/10.1017/S0952836902001607

Guerrero-Sanchez S, Burger R, Rosli M, Natsyir N, Rahman R, Saimin S, Orozco-terWengel P, Goossens B (2021) Homing: a case-study on the spatial memory of the Asian water monitor lizard Varanus salvator in the Kinabatangan floodplain. Herpetological Bulletin 158:37–39. https://doi.org/10.33256/hb158.3739.

Guerrero-Sanchez S, Frias L, Saimin S, Orozco-terWengel P, Goossens B (2023) The fast-food effect: costs of being a generalist in a human-dominated landscape. Conservation Physiology 11(1):coad055. https://doi.org/10.1093/conphys/coad055

Guerrero-Sanchez S, Majewski K, Orozco-terWengel P, Saimin S, Goossens B (2022) The effect of oil palm-dominated landscapes on the home range and distribution of a generalist species, the Asian water monitor. Ecology and Evolution 12:e8531. https://doi.org/10.1002/ece3.8531

Haacke WD, Groves D (1995) *Varanus niloticus niloticus* Nile monitor size. African Herp News 22:45–46.

Hayes Hursh S, Bauder JM, Fidino M, Drake D (2023) An urban cast of characters: Landscape use and cover influencing mammal occupancy in an American midwestern city. Landscape and Urban Planning 229:104582. https://doi.org/10.1016/j.landurbplan.2022.104582

Ibrahim A (2002) Activity area, movement patterns, and habitat use of the desert monitor, *Varanus griseus*, in the Zaranik Protected Area, North Sinai, Egypt, African Journal of Herpetology 51:35–45. https://doi.org/10.1080/21564574.2002.9635460.

Johnson C, Nielsen S, Merrill E, Mcdonald T, Boyce M (2009) Resource selection functions based on use–availability data: Theoretical motivation and evaluation methods. Journal of Wildlife Management 70: 347–357. https://doi.org/10.2193/0022-541X(2006)70[347:RSFBOU]2.0.CO;2

Josiah KK, Downs CT (2023) Facets of the nesting ecology of a ground-nesting bird, the Spotted Thick-knee, in an urban mosaic landscape. Global Ecology and Conservation 44:e02500. https://doi.org/10.1016/j.gecco.2023.e02500

Karunarathna S, Surasinghe T, Madawala M, Somaweera R, Amarasinghe AAT (2017) Ecological and behavioural traits of the Sri Lankan water monitor (*Varanus salvator*) in an urban landscape of Western Province, Sri Lanka. Marine and Freshwater Research 68(12):2242–2252. https://doi.org/10.1071/MF17038

Kondratyeva A, Knapp S, Durka W, Kühn I, Vallet J, Machon N, Martin G, Motard E, Grandcolas P, Pavoine S (2020) Urbanization effects on biodiversity revealed by a two-scale analysis of species functional uniqueness versus redundancy. Frontiers in Ecology and Evolution 8:73. https://doi.org/10.3389/fevo.2020.00073

Kruger AC, Nxumalo MP (2017) Historical rainfall trends in South Africa: 1921–2015. Water SA 43(2):285–297. https://doi.org/10.4314/wsa.v43i2.12

Lebepe J, Khumalo N, Mnguni A, Pillay S, Mdluli S (2022) Macroinvertebrate assemblages along the longitudinal gradient of an urban Palmiet River in Durban, South Africa. Biology 11(5):705. https://doi.org/10.3390/biology11050705

Lei J, Booth DT (2018) Intraspecific variation in space use of a coastal population of lace monitors (*Varanus varius*). Australian Journal of Zoology 65:398-407.

Maphalala M, Monadjem A, Bildstein K, McPherson S, Hoffman B, Downs C (2020) Ranging behaviour of Long-crested Eagles *Lophaetus occipitalis* in human-modified landscapes of KwaZulu-Natal, South Africa. Ostrich 91:1–7. https://doi.org/10.2989/00306525.2020.1770888.

Markovchick-Nicholls L, Regan HM, Deutschman DH, Widyanata A, Martin B, Noreke L, Hunt TA (2008) Relationships between human disturbance and wildlife land use in urban habitat fragments. Conservation Biology 22(1):99–109.

McPherson S, Brown M, Downs CT (2016) Crowned eagle nest sites in an urban landscape: Requirements of a large eagle in the Durban Metropolitan Open Space System. Landscape and Urban Planning 146:43–50. https://doi.org/10.1016/j.landurbplan.2015.10.004

Mendoza-Roldan JA, Modry D, Otranto D (2020) Zoonotic parasites of reptiles: A crawling threat. Trends in Parasitology 36(8):677–687. https://doi.org/10.1016/j.pt.2020.04.014

Mlangeni LN, Ramatla T, Lekota KE, Price C, Thekisoe O, Weldon C (2024) Occurrence, antimicrobial resistance, and virulence profiles of *Salmonella* Serovars isolated from wild reptiles in South Africa. International Journal of Microbiology 2024:5213895. https://doi.org/10.1155/2024/5213895

Mo M, Mo E (2022) Using the iNaturalist application to identify reports of green iguanas (*Iguana iguana*) on the mainland United States of America outside of populations in Florida. Reptiles and Amphibians 29:85–92. https://doi.org/10.17161/randa.v29i1.16269

Mowry CB, Lee A, Taylor ZP, Hamid N, Whitney S, Heneghan M, Russell J, Wilson LA. (2021) Using community science data to investigate urban coyotes (*Canis latrans*) in

Atlanta, Georgia, USA. Human Dimensions of Wildlife 26:163–178. https://doi.org/10.1080/10871209.2020.1806415
Mugiyo H, Chimonyo VGP, Kunz R, Sibanda M, Nhamo L, Ramakgahlele Masemola C, Modi AT, Mabhaudhi T (2022) Mapping the spatial distribution of underutilised crop species under climate change using the MaxEnt model: A case of KwaZulu-Natal, South Africa. Climate Services 28:100330. https://doi.org/10.1016/j.cliser.2022.100330
Ndlovu M, Demlie M (2020) Assessment of meteorological drought and wet conditions using two drought indices across KwaZulu-Natal Province, South Africa. Atmosphere 11:623. https://doi.org/10.3390/atmos11060623
Ndlovu M, Clulow AD, Savage MJ, Nhamo L, Magidi J, Mabhaudhi T (2021) An assessment of the impacts of climate variability and change in KwaZulu-Natal Province, South Africa. Atmosphere 12:427. http://dx.doi.org/10.3390/atmos12040427
Ngcobo S.P, Wilson A-L, Downs CT (2018) Home ranges of Cape porcupines on farmlands, peri-urban and suburban areas in KwaZulu-Natal, South Africa. Mammalian Biology 96:102–109. https://doi.org/10.1016/j.mambio.2018.10.006
Ofstad EG, Herfindal I, Solberg EJ, Sæther B-E (2016) Home ranges, habitat and body mass: simple correlates of home range size in ungulates. Proceedings of the Royal Society B: Biological Sciences 283:20161234. https://doi.org/10.1098/rspb.2016.1234
Okeke DC (2021) Prospects for sustainable urban development in Africa – (re)viewed from a planning perspective. International Planning Studies 26(2):198–217. https://doi.org/10.1080/13563475.2020.1785278
Pascoe JH, Flesch JS, Duncan MG, Le Pla M, Mulley RC (2019) Territoriality and seasonality in the home range of adult male free-ranging lace monitors (*Varanus varius*) in South-eastern Australia. Herpetological Conservation and Biology 14:97-104.
Pernetta A (2009) Monitoring the trade: using the CITES trade database to examine the dynamics of the trade in live monitor lizards (*Varanus* spp.). Biawak: Quarterly Journal of Varanid Biology and Husbandry 3:37–45.
Phillips JA (1995) Movement patterns and density of *Varanus albigularis*. Journal of Herpetology 29:407-416.
Pillay KR, Streicher JP, Downs CT (2024) Trends in vervet monkey admissions to a wildlife rehabilitation centre: a reflection of human-wildlife conflict in an urban-forest mosaic landscape. Mammalian Biology. https://doi.org/10.1007/s42991-024-00447-x
Price C, Ezat MA, Hanzen C, Downs CT (2022a) Never smile at a crocodile: Gaping behaviour in the Nile crocodile at Ndumo Game Reserve, South Africa, Behavioural Processes 203:104772. https://doi.org/10.1016/j.beproc.2022.104772
Price C, Burnett M, O'Brien GC, Downs C (2022b) Presence and temporal activities of serrated hinged terrapin (*Pelusios sinuatus*) and marsh terrapin (*Pelomedusa galeata*) in KwaZulu-Natal, South Africa, assessed using telemetry. Tropical Conservation Science 15:194008292210742. https://doi.org/10.1177/19400829221074241
Price C, Hanzen C, Downs C (2021) Demographics and morphometrics of marsh terrapins (*Pelomedusa galeata*) and serrated hinged terrapins (*Pelusios sinuatus*) populations in KwaZulu-Natal, South Africa: skewed size-class bias concerns. Zoomorphology 140:1–9. https://doi.org/10.1007/s00435-021-00518-4
Purwandana D, Ciofi C, Imansyah M, Ariefiandy A, Rudiharto H, Jessop T (2021) Prey preferences and body mass most influence movement behavior and home range area of Komodo dragons. Ichthyology and Herpetology 109. https://doi.org/10.1643/h2020028
Ramesh T, Downs CT (2014) Modelling large spotted genet (*Genetta tigrina*) and slender mongoose (*Galerella sanguinea*) occupancy in a heterogeneous landscape of South

Africa. Mammalian Biology 79(5):331–337. https://doi.org/10.1016/j.mambio.2014.05.001.
R Core Team (2023) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
Reaney LT, Whiting MJ (2003) Picking a tree: habitat use by the tree agama, *Acanthocercus atricollis atricollis*, in South Africa. African Zoology 38(2):273–278. https://doi.org/10.1080/15627020.2003.11407281
Roberts CJ, Vergés A, Callaghan CT, Poore AGB (2022) Many cameras make light work: opportunistic photographs of rare species in iNaturalist complement structured surveys of reef fish to better understand species richness. Biodiversity and Conservation 31(4):1407–1425. https://doi.org/10.1007/s10531-022-02398-6
Roberts D, Diederichs Mander N (2002) Durban's Local Agenda 21 programme: Tackling sustainable development in a post-apartheid city. Environment and Urbanization 14:189–201. https://doi.org/10.1177/095624780201400116
Rowe-Rowe DT, Taylor PJ (1996) Distribution patterns of terrestrial mammals in KwaZulu-Natal. South African Journal of Zoology 31:131–144. https://doi.org/10.1080/02541858.1996.11448404
Seto KC, Güneralp B, Hutyra LR. (2012) Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proceedings of the National Academy of Sciences 109:16083–16088. https://doi.org/10.1073/pnas.1211658109
Shin Y, Kim K, Groffen J, Woo D, Song E, Borzée A (2022) Citizen science and roadkill trends in the Korean herpetofauna: The importance of spatially biased and unstandardized data. Frontiers in Ecology and Evolution 10:944318. https://www.frontiersin.org/articles/10.3389/fevo.2022.944318
Singh N, Price C, Downs CT (2021) Aspects of the ecology and behaviour of a potential urban exploiter, the southern tree agama, *Acanthocercus atricollis*. Urban Ecosystems 24:905–914. https://doi.org/10.1007/s11252-020-01078-z
Smith JG, Griffith, AD (2009) Determinants of home range and activity in two semi-aquatic lizards. Journal of Zoology 279(4):349–357. https://doi.org/10.1111/j.1469-7998.2009.00623.x.
Stanner M, Mendelssohn H (1987) Sex ratio, population density and home range of the desert monitor (*Varanus griseus*) in the Southern Coastal Plain of Israel. Amphibia-Reptilia 8:153-163. https://doi.org/10.1163/156853887X00414
Steen D, Barbour M, McClure C, Wray K, Macey J, Stevenson D (2015) Landscape scale habitat selection of harlequin coralsnakes (*Micrurus fulvius*) in three large, protected areas in the southeastern United States. Copeia 103:1037–1042. https://doi.org/10.1643/CE-15-235
Streicher JP, Ramesh T, Downs CT (2020) Home range and core area utilisation of three co-existing mongoose species: large grey, water and white-tailed in the fragmented landscape of the KwaZulu-Natal Midlands, South Africa. Mammalian Biology 100(3):273–283. https://doi.org/10.1007/s42991-020-00028-8
Sutherland K, Ndlovu M, Pérez-Rodríguez A (2018) Use of artificial waterholes by animals in the southern region of the Kruger National Park, South Africa. African Journal of Wildlife Research 48(2):1–14. https://doi.org/10.3957/056.048.023003
Sweet SS (1999) Spatial ecology of *Varanus glauerti* and *V. glebopalma* in northern Australia. Mertensiella 11:317-366.
Thaker M, Amdekar M, Mohanty N, Nageshkumar A, Prakash H, Seshadri K (2022) An expanding cityscape and its multi-scale effects on lizard distribution. Frontiers in Conservation Science 3. https://doi.org/10.3389/fcosc.2022.839836

Thamaga KH, Dube T, Shoko C (2022) Evaluating the impact of land use and land cover change on unprotected wetland ecosystems in the arid-tropical areas of South Africa using the Landsat dataset and support vector machine. Geocarto International 37(25):10344–10365. https://doi.org/10.1080/10106049.2022.2034986

Thatcher HR, Downs CT, Koyama NF (2020) Understanding foraging flexibility in urban vervet monkeys, *Chlorocebus pygerythrus*, for the benefit of human-wildlife coexistence. Urban Ecosystems, 23(6):1349–1357. https://doi.org/10.1007/s11252-020-01014-1

Thompson G. (1994) Activity area during the breeding season of *Varanus gouldii* (Reptilia: Varanidae) in an urban environment. Wildlife Research 21:633-641.

Thompson GG, de Boer M, Pianka ER (1999) Activity areas and daily movements of an arboreal monitor lizard, *Varanus tristis* (Squamata: Varanidae) during the breeding season. Australian Journal of Ecology 24(2):117-122.

Tolley K, Alexander G, Branch W, Bowles P, Maritz B (2016) Conservation status and threats for African reptiles. Biological Conservation 204: 63-71. https://doi.org/10.1016/j.biocon.2016.04.006

Trivalairat P, Srikosamatara S (2023) Daily activities of water monitors (*Varanus salvator macromaculatus* Deraniyagala, 1944) in urban wetland, Bangkok, Thailand. Herpetozoa 36:189–201. https://doi.org/10.3897/herpetozoa.36.e93492

Uyeda L (2009) Garbage appeal: Relative abundance of water monitor lizards (Varanus salvator) correlates with presence of human food leftovers on Tinjil Island, Indonesia. Biawak 3:9–17.

Waldner F, Hansen M, Potapov P, Löw F, Newby T, Ferreira S, Defourny P (2017) National-scale cropland mapping based on spectral-temporal features and outdated land cover information. PLoS One 12:e0181911. https://doi.org/10.1371/journal.pone.0181911

Warrier R, Noon B, Bailey L (2021) A framework for estimating human-wildlife conflict probabilities conditional on species occupancy. Frontiers in Conservation Science 2:679028. https://doi.org/10.3389/fcosc.2021.679028

Weavers BW (1993) Home range of male lace monitors, *Varanus varius* (Reptilia : Varanidae), in south-eastern Australia. Wildlife Research 20:303–313. https://doi.org/10.1071/wr9930303

Western D (1974) The distribution, density and biomass density of lizards in a semi–arid environment of northern Kenya. African Journal of Ecology 12(1):49–62. https://doi.org/10.1111/j.1365-2028.1974.tb00106.x

Whiting M, Williams V, Hibbitts T (2011) Animals traded for traditional medicine at the Faraday Market in South Africa: Species diversity and conservation implications. Journal of Zoology 284:84–96. https://doi.org/10.1007/978-3-642-29026-8_19

Wilms T, Wagner P, Luiselli L, Branch WR, Penner J, Baha El Din S, Beraduccii J, Msuya CA, Howell K, Ngalason W (2021) *Varanus niloticus*. The IUCN Red List of Threatened Species 2021:e.T198539A2531945.

Wongtienchai P, Lapbenjakul S, Jangtarwan K, Areesirisuk P, Mahaprom R, Subpayakom N, Singchat W, Sillapaprayoon S, Muangmai N, Songchan R, Baicharoen S, Duengkae P, Peyachoknagul S, Srikulnath K. (2020) Genetic management of a water monitor lizard (*Varanus salvator macromaculatus*) population at Bang Kachao Peninsula as a consequence of urbanization with Varanus Farm Kamphaeng Saen as the first captive research establishment. Journal of Zoological Systematics and Evolutionary Research 59:1–14. https://doi.org/10.1111/jzs.12436

Woolley CK, Hartley S, Innes JG, Shanahan DF, van Heezik Y, Wilson DJ, Nelson NJ (2023) Conservation of skinks in New Zealand cities. Urban Ecosystems 26:1493–1508. https://doi.org/10.1007/s11252-023-01398-w

CHAPTER 3

Population dynamics and morphometrics of Nile monitors along a gradient of urbanisation in KwaZulu-Natal, South Africa

Euan E.Z. Genevier[1], Cormac Price[1,2], Nick Evans[1,3] and Colleen Downs[1*]

[1]*Centre for Functional Biodiversity, School of Life Sciences, P/Bag X01, Scottsville, Pietermaritzburg, 3209, South Africa.*

[2] *The HerpHealth lab. Unit for Environmental Sciences and Management, North-West University, Potchefstroom, South Africa 2531*

[3] *KZN Amphibian and Reptile Conservation.*

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*** Corresponding author:** Colleen T. Downs

Email: downs@ukzn.ac.za; ORCID: http://orcid.org/0000-0001-8334-1510

Other emails and ORCIDs:

Euan Genevier ORCID: https://orcid.org/0009-0006-4457-3195

Cormac Price Email: [redacted]; ORCID: https://orcid.org/0000-0002-9092-0796

Nick Evans Email: [redacted]

Running header: Population size and morphology of Nile monitors

3.1 Abstract

Globally, with the continuous increase in human populations, land use is changing at an unprecedented rate. Little is known about how African reptiles respond to urbanisation. We used mark-resight and baited camera traps to compare population density and demographics of Nile monitors (*Varanus niloticus*) on a gradient of land use, namely a golf course, a farm and a nature reserve in KwaZulu-Natal, South Africa, during 2022-2023. We also collected detailed morphometrics on 112 Nile monitors throughout KwaZulu-Natal. We compared morphometric data on limbs, head, body and mass to establish whether the different ecological factors in urban and rural environments and associated behavioural shifts have resulted in unique phenotypes. No significant morphometric differences were found, but urban individuals were generally smaller and lighter than their rural counterparts. Population size estimates varied between sites. Overall, population size estimates increased with anthropogenic disturbance between our three focal sites, while the mean estimated snout-vent length decreased. The naïve presence of competing diurnal mammals did not follow a specific pattern. The overall insignificant variations in morphometrics throughout urban and rural areas of KwaZulu-Natal suggests that Nile monitors have adapted to urban mosaic landscapes but are not as successful as some urban-exploiting varanids globally. Future research should investigate Nile monitor interactions with people and pets as well as conflict mitigation strategies.

Keywords: *Varanus*; urban ecology; morphometrics; population estimate; mark-resight; urban adapter

3.2 Introduction

Humans have directly transformed more than half of the Earth's land surface and continue to do so in conjunction with a fast-growing population (Hooke et al., 2012). Sub-Saharan Africa is projected to experience the largest population growth of all continents (Gu et al., 2021; Brum et al., 2022). Much of Africa's urban development is fast and unplanned, posing a threat to sustainable development (Cobbinah et al., 2015; Okeke, 2021). The conservation of persisting wildlife is key to sustainable anthropogenic development and human well-being (Kowarik et al., 2020; Brum et al., 2022). However, little is documented about the response of African wildlife to urbanisation and agriculture, especially in reptiles (Tolley et al., 2016; Downs et al., 2021; Brum et al., 2022; Woolley et al., 2023). Studies on urban reptiles often lack detailed morphological and behavioural data (French et al., 2018).

Nile monitors (*Varanus niloticus* [Linnaeus, 1758]) are the largest lizard in Africa and the fifth largest lizard in the world. They are widespread, yet understudied compared with other large charismatic lizards (Angelici & Luiselli, 1999; de Buffrénil & Rimblot-Baly, 1999; de Buffrénil & Castanet, 2000; de Buffrénil & Hémery, 2002; Enge et al., 2004; Dowell et al., 2015). Large females can produce up to 60 eggs, showing one of the largest single clutch sizes among varanids (Thompson & Pianka, 2001; Auliya & Koch, 2020). However, many species' clutch sizes remain unknown, and others can breed year-round, produce multiple clutches a year and possibly exceed the reproductive output of Nile monitors (de Buffrénil & Rimblot-Baly, 1999; Retes & Bennett, 2001; Zeeuw, 2010; Auliya & Koch, 2020).

Morphology can be affected by multiple aspects of a species' biology, such as retreat choice, habitat preference, physiology and feeding strategy (Thompson & Withers, 1997; Thompson et al., 2008; Collar, Schulte & Losos, 2011). There is a trend towards terrestrial monitors being exceptionally large and both rock-dwelling and arboreal species being small, suggesting that habitat use strongly influences the evolution of size and shape in monitor lizards

(Collar et al., 2011). Nile monitors are amongst Africa's most heavily exploited tetrapods, with estimates in Sahelian Africa reaching 2 million individuals hunted in peak years (de Buffrénil & Hémery, 2002; Dowell et al., 2015). Heavily exploited populations in Sahelian Africa have reduced longevity, mean age and snout-vent lengths (SVL) and accelerated initial growth with early sexual maturation compared with less heavily exploited populations (de Buffrénil & Hémery, 2002). Populations in West Africa appear to be declining because of habitat loss and extensive hunting for food, leather and the pet trade (Bio Oure et al., 2016).

Differences in food availability in anthropogenic landscapes are known to affect varanid populations (Uyeda, 2009; Jessop, 2012; Guerrero-Sanchez et al., 2021, 2022, 2023). Asian water monitors (*V. salvator*) and lace monitors (*V. varius*) are known to exploit refuse tips and grow considerably in body mass and population size (Uyeda, 2008; Jessop, 2012). The abundance of Asian water monitors in areas with anthropogenic food waste and wild areas of Tinjil Island, Indonesia, differed greatly with 4 individuals/km^2 in wild areas and 1400 individuals/km^2 in areas inhabited by humans (Uyeda, 2009; Uyeda et al. 2013; 2015). Habitat occupancy of Komodo dragons in the Flores region is negatively affected by proximity to farms and villages (Ariefiandy et al., 2021). There is generally an abundance of prey associated with urbanisation but likely at a lesser level than refuse tips (Singh & Downs, 2016a, b; Singh et al., 2021; Downs et al., 2021; Lettoof et al., 2023). The extent of population growth in Nile monitors in response to human food sources is unknown, but dense populations have been recorded in areas of Kenya and Florida, USA (Western, 1974; Enge et al., 2004; Wood et al., 2016). The well-established invasive population in Florida shows Nile monitors' ecological plasticity (Enge et al., 2004; Savarie et al., 2011; Arida, 2008; Dowell et al., 2016; Eckles et al., 2017; Briggs-Gonzalez et al., 2022). They are breeding successfully and consume a wide array of native and invasive species found locally (Eckles et al., 2017; Mazzotti et al., 2020).

Both mark-resight and mark-recapture methods are commonly used to estimate abundance, monitor trends, and study the movement of wildlife (McClintock et al., 2014; Erb et al., 2015). Camera traps are low cost, low maintenance, and create minimal disturbance to the animal and can provide data on date, time, activity patterns, temperature, and individual identification of animals if they have natural or artificial individually distinct marks (Alonso et al., 2015; Moore et al., 2020; Purwandana et al., 2021).

There is a need to further understand urban and rural population trends and dynamics of Nile monitors to inform management (de Buffrénil & Hémery, 2002; Ejigu & Tassie, 2020). In this study, we investigate the demographics and abundance of Nile monitors on a gradient of anthropogenic disturbance using mark-resight and camera traps at three sites in KwaZulu-Natal, South Africa. Additionally, we collected morphological data from Nile monitors throughout KwaZulu-Natal to identify province-wide trends between urban and rural populations and recorded negative human/domestic animal interactions with Nile monitors. Our research questions were: Do Nile monitor population sizes change with increased anthropogenic activity? Are their demographics affected by anthropogenic activity? Have Nile monitors adapted morphologically to new ecological factors associated with urban environments? We predicted that rural Nile monitor populations would have larger individuals and less variation in body condition, that some urban areas would have a higher population size than in nature reserves, that Nile monitors have adapted morphologically to new ecological factors associated with urban environments, and that there is more evidence of potentially competing diurnal mesocarnivores in rural areas than urban habitats.

3.3 Methods

3.3.1 Study area for morphometrics

The province of KwaZulu-Natal is located on the east coast of South Africa and borders Lesotho, Eswatini, and Mozambique internationally, and the provinces of the Free State, Eastern Cape and Mpumalanga nationally, as well as a coastline of some 600 km along the Indian Ocean (Ndlovu & Demlie, 2020). Its climate is heavily influenced by the Indian Ocean, with high temperatures and rainfall in the austral summer that often rise above 25°C and can exceed 40°C during the day and cool, while there are dry winters that average ~ 20°C (Fairbanks & Benn, 2000; Ndlovu & Demlie, 2020). The mean annual rainfall in KwaZulu-Natal ranges from 650 mm in eastern grasslands to 1400 mm in eastern Coastal Bushveld (Kruger & Nxumalo, 2017; Ndlovu et al., 2021; Mugiyo et al., 2022). This influence of relatively mild winters and high humidity creates favourable conditions for relatively large reptiles in KwaZulu-Natal, eastern Limpopo and eastern Mpumalanga (Alexander, 2007; Bates et al., 2014). All the Nile monitors measured in this study were caught in Pietermaritzburg, Durban and surrounding areas (Supplementary information Fig. S3.1).

3.3.2 Study areas for population density

Tala Collection Game Reserve (hereafter Tala) is a 3300 ha protected area situated south of Pietermaritzburg at approximately 650-700 m.a.s.l (Supplementary information Fig. S3.1). It is surrounded by agricultural land (sugar cane and cattle) and settlements with nearby impoundments that feed into the Umhlatuzana River and its catchment area (Humphries et al., 2016; Price et al., 2022). There are 13 artificial dams and pans, with others outside the reserve boundary.

Maritzburg Golf Course (hereafter golf course) is situated within the city of Pietermaritzburg, the provincial capital of KwaZulu-Natal (Supplementary information Fig.

S3.1). It is surrounded by residential and industrial areas and borders a wastewater treatment site. We have included Ascot Bush Lodge (29°36'12.1"S 30°24'38.6"E) in this field site as we believe the population is connected through proximity and the Msunduzi River. It borders an industrial area and the N3 motorway. Together, this is approximately a 134 ha area. The golf course has nine small ponds and is traversed by the Blackborough stream, which flows into the uMsunduzi River nearby, both of which are highly polluted (Bulannga & Schmidt, 2022; Ngubane et al., 2023). Ascot Bush Lodge borders the uMsunduzi River and has two artificial ponds. The mean monthly maximum and minimum temperatures in Pietermaritzburg range from 30.7°C to 17.9°C, and rainfall is greater in summer (Singh & Downs, 2021).

Bar Circle Ranch (hereafter farm) is an approximately 320 ha cattle and game farm (Supplementary information Fig. S3.1). It is situated 6 km south of Pietermaritzburg (29°40'41.9"S 30°28'51.6"E) and is predominantly grazed grassland with nine different artificial dams fed by a tributary of the Mpushini River. All nine dams are close by, ranging from 130 m to 390 m distance from each other. The farm is surrounded by other cattle farms, is near a motorway and borders a conservancy. It is thus more modified than Tala and less modified than the golf course. All three of these field sites are within 25 km of each other.

3.3.3 Nile monitor captures

Nile monitors were captured between July 2022 and October 2023 either by hand, sometimes using welding gloves (Johnsons, South Africa), or using baited step plate traps. Trap dimensions ranged from 300 x 300 x 900 to 700 x 700 x 1350 mm. The main bait used was chicken intestines (~400 g). Additionally, chicken necks or fish were hung from the trigger mechanism to increase our ability to trap small individuals. Each trap was faced with a camera trap (Moultrie MCG-13098, USA or Browning BTC-5HDPX, USA) to prevent theft and poaching, monitor by-catch, and record animal behaviour.

Traps were placed on water edges, covered for shade, or placed in attics and gardens if targeting particular urban individuals. Dead specimens were included if they were recently deceased and were in reasonable condition. Most urban specimens were caught by hand in people's gardens or homes and relocated to suitable green spaces in a 10 km radius to avoid human/domestic animal-Nile monitor conflict. Many of these removals happened during conflicts, which may affect Nile monitor populations and were therefore recorded as evidence of negative human interactions. This study was approved by the University of KwaZulu-Natal Animal Ethics Committee (reference number: AREC/00004360/2022) and Ezemvelo KZN Wildlife (permit number: OP 2258/2022).

3.3.4 Population estimates

Population size was estimated at three sites using mark-resight. Nile monitors were marked using unique colour combinations of 10 mm x 10 mm x 3 mm glass beads sutured to the tail crest (Fig. 3.1). They were sutured about a third of the way down the tail using two threads of sterilised 19 kg strength 0.6 mm thick copolymer monofilament fishing line (Adrenalin, South Africa) and further secured with cyanoacrylate glue (Q-Bond, South Africa). Animals were restrained by hand and given a glove to bite on. A total of two or four glass beads of one or two different colours were used per individual and mirrored on both sides of the tail, maximising visibility from multiple angles. Four easily distinguishable colours were used, allowing 20 unique combinations per field site. Hatchlings were not externally marked, and sub-adults under 375 mm SVL only received one pair of beads. The yellow stripe on which the beads were placed also varied and was noted in case any beads came off after release.

After marking individuals, one to two baited camera trap stations were deployed near water edges at each site. We opted for baited camera traps to maximise detection, an effective method used for Komodo dragons (*V. komodoensis*; Purwanda et al., 2021). Each station

comprised of a wire mesh cage baited with either chicken intestines, chicken necks or goat, and two camera traps. Camera traps were set on motion detect with minimal trigger delay (1 to 5 seconds delay, depending on the model) and were placed about 2 m away from the bait, one facing from above and one from the side. These stations were rebaited on the same day every two to four weeks. The golf course and farm each had one station, while Tala had two stations to account for the difference in size and distance between marked individuals. Stations were placed in suitable areas within 5 m of the edge of dams where monitors were regularly seen (pers. comm.).

Figure 3.1: An example of our Nile monitor bead marking system (left) and a resighting 46 days after release (right).

We attempted to use version 5b of MegaDetector to sort through photographs using pattern recognition (Beery et al., 2019; Fennell et al., 2022). However, the threshold had to be set very low (>0.05 confidence) because of questionable detection rates for reptiles, and

ultimately, all daytime photographs were viewed by the authors. Although some varanids exhibit nocturnal activity, we have found no evidence of such behaviour in Nile monitors (Rhind et al., 2013; Uyeda et al., 2013; Clarkson & Massyn, 2020). Initially, all photographs were inspected in the field, where the lack of nocturnal sightings was noted. Due to the lack of evidence of nocturnal activity, photographs were removed if they were taken between 23h00 and 03h59. We used photographs from 16 February to 28 August 2023 for the golf course. We used photographs from 08 March to 21 November 2023 for the farm. We used photographs from 16 February 2023 to 20 August 2023 for Tala. Camera malfunctions were considered to ensure each site had working cameras for 174 days between these timeframes. We defined an independent detection event as consecutive triggers separated by over 15 min. difference unless the individual seen was visibly different (Moore et al., 2020). When possible, we used the known dimensions of each bait cage and of marked individuals to estimate the SVL of each sighting. Estimates of SVL in perenties (*Varanus giganteus*) based on camera trap photographs varied between 6 and 78 mm (Moore et al., 2020). Due to varying positions, distances and angles of monitors towards the camera, we estimated SVL at ± 50 mm accuracy. We confirmed our accuracy by estimating the SVL of marked individuals, all of which were within the ± 50 mm accuracy threshold. When processing photographs, monitors were labelled as "marked", "unmarked" or "unconfirmed" if the tail was not visible during the independent trigger event. Unconfirmed trigger events were excluded from analyses.

We used three equations to calculate population density to mitigate potential biases (Mercer et al., 2013). These assume population closure, equal likelihood of capture amongst individuals and sampling efforts, and that the marking does not come off nor affect their survival or mobility (Mercer et al., 2013). We used the Peterson estimate (Krebs, 1999):

$$\mathbf{N_{est} = (M \times S)/R}$$

Where N_{est} is the estimated population size index, M is the number of individuals marked, S is the total number of sightings, and R is the number of resightings. The Peterson equation is widely used, is therefore repeatable, and allows comparison (Krebs, 1999). However, it can result in biased estimates, which can be mitigated by the inclusion of the following equations: The Seber equation (Seber, 1982; Krebs, 1999; Mercer et al., 2012):

$$\mathbf{N_{est} = [(M + 1)(S + 1)]/(R + 1) - 1}$$

Lastly, we used the Bailey equation (Bailey, 1951):

$$\mathbf{N_{est} = [M(S + 1)]/(R + 1)}$$

We recorded detections of some potentially competing mesocarnivores (e.g., mongooses, genets, servals, jackals and otters). These were quantified as the number of independent trigger events per day of trapping.

3.3.5 Morphometric data collection

We recorded the geographical location coordinates for each Nile monitor capture with a GPS (Garmin, Olathe, USA). We took a total of 11 measurements similar to those of Thompson et al. (2008)(Fig. 3.2). Tail length and SVL were measured using a measuring tape at ±2 mm accuracy (Gelmar, South Africa). Upper limbs were measured using small 280 mm analogue callipers at ± 1 mm accuracy (SKU KEN35311, Kendo, China). Tail, head and body widths, head and lower limb lengths, and head depth were measured using large 650 mm analogue callipers (Haglof, S-88200, Sweden) at ± 1 mm accuracy. Body mass was recorded using a digital spring scale at 10 g accuracy (Sensation Tom Thumb 490005, South Africa) by weighing the monitor in a closed container and subtracting the container's mass.

Varanids are notoriously hard to sex in the field (Mayes et al., 2005; Rovatsos et al., 2015). Nile monitors are particularly hard to sex, thus studies that include sex determination are limited to dead specimens (e.g., De Buffrenil, 1999; Awad Alshehri & Hassaneen, 2021;

Dalhuijsen, Branch & Alexander, 2014). We could only sex individuals if they were gravid or, in rare cases, of non-partial hemipenes or hemiclitoris eversion (Funk, 2002). This was rare, so sex could not be included in our analyses.

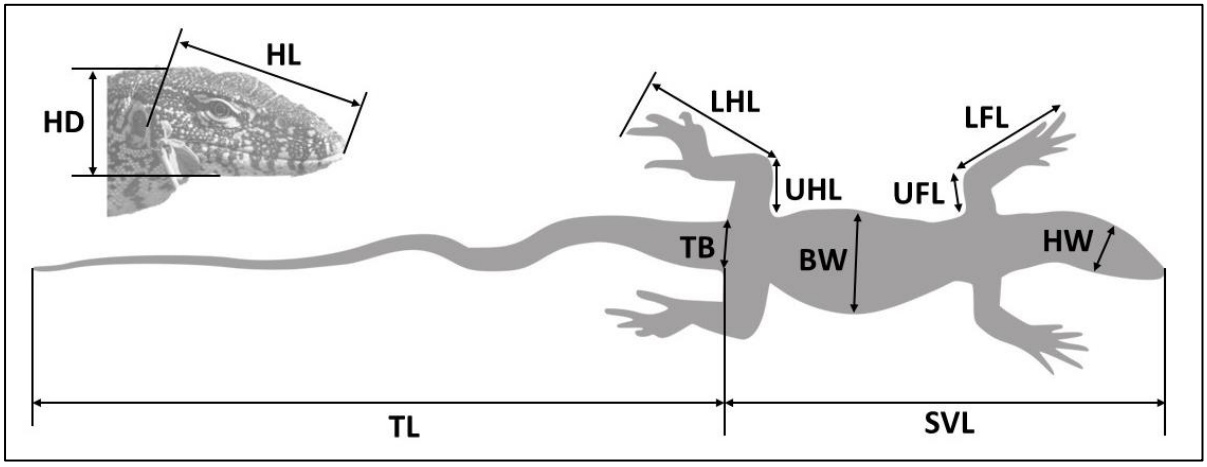

Figure 3.2: Morphometric data recorded in this study with TL = tail length; SVL = snout-vent length; TB = tail base; BW = body width; LHL = lower hindlimb; UHL = upper hindlimb; LFL = lower forelimb; UFL = upper forelimb; HL = head length; HW = head width; HD = head depth. (Silhouette source: https://www.phylopic.org/).

Each individual was marked using a BackHome Super Mini Passive Integrative Transponder (PIT) tag to avoid double counts (Virbac, South Africa). Recaptured individuals were remeasured to gain insight into their growth, but we excluded these from morphometric analyses.

We used descriptive statistics and analysis of covariance (ANCOVA) with SVL as a covariate to account for body size to summarise data and test the variance in means of morphological data and body mass of adults in IBM SPSS Statistics (version 22_2022, IBM Inc., Chicago). We used linear regressions between log transformed body mass and log transformed SVL for body condition indices (BCI) (Green, 2001; Labocha et al., 2014).

3.4 Results

3.4.1 Population size estimates

We externally marked 29 Nile monitor individuals across three usable sites: 11 at the golf course, five at the farm and 13 at Tala. One marked individual from Tala was translocated to the site after receiving veterinary treatment. It was resighted once 56 days later, only 38 m from its release. Another from the farm was translocated from an urban garden 6850 m away from its release location. This individual was never resighted or recaught. When cumulating distances between sightings, a 4095 g and 555 mm SVL marked individual travelled a strict minimum of 2590 m between 11 January 2023 and 16 May 2023 (125 days; Supplementary information Fig. S3.1).

We obtained 203,939 photographs between 16 February 2023 and 11 October 2023. Most came from Tala (81%), followed by 14% from the farm and 5% from the golf course. After excluding sightings outside of our used timeframe of 174 days, we had 90 Nile monitor sightings at the golf course, of which 28% (n = 25) were marked, and 18% (n = 16) were unconfirmed. The farm had 76 sightings, of which 16% (n = 12) were marked, and 21% (n = 16) were unconfirmed. Tala had 175 sightings, with 61% (n = 82) marked and 10% (n = 13) unconfirmed. We calculated the highest population density estimates at the golf course (mean index = 32.6), followed by the farm (mean index = 25.2), and the lowest estimate was in Tala (mean index = 21.1; Table 3.1). The farm had the most independent trigger events of mongooses per day. The golf course had the lowest number of mammalian mesocarnivore trigger events (Table 3.1). The average estimated SVLs per site were 386 ± 93 mm (n = 83) at the golf course, 434 ± 121 mm (n = 70) at the farm and 452 ± 68 mm (n = 193) at Tala.

Table 3.1: Population size estimates of Nile monitors and independent trigger events per day of several mammal species in the three sites used in this study. Population estimates are given with minimum and maximum 95 % bootstrap confidence intervals.

Site	Lincoln–Petersen estimate (min-max)	Seber estimate (min-max)	Bailey estimate (min-max)	Mongoose /day	Genet /day	Serval /day	Jackal /day	Otter /day
Golf	32.6 (24.7 - 47.9)	33.6 (21.0 - 44.8)	31.7 (24.3 - 45.8)	0.06	0.00	0.00	0.00	0.00
Farm	25.0 (16.7 - 50.0)	27.2 (13.1 - 38.0)	23.5 (16.1 - 43.6)	0.85	0.10	0.01	0.04	0.02
Tala	21.0 (18.8 - 23.8)	21.5 (16.5 - 24.5)	20.9 (18.8 - 23.7)	0.06	0.08	0.00	0.01	0.02

3.4.2 Urban and rural morphometrics

We measured and weighed 112 Nile monitors between July 2022 and October 2023. Rural individuals ranged from 25 to 10340 g (n = 32, mean = 3855, SD = 2626) and 110 to 735 mm in SVL (n = 32, mean = 511, SD = 142). Urban individuals ranged from 25 to 9550 g (n = 79, mean = 2676, SD = 2248) and 114 to 700 mm in SVL (n = 80, mean = 455, SD = 147). Individual body mass increased with SVL, with more variation observed in the urban adult population than in the rural adult population (Fig. 3.3). On average, the body condition index was slightly higher in rural individuals (BCI = 0.960, SE = 0.095) than urban (BCI = 0.957, SE = 0.122).

Three urban individuals were gravid, two of whom were kept in captivity until eggs were laid; dogs killed the third, and eggs were removed. In all three cases, the mass without eggs was used for analysis. Due to the low sample size of rural juveniles, we had to compare adults only for analysis (SVL > 375 mm). We found no significant differences in morphology

(P > 0.05 throughout) or body mass (P = 0.53) between urban and rural individuals. There was, however, a general trend of bulkier and heavier individuals in rural areas (Fig. 3.3; Fig 3.4; Fig 3.5; Fig. 3.6; Supplementary Table S3.1).

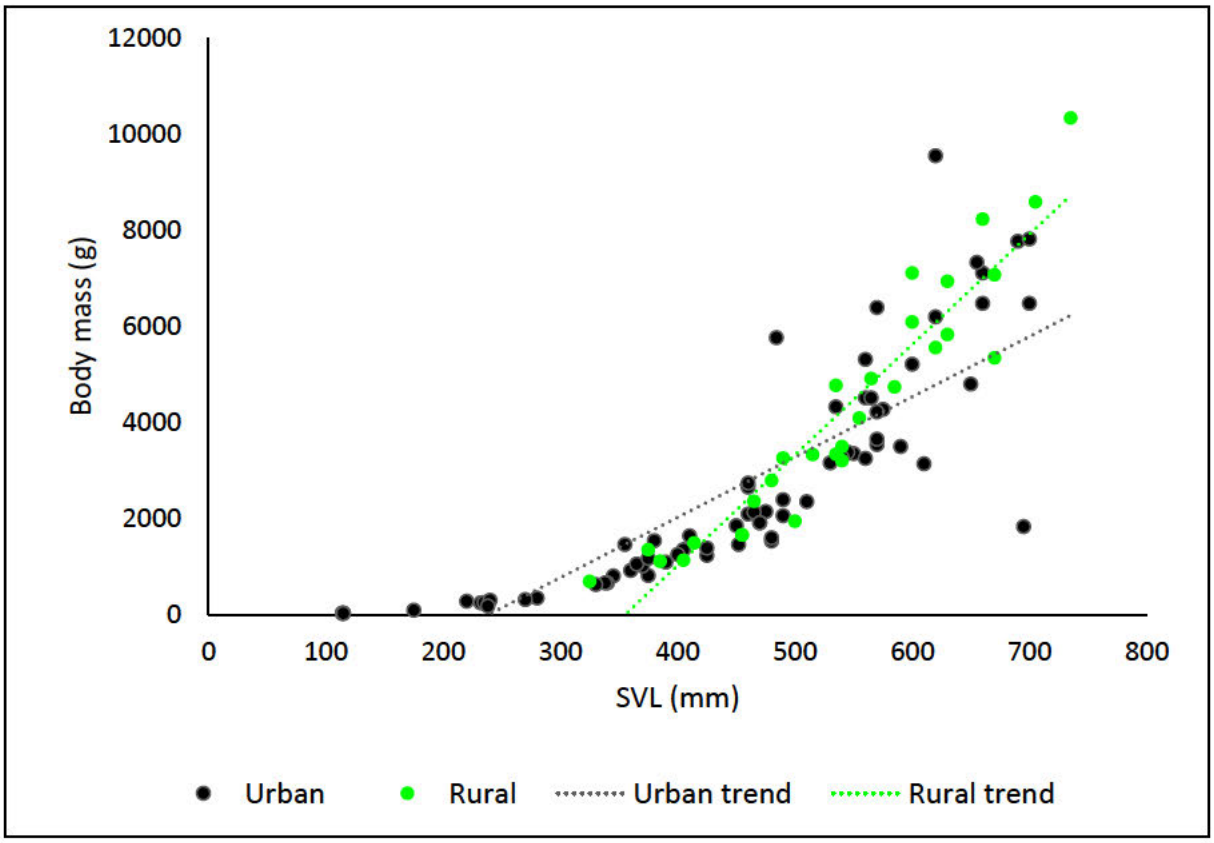

Figure 3.3: Nile monitor body mass and snout-vent length (SVL) in urban (n = 80) and rural (n = 32) areas of KwaZulu-Natal in the present study.

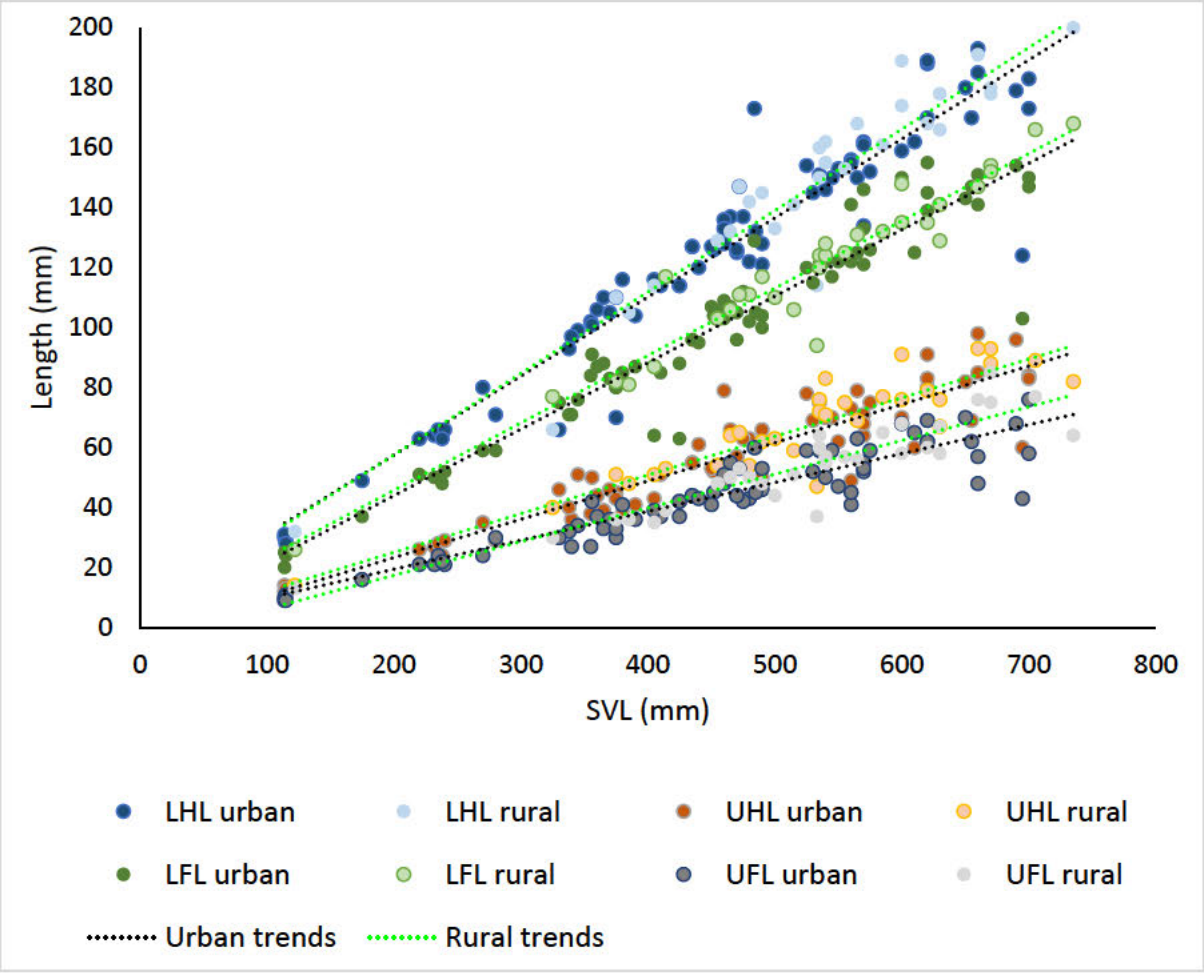

Figure 3.4: Nile monitor morphometric data by snout-vent length in urban and rural individuals used in this study. (Note: LHL = lower hindlimb; UHL = upper hindlimb; LFL = lower forelimb; UFL = upper forelimb).

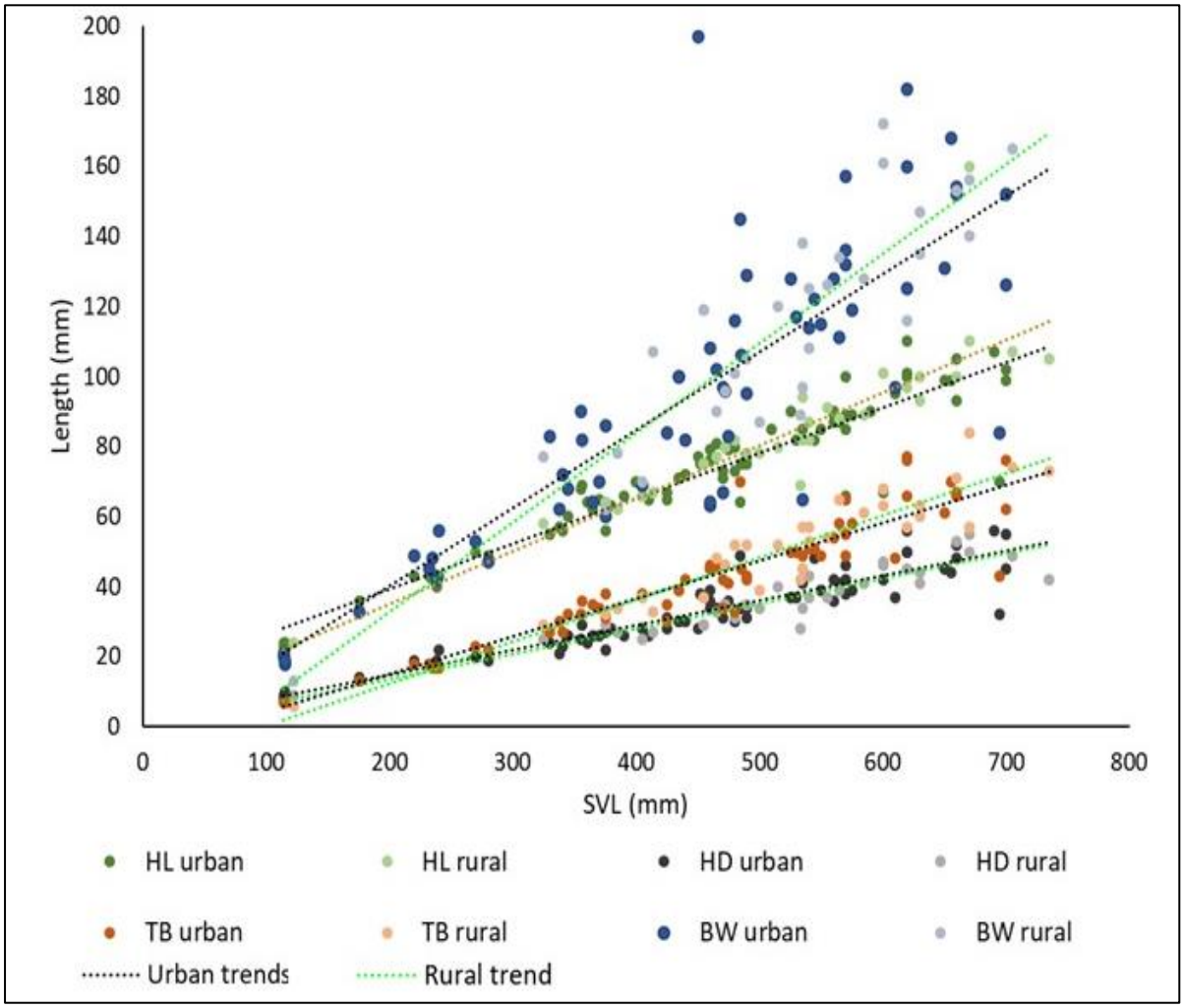

Figure 3.5: Nile monitor morphometric data by snout-vent length in urban and rural individuals used in this study. (Note: with HL = head length; HD = head depth; TB = tail base; BW = body width).

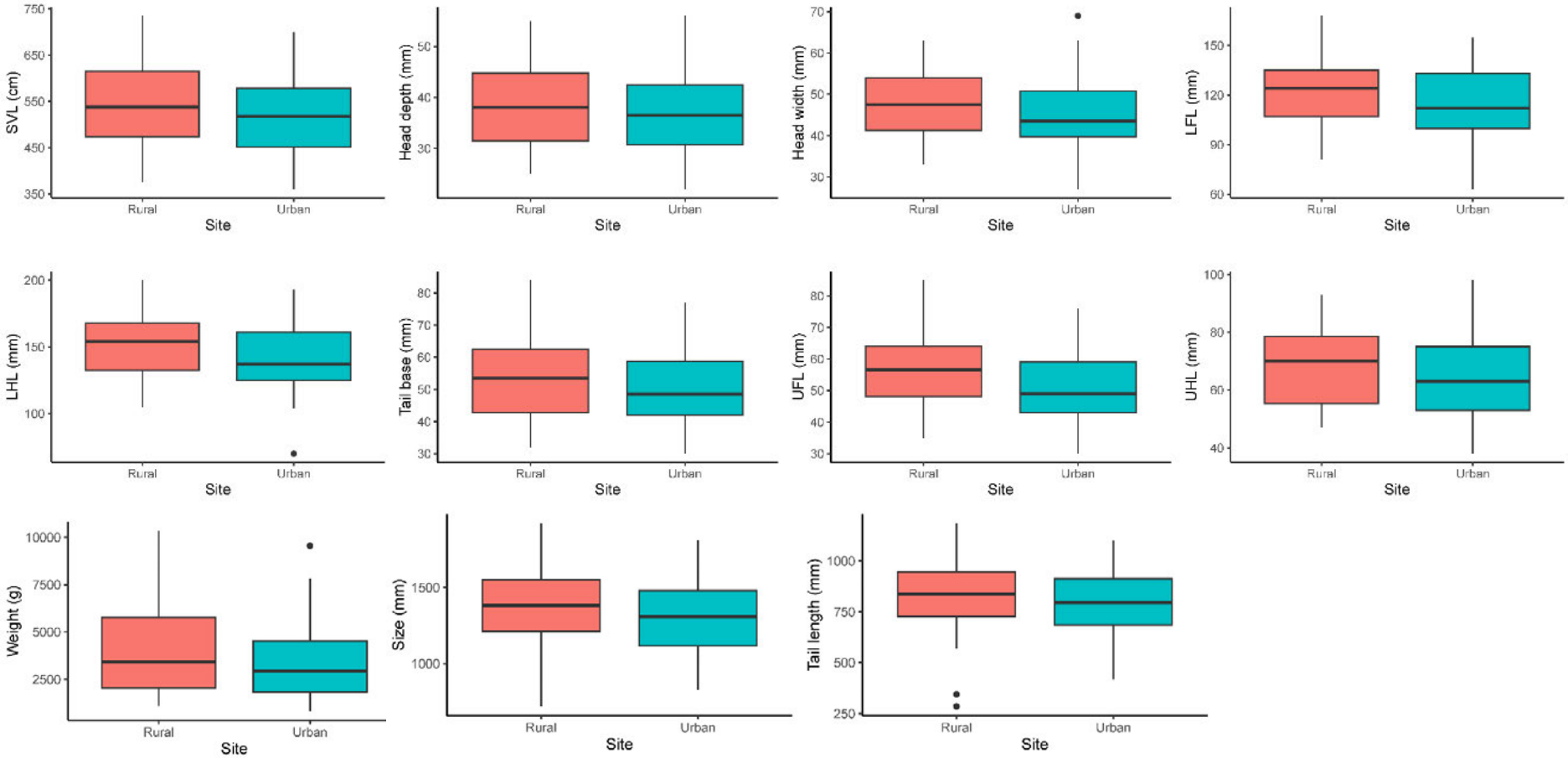

Figure 3.6: Comparative means of morphometric lengths and body mass between urban and rural adult Nile monitors. (Note: Whiskers show 95 % confidence intervals. SVL = snout-vent length; LFL = lower forelimb; LHL = lower hindlimb, UFL = upper forelimb; UHL = upper hindlimb; Red = rural, Blue = urban).

3.4.3 Negative human-Nile monitor interactions

During our study, we found evidence of illegal hunting of Nile monitors on the golf course and neighbouring wastewater treatment facility. An unidentified person was caught on a camera trap, removing a young adult monitor from our trap on 2nd June 2023. He was then seen killing and bagging the carcass. The whole event happened within 45 min. of us visiting the trap. We reported the incident and removed our traps immediately. On 2nd September 2023, we found and removed five snare traps made of nylon builders' line set around the edges of golf ponds, some in the open, tethered to sticks pushed into the ground, others amongst bushes, tethered to branches. We believe these targeted monitors because of their placement, size, previous incidence of poaching, and discussion with the golf course manager. More snares were found on the next three consecutive visits. Illegal hunters and their hunting dogs (*Canis lupus familiaris*) have also been reported from the neighbouring wastewater treatment plant (pers. comm. Gula, Downs unpublished data).

We found Nile monitor individuals (n = 11) with severe or fatal injuries caused by domestic dog conflicts, severe or fatal injuries likely from interactions with people (n = 4), and others from fatal vehicle collisions (n = 3; Supplementary information Fig. S3.2).

3.5 Discussion

The population size of Nile monitors at our three sites varied with increased anthropogenic activity. The golf course had an estimated 32.6 (± 11.6) Nile monitors, which was 8.0 (± 16.7) more than the farm and 11.0 (± 2.5) more than Tala. Increased abundance with anthropogenic activities was also recorded in Asian water monitors, even in areas with high levels of harvesting for local and international trade (Twining et al., 2017; Khadiejah et al., 2019). Nile monitors were both consistently more abundant and smaller on average, with increased anthropogenic intensity between our field sites, not accounting for area. Nile monitors in Lake

Chad reach sexual maturity at 360 ± 20 mm SVL, and males in Chad and Mali reach sexual maturity at 360 mm and 400 mm, respectively (de Buffrénil & Rimblot-Baly, 1999; de Buffrénil et al., 1999). Although this may differ slightly in South African individuals, it suggests that the average Nile monitor at the golf course is just reaching sexual maturity. Moreover, reproductive output in the species increases considerably with female size (de Buffrénil & Rimblot-Baly, 1999; Auliya & Koch, 2020).

Overall, our results suggest that the farm and Tala may have a higher proportion of large Nile monitor individuals capable of producing large clutches than the golf course. Despite this, the overall abundance was lower. We suggest that there may be less predation on juveniles at the golf course than in the less intensive anthropogenic sites but that larger specimens are more at risk of persecution. Our study found evidence of illegal hunting on the golf course and a neighbouring wastewater treatment facility. It is difficult to say whether the perceived reduced size class in the population results from direct persecution or other anthropogenic factors. In Sahelian regions of Africa, the average age of Nile monitors decreases with increasing levels of exploitation (De Buffrénil & Hémery 2002). This may be a factor in urban areas, where larger individuals are targeted but also more likely detected by dogs, people and vehicles. We could not assess the age of individuals as this requires dead specimens for skeletochronology (De Buffrénil & Castanet, 2000). Tree agamas (*Acanthocerus atricollis atricollis*) in a communal settlement in South Africa were found significantly more abundant in villages than adjacent communal rangelands despite an overwhelmingly negative perception of people towards and 84% of respondents actively deterring the species (Whiting et al., 2009). Another event that may have affected population demographics at the golf course was an intensive clearing of trees and debris in April 2023, including some dead trees which were frequently used for basking (pers. obs.). Future studies would benefit from more study sites for population size estimates to identify broader trends. There is a need to assess levels of

exploitation of this protected yet targeted species in South Africa (Alexander, 1990; Whiting et al., 2011). We also recommend investigating the potential impacts of chemical runoff and other forms of pollution associated with rivers in cities, such as Durban, and increased flooding events, which have likely doubled in frequency in Durban in the last century (Lebepe et al., 2022; Grab & Nash, 2023).

The golf course had the least evidence of potentially competing diurnal mesocarnivores. The main mesocarnivores found were mongooses with considerable overlap in their opportunistic diet with Nile monitors, such as crustaceans (Dalhuijsen et al., 2014; Streicher et al., 2022). This potentially reflects a similar pattern to Southeast Asian water monitors (*V. salvator macromaculatus*) in which their relative occupancy increased with land use intensity (oil palm plantation > logged forest > primary forest) while Malay civets (*Viverra tangaluna*) noticeably decreased, albeit insignificantly and with variability (Twining et al., 2017).

We marked and released 60 Nile monitor individuals throughout the urban mosaic landscape of Durban with PIT tags and only one individual has been recaught. This lack of recaptured monitors in Durban suggests a considerable population size, low survivorship of translocated animals, or that individuals remain within the boundaries of their release location (urban green spaces). We cannot make any conclusions regarding the Durban population size; however, this outlined the benefits of mark-resight over mark-recapture for this elusive species. An individual was sighted in Tala with noticeable notches where beads had been sutured. Three individuals were subsequently recaptured with these notches, confirming that some of our sutured beads were ripped off, which systematically resulted in two distinct marks. Fortunately, we varied the yellow band on which beads were placed between individuals and sites and kept photographs. These individuals were marked again and successfully identified on photographs with and without beads. One large individual at Tala (SVL = 705 mm) was resighted, with all four beads still attached 493 days after initial release. Most sightings throughout the camera

trapping in Tala had beads (60%). Labelling individual animals with unique and distinct markers is necessary in many ecological studies (Johnson, 2005). Permanent markings by morphological altercation, such as toe clipping, have been widely used in amphibians and reptiles, including monitor lizards, but are often hard to see from a distance reliably, often lack ethical considerations and reduce survivability in many species (Perry et al., 2011; Plummer & Fernet, 2013; Galdino, Horta & Young, 2014; Olivera-Tlahuel et al., 2017). Another permanent varanid marking is tail notching (Bennett, 2000). This method involves cutting a "V" shaped notch from the lizards' tail crest. However, Nile monitors often have similar notches in the wild from natural or anthropogenic causes (pers. obs.). Pattern recognition software can also be used in species with naturally occurring unique pattern variations between individuals (Gardiner et al., 2014; Moore et al., 2020).

The most common method of marking monitor lizards is using paint, but it is not permanent (e.g., Phillips, 1995; Phillips & Millar, 1998; Jessop et al., 2012). Sutured beads were first used to permanently mark a lizard in 1989 (Fisher & Muth, 1989; Galdino et al., 2014). Many studies have used and improved the method (Galdino et al., 2014; Singh et al., 2021). To our knowledge, this is the first record of beads used on a monitor lizard. We were able to recognise marked and most unmarked Nile monitor individuals. One large individual was resighted, with all four of its beads attached after over a year. Most sightings throughout the camera trapping in Tala had beads. We believe the combination of PIT tagging and sutured beads (if the placement is varied along the tail and recorded) is a successful method to mark semi-aquatic varanids for extensive periods. However, the ability of individuals to bite them off should not be overlooked.

Despite the decrease in estimated size classes found between our sites with increased anthropogenic intensity, there was no significant difference in morphology between urban and rural individuals at a provincial level. However, a general trend was toward heavier, bulkier

adults in rural areas. Globally, urban lizards tend to have access to better resources and more time for foraging, resulting in larger body sizes than their rural counterparts (Putman & Tippie, 2020). This was observed in lace monitors, where individuals were significantly bigger and bolder in anthropogenically modified habitats and areas with human-mediated trophic subsidies (Jessop et al., 2012; Pettit et al., 2021a,b). Asian water monitors are also known to increase in size with land use intensity and the presence of human food scraps (Uyeda 2009; Twining et al., 2017). They can consume human food waste (~ 36.7 % of the diet mass of the population in Malaysia) and even include unusual items like cooked rice (Uyeda 2009; Karunarathna et al., 2012, 2015; Yu et al., 2021). Although the diet of urban Nile monitors is unknown, a broader diet which includes abundant and highly accessible items may facilitate urban exploitation and increased body mass more for Asian water monitors than Nile monitors. Urban dugites (*Pseudonaja affinis*) in Western Australia also don't differ greatly to their rural counterparts and are generally smaller (Wolfe et al., 2018). Similar to these snakes, Nile monitors have adapted and are persisting in urban environments, but do not seem to greatly benefit from urban resources like true urban exploiters do. However, details of their urban diet are needed to reach such conclusions, as other factors may affect their body condition such as pollution or parasite load (Lazić et al., 2017; Lettoof et al., 2021). Our results suggest that despite persisting in urban environments, Nile monitors are not as successful in anthropogenic areas as some other varanids globally. We found individuals with severe wounds from negative interactions with people, dogs, and vehicle collisions. Differences in diet and anthropogenic stressors such as conflicts, heavy pollution and poor diet quality may be linked to the slight trend in smaller and lighter individuals in urban areas. The overall insignificant differences suggest that Nile monitors are urban adapters but are not as significantly different from their rural counterparts as in some other varanid species.

3.6 Conclusions

This study found evidence of localised response to anthropogenic land use in KwaZulu-Natal. The golf course had the largest population size estimate of Nile monitors, the farm had the second largest population estimate, and the game reserve had the smallest. However, the estimated size of individuals decreased with anthropogenic intensity. The average estimated size of Nile monitors at the golf course was 386 mm, just over the size of sexual maturity in females. We also found poaching of Nile monitors at the golf course and the least evidence of potentially competing diurnal mesocarnivores. We suggest juveniles have a higher survival rate at the golf course than in rural areas, while larger individuals may be more at risk. The game reserve had the largest average size class, the smallest population size estimate and the highest diversity of potentially competing diurnal mesocarnivores. The morphology of adults at a provincial scale did not vary significantly but showed a general trend towards larger and heavier individuals in rural areas than in urban. Overall, the lack of significant differences in urban morphology at a provincial level suggests that Nile monitors are successful urban adapters but not significantly different from their rural counterparts as in some other similar varanid species globally. Future studies should focus on the impact of environmental pollution, diet, and human dimensions of urban monitors to better understand the perceived localised trends in urban mosaic landscapes.

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3.8 References

Alexander, G.J. (1990). Reptiles and amphibians of Durban. *Durban Museum Novitates*, 15(1), 1–41. https://doi.org/10.10520/AJA0012723X_1939.

Alexander, G.J. (2007). Thermal biology of the Southern African Python (*Python natalensis*): does temperature limit its distribution. In: Henderson, R.W. and Powell, R. (Eds), *Biology of the boas and pythons.* Eagle Mountain Publishing, Eagle Mountain, Utah. pp. 50–75.

Alonso, R.S., McClintock, B.T., Lyren, L.M., Boydston, E.E. & Crooks, K.R. (2015). Mark-recapture and mark-resight methods for estimating abundance with remote cameras: a carnivore case study. *PLoS One*, 10(3), pe0123032. https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0123032

Angelici, F.M. & Luiselli, L. (1999). Aspects of the ecology of *Varanus niloticus* (Reptilia, Varanidae) in Southeastern Nigeria, and their contribution to the knowledge of the evolutionary history of *V. niloticus* species complex. *Revue d'Ecologie, Terre et Vie*, 54(1), 29–42.

Arida, E. & Evy (2008). An overview of the ecology of varanid lizards. *Zoo Indonesia*, 17, 65–82.

Ariefiandy, A., Purwandana, D., Azmi, M., Nasu, S.A., Mardani, J., Ciofi, C. & Jessop, T.S. (2021). Human activities associated with reduced Komodo dragon habitat use and range loss on Flores. *Biodiversity and Conservation*, 30(2), 461–479. https://doi.org/10.1007/s10531-020-02100-8.

Auliya, M. & Koch, A. (2020). Visual identification guide for the monitor lizard species of the world (Genus *Varanus*) : guidance for the identification of monitor lizards with current distribution data as well as short explanations on reproductive characteristics and captive breeding to support CITES authorities. Deutschland / Bundesamt für Naturschutz. https://doi.org/10.19217/skr552.

Awad, M., Alshehri, M. & Hassaneen, A.S.A. (2021). Histological, histochemical, and ultrastructural approach to the ductus deferens in male Nile monitor lizard (*Varanus niloticus*). *Microscopy and Microanalysis*, 27(4), 935–940. https://doi.org/10.1017/S1431927621012046.

Bailey, N.T.J. (1951). On estimating the size of mobile populations from capture-recapture data. *Ecology*, 63. 103-112.

Bates, M.F., Branch, W.R., Bauer, A.M., Burger, M., Marais, J., Alexander, G.J. & de Villiers, M.S. (2014). Atlas and Red List of Reptiles of South Africa, Lesotho and Swaziland. Animal Demography Unit (Cape Town) and the South African National Biodiversity Institute (Pretoria).

Beery, S., Morris, D., Yang, S., Simon, M., Norouzzadeh, A. & Joshi, N. (2019). Efficient pipeline for automating species ID in new camera trap projects. *Biodiversity Information Science and Standards*, 3. https://doi.org/10.3897/biss.3.37222.

Bennett, D. (2000) The density and abundance of juvenile *Varanus exanthematicus* (Sauria: Varanidae) in the coastal plain of Ghana. *Amphibia-Reptilia*, 21(3), 301–306. https://doi.org/10.1163/156853800507499.

Bennett, D. & Retes, F. (2001). Multiple generations, multiple clutches, and early maturity in four species of monitor lizards (Varanidae) bred in captivity. *Herpetological Review*, 32, 244–255.

Bio Oure, R., Daouda, I.-H., Valentin, K. & Mensah, G. (2016). Inventaire, structure morpho-métrique et importance des varans sacrés de Kandi (Nord-Est Bénin). *International Journal of Biological and Chemical Sciences*, 9, 2663. https://doi.org/10.4314/ijbcs.v9i6.12.

Brum, P.H.R., Gonçalves, S.R.A., Strüssmann, C. & Teixido, A.L. (2022). A global assessment of research on urban ecology of reptiles: patterns, gaps and future directions. *Animal Conservation*, 26(1), 1–13. https://doi.org/10.1111/acv.12799.

Bulannga, R. & Schmidt, S. (2022). Uptake and accumulation of microplastic particles by two freshwater ciliates isolated from a local river in South Africa. *Environmental Research*, 204, 112123. https://doi.org/10.1016/j.envres.2021.112123.

Cissé, M. (1972). L'alimentation des Varanides au Senegal. Bulletin de l'Institut François d'Afrique Noire 34: 503515.

Clarkson, M. & Massyn, D. (2020): Nocturnal hunting activity of *Varanus salvator* in Goa Lalay Cave, Pelabuhan Ratu, Indonesia. *Biawak*, 14(1–2), 79–91

Cobbinah, P.B., Erdiaw-Kwasie, M.O. & Amoateng, P. (2015). Africa's urbanisation: Implications for sustainable development. *Cities*, 47, 62–72. https://doi.org/10.1016/j.cities.2015.03.013.

Collar, D.C., Schulte, J.A. & Losos, J.B. (2011). Evolution of extreme body size disparity in monitor lizards (*Varanus*). *Evolution; International Journal of Organic Evolution*, 65(9), 2664–2680. https://doi.org/10.1111/j.1558-5646.2011.01335.x.

Dalhuijsen, K., Branch, W.R. & Alexander, G.J. (2014). A comparative analysis of the diets of *Varanus albigularis* & *Varanus niloticus* in South Africa. *African Zoology*, 49, 83–93. https://doi.org/10.1080/15627020.2014.11407621

de Buffrénil, V., Castanet, J. & Rimblot, F. (1999). Maturation génitale des varans du nil mâles (*Varanus niloticus*) dans trois populations du Sahel. *Canadian Journal of Zoology*, 77(2), 222–232. https://doi.org/10.1139/z98-216.

de Buffrénil, V. & Rimblot-Baly, F. (1999). Female reproductive output in exploited Nile monitor lizard (*Varanus niloticus* L.) populations in Sahelian Africa. *Canadian Journal of Zoology*, 77, 1530–1539. https://doi.org/10.1139/z99-130

de Buffrénil, V. & Castanet, J. (2000). Age estimation by skeletochronology in the Nile monitor (*Varanus niloticus*), a highly exploited species. *Journal of Herpetology*, 34, 414. https://doi.org/10.2307/1565365

de Buffrénil, V. & Hémery, G. (2002). Variation in longevity, growth, and morphology in exploited Nile monitors (*Varanus niloticus*) from Sahelian Africa. *Journal of Herpetology*, 36, 419. https://doi.org/10.2307/1566186

Dowell, S.A., de Buffrénil, V., Kolokotronis, S.-O. & Hekkala, E.R. (2015). Fine-scale genetic analysis of the exploited Nile monitor (*Varanus niloticus*) in Sahelian Africa. *BMC Genetics*, 16, 32. https://doi.org/10.1186/s12863-015-0188-x

Dowell, S.A., Portik, D.M., de Buffrénil, V., Ineich, I., Greenbaum, E., Kolokotronis, S.-O. & Hekkala, E.R. (2016). Molecular data from contemporary and historical collections

reveal a complex story of cryptic diversification in the *Varanus* (Polydaedalus) *niloticus* Species Group. *Molecular Phylogenetics and Evolution*, 94(Pt B), 591–604. https://doi.org/10.1016/j.ympev.2015.10.004.

Eckles, J.K., Mazzotti, F., Giardina, D., Hazelton, D. & Rodgers, H.L. (2017). First evidence for reproduction of Nile monitors (*Varanus niloticus*) in Palm Beach County. *Southeastern Naturalist*, 15, 114–119.

Ejigu, D. & Tassie, N. (2020). Present and future suitability of the Lake Tana Biosphere Reserve in Ethiopia for the Nile monitor (*Varanus niloticus*) using the MaxEnt model. *Environmental Systems Research*, 9(1), 1–15. https://doi.org/10.1186/s40068-020-00197-y.

Enge, K., Krysko, K., Hankins, K., Campbell, T. & King, A. (2004). Status of the Nile monitor (*Varanus niloticus*) in Southwestern Florida. *Southeastern Naturalist*, 3, 571–582. https://doi.org/10.1656/1528-7092(2004)003[0571:SOTNMV]2.0.CO;2.

Erb, L.A., Willey, L.L., Johnson, L.M., Hines, J.E. & Cook, R.P. (2015). Detecting long-term population trends for an elusive reptile species. *Journal of Wildlife Management*, 79(7), 1062–1071. https://doi.org/10.1002/jwmg.921.

Fairbanks, D.H.K. & Benn, G.A. (2000). Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. *Landscape and Urban Planning*, 50, 237-257. https://doi.org/10.1016/S0169-2046(00)00068-2

Fennell, M., Beirne, C. & Burton, A.C. (2022). Use of object detection in camera trap image identification: Assessing a method to rapidly and accurately classify human and animal detections for research and application in recreation ecology. *Global Ecology and Conservation*, 35, e02104. https://doi.org/10.1016/j.gecco.2022.e02104.

Fisher, M., & Muth, A. 1989. A technique for permanently marking lizards. *Herpetological Review*. 20:45–46.

French, S.S., Webb, A.C., Hudson, S.B. & Virgin, E.E. (2018). Town and country reptiles: A review of reptilian responses to urbanization. *Integrative and Comparative Biology*, 58(5), 948–966. https://doi.org/10.1093/icb/icy052.

Funk, R.S. (2002). Lizard reproductive medicine and surgery. *Veterinary Clinics: Exotic Animal Practice*, 5(3), 579–613. https://doi.org/10.1016/S1094-9194(02)00018-X.

Galdino, C., Horta, G. & Young, R. (2014). An update to a bead-tagging method for marking lizards. *Herpetological Review*, 45, 587.

Gardiner, R., Doran, E., Strickland, K., Carpenter-Bundhoo, L. & Frere, C. (2014). A Face in the Crowd: A Non-Invasive and Cost Effective Photo-Identification Methodology to Understand the Fine Scale Movement of Eastern Water Dragons. *PLoS One*, 9, e96992. https://doi.org/10.1371/journal.pone.0096992.

Grab, S.W. & Nash, D.J. (2023). A new flood chronology for KwaZulu-Natal (1836–2022): the April 2022 Durban floods in historical context. *South African Geographical Journal*, 106(4), 476–497. https://doi.org/10.1080/03736245.2023.2193758.

Gu, D., Andreev, K. & Dupre, M.E. (2021). Major trends in population growth around the World. *China CDC Weekly*, 3(28), 604–613. https://doi.org/10.46234/ccdcw2021.160.

Guerrero-Sanchez, S., Goossens, B., Saimin, S. & Orozco-terWengel, P. (2021). The critical role of natural forest as refugium for generalist species in oil palm-dominated landscapes. *PLoS One*, 16(10), e0257814. https://doi.org/10.1371/journal.pone.0257814

Guerrero-Sanchez, S., Majewski, K., Orozco-terWengel, P., Saimin, S., Goossens, B. (2022). The effect of oil palm-dominated landscapes on the home range and distribution of a generalist species, the Asian water monitor. *Ecology and Evolution*, 12, e8531. https://doi.org/10.1002/ece3.8531

Guerrero-Sanchez, S., Frias, L., Saimin, S., Orozco-terWengel, P. & Goossens, B. (2023). The fast-food effect: costs of being a generalist in a human-dominated landscape. *Conservation Physiology*, 11(1), coad055. https://doi.org/10.1093/conphys/coad055.

Green, A.J. (2001) Mass/length residuals: Measures of body condition or generators of spurious results? *Ecology*, 82(5), 1473–1483. https://doi.org/10.1890/0012-9658(2001)082[1473:MLRMOB]2.0.CO;2.

Hooke, R.LeB. & Martín-Duque, J.F. (2012). Land transformation by humans: A review. *Geological Society of America Today*, 12(12), 4–10. https://doi.org/10.1130/GSAT151A.1.

Humphries, B.D., Ramesh, T., Hill, T.R. & Downs, C.T. (2016). Habitat use and home range of black-backed jackals (*Canis mesomelas*) on farmlands in the Midlands of KwaZulu-Natal, South Africa. *African Zoology*, 51(1), 37–45. https://doi.org/10.1080/15627020.2015.1128356.

Jessop, T.S., Smissen, P., Scheelings, F. & Dempster, T. (2012). Demographic and phenotypic effects of human mediated trophic subsidy on a large Australian Lizard (*Varanus varius*): Meal ticket or last supper? *PLoS One*, 7(4), e34069. https://doi.org/10.1371/journal.pone.0034069.

Johnson, M. (2005). A new method of temporarily marking lizards. *Herpetological Review*, 36, 277–279.

Karunarathna, S., Amarasinghe, A.A.T., Madawala, M. & Kandambi, D. (2012) Population status of two Varanus species (Reptilia: Sauria: Varanidae) in Sri Lanka's Puttalam Lagoon System, with notes on their diet and conservation status. *Biawak*, 6, 22–33.

Karunarathna, S., Surasinghe, T., Silva, M.C., Madawala, M., Gabadage, D. & Botejue, M. (2015) Dietary habits of *Varanus salvator salvator* in Sri Lanka with a new record of predation on an introduced clown knifefish *Chitala ornata*. *Herpetological Bulletin*, 133, 23–28.

Khadiejah, S., Razak, N., Ward-Fear, G., Shine, R. & Natusch, D.J.D. (2019). Asian water monitors (*Varanus salvator*) remain common in Peninsular Malaysia, despite intense harvesting. *Wildlife Research*, 46(3), 265–275. https://doi.org/10.1071/WR18166.

Kowarik, I., Fischer, L.K. & Kendal, D. (2020). Biodiversity conservation and sustainable urban development. *Sustainability*, 12(12), 4964. https://doi.org/10.3390/su12124964.

Krebs, C.J. (1999). *Ecological methodology*. 2. Addison Wesley Longman, Inc; Menlo Park, CA.

Kruger, A.C. & Nxumalo, M.P. (2017). Historical rainfall trends in South Africa: 1921–2015. *Water SA*, 43(2), 285–297. https://doi.org/10.4314/wsa.v43i2.12.

Labocha, M.K., Schutz, H. & Hayes, J.P. (2014). Which body condition index is best? *Oikos*, 123(1), 111–119. https://doi.org/10.1111/j.1600-0706.2013.00755.x.

Lazić, M.M., Carretero, M.A., Živković, U. & Crnobrnja-Isailović, J. (2017). City life has fitness costs: reduced body condition and increased parasite load in urban common wall lizards, *Podarcis muralis*. *Salamandra*, *53*(1).

Lebepe, J., Khumalo, N., Mnguni, A., Pillay, S. & Mdluli, S. (2022). Macroinvertebrate assemblages along the longitudinal gradient of an urban Palmiet River in Durban, South Africa. *Biology*, 11(5), 705. https://doi.org/10.3390/biology11050705.

Lettoof, D.C., Cornelis, J., Jolly, C.J., Aubret, F., Gagnon, M.M., Hyndman, T.H., Barton, D.P. & Bateman, P.W. (2022). Metal(loid) pollution, not urbanisation nor parasites predicts low body condition in a wetland bioindicator snake, *Environmental Pollution*, 295, 118674. https://doi.org/10.1016/j.envpol.2021.118674.

Lettoof, D.C., Parkin, T., Jolly, C.J., de Laive, A. & von Takach, B. (2023). Snake life history traits and their association with urban habitat use in a tropical city. *Urban Ecosystems*, 26(2), 433–445. https://doi.org/10.1007/s11252-023-01327-x.

Mayes, P., Bradshaw, D. & Bradshaw, F. (2005). Successfully Determining the sex of adult *Varanus mertensi* (Reptilia: Varanidae) using a combination of both hemipenile eversion and the ratio of androgens: estradiol in plasma. *Annals of the New York Academy of Sciences*, 1040, 402–5. https://doi.org/10.1196/annals.1327.074.

Mazzotti, F.J., Nestler, J.H., Cole, J.M., Closius, C., Kern, W.H., Rochford, M.R., Suarez, E., Brubaker, R., Platt, S.G., Rainwater, T. & Ketterlin, J.K. (2020). Diet of Nile monitors (*Varanus niloticus*) removed from Palm Beach and Broward Counties, Florida, USA. *Journal of Herpetology*, 54(2), 189–195. https://doi.org/10.1670/18-115.

McClintock, B.T., Hill, J.M., Fritz, L., Chumbley, K., Luxa, K. & Diefenbach, D.R. (2014). Mark-resight abundance estimation under incomplete identification of marked individuals. *Methods in Ecology and Evolution*, 5(12), 1294–1304. https://doi.org/10.1111/2041-210X.12140.

McClintock, B.T., Lyren, L.M., Boydston, E.E. & Crooks, K.R. (2015). Mark-recapture and mark-resight methods for estimating abundance with remote cameras: A carnivore case study. *PLoS One*, 10(3), e0123032. https://doi.org/10.1371/journal.pone.0123032.

Mercer, D. R., Marie, J., Bossin, H., Faarula, M., Tetuanula, A., Sang, M. C. & Dobson, S. L. (2012). Estimation of population size and dispersal of *Aedes polynesiensis* on Toamaro Motu, French Polynesia. *Journal of Medical Entomology*, 49(5), 971–980.

Moore, H.A., Champney, J.L., Dunlop, J.A., Valentine, L.E. & Nimmo, D.G. (2020). Spot on: using camera traps to individually monitor one of the world's largest lizards. *Wildlife Research*, 47(4), 326–337. https://doi.org/10.1071/WR19159.

Mugiyo, H., Chimonyo, V.G.P., Kunz, R., Sibanda, M., Nhamo, L., Ramakgahlele Masemola, C., Modi, A.T. & Mabhaudhi, T. (2022). Mapping the spatial distribution of underutilised crop species under climate change using the MaxEnt model: A case of KwaZulu-Natal, South Africa. *Climate Services*, 28, 100330. https://doi.org/10.1016/j.cliser.2022.100330.

Ndlovu, M. & Demlie, M. (2020). Assessment of meteorological drought and wet conditions using two drought indices across KwaZulu-Natal Province, South Africa. *Atmosphere*, 11, 623. https://doi.org/10.3390/atmos11060623.

Ndlovu, M., Clulow, A.D., Savage, M.J., Nhamo L., Magidi, J. & Mabhaudhi, T. (2021). An assessment of the impacts of climate variability and change in KwaZulu-Natal Province, South Africa. *Atmosphere* 12, 427. . http://dx.doi.org/10.3390/atmos12040427

Ngubane, Z., Dzwairo, B., Moodley, B., Stenström, T.A. & Sokolova, E. (2023). Quantitative assessment of human health risks from chemical pollution in the uMsunduzi River, South Africa. *Environmental Science and Pollution Research*, 30(55), 118013–118024. https://doi.org/10.1007/s11356-023-30534-4.

Okeke, D.C. (2021). Prospects for sustainable urban development in Africa – (re)viewed from a planning perspective. *International Planning Studies*, 26(2), 198–217. https://doi.org/10.1080/13563475.2020.1785278.

Olivera-Tlahuel, C., Pérez-Mendoza, H.A., Zúñiga-Vega, J.J., Rubio-Rocha, L.C., Bock, B.C., Rojas-González, R.I., Zamora-Abrego, J.G., Alzate, E., Ortega-León, A.M., Maceda-Cruz, R.J., la Cruz, F.R.M., Siliceo-Cantero, H.H. & Serna-Lagunes, R. (2017). Effect of toe-clipping on the survival of several lizard species. *Herpetological Journal*, 27(3), 266–275.

Perry, G., Wallace, M.C., Perry, D., Curzer, H. & Muhlberger, P. (2011). Toe clipping of amphibians and reptiles: Science, ethics, and the law. *Journal of Herpetology*, 45(4), 547–555.

Pettit, L., Brown, G.P., Ward-Fear, G. & Shine, R. (2021a). Anthropogenically modified habitats favor bigger and bolder lizards. *Ecology and Evolution*, 11(4), 1586–1597. https://doi.org/10.1002/ece3.7124.

Pettit, L., Crowther, M.S., Ward-Fear, G. & Shine, R. (2021b). Divergent long-term impacts of lethally toxic cane toads (*Rhinella marina*) on two species of apex predators (monitor lizards, *Varanus* spp.). *PLoS One*, 16(7), e0254032. https://doi.org/10.1371/journal.pone.0254032.

Phillips, J.A. (1995). Movement patterns and density of *Varanus albigularis. Journal of Herpetology*, 29(3), 407–416. https://doi.org/10.2307/1564991.

Phillips, J.A. & Millar, R.P. (1998). Reproductive biology of the white-throated savanna Monitor, *Varanus albigularis. Journal of Herpetology*, 32(3), 366–377. https://doi.org/10.2307/1565451.

Plummer, M. & Ferner, J. (2012). Marking Reptiles. In: Dodd C.K. Jr. (Ed.), *Reptile Ecology and Conservation: A Handbook of Techniques*. Oxford University Press, Oxford. 143–150.

Price, C., Burnett, M.J., O'Brien, G. & Downs, C.T. (2022). Presence and temporal activities of serrated hinged terrapin (*Pelusios sinuatus*) and marsh terrapin (*Pelomedusa galeata*) in KwaZulu-Natal, South Africa, assessed using telemetry. *Tropical Conservation Science*, 15(1). https://doi.org/10.1177/19400829221074241.

Purwandana, D., Ariefiandy, A., Azmi, M., Nasu, S.A., Sahudin, Dos, A.A., Jessop, T.S., Purwandana, D., Ariefiandy, A., Azmi, M., Nasu, S.A., Sahudin, Dos, A.A. & Jessop, T.S. (2021). Turning ghosts into dragons: improving camera monitoring outcomes for a cryptic low-density Komodo dragon population in eastern Indonesia. *Wildlife Research*, 49(4), 295–302. https://doi.org/10.1071/WR21057.

Putman, B.J. & Tippie, Z.A. (2020). Big city living: A global meta-analysis reveals positive impact of urbanization on body size in lizards. *Frontiers in Ecology and Evolution*, 8. https://www.frontiersin.org/article/10.3389/fevo.2020.580745

Rhind, D., Doody, J., Ellis, R., Ricketts, A., Scott, G., Clulow, S. and McHenry, C. (2013). *Varanus glebopalma* (black-palmed monitor) nocturnal activity and foraging, *Herpetological Review*, 44, 687–688.

Rovatsos, M., Johnson Pokorná, M., Altmanová, M., Kratochvil, L., Velensky, P., Vodička, R. & Rehak, I. (2015). Sexing of Komodo Dragons, *Varanus komodoensis*. *Gazella*, 42, 92–107.

Savarie, P., Engeman, R., Mauldin, R., Mathies, T. & Tope, K. (2011). Tools for managing invasions: acceptance of non-toxic baits by juvenile Nile monitor lizards and Burmese pythons under laboratory conditions. *International Journal of Pest Management*, 57, 309–314. https://doi.org/10.1080/09670874.2011.598581.

Seber, G.A.F. (1982). The Estimation of Animal Abundance and Related Parameters. 2nd edition. Griffin, London.

Singh, N., Price, C. & Downs, C.T. (2021). Aspects of the ecology and behaviour of a potential urban exploiter, the southern tree agama, *Acanthocercus atricollis. Urban Ecosystems*, 24(5), 905–914. https://doi.org/10.1007/s11252-020-01078-z.

Singh, P. & Downs, C.T. (2016a). Hadedas in the hood: Hadeda ibis activity in suburban neighbourhoods of Pietermaritzburg, KwaZulu-Natal, South Africa. *Urban Ecosystems*, 19(3), 1283–1293. https://doi.org/10.1007/s11252-016-0540-6.

Singh, P. & Downs, C.T. (2016b). Hadeda ibis (*Bostrychia hagedash*) urban nesting and roosting sites. *Urban Ecosystems*, 19(3), 1295–1305. https://doi.org/10.1007/s11252-016-0541-5.

Thompson, G.G. and Withers, P.C. (1997). Comparative morphology of Western Australian varanid lizards (Squamata: Varanidae). *Journal of Morphology*, 233(2), 127–152. https://doi.org/10.1002/(SICI)1097-4687(199708)233:2<127::AID-JMOR4>3.0.CO;2-3.

Thompson, G.G. and Pianka, E.R. (2001). Allometry of clutch and neonate sizes in monitor lizards (Varanidae: *Varanus*). *Copeia*, 2001(2), 443–458. https://doi.org/10.1643/0045-8511(2001)001[0443:AOCANS]2.0.CO;2

Thompson, G., Clemente, C., Withers, P. & Norman, J. (2008). Is body shape of Varanid lizards linked with retreat choice? *Australian Journal of Zoology*, 56, 351–362. https://doi.org/10.1071/ZO08030.

Tolley, K., Alexander, G., Branch, W., Bowles, P. & Maritz, B. (2016). Conservation status and threats for African reptiles. *Biological Conservation*, 204, 63-71. https://doi.org/10.1016/j.biocon.2016.04.006.

Uyeda, L. (2009). Garbage appeal: Relative abundance of water monitor lizards (*Varanus salvator*) correlates with presence of human food leftovers on Tinjil Island, Indonesia. *Biawak*, 3, 9–17.

Uyeda, L.T., Iskandar, E., Wirsing, A., Kyes, R. (2013). Nocturnal activity of *Varanus salvator* on Tinjil Island, Indonesia. *Biawak* 7, 25–30.

Uyeda, L.T., Iskandar, E., Kyes, R.C., Wirsing, A.J. 2015. Encounter rates, agonistic interactions, and social hierarchy among garbage feeding water monitor lizards (*Varanus salvator bivittatus*) on Tinjil Island, Indonesia. *Herpetological Conservation and Biology* 10, 753–764.

Western, D. (1974). The distribution, density and biomass density of lizards in a semi–arid environment of northern Kenya. *African Journal of Ecology*, 12(1), 49–62. https://doi.org/10.1111/j.1365-2028.1974.tb00106.x.

Whiting, M., Chetty, K., Twine, W. & Carazo, P. (2009). Impact of human disturbance and beliefs on the tree agama *Acanthocercus atricollis atricollis* in a South African communal settlement. *Oryx*, 43, 586. https://doi.org/10.1017/s0030605309990160.

Whiting, M, Williams, V. & Hibbitts, T (2011). Animals traded for traditional medicine at the Faraday Market in South Africa: Species diversity and conservation implications. *Journal of Zoology*, 284, 84-96. https://doi.org/10.1007/978-3-642-29026-8_19

Wolfe, A.K., Bateman, P.W. & Fleming, P.A. (2018). Does urbanization influence the diet of a large snake? *Current Zoology*, 64(3), 311–318. https://doi.org/10.1093/cz/zox039.

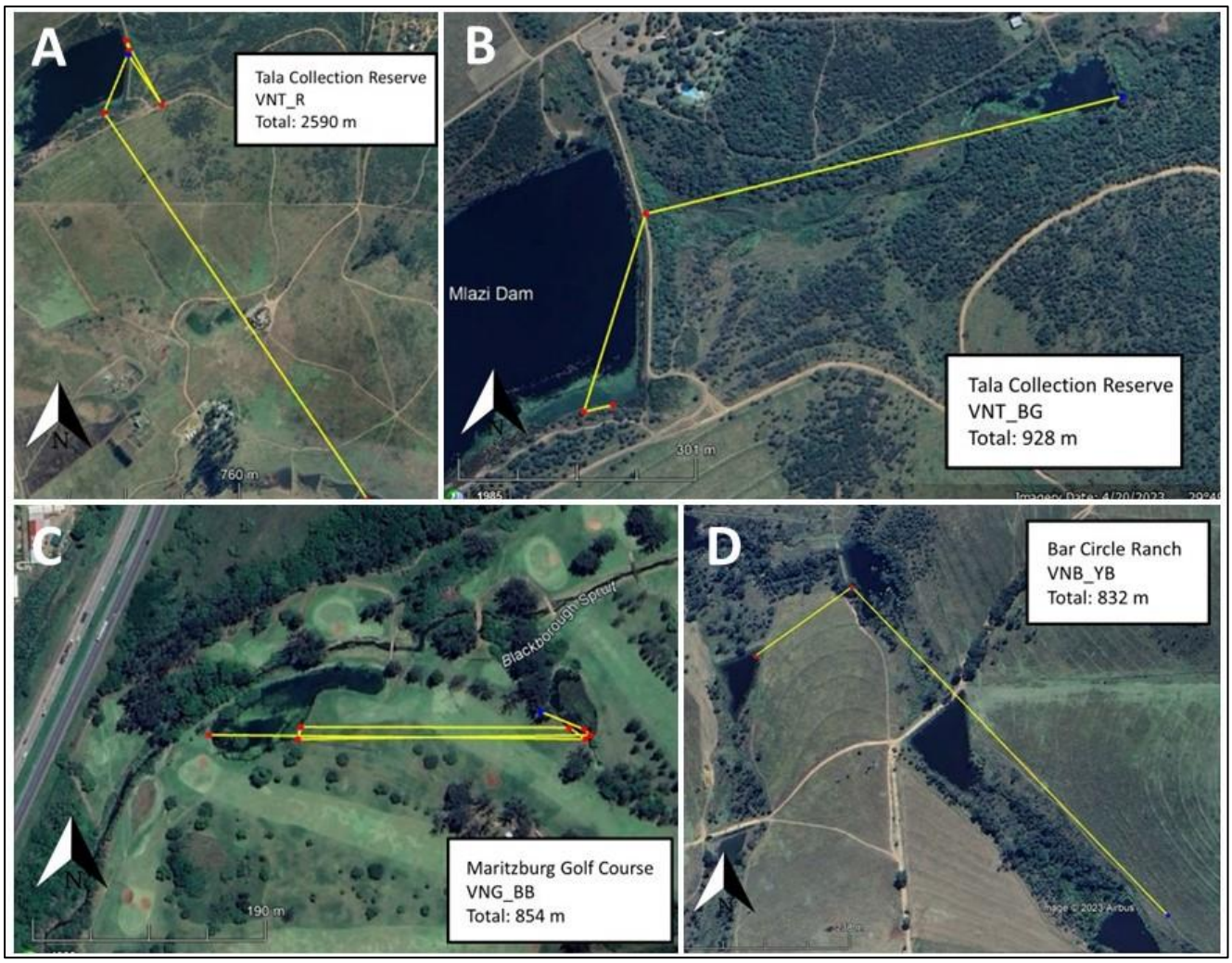
Wood, J.P., Dowell, S.A., Campbell, T.S. & Page, R.B. (2016). Insights into the introduction history and population genetic dynamics of the Nile Monitor (*Varanus niloticus*) in Florida. *Journal of Heredity*, 107(4), 349–362. https://doi.org/10.1093/jhered/esw014.

Woolley, C., Hartley, S., Innes, J., Shanahan, D., van Heezik, Y., Wilson, D. & Nelson, N. (2023). Conservation of skinks in New Zealand cities. *Urban Ecosystems*, 26, 1–16. https://doi.org/10.1007/s11252-023-01398-w.

Yu, X., Zanudin, A.B.T.M., Rusli, M.U., Booth, D.T. & Lei, J. (2021). Diet reflects opportunistic feeding habit of the Asian water monitor (*Varanus salvato*r). *Animal Biology* 72, 27–37.

Zeeuw, M.D. (2010). Husbandry and reproduction of *Varanus glauerti* in Captivity. *Biawak*, 4(3), 103-107.

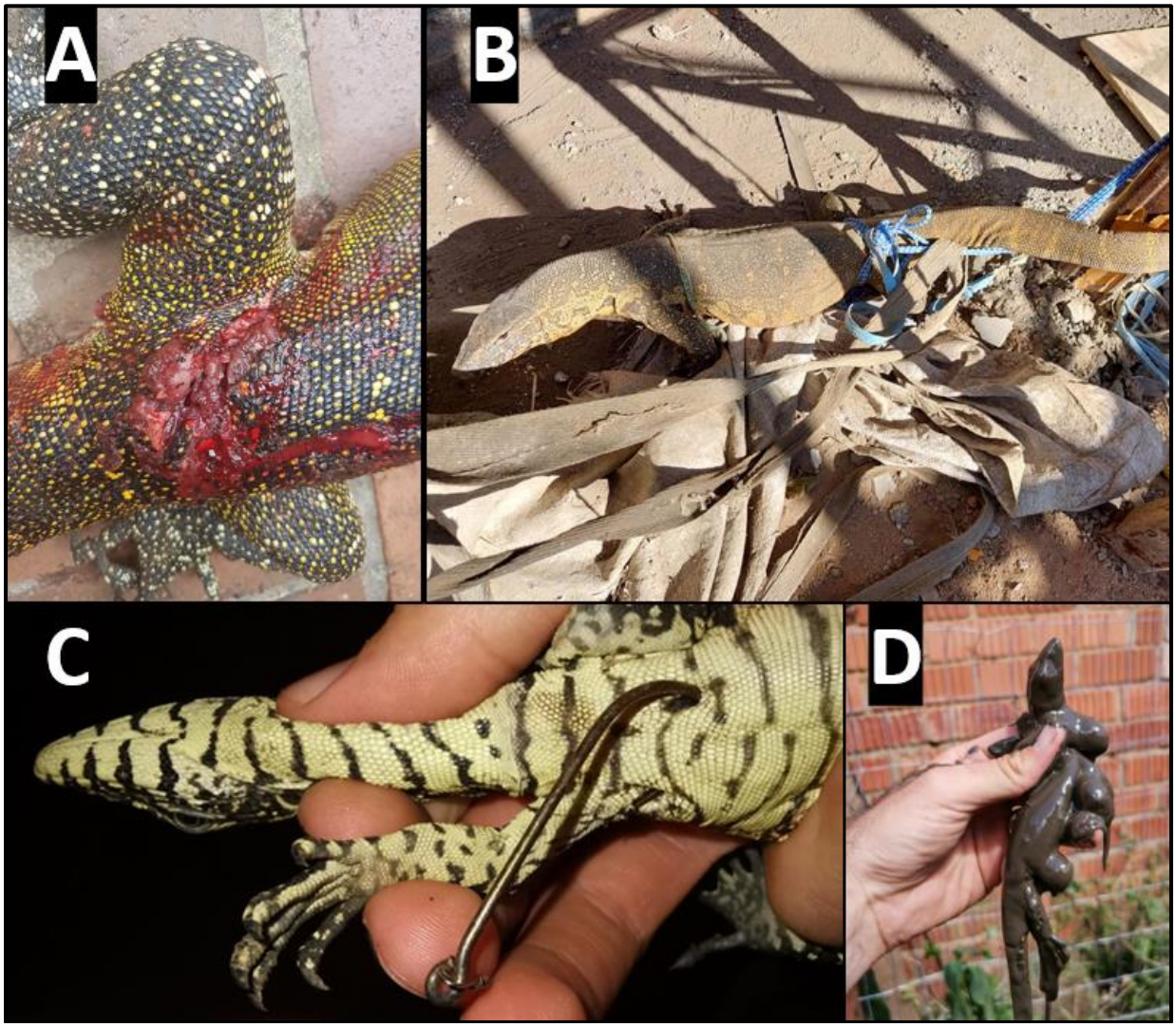
3.9 Supplementary information



Supplementary information Figure S3.1: Chronological path between sightings of marked Nile monitors, including initial capture location, opportunistic sightings, transect sightings and camera trap observations. Blue dots are the last location seen. A = between 11/01/2023 and 16/05/2023. B = between 10/03/2023 and 03/10/2023. C = between 30/10/2022 and 10/03/2023.

Supplementary information Table S3.1: ANCOVA of morphometric lengths, body mass and tick load with SVL as a covariate between urban, rural and total KZN adult Nile monitors. Lengths are given in mm and mass is in grams. The trend shows the relation to urbanisation.

	Mean rural	Mean urban	Trend	Mean KZN	F value	P value
Tail	807.6 ± 75.10	811.2 ± 47.90	↑	809.7 ± 40.72	0.25	0.62
HW	47.1 ± 3.40	46.3 ± 2.62	↓	46.6 ± 2.03	0.00	0.96
HL	86.3 ± 7.33	81.3 ± 4.08	↓	83.4 ± 1.89	2.37	0.13
HD	38.2 ± 3.20	38.1 ± 2.43	↓	38.1 ± 2.99	0.21	0.65
TB	52.8 ± 5.03	50.2 ± 3.81	↓	51.3 7.19	0.82	0.37
BW	118.8 ± 10.98	110.7 ± 9.77	↓	114.0 ± 5.99	1.42	0.24
LHL	150.2 ± 9.31	144.1 ± 8.08	↓	146.6 ± 5.99	1.57	0.21
UHL	69.0 ± 5.42	66.1 ± 3.13	↓	67.3 ± 5.36	0.87	0.36
LFL	121.3 ± 8.23	115.3 ± 7.26	↓	117.8 ± 5.36	2.51	0.12
UFL	55.6 ± 2.73	51.7 ± 3.13	↓	53.3 ± 2.73	3.36	0.07
Mass	4005.2 ± 846.32	3667.0 ± 690.91	↓	3807.1 ± 523.19	0.33	0.57
Ticks	18.7 ± 7.37	19.5 ± 14.17	↑	19.2 ± 8.72	0.02	0.88



Supplementary information Figure S3.2: Examples of anthropogenic threats to Nile monitors encountered in Durban, KwaZulu-Natal. (A = Injuries from a dog conflict. B = Live Nile monitor on sale and tied to a street post before being confiscated. C = Hatchling found in a residential garden with two fishing hooks implanted in its thorax and abdomen. D = One of two hatchlings found live in an oil container). (Photographs © Nick Evans).

CHAPTER 4

General discussion, conclusions and management recommendations

This chapter serves as a synthesis of the previous data chapters above. We aim to make recommendations based on our research on aspects of the urban and rural ecology of Nile monitors (*Varanus niloticus*) in KwaZulu-Natal, South Africa, conducted between July 2022 and December 2023. Overall, this can be seen as a case study of a highly exploited and persecuted species that is beating the odds by successfully persisting in anthropogenic ecosystems, even in human population-dense areas. However, they are not as successful as other urban exploiting varanids globally and would benefit from the recommendations hereafter described.

4.1 Background

Nile monitors (*Varanus niloticus*) are the largest lizard in Africa, are widespread and are one of the most exploited tetrapods in Africa, mainly for bushmeat, leather, the pet trade, and traditional medicines (De Buffrénil & Castanet, 2000; de Buffrénil & Hémery, 2002; Enge, 2004; Dowell et al., 2015). They are projected to become threatened in parts of their range without successful management (Ejigu & Tassie, 2020). However, they are greatly understudied and much of their ecology is unknown. Nile monitors occur in a wide variety of habitats in KwaZulu-Natal, including the province's urban areas. Little is known about their persistence, behaviour, and movement patterns in urban ecosystems. Successful management and a better understanding of their urban ecology may be key to the population maintenance of such varanids in an increasing anthropogenic changing land use scenario (Uyeda, 2009). Many see them as vermin, which results in human-wildlife conflict. There are also prominent myths

and folklore about them (EG unpublished data). Overall, there is a need for research-based management and conflict mitigation.

4.2 Findings

Personal sightings supplemented with citizen science records and reptile relocation groups were used to investigate Nile monitors' habitat use and distribution in KwaZulu-Natal (Chapter 2). Two areas of KwaZulu-Natal where most of the sightings occurred were compared. One area was dominated by natural land, while the other was dominated by anthropogenic land use (Chapter 2). By using resource selection functions, the relative selection strengths of various land-cover types over anthropogenic land use were determined. Nile monitors in the human-dominated site only showed a preference for grassland and overall little preference for other land cover types over anthropogenic land (Chapter 2). In contrast, the Nile monitors in the area dominated by natural land showed a preference for all tested land cover types over anthropogenic land use (Chapter 2). This showed that Nile monitors are persisting in some modified areas of KwaZulu-Natal and that their urban use is driven by habitat modification and/or loss (Chapter 2). This suggests that continued land-use change as a response to growing human populations is likely to increase human-monitor interactions and conflicts. Monitors showed a strict avoidance of exotic timber plantations throughout. They had a highly significant association with distance to rivers throughout, but their use of swimming pools and storm drains in urban ecosystems has reduced their reliance on rivers, implying both managed and unmanaged areas are important for their persistence in mosaic urban landscapes (Chapter 2).

Three sites on a gradient of anthropogenic intensity (a golf course, a cattle and game farm, and a game reserve) were used to monitor Nile monitors in KwaZulu-Natal (Chapter 3). Mark-resight with baited camera traps was used to estimate population size and estimated

snout-vent length (Chapter 3). Overall, population size estimates increased with anthropogenic intensity, while the average estimated snout-vent length decreased. Evidence of poaching of Nile monitors was found at the golf course (Chapter 3). It is suggested that juveniles might have a greater survival rate in urban areas than in rural ones, but adults are at an increased risk of anthropogenic threats. A general trend of rural Nile monitors being bulkier, heavier, and reaching bigger sizes than urban ones was found (Chapter 3). However, there was no significant difference between urban and rural individuals when comparing the means of various morphological data and body mass of adult Nile monitors at a provincial scale (Chapter 3). It is concluded that some Nile monitor populations are negatively affected by anthropogenic activities but that, at a provincial level, they are successful urban adapters (Chapter 3).

In-person interviews were conducted, and an online questionnaire was shared (both in English and isiZulu), to document people's perceptions, their myths, and the folklore on Nile monitors in KwaZulu-Natal. Online questionnaires showed a general appreciation for the species and their conservation, with little direct evidence of traditional use or hunting, and were mainly presented as rumours. However, when conducting in-person interviews, conflicting results with many myths were reported. Some of the beliefs reported in the interviews included: *They drink cows' milk either by climbing up the cow's leg or by shapeshifting into a calf; They can shapeshift into many different animals (usually snakes when they feel threatened); If they whip you with their tail it will break bone and create an injury that will never heal and causes a pain that reoccurs every year on the same day; Injecting monitor lizard blood into a patient will cure some diseases including HIV; Carrying a vial of their fat or applying it cutaneously brings money, strength, good luck and/or will manifest an easier life; Wearing an item made from their skin (usually a bracelet) makes you invincible, with one person detailing "their skin is tough, a person wearing the skin won't get hurt even if a large rock falls on them."; Their internal organs have medicinal properties; Consuming the meat of a Nile monitor makes you*

stronger but also cancels "love spells" which has resulted in many divorces; All lizards found in or near the house bring bad luck and they must be killed; They sometimes sit on people's chests while they are sleeping and use their forked tongue to eat the brain, going through the nose.'

One young adult interviewed said "*I think they are quite beautiful; I like them actually. But we can sell them and get something from them. It's an exchange. When I catch them, I get money. Even as a child, I would get money and I would wear their skin as a bracelet which gave me strength*". A woman also answered "*I heard that our ancestors used to kill these animals for making traditional medicines and that a child was caught by a crocodile while crossing a river and was saved by a Nile monitor. That made me understand that they are good animals and they are good to the people if they are also good to them*". This woman continued to explain that she had stopped purchasing traditional medicine derived from monitor lizards since being told that story. Despite the many misbeliefs and the general negative nature of them towards Nile monitors, this later statement is promising evidence that, at least for some people, public outreach could be a successful measure to reduce demand. Furthermore, one person explained that a Nile monitor, dead or alive, can be sold to traditional healers for 200 ZAR/kg (~10.63 USD/kg). The average body mass of all Nile monitors caught in this study was 3.1 kg, and the heaviest was 10.3 kg. This suggests that on average, a person can sell their catch for 617 ZAR (~32.7 USD) and can earn up to 2068 ZAR (~109.7 USD) from a single lizard. Most (68%) of online respondents knew that they were a protected species in South Africa, while only 6% of people interviewed in person knew of their protected status. Public outreach may help reduce the demand for Nile monitor products, but an impact on active persecution may be harder to achieve. We presently have little to no evidence of implementations or enforcement of their protected status in KwaZulu-Natal.

4.3 Limitations

There were some limitations in our study on the habitat use of Nile monitors in KwaZulu-Natal (Chapter 2). Citizen science records can have spatial and temporal bias and must be addressed with caution (Di Cecco et al., 2021; Roberts et al., 2022; Shin et al., 2022; Geurts et al., 2023). Despite these limitations, it can provide much-needed data with limited resources, and when combined with professional data, can increase spatial coverage and produce more comprehensive datasets overall (Farhadinia et al., 2018; Mo & Mo, 2022; Forti & Szabo, 2023; Dimson et al., 2023). After reviewing each iNaturalist observation, rejecting dubious records, and combining them with our sightings, we were able to successfully analyse the distribution and habitat use of Nile monitors between two sites in KwaZulu-Natal. The lack of studies on their home range size was a limiting factor in the choice of buffer area size (Chapter 2). However, the comparison with our analysis using point land-cover type showed similar results, suggesting that our 500 m buffer area could be used to assess land-cover types associated with Nile monitor presence. Studying their home range would benefit future studies and the conservation of Nile monitors by improving our understanding of their environmental requirements and their adaptation to anthropogenic landscapes (Uyeda et al., 2012; McGregor et al., 2023).

Concerns about theft of equipment and safety for monitors in traps limited our options for urban field sites for mark-resight (Chapter 3). We opted for Maritzburg Golf Course as previous studies have successfully used it as a site (Pillay et al., 2018; Singh et al., 2021). Despite being fully fenced and having private security, a total of two camera traps, one step-plate trap and two bait cages were stolen between February and December 2023. Such limitations in finding field sites when studying the urban ecology of exploited species may result in an underrepresentation of the impact of poaching (Meek et al., 2018). Another limitation was the appearance of individuals having lost their beads. These were rare, and

multiple individuals were still marked after over a year (Chapter 3). Furthermore, these rare incidences left a distinctive notch in the tail crest, which we confirmed by recapturing individuals and scanning their PIT tag. We varied the placement of the sutured beads along the tail and were able to recognise marked and most unmarked Nile monitor individuals from both the presence/absence of notches and some distinct natural colour pattern variations or scars.

4.4 Recommendations

As mentioned above, Nile monitors are projected to become threatened in parts of their range without successful management (Ejigu & Tassie, 2020). The present knowledge of Nile monitor ecology is limiting research-based management decision-making. Asian water monitors (*Varanus salvator*) were the third most internationally exploited monitors in the world between 1975 and 2005 and continue to be among the most internationally traded varanids (Pernetta, 2009; Arida et al., 2020; Rifaie et al., 2023). Some countries in their range have harvest quotas, mainly for the international luxury leather trade (Koch et al., 2013; Khadiejah et al., 2019; Rifaie et al., 2023). An assessment of population size and demographics where intense harvesting occurs suggested that harvest levels were sustainable despite being high (Khadiejah et al., 2019). This was partly because of the preference for a specific body size range and that traders claim to release individuals with inferior skins (Khadiejah et al., 2019). Many other species are harvested at unsustainable levels (Koch et al., 2013). It is believed that a sustainable quota system is presently not feasible in South Africa because of the localised nature of exploitation, the observed lack of enforcement of their protected status and the general lack of knowledge on Nile monitor population dynamics, among other aspects of their ecology. It is recommended more research on people's perception of Nile monitors using in-person interviews in conjunction with continued public outreach efforts that promote the much-needed equilibrium between reptile conservation and human well-being in Africa is undertaken (Tolley

et al., 2016; Brum et al., 2022). The priority for Nile monitor research should be to fill knowledge gaps which directly impact the ability to make research-based successful management decisions.

Wetlands are presently declining in size and distribution in southern Africa and are projected to continue, which could affect semi-aquatic reptiles like Nile monitor (Grundling et al., 2013; Ejigue & Tassie, 2020; Price et al., 2021, 2022; Thamaga et al., 2022). Our results suggest that urban Nile monitors can partly meet their ecological needs using small artificial water bodies like swimming pools and storm drains (Chapter 2). Many nature reserves in South Africa presently use artificial waterholes to mitigate water scarcity or unpredictability (Sutherland et al., 2018). Future research should investigate the use of such structures by Nile monitors and their feasibility as a management tool in increasingly arid parts of their range. More research is also needed on the potential impacts of chemical runoff and other forms of pollution associated with rivers in cities, such as Durban, and increased flooding events, which have likely doubled in frequency in Durban in the last century (Lebepe et al., 2022; Grab & Nash, 2023).

There is a lack of knowledge on African varanid parasites (Cook et al., 2016). Nile monitors in Nigeria were found to have a relatively heavy parasitic burden in Nigeria (Omonona et al., 2019). During our fieldwork, we recorded a greater proportion of monitors with the presence of ticks (*Amblyomma* spp.) in rural areas (90 % rural, 75 % urban), while urban monitors reached higher tick loads with multiple individuals surpassing 200 ticks. We suggest more research be done on the burden of parasites in urban and rural Nile monitors in KwaZulu-Natal with the inclusion of blood parasites.

The onset of breeding of Nile monitors varies throughout their extensive geographical range with little information on where and when, with only suggestions that it is linked to seasonal rainfall (Branch & Erasmus, 1982; de Buffrénil & Rimblot-Baly, 1999). An evaluation

of the onset of breeding in Nile monitors would be of value to future research and management of locally declining Nile monitor populations. The lack of knowledge on the home range size was somewhat limiting during this study. With the ever-improving technologies and methodologies in wildlife tracking and the recent increased success in the use of global positioning systems (GPS) tracking for varanids, a study of Nile monitor spatial and temporal activity in various land use types is recommended (Chapter 2; Flesch, 2009; Guerrero-Sanchez et al., 2022; Fletcher et al., 2023). Ideally, this would include sex determination, which may be achievable using blood samples (Mayes et al., 2006; Sulandari et al., 2014).

4.5 References

Arida, E., Hidayat, A., Mulyadi, M., Maireda, N.L., Subasli, D.R. and Mumpuni, M. (2020). Consumption and trade of Asian water monitor, *Varanus salvator* as reliance on wildlife for livelihoods among rural communities in North Sumatra, Indonesia. *Journal of Tropical Ethnobiology*, 3(2), 81–92. https://doi.org/10.46359/jte.v3i2.40.

Branch, W. R., and Erasmus, H. (1982). Notes on reproduction in South African water monitors, *Varanus niloticus niloticus* (Sauria: Varanidae). *Journal of the Herpetological Association of Africa*, 28(1), 4–9. https://doi.org/10.1080/04416651.1982.9650104.

Brum, P.H.R., Gonçalves, S.R.A., Strüssmann, C. and Teixido, A.L. (2022). A global assessment of research on urban ecology of reptiles: patterns, gaps and future directions. *Animal Conservation*, 26(1), 1–13. https://doi.org/10.1111/acv.12799.

Buffrénil, V., Hémery, G. (2002). Variation in longevity, growth, and morphology in exploited Nile Monitors (*Varanus niloticus*) from Sahelian Africa. *Journal of Herpetology*, 36, 419. https://doi.org/10.2307/1566186.

Cook, C., Netherlands, E. and Smit, N. (2016). Redescription, molecular characterisation and taxonomic re-evaluation of a unique African monitor lizard haemogregarine *Karyolysus paradoxa* (Dias, 1954) n. comb. (Karyolysidae). *Parasites & Vectors*, 9, 347. https://doi.org/10.1186/s13071-016-1600-8.

de Buffrénil, V. and Castanet, J. (2000). Age estimation by skeletochronology in the Nile Monitor (*Varanus niloticus*), a highly exploited species. *Journal of Herpetology*, 34, 414. https://doi.org/10.2307/1565365.

de Buffrénil, V. and Rimblot-Baly, F. (1999). Female reproductive output in exploited Nile monitor lizard (*Varanus niloticus* L.) populations in Sahelian Africa. *Canadian Journal of Zoology*, 77(10), 1530–1539. https://doi.org/10.1139/z99-130.

Di Cecco, G.J., Barve, V., Belitz, M.W., Stucky, B.J., Guralnick, R.P. and Hurlbert, A.H. (2021). Observing the observers: How participants contribute data to iNaturalist and implications for biodiversity science. *BioScience*, 71(11), 1179–1188. https://doi.org/10.1093/biosci/biab093

Dimson, M., Berio Fortini, L., Tingley, M.W. and Gillespie, T.W. (2023). Citizen science can complement professional invasive plant surveys and improve estimates of suitable habitat. *Diversity and Distributions*, 29(9), 1141–1156. https://doi.org/10.1111/ddi.13749.

Dowell, S.A., de Buffrénil, V., Kolokotronis, S.-O. and Hekkala, E.R. (2015). Fine-scale genetic analysis of the exploited Nile monitor (*Varanus niloticus*) in Sahelian Africa. *BMC Genetics*, 16, 32. https://doi.org/10.1186/s12863-015-0188-x.

Ejigu, D. and Tassie, N. (2020). Present and future suitability of the Lake Tana Biosphere Reserve in Ethiopia for the Nile monitor (*Varanus niloticus*) using the MaxEnt model. *Environmental Systems Research*, 9(1), 1–15. https://doi.org/10.1186/s40068-020-00197-y.

Enge, K., Krysko, K., Hankins, K., Campbell, T. and King, A. (2004). Status of the Nile Monitor (*Varanus niloticus*) in Southwestern Florida. *Southeastern Naturalist*, 3, 571–582. https://doi.org/10.1656/1528-7092(2004)003[0571:SOTNMV]2.0.CO;2.

Farhadinia, M.S., Moll, R.J., Montgomery, R.A., Ashrafi, S., Johnson, P.J., Hunter, L.T.B. and Macdonald, D.W. (2018). Citizen science data facilitate monitoring of rare large carnivores in remote montane landscapes. *Ecological Indicators*, 94, 283–291. https://doi.org/10.1016/j.ecolind.2018.06.064.

Flesch, J., Duncan, M., Pascoe, J. and Mulley, R. (2009). A simple method of attaching GPS tracking devices to free-ranging Lace Monitors (*Varanus varius*). *Herpetological Conservation and Biology*, 4(3), 411-414.

Fletcher, D.B., Guarino, F., Brickhill, J.G. and Green, B. (2023). Success with GPS on Rosenberg's monitor (*Varanus rosenbergi*) using modified harness design and altered monitoring procedures. *Herpetological Conservation and Biology*, 18, 100-110.

Forti, L. and Szabo, J. (2023). The iNaturalist platform as a source of data to study amphibians in Brazil. *Anais da Academia Brasileira de Ciências*, 95, 20220828. https://doi.org/10.1590/0001-3765202320220828

Geurts, E., Reynolds, J. and Starzomski, B. (2023). Turning observations into biodiversity data: Broadscale spatial biases in community science. *Ecosphere*, 14. https://doi.org/10.1002/ecs2.4582

Grab, S. and Nash, D. (2023). A new flood chronology for KwaZulu-Natal (1836–2022): the April 2022 Durban floods in historical context. *South African Geographical Journal*, 1–22. https://doi.org/10.1080/03736245.2023.2193758.

Grundling, A.T., Van den Berg, E.C. and Price, J.S. (2013). Assessing the distribution of wetlands over wet and dry periods and land-use change on the Maputaland Coastal Plain, north-eastern KwaZulu-Natal, South Africa. *South African Journal of Geomatics*, 2(2), 120–138.

Guerrero-Sanchez, S., Majewski, K., Orozco-terWengel, P., Saimin, S. and Goossens, B. (2022). The effect of oil palm-dominated landscapes on the home range and distribution of a generalist species, the Asian water monitor. *Ecology and Evolution*, 12(1), e8531. https://doi.org/10.1002/ece3.8531.

Khadiejah, S., Razak, N., Ward-Fear, G., Shine, R. and Natusch, D.J.D. (2019). Asian water monitors (*Varanus salvator*) remain common in Peninsular Malaysia, despite intense harvesting. *Wildlife Research*, 46(3), 265–275. https://doi.org/10.1071/WR18166.

Koch, A., Ziegler, T., Böhme, W., Arida, E. and Auliya, M. (2013). Pressing problems: Distribution, threats, and conservation status of the monitor lizards (Varanidae: *Varanus* ssp.) of Southeast Asia and the Indo-Australian Archipelago. *Herpetological Conservation and Biology*, 8, 1–62.

Lebepe, J., Khumalo, N., Mnguni, A., Pillay, S. and Mdluli, S. (2022). Macroinvertebrate assemblages along the longitudinal gradient of an urban Palmiet River in Durban, South Africa. *Biology*, 11(5), 705. https://doi.org/10.3390/biology11050705.

Mayes, P.J., Bradshaw, S.D. and Bradshaw, F.J. (2005). Successfully determining the sex of adult *Varanus mertensi* (Reptilia: Varanidae) using a combination of both hemipenile eversion and the ratio of androgens: estradiol in plasma. *Annals of the New York Academy of Sciences*, 1040(1), 402–405. https://doi.org/10.1196/annals.1327.074.

McGregor, D., Nordberg, E., Yoon, H.-J., Youngentob, K., Schwarzkopf, L. and Krockenberger, A. (2023). Comparison of home range size, habitat use and the influence of resource variations between two species of greater gliders *(Petauroides minor* and *Petauroides volans*). *PLoS One*, 18(10), e0286813. https://doi.org/10.1371/journal.pone.0286813.

Meek, P.D., Ballard, G.A., Sparkes, J., Robinson, M., Nesbitt, B. and Fleming, P.J.S. (2019) Camera trap theft and vandalism: occurrence, cost, prevention and implications for wildlife research and management. *Remote Sensing in Ecology and Conservation*, 5(2). 160–168. https://doi.org/10.1002/rse2.96.

Mo, M. and Mo, E. (2022). Using the iNaturalist application to identify reports of green iguanas (*Iguana iguana*) on the mainland United States of America outside of populations in Florida. *Reptiles & Amphibians*, 29, 85–92. https://doi.org/10.17161/randa.v29i1.16269

Omonona, A., Kareem, Y., Odeniran, P. and Ademola, I. (2019). Parasites and blood profile of Nile Monitor Lizards (*Varanus niloticus*) in Ibadan, Nigeria. *Nigerian Journal of Parasitology*, 40, 303–308. https://doi.org/10.4314/njpar.v40i2.30.

Pernetta, A. (2009). Monitoring the trade: using the CITES trade database to examine the dynamics of the trade in live monitor lizards (*Varanus* spp.). *Biawak: Quarterly Journal of Varanid Biology and Husbandry*, 3(2), 37–45.

Pillay, K.R., Streicher, J. and Downs, C.T. (2018). Home range and habitat use of feral cats in an urban mosaic in Pietermaritzburg, KwaZulu-Natal, South Africa. *Urban Ecosystems*, 21(5), 999–1009. https://doi.org/10.1007/s11252-018-0766-6.

Price, C., Burnett, M.J., O'Brien, G. and Downs, C.T. (2022). Presence and temporal activities of serrated hinged terrapin (*Pelusios sinuatus*) and marsh terrapin (*Pelomedusa galeata*) in KwaZulu-Natal, South Africa, assessed using telemetry. *Tropical Conservation Science*, 15(1). https://doi.org/10.1177/19400829221074241.

Price, C., Hanzen, C. and Downs, C. (2021). Demographics and morphometrics of marsh terrapins (*Pelomedusa galeata*) and serrated hinged terrapins (*Pelusios sinuatus*) populations in KwaZulu-Natal, South Africa: skewed size-class bias concerns. *Zoomorphology*, 140, 1–9. https://doi.org/10.1007/s00435-021-00518-4.

Rifaie, F., Yafi, K.M., Maireda, N.L. and Arida, E. (2023). Human-water monitor conflicts in Indonesia: Spatial patterns and mitigation alternatives. *Journal of Tropical Ethnobiology*, 6, 1–15. https://doi.org/10.46359/jte.v6i1.160.

Roberts, C.J., Vergés, A., Callaghan, C.T. and Poore, A.G.B. (2022). Many cameras make light work: opportunistic photographs of rare species in iNaturalist complement structured surveys of reef fish to better understand species richness. *Biodiversity and Conservation*, 31(4), 1407–1425. https://doi.org/10.1007/s10531-022-02398-6

Shin, Y., Kim, K., Groffen, J., Woo, D., Song, E. and Borzée, A. (2022). Citizen science and roadkill trends in the Korean herpetofauna: The importance of spatially biased and unstandardized data. *Frontiers in Ecology and Evolution*, 10, 944318. https://www.frontiersin.org/articles/10.3389/fevo.2022.944318

Singh, N., Price, C. and Downs, C.T. (2021). Aspects of the ecology and behaviour of a potential urban exploiter, the southern tree agama, *Acanthocercus atricollis. Urban Ecosystems*, 24(5), 905–914. https://doi.org/10.1007/s11252-020-01078-z.

Sulandari, S., Zein, M.S.A., Arida, E.A. and Hamidy, A. (2014). Molecular sex determination of captive Komodo Dragons (*Varanus komodoensis*) at Gembira Loka Zoo, Surabaya Zoo, and Ragunan Zoo, Indonesia. *HAYATI Journal of Biosciences*, 21(2), 65–75. https://doi.org/10.4308/hjb.21.2.65.

Sutherland, K., Ndlovu, M. and Pérez-R.A. (2018). Use of artificial waterholes by animals in the southern region of the Kruger National Park, South Africa. *African Journal of Wildlife Research*, 48(2), 1–14. https://doi.org/10.3957/056.048.023003.

Thamaga, K.H., Dube, T. and Shoko, C. (2022). Evaluating the impact of land use and land cover change on unprotected wetland ecosystems in the arid-tropical areas of South Africa using the Landsat dataset and support vector machine. *Geocarto International*, 37(25), 10344–10365. https://doi.org/10.1080/10106049.2022.2034986.

Tolley, K., Alexander, G., Branch, W., Bowles, P. and Maritz, B. (2016). Conservation status and threats for African reptiles. *Biological Conservation*, 204, 63-71. https://doi.org/10.1016/j.biocon.2016.04.006.

Uyeda, L. (2009). Garbage appeal: Relative abundance of water monitor lizards (*Varanus salvator*) correlates with presence of human food leftovers on Tinjil Island, Indonesia. *Biawak*, 3, 9–17.

Uyeda, L., Iskandar, E., Kyes, R. and Wirsing, A.J. (2012). Proposed research on home ranges and resource use of the water monitor lizard, *Varanus salvator*. *Forestry Chronicle*, 88, 542-546. https://doi.org/10.5558/tfc2012-103