Medium and large mammal site use and environmental correlates in riparian habitats of the Western Cape.

by

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DECLARATION

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Exact wording of the title of the dissertation as appearing on the electronic copy submitted for examination: Medium and large mammal site use and environmental correlates in riparian habitats of the Western Cape.

I declare that the above dissertation is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references.

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I further declare that I have not previously submitted this work, or part of it, for examination at Unisa for another qualification or at any other higher education institution.

SIGNATURE

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DATE

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Abstract

Riparian zones are vital for biodiversity and ecosystem health. This study investigated the site use and detection of riparian mammals at two sites in the Breede–Gouritz Water Management Area using camera traps. Multispecies occupancy models were used to evaluate how variation in habitat characteristics such as vegetation and water quality influenced species site use across two sites and seasons. Site use responses varied across species, sites and seasons: both vegetation structure and water-quality covariates were consistently associated with higher site use of species in summer, while in winter the association was weaker. Water quality covariates overall showed more variable, often weaker effects across both seasons. The findings suggest that riparian mammals are possibly more strongly influenced by vegetation structure than by water quality at both study areas. Future research should focus on additional drivers of site use, such as prey availability and diet composition, especially for semi-aquatic species such as African clawless otter (Aonyx capensis) and water mongoose (Atilax paludinosis).

Key terms:
Riparian mammals, occupancy modelling, camera traps, vegetation structure, water quality, Breede–Gouritz Water Management Area, freshwater ecosystems

Table of Contents

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

ASPT	Average Score Per Taxon
CFR	Cape Floral Region
CT-DS	Camera trap Distance Sampling Models
DO	Dissolved oxygen
ECI	Ecological Condition Index
FBIS	Freshwater Biodiversity Information System
FRAI	Fish Assemblage Integrity Index
REM	Random Encounter Model
REMP	River EcoStatus Monitoring Programme
PCB	Polychlorinated Biphenyls
RAI	Relative Abundance Index
REST	Random Encounter and Staying Time models
SASS5	South African Scoring System version 5
TDS	Total dissolved solids
VEGRAI	Riparian Vegetation Response Assessment Index

CHAPTER 1: INTRODUCTION

1.1 Title

Medium and large mammal site use and environmental correlates in riparian habitats of the Western Cape.

1.2 Background

1.2.1 Riparian zones

Riparian zones are important transitional habitats where terrestrial and aquatic ecosystems meet and interact, usually extending from the edges of rivers to the edge of adjoining terrestrial habitats (Décamps et al., 2009). They provide essential ecosystem services such as erosion control, water filtration, nutrient retention, and flood regulation (Lowrance et al., 1984; Crous et al., 2011; Maestas et al., 2023). The high productivity of these areas and proximity to water, means that riparian zones support a high diversity of animal communities (Meek et al., 2010). They also act as ecological corridors that connect habitats across landscapes (Graziano et al., 2022).

There are many mammal species that depend on these areas for shelter and food. In South Africa, mammal species that are often found in riparian zones are African clawless otter (*Aonyx capensis*) and spotted-necked otter (*Hydrictis maculicollis*), riverine rabbits (*Bunolagus monticularis*); water mongoose (*Atilax paludinosis*) and hippopotamus (*Hippopotamus amphibicus*) (Lloyd, 2000; Hanekom and Randall, 2015). In the Western Cape, African clawless otters, water mongoose and riverine rabbits are often found in riparian zones, along with forest species such as Cape bushbuck *(Tragelaphus scriptus*), Cape genets (*Genetta tigrina*) and bushpigs (*Potamochoerus larvatus*) (Hanekom and Randall, 2015).

1.2.2 Freshwater aquatic systems and global threats

Aquatic ecosystems such as rivers, lakes and dams are vital resources of ecological services, human use and habitat for many plants and animals. Globally, the biggest threats aquatic systems face are pollution, water-abstraction, riparian vegetation destruction, and climate change (Bates et al., 2008; Woodward, et al., 2012; Dallas et al., 2017; Bashir et al., 2020). Urban areas, industrial areas and agriculture are

the major sources of water pollution world-wide (Mateo-Sagasta et al., 2017). Climate change exacerbates these impacts by altering rainfall, runoff, water quality, and promoting invasive species (Poff et al., 2002; Bunn and Arthington, 2002; Dallas and Rivers-Moore, 2014; Dallas et al., 2017; Pörtner et al., 2022). This alters the fragile dynamics of aquatic ecosystems; water chemistry, and species distributions, highlighting the vulnerability of freshwater habitats (Cianfrani et al., 2011; Malhi et al., 2020).

1.2.3 Freshwater systems in South Africa and specific threats to them

These challenges are mirrored in South Africa, a water-stressed, semi-arid country with limited water resources (Naidoo et al., 2016). The 2018 National Biodiversity Assessment found that 64% of river ecosystem types in South Africa are threatened (Nel and Driver, 2015; Skowno et al., 2019; Cape Nature, 2025). The main threats to aquatic habitats in South Africa include damming; water abstraction, invasive plants; removal of riparian vegetation, and pollution (Somers and Nel, 2004; Chamier et al., 2012; Okes et al., 2016). As with freshwater systems globally, climate changes are predicted to increase current negative impacts in these habitats (New, 2002). Tracking and monitoring changes in aquatic systems are complex. South Africa has a national monitoring system in place called the River EcoStatus Monitoring Programme (REMP) (Dallas, 2007; Department of Water and Sanitation, 2016). The REMP uses biological indices of fish communities, riparian vegetation and macroinvertebrate assemblies such as FRAI (Fish Assemblage Integrity Index); Riparian Vegetation Response Assessment Index (VEGRAI) and South African Scoring System (SASS5) (Dickens and Graham, 2002), as well as physicochemical parameters, such as pH and dissolved oxygen, to assess and monitor the ecological condition and health of rivers.

1.2.4 Riparian mammals as sentinels

Sentinel species are animals that respond to changes in an environment through changes in presence, health or behaviour (Bossart, 2006; Hazen et al., 2019; Clark-Wolf et al., 2024). They serve as early warning systems for ecosystem degradation and pollution, therefore helping to detect issues before wider ecological effects occur (National Research Council, 1991; Baos et al., 2022). While aquatic

macroinvertebrates and fish are often used as bioindicators in South Africa's freshwater systems by SASS (South African Scoring System) and FRAI (Fish Response Assessment Index) (Dickens and Graham, 2002; Levin et al., 2019), riparian mammals, such as otters, also exhibit sensitivity to both terrestrial and aquatic conditions, making them valuable sentinels of riparian ecosystem health (Stevens et al., 2011; Baos et al., 2022). Knowing the abundance and occupancy of riparian mammals in relation to different riparian covariates is important in assessing their potential as sentinel species.

1.2.5 Environmental influences on riparian mammals

Riparian zones are heterogenous habitats with complex vegetation that is often taller and thicker than surrounding areas (Catterall et al., 2007; Coverdale and Davies, 2023). The species richness of riparian zones could be attributed to this habitat complexity which increases the variety of ecological niches available (Martin, 2002; Stoffyn-Egli and Willison, 2011; Catterall et al., 2007; Hamilton et al., 2015; Maestas et al., 2023). Canopy cover and vegetation structure play a role in the occurrence of riparian mammals by providing food, shelter, and protection from predators (Catterall et al., 2007; Regmi et al, 2023). More complex and continuous vegetation generally has higher species richness and occupancy, while fragmented or sparse vegetation can limit mammal presence and biodiversity (Monroy-Vilchis et al., 2009; Regmi et al., 2023; Oladimeji et al., 2024). This highlights the importance of maintaining continuous riparian corridors and keeping riparian vegetation intact (Regolin et al., 2017; Basile et al, 2021; Maestas et al., 2023). Because riparian mammals often respond to changes in habitat structure (Hamilton et al., 2015; Zimbres et al., 2017), their presence and distribution can serve as indications of riparian ecosystem health.

1.2.6. Research on riparian zones in South Africa

There is a limited amount of research on mammal communities in riparian zones in South Africa. Research on South African riparian zones has largely focused on plant communities and their responses to environmental pressures (Meek et al., 2010; Schachtschneider & Reinecke, 2014; Dube et al., 2017; Scott-Shaw and Everson, 2019; Reinecke et al., 2022; Mpanyaro et al., 2024), as well as restoration and

management practices (Quin et al., 2003; Holmes et al., 2005; Malan et al., 2007; Homes et al., 2008; Du Plessis et al., 2021). The influence of riparian vegetation on aquatic macroinvertebrate communities is also relatively well researched (Swanepoel et al., 2014; Rivers-Moore et al., 2018; Agboola et al., 2019; Agboola et al., 2020). The studies that have focused on mammals are limited in scope (Hanekom and Randall, 2015; Simelane et al., 2018; Janecke and Bolton, 2020). There is limited research done on other animals in South African riparian zones, such as birds (Malan, 2001; Malan et al., 2007; Mangachena and Geerts, 2017), and reptiles (Maritz and Alexander, 2007; Hunt et al., 2013). .

Therefore, there remains a limited understanding of mammal communities in riparian zones and the environmental factors that influence their occupancy, highlighting a potential knowledge gap for conservation and management.

1.2.7 Camera trapping in ecology

Camera trapping is an important cost-effective tool for researching cryptic species (Claridge et al., 2004; Rowcliffe, 2017), giving information about species behaviour, movement patterns, abundance, and occupancy (Silveira et al., 2003; Cove et al., 2013; Burton et al., 2015; Palencia et al., 2021).

1.2.8 Relative abundance indices and occupancy modelling

A relative abundance index (RAI) is an index derived from the count of animal signs or camera trap capture rates that can be potentially correlated with population size (Carbone et al., 2001; Anderson, 2003; Palmer et al., 2018; Martin-Garcia et al., 2022). However, there are limitations to RAIs: they do not compensate for detection probability and are prone to bias and therefore they cannot be reliably used to estimate population abundance (Anderson, 2003; Ulrich and Ollik, 2005; Sollman et al., 2013). Occupancy models that investigate habitat use, and distribution rather than abundance are a better alternative to relative abundance indices (RAIs) (Sollman et al., 2013). Although not a direct measure of abundance, occupancy can be seen as a surrogate for abundance and a robust alternative to RAIs (MacKenzie et al., 2002; Gaston and Blackburn, 2003; MacKenzie and Nichols, 2004; Sollman et al., 2013). They estimate the probability of a species occupying a site, while accounting for imperfect detection (MacKenzie et al., 2002; Bailey and Adams,

2005). Occupancy models are fitted to investigate how environmental covariates such as habitat structure and water quality influence the probability of occupancy (MacKenzie et al., 2002; Sollmann, 2018; Gould et al., 2019).

1.2.10 Study design

This study aimed to evaluate the site use of riparian mammals across two river systems in the Breede-Gouritz water management area in the Western Cape, allowing the assessment of the influence of environmental covariates such as vegetation structure and water quality, and providing insights in the factors that affect the occupancy of riparian mammals, which can inform species distribution and conservation management (MacKenzie et al., 2002; Wearn et al., 2022).

1.2.11 Expected ecological patterns

It can be expected that semi-aquatic species such as African clawless otters and water mongooses will decrease in site use as water quality declines, since it could potentially affect prey populations such as fish and crabs (Mason, 1995). This is without taking into account the direct impact fishing has on prey populations (Jacques et al., 2015). Other species may be less directly affected by water quality since they are not directly dependant on rivers for food, although it still reflects an aspect of the health of riparian zones. An increase in the canopy cover and ground vegetation cover of indigenous plants are expected to be positively correlated with the site use of all riparian mammal species found in the area, as it indicates less fragmented habitat and provides greater shelter, food availability, and protection from predators. However, the presence of natural clearings within riparian vegetation may also be important, as a heterogeneous habitat structure that includes both dense cover and open areas support a wider range of species who all have differing ecological requirements (Tews et al., 2003). Understanding these relationships will help clarify how vegetation structure and water quality jointly influence the distribution of riparian mammals, providing insight into their potential as indicators of ecosystem health.

1.3 Project outline

Riparian zones are vitally important for the ecosystem services they provide, as well as the essential habitat they provide for different riparian mammals. The site use of riparian mammals can be indicators of ecosystem health, and the goal of this study was to determine how vegetation structure and water quality influence the probability of site use of riparian mammals in the Breede River catchment area. The study used camera traps to determine the probability of site use of riparian mammals of two rivers in the Breede River catchment area. These two study areas are similar in land use, vegetation and climate. Camera traps were set up in the riparian vegetation zone, 15 m from the riverbank. Resulting data was used to run occupancy models which was used to estimate site use and detection probability. Camera trap covariates that were collected were canopy cover, ground vegetation cover, and water quality. Camera trap effort was used as the main survey covariate. Water quality was assessed using the South African Scoring System (SASS5) data gathered by the Grootvadersbosch Conservancy. Additionally, water quality parameters such as pH, dissolved oxygen and conductivity were collected on site during the survey period. Canopy cover was determined using Google Earth imagery, while ground vegetation cover was determined using quadrats. Multispecies occupancy models were run in R using the spOccupancy package (Doser et al., 2022; R Core Team, 2025).

1.4 Research problem

There is a need for estimation and confirmation of riparian mammals' site use and distribution in freshwater ecosystems in South Africa. Additionally, it is important to know how their presence relates to habitat variables such as vegetation structure and water quality to help inform conservation decisions in riparian zones. By identifying the factors that influence the species the most – it helps highlight what habitat features should be prioritized in restoration and protection and monitor the effects of changes in these habitat covariates over the long term. This also helps see how environmental restoration and degradation affect the site use of riparian species. The specific research problem being addressed is the influence that water quality and vegetation structure has on the site use of riparian mammals in the

Breede-Gouritz water management area, to assess these mammals' potential as sentinels of freshwater ecosystem health.

1.5 Significance of the study

This research on riparian mammal populations along rivers of the Western Capel provides estimates of the species' probability of site use and investigates the potential effects of vegetation covariates and water quality on the site use of medium to large riparian mammals. This will help inform future conservation planning of the species and assessment of their conservation status. Riparian mammals have potential as sentinels in rivers and reflect the overall health of the river habitat (Stevens et al., 2011; Baos et al., 2022). Therefore, it will also provide information on the health of these portions of river and how management of these sections can be improved. There has been little research done on riparian mammals in the Overberg so this study will shed light on the distribution patterns of riparian mammals in this area and raise awareness in local communities about the importance of riparian zones and their role in the health of freshwater ecosystems.

1.6 Aims and objectives

Research Aims

The project aimed to determine the probability of site use and detection probability of medium to large riparian mammals in the riparian zones of two rivers (Buffeljags River and Grootvadersbosch River) in the Breede-Gouritz water management area of the Western Cape.

1.6.1 Aim 1: To determine the probability of site use and detection probabilities of riparian mammals in riparian zones using data obtained via camera traps.

Research questions:

- How does mammal probability of site use differ between the two study areas?
- How does mammal probability of site use differ between seasons?

1.6.2 Aim 2: To assess how differences in water quality influence riparian mammals' probability of site use.

Research questions:

- How does the water quality (as measured by SASS5 and physicochemical parameters) differ between the two study areas?
- How does water quality correlate with riparian mammal probability of site use?

1.6.3 Aim 3: To evaluate the influence vegetation structure has on the probability of site use of riparian mammals.

Research questions:

- How does canopy cover influence riparian mammal probability of site use?
- How does ground vegetation cover influence riparian mammal probability of site use?

CHAPTER 2: LITERATURE REVIEW

2.1 Riparian habitats

2.1.1 Definition of riparian zones

There are a variety of definitions for riparian zones (Verry et al., 2004) but fundamentally it is the area of land surrounding a river and its associated vegetation and wildlife. The riparian zone is defined by Ollis et al. (2013) as the part of land right next to the 'active channel' (main water-containing channel) of the river. According to Naiman et al., (1993) the riparian zone includes the river itself and stretches from the high-water mark to the area where flooding or the water table influences the plants and soil's water retention ability. The width of riparian areas varies but it is often relatively narrow in comparison with surrounding habitat (Phillips et al., 2000).

2.1.2 Ecological role of riparian zones

Riparian zones play various ecological roles: they act as buffer zones; play a role in flood mitigation, provide important habitats for plants and animals, and play a role in temperature regulation of water (Karr and Schlosser, 1978). They prevent erosion by stabilizing the ground next to the river and filter the nutrients from the river for use in the ecosystem (Holmes et al., 2005; Caterelle et al., 2007; Crous, et al., 2011). Their proximity to the river and the natural beauty that riparian zones have, mean they are often used for recreation by humans (Crous et al., 2011; Saklaurs et al., 2022).Riparian zones have alluvial soils and unique vegetation in comparison with habitat further away from rivers (Naiman and Décamps, 1997; Ollis et al., 2013). The soil is high in nutrients which is advantageous for agricultural purposes (Crous et al., 2011). These areas can vary between well-drained and dry. Well-drained riparian zones are often classified as wetlands and riparian zones that are rich in tree species are sometimes called riparian forest (Ollis et al., 2013). Riparian zones are considered as some of the most productive areas on earth (Nilsson and Berggren, 2000) because of the nutrient rich, moist soils and proximity of water (Catterall et al., 2007; Crous et al., 2011). This allows abundant plant growth meaning there is often a rich plant diversity (Meek et al., 2010; Kinnoumè et al., 2024) and unique plants (Sabo et al, 2005). Riparian vegetation is also well adapted

to disturbances caused by varying waterflow, flooding, and consequent changes in water, oxygen, and nutrient levels in the soil (Holmes et al., 2005).

Additionally, riparian vegetation plays a critical role in anchoring soil and regulating erosion (Crous et al. 2011). It also lessens flood damage by absorbing water, slowing down the flood speed, and stabilizing the banks (Crous et al., 2011). Riparian zones are 'critical transition zones' (Holmes et al., 2005). There is a big exchange of materials (e.g. nutrients and soils) between the river and the habitat through which the river flows (Verry et al., 2004; Stoffyn-Egli and Willison, 2011). They are thus 'filters, sources and sinks' of nutrients and other biological materials (Malanson,1993). Riparian vegetation filter and purify water by trapping pollutants with their roots (Lowrance et al., 1984; Crous et al., 2011). Riparian vegetation also captures physical trash such as plastics and glass thus collecting it in one place, lessening the spread of litter in the environment (Cesarini and Scalici, 2022). The plant species that are present determine how effective a riparian zone is in filtering nutrients and other substances (Naiman and Décamps, 1997). For example, poplars (*Populus* spp.) are better at retaining nitrates than grass (Haycock and Pinay, 1993).

Riparian habitats are often shifting and changing as the flow of the river changes, affecting the nutrients and soil depths (Hood, 2020). Riparian vegetation that is maintained, creates better habitat with lower turbidity and suspended solids, and an increase in dissolved oxygen, and therefore has a direct positive effect on water quality (Alemu et al., 2017). If a riparian zone is tree rich and has dense canopy cover, it helps regulate the temperature of the water because of the shade effect of the trees (Moore et al., 2005; Dugdale et al., 2018).

2.1.3. Riparian zones as habitat

Riparian zones act as habitat and corridors for various birds, mammals, and other animals (Decamps et al., 1988; Meek et al., 2010; Graziano et al., 2022). These zones are often richer in biodiversity than other surrounding areas despite being relatively small (Gregory et al., 1991; Naiman et al. 1993; Naiman and Décamps, 1997; Catterall et al., 2007; Crous et al., 2011; Stoffyn-Egli and Willison, 2011). There are arguments that higher biodiversity is not necessarily the case, rather that there is more unique species in comparison with surrounding areas (Sabo et al.,

2005). Either way, there is often a large variety of species in riparian zones (Anthony et al., 2003).

While many species observed in riparian areas also use surrounding habitats, certain species are riparian specialists or exhibit a strong preference for riparian zones, such as beavers (*Castor* spp.), otters (e.g. Eurasian otter *(Lutra lutra*)) and riverine rabbits *(Bunolagus monticularis)* (Perrin and D'Inzillo Carranza, 2000; Catterall et al., 2007; Collins and Du Toit, 2016; Giriat et al., 2016; Ramey and Richardson, 2017). The high biodiversity of animals can be attributed to a mixture and diversity of biological and physical features providing a variety of benefits (Kauffman et al., 2001; Anthony et al., 2003; Capon and Dowe, 2007). The thick vegetation and proximity to water provide shelter for animals and birds (Catterall et al., 2007; Weinberger et al., 2019). Their use of the area varies with the species, whether as foraging/hunting grounds, shelter, nesting or roosting areas or as a corridor (Catterall et al., 2007; Dickie et al., 2019). Riparian zones are important as corridors for animal movement (Machtans et al., 1996; Green, 2007) especially in dry areas or where the surrounding areas are more open (Catterall et al., 2007). Riparian vegetation allows the creation of microclimates that create refuges for invertebrates and small mammals (Moore et al., 2005; Williamson et al., 2021). It provides safety from harsh conditions that might occur outside the riparian zones (Catterall et al., 2007). Changes in riparian vegetation such by removal of trees or shrubs can alter the microclimates and take away the shelter and thus upset the fine balance of these areas (Moore et al., 2005).

2.1.4 Classification of riparian zones

The exact classification of riparian zones differs between countries. It is primarily based on riparian vegetation. For example, riparian zones in the United States are classified based on vegetation, whether it is dominated by forest or shrub-scrub, and then further grouped based on whether deciduous trees or coniferous trees are dominant and the species that occur frequently (U.S. Fish and Wildlife Service, 2019).

Riparian zones are classified as alluvial zones in South Africa (Mucina et al., 2006). The riparian zones in the Westen Cape are called Cape Lowland Alluvial Vegetation (Mucina et al., 2006). Cape Lowland Alluvial zones have shared features

with many other alluvial/riparian zones across South Africa (Mucina et al., 2006). Generally, this habitat type is represented by flat landscapes with sandy or silty fine soils, occupied by slow flowing rivers. Banks are often fringed by varying thick reeds, tall riparian thickets and grasslands There is a gradient of vegetation as you move from the river upland. Lower banks that often experience flooding are mainly occupied by annuals that have high nutrient requirements, as well as reeds like *Phragmites australis* and *Typha capensis.* On the low and middle banks grass is common while riparian thickets and trees occur higher up on the banks because of less disturbance by flooding and other factors (Mucina et al., 2006).

2.1.5 Research on riparian zones in South Africa

There are limited studies on mammals in riparian zones in South Africa (Hanekom and Randall., 2015; Simelane et al., 2018; Janecke and Bolton, 2020). These research papers are often focused on small mammals such as rodents (Hanekom and Randall., 2015) or are part of larger scale mammal-habitat studies of which riparian zones are one of the sections studied e.g. a granite catena in Kwazulu-Natal (Janecke and Bolton, 2020). There are some studies on birds (Malan, 2001; Malan et al., 2007; Mangachena and Geerts, 2017) and reptiles (Maritz and Alexander, 2007; Hunt et al., 2013) in riparian zones, although this is also limited.

A lot of the research on riparian zones in South Africa. focus on the plant communities of these areas, especially their response to environmental and human pressures such as droughts, pollution and alien plant invasions (Meek et al., 2010; Schachtschneider and Reinecke, 2014; Dube et al., 2017; Scott-Shaw and Everson, 2019; Reinecke et al., 2022; Mpanyaro et al., 2024). Research on riparian zones also often focused on restoration of these areas, especially alien plant removal, as well as other conservation management practices (Quin et al., 2003; Holmes et al., 2005; Malan et al., 2007; Homes et al., 2008; Du Plessis et al., 2021). There is also considerable research on the influence of riparian vegetation of macro-invertebrate communities (Swanepoel et al., 2014; Rivers-Moore et al., 2018; Agboola et al., 2019; Agboola et al., 2020).

While this research provides valuable insights into riparian ecosystem processes in South Africa and conservation of these areas, there is still limited

understanding of mammal communities in these zones and the environmental factors that influence their occurrence.

2.1.6 Threats to riparian zones in South Africa

Cape Lowland Alluvial Vegetation is on the revised national list of ecosystems that are threatened and in need of protection (Du Toit et al., 2020). It was classified as critically endangered in 2016 (Pool-Stanvliet et al., 2017) but it's threat status in 2021 was 'Endangered' with the reason being that it 'is narrowly distributed with high rates of habitat loss in the past 28 years (1990-2018), placing the ecosystem type at risk of collapse' (Creecy, 2022).

The main threats to riparian zones include alien vegetation, clearing of vegetation for building or agricultural purposes, as well as destruction by cattle and other livestock in farming (Holmes et al., 2005). Riparian zones are especially vulnerable to alien plant invasion because rivers often carry and distribute the seeds of alien plants and the fertile, well-drained soils of riparian zones mean that it is the perfect nursery grounds for alien plants (Holmes et al., 2005). Many alien plants have high water requirements (Dye and Jarmain, 2004; Holmes et al., 2005; Chamier et al., 2012; Cavaleri et al., 2014), which negatively influences water levels and flow in rivers (Tickner et al., 2001; Dudgeon et al., 2006). For example, uncontrolled alien vegetation in the Western Cape was projected in 2011 to reduce water supply to the City of Cape Town by 34% (Crous et al., 2011). In 2020 it was estimated that invasive plants can decrease surface water runoff by 1.44–2.44 billion m^3 per year, with catchments in the Western Cape, Eastern Cape and Kwazulu Natal being the most vulnerable to this (Le Maitre et al., 2020). This number could increase to 2.59–3.15 billion m^3 per year, if nothing is done to reduce invasive plants in riparian zones (Le Maitre et al., 2020). Removal of alien plants is also important for restoration of the indigenous riparian vegetation and ecosystem integrity (Holmes et al., 2008).

Human activities also affect riparian ecosystems indirectly. Runoff from agriculture introduces pesticides and sediments, further degrading water quality and prey availability (Rowe-Rowe, 1986; Roos et al., 2012). Habitat modifications, including river channelling, vegetation removal, draining of wetlands, and dam construction, alter the river flow and destroy habitat, negatively impacting otters and

other species (Power et al., 1996; Kubekha et al., 2013; Okes et al., 2016). Alien plant invasions exacerbate these effects by outcompeting indigenous vegetation and their high consumption of water, however programs like South Africa's *Working for Water* can help restore ecosystem integrity (Hood and Naiman, 2000; Blanchard and Holmes, 2008).

Urbanization is another major driver of habitat loss, leading to declines in species numbers and diversity (Bettigole et al., 2013; Widdows et al., 2015). Riparian mammals, such as the African clawless otter (*Aonyx capensis*), are especially vulnerable (O'Riain et al., 2016). Otters' dependence on freshwater habitats makes them sensitive to human impacts on rivers, including pollution, habitat modification, and water abstraction (Kubekha et al., 2013; Okes and O'Riain, 2017). Pollutants such as organochlorines and polychlorinated biphenyls (PCBs) can bioaccumulate, reducing reproduction and causing mortality (Foster-Turley, 1990; Mason and Macdonald, 1986; Roos et al., 2012). Despite these threats, otters have shown resilience, recovering after toxic spills and other environmental disturbances (Fox, 1999; Narváez et al., 2020).

Restoration and protection of riparian habitats, particularly freshwater systems, remain central to conserving otters and other riparian mammals in South Africa (Brancalion et al., 2013; Okes et al., 2016; Okes and O'Riain, 2017).

2.2 Water quality

2.2.1 Aquatic ecosystems overview

Aquatic ecosystems are vital for providing ecological services, clean water and habitats for many organisms. They are sensitive to a variety of different factors. Small changes in nutrient cycles, runoff and other variables, such as water temperature and pH, can have profound effects on the lifecycles of aquatic organisms (Woodward, et al., 2012). Globally, aquatic systems are increasingly threatened by climate change (Bates et al., 2008; Dallas et al., 2017) and human impacts such as pollution, water-abstraction, and destruction of riparian vegetation (Intergovernmental Panel on Climate Change, 2014; Mateo-Sagasta et al., 2017; Bashir et al., 2020). Pollution includes industrial waste (including heavy metals), plastics, sewage, and agricultural waste (such as pesticides, herbicides, fertilizers)

(Mateo-Sagasta et al., 2017; United Nations World Water Assessment Programme, 2017). An increase in untreated sewage; agricultural fertilizer, and runoff can lead to eutrophication, which is caused by too many nutrients in a water source (Roessig et al., 2004; Bashir et al., 2020). It is a major source of decline in aquatic habitats worldwide (Bashir et al., 2020). Climate change alters rainfall patterns, water temperature, and flow, which together increases eutrophication, sedimentation, and pollutant presence in rivers (Poff et al., 2002; Bates et al., 2008; Dallas and Rivers-Moore, 2014; Nimma et al., 2024).

Climate change means that there are more extreme fluctuations or permanent changes in the physical and biotic factors within a freshwater habitat (Nimma et al., 2024). This influences the fragile dynamics of ecosystems (Cianfrani et al., 2011; Malhi et al., 2020). For example, higher rainfall can lead to increased turbidity in lakes and dams because of higher erosion rates and increased movement of pollutants, such as pesticides that were previously dissolved in the soil, to surface and groundwater (Abdullah et al., 2025). The rise of sea levels increases the salinity of ground water and estuaries; this decreases available fresh water globally (Bates et al., 2008; Zamrsky et al., 2024). Higher water temperature, one of the estimated impacts of climate change, increase the thermal stability and therefore change the patterns of mixing within water bodies such as dams and lakes. This leads to lower oxygen concentrations and higher phosphorous release from sediments (George et al., 2007; Bates et al., 2008). Changes in water temperature also influence fish species. Many fish species are dependent on external temperatures for their own body temperature (poikilothermic) and are therefore sensitive to water temperature changes (Roessig et al., 2004) e.g., the endangered Clanwilliam Sawfin (*Barbus serra)* requires specific flow and water temperature regimes for reproduction (Paxton, 2008). Generally, cold water species are predicted to be affected negatively, while warm water species will benefit from climate change (Cianfrani et al., 2011).

2.2.2 Threats to aquatic ecosystems in South Africa

South Africa is a country with a semi-arid climate (Pitman, 2011). South Africa's classification as a water stressed country (Hedden and Cilliers, 2014; Donnenfeld et al., 2018; Skowno et al., 2019), is because of erratic rainfall; droughts; over-use

of water, and lack of adequate water saving habits and protocols (Poff and Ward, 1989; Schachtschneider and Reinecke, 2014). There is a lot of pressure on water resources because of population growth, habitat destruction and other factors (Hedden and Cilliers, 2014). Therefore, aquatic systems in South Africa are especially vulnerable to climate change and over abstraction and to the associated effect on flow and water quality (Ashton, 2007; Viljoen and van der Walt, 2018). Only about 8.6% of annual rainfall becomes runoff (Crous et al., 2011; Schachtschneider and Reinecke, 2014). Apart, from water scarcity, water quality is also a concern in South Africa (Hedden and Cilliers, 2014; Knight, 2018). In South Africa, drinking water quality is thought to be decreasing as wastewater treatment infrastructure and ecological (including wetlands etc.) are becoming less effective (Cullis et al., 2018; Carnie, 2023). There is good water quality infrastructure and management plans in South Africa (Griffin et al., 2014). However, there was a decline in drinking water quality between 2014 and 2023 (Carnie, 2023) confirming that there is still need for better execution of the water management plans (Knight, 2018). Declining water quality is further worsened by poor wastewater treatment performance. The 2023 Blue Drop Report showed that only 3% of South Africa's water supply systems met the required standards, while nearly a third were in critical condition (Department of Water and Sanitation, 2023). Overall, changes in river flow, the deposition of raw or poorly treated sewage in rivers and high levels of phosphorus allowed to remain after treatment of water (Coetzee and Hill, 2011) mean there is high eutrophication that occur in South Africa's water bodies (Lukhele and Alfred, 2024). In fact, South Africa is thought to have some of the highest levels of eutrophication worldwide (Coetzee and Hill, 2011; Bega, 2024). Eutrophication occurs when unnaturally high amounts of nutrients in aquatic ecosystems leads to high levels of primary production, including algae blooms (Akinnawo, 2023). Algae blooms can lead to the release of toxins, which can cause nausea, diarrhoea and stomach pain amongst other adverse effects when consumed (Hwang, 2020; National Institute of Environmental Health Sciences, 2023). Eutrophication also leads to oxygen depletion within aquatic habitats (Suresh et al., 2023). In addition, eutrophication and algae blooms affect the taste and appearance of water systems and can cause unpleasant smells (Srinivasan and Sorial, 2011). All this is hazardous to both humans and organisms living in or dependant on water.

2.2.3 River ecosystems in South Africa

There is a high diversity of river and wetland ecosystems in South Africa (Skowno et al., 2019). The river ecosystems are however at high risk (Department of Water Affairs, 2013; Skowno et al., 2019). In South Africa, rivers are grouped in “river ecosystem categories”. River ecosystem categories is a way of classifying rivers according to similar ‘ecological characteristics and functions’ (Dayaram et al., 2021). This is created by delineating three hierarchical categories, namely landscape classification (31 ecoregions, e.g. Eastern Coastal Belt-rivers flowing in deep valleys), river flow variability (two categories) and slope/longitudinal categories (four longitudinal zones, for example, mountain streams vs lowland rivers) (Dayaram et al., 2021). According to Skowno et al. (2019) (in the National Biodiversity Assessment of 2018), there are 222 river ecosystem categories in South Africa, 64% of which are classified as threatened, while 42% of all river ecosystems are critically endangered. This shows some improvement over reports by Nel et al. (2007) in 2007, which stated that 84% of main river ecosystems are threatened and more than half critically endangered. In the Western Cape, 101 of the 138 freshwater ecosystems are threatened (73%); 64 of the 101 threatened ecosystems are river ecosystems, while the other are wetland types (Cape Nature, 2025). The most significant threats to freshwater ecosystems in the Western Cape include habitat loss, agricultural pollution (invasive plants, overuse of water, climate change, and change of flow regime (Midgley and Lötze, 2011; Skowno et al., 2019; Cape Nature, 2023b).

2.2.4 Monitoring aquatic ecosystems

Aquatic organisms are sensitive to changes in the environment. Tracking and monitoring changes in aquatic systems is important in understanding their distribution and other aspects of their ecology, but can be complex (Dodds and Whiles, 2010). As mentioned before, fluctuation in circumstances leads to changes in the turbidity and flow of rivers, as well as nutrient cycling and the flow of energy, etc. which affect parameters such as pH, dissolved oxygen, etc., as well as the biological communities within the aquatic habitat. All these changes can happen in a short period of time. Given this natural variability, assessing water quality, although useful for a given point in time, does not always account for variation over

time (González et al., 2024), depending on survey length and what metric is used.

2.2.4.1 Biological indices

Water quality is a critical determinant of river health, influencing both biodiversity and the ecosystem services rivers provide (Dudgeon et al., 2006). It is normally assessed using physicochemical indicators such as pH, water temperature, dissolved oxygen, conductivity, and turbidity, together with biological indices (Dallas, 2007). Using biological indicators allows one to monitor water quality over time and see the direct effect of conservation concerns like pollution, habitat modifications, and invasive plants on riverine areas (Chaves et al., 2006; Moog et al., 2018). There are a few national programmes used for long term monitoring of water quality in South Africa, including the River EcoStatus Monitoring Programme (REMP), formerly known as the River Health Programme (RHP) (Dallas, 2007; Department of Water and Sanitation, 2016). The REMP is an umbrella project that monitors the ecological condition of rivers by assessing different aspects of the ecology of rivers such as fish, riparian vegetation, and macroinvertebrates to monitor river health, through biological indices such as the FRAI (Fish Assemblage Integrity Index); VEGRAI and South African Scoring System (SASS5) (Dickens and Graham, 2002). The South African Scoring System is a widely applied tool for evaluating the condition of rivers using benthic macroinvertebrate communities as indicators since these are sensitive to pollution, flow alteration, and habitat degradation (Dickens and Graham, 2002). The method provides a rapid and cost-effective indication of ecological condition (Dickens and Graham, 2002; Dallas, 2007). Macroinvertebrates are assigned taxon-specific quality scores, and the total of these scores per site makes up the SASS score, while the average score per taxon (ASPT) is the SASS score divided by the number of taxa (Dickens and Graham, 2002). Higher SASS and ASPT scores indicate better water quality (Dickens and Graham, 2002; Jordaan et al., 2020). Using SASS5 alongside habitat covariates allows for a more comprehensive understanding of the ecological drivers influencing riparian mammal probability of site use. Pollution and human modification do influence macroinvertebrate communities, but they vary considerably with changes in natural conditions such as droughts and floods (Agboola et al., 2020). That is why it is preferable that it is used in conjunction with

other indices, as it is with the REMP. A key comprehensive outcome of the REMP is the EcoStatus (Department of Water and Sanitation, 2016). The EcoStatus refers to all the different characteristics of the river and surrounding riparian vegetation, and how that reflects the ability of the river ecosystem to support the animals and plants and to provide ecosystem services (Iversen et al., 2000; Kleynhans et al., 2007). At a broader national scale, the Ecological Condition Index (ECI) was developed as part of South Africa's ecosystem accounting framework to assess ecosystem integrity and change across South Africa (Nel and Driver, 2015).

The ECI consists of a composite of flow, water quality, instream condition, and riparian condition which helped to establish reference conditions and set up monitoring by rating the degree of modification on a scale between 0 (heavily modified) to 100 (natural). The ECI of freshwater habitats in South Africa was found to have decreased between 1999 and 2011 with 10.12%, which indicates overall degradation of water resources in that time period (Nel and Driver, 2015). There have not been recent reports of the ECI since 2011.

2.2.4.2 Physicochemical parameters

As previously mentioned, in addition to biological indices, water quality is also assessed using physicochemical parameters such as pH, dissolved oxygen, and water temperature (Dallas, 2007). These parameters give information about the health of river ecosystems and help to inform the monitoring and conservation management of water resources (Wetzel, 2001; Dudgeon et al., 2006). Overall, both biological indices and physicochemical parameters are necessary for a comprehensive overview of aquatic ecosystem health. A few of the main physicochemical parameters are discussed here.

pH is an important parameter. It refers to the negative logarithm of the hydrogen ion concentration and shows to what measure a solution is acidic or basic. The activity of the hydrogen ions is important in maintenance of biological activity (Dodds and Whiles. 2010). The pH can be linked to soil type, geology and vegetation surrounding the river. For example, due to high tannin content in leaves, rivers in Fynbos areas can be more acidic, while limestone rocks can make water more alkaline. The link between pH and ions in the water means that salinity, mine waste and fertilizer can affect pH (Chapman, 1996).

Dissolved oxygen (DO) refers to the amount of oxygen in the water. Generally, the higher the oxygen concentration is, the higher the water quality is considered to be, since there is more oxygen available for aquatic organisms (McCaffrey, n.d.). It is linked to the pressure, salinity, and temperature of the water.

Water temperature influences a variety of different processes in aquatic habitats. As water temperature increases, dissolved oxygen decreases, which means less oxygen is available to organisms within the environment. Water temperature also has an influence on the rate of photosynthesis in plants, aquatic animals' metabolic rate, and many other factors such as migration, parasitic infection, and occurrence of diseases (McCaffrey, n.d.).

The salinity of water is the number of ions/salts in the water and is measured by electrical conductivity (EC) or total dissolved solids (TDS) (Dodds and Whiles, 2010). These two parameters are thus closely related. Electrical conductivity refers to the ability of water to carry an electrical current over a certain distance. Water with a lot of nutrients has higher conductivity because a higher concentration of dissolved solids means more electricity gets conducted and vice versa (Dodds and Whiles, 2010). TDS as the name implies is the amount of organic and inorganic solutes in the water (Hancock, 2016). This includes nutrients such as nitrogen and phosphorus, two nutrients which are often focused on in water quality. They are both parts of nutrient cycles in aquatic ecosystems. Nitrogen plays an important role in protein synthesis and chlorophyll production (Fathi, 2022). Phosphorus plays an important role in DNA and RNA synthesis, as well as energy transfer (Conley et al., 2009). However, they can be detrimental if in excess. Nitrogen and phosphorus are present in many fertilizers and other agricultural runoff. If water is too high in nitrogen and phosphorus, eutrophication can occur, which can lead to toxin production and oxygen depletion within aquatic habitats (Suresh et al., 2023).

Turbidity is related to the amount of suspended soil or algae within the water and influences light absorption or scattering of light by water. Lower turbidity means higher visibility and vice versa. This affects predator-prey dynamics, with high turbidity making it harder for predators to see prey, even affecting those predators that rely on non-visual cues (Ortega et al., 2020). High turbidity also affects photosynthesis of aquatic plants and algae since it affects the amount of light that passes through to the river bottom. Turbidity affects the taste, smell and appearance of water. Temperature increases as turbidity increases, as an increase in the

number of solutes means more sunlight is absorbed by these particles (Birmachu et al., 2024). Generally, water with higher turbidity is considered to have lower water quality. As you move from a river's headwaters downstream to its lower reaches, water quality tends to decline (Dallas and Day, 2004). Dissolved oxygen often decreases, while conductivity and total dissolved solids increase. pH also tends to increase as you move downstream. This pattern reflects not only the cumulative inputs from upstream catchments but also higher human presence in downstream areas, greater bank erosion, agricultural runoff, slower velocity, and other disturbances from surrounding land uses (Dodds and Oakes, 2007). These combined factors contribute to higher nutrient and sediment loads and overall reduced water quality.

2.3 Breede River Catchment

The study area falls within the Breede Gouritz Water Management Area, specifically in the Breede river Catchment Area. This is a predominantly rural region where agriculture and tourism are the main economic drivers (Breede-Overberg Catchment Management Agency, 2011; Pool-Stanvliet et al., 2017). Intensive agriculture activities – especially grain, fruit and diary production – place pressure on the water resources within this catchment area (Breede-Overberg Catchment Management Agency, 2011) because of agricultural pollution (herbicides and pesticide runoff). Due to the importance of agricultural exports to other countries (Oberholster and Botha, 2014), as well as the high occurrence of tourism in the area, there are economical concerns regarding declining water quality of the Breede River. As water quality declines, tourism and the other industries will suffer as well.

A management plan has been created to help steward the Breede River Catchment Area, called the Environmental Resources Protection Plan (ERPP) (Seeliger et al., 2018). It looks at the ecological based approach to conservation of these resources and emphasizes sustainable use (Pool-Stanvliet et al., 2017). Apart from agricultural runoff, other key issues are untreated sewage deposition (Pool-Stanvliet et al., 2017; Seeliger et al., 2018), as well as invasion of alien plants and fish (Breede-Overberg Catchment Management Agency, n.d.; Dallas et al., 2017; Seeliger et al., 2018) both which influence riparian structure and aquatic biodiversity

respectively (Department of Water and Sanitation, 2011; Dallas et al., 2017; Cape Nature, 2020).

Invasive fish species found in the Breede River Catchment Area include sharptooth catfish (*Clarias gariepinus*) and largemouth bass (*Micropterus salmoides)* which threaten the indigenous fish species and influence habitat integrity. Other invasive fish species include rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), spotted bass *(Micropterus punctulatus) and* smallmouth bass *(Micropterus dolomieu*), and banded tilapia (*Tilapia sparrmanii*) (Tweddle et al., 2009; Dallas et al., 2017). Where bass species occur, especially smallmouth bass (*Micropterus dolomieu),* there are relatively few indigenous species present (Tweddle et al., 2009). Problematic alien trees in the Cape Fold Ecoregion areas are black wattle (*Acacia mearnsii*), Port Jackson *(Acacia saligna)*, pine trees (*Pinus* spp.) and beefwoods (*Casuarina* spp.) (Dallas et al., 2017). Generally, rivers in the catchment area consist of small catchments and large estuaries except for the Breede River which is relatively large. There is considerable variability in stream flow between seasons (Breede-Overberg Catchment Management Agency, n.d.).

The head water of the Breede river catchment is in good condition because it occurs in the mountains and conservation areas and is not as easy to access (Breede-Overberg Catchment Management Agency, 2011). Water quality in the Breede-Overberg area decreases as water moves into main streams and rivers (Department of Water and Sanitation, 2011) because of increased human settlements and pollution. The catchment area's water quality is monitored monthly since 2010 by the Breede-Gouritz Catchment Management Agency (BGCMA) along 23 different points of the Breede River and tributaries (Cullis et al., 2018). The pH levels of the Breede River close to Buffeljagsrivier, Swellendam and Bontebok National Park are generally ideal. However, phosphorous and nitrogen levels, as well as chemical oxygen demands are all classified as not acceptable at this point (Cullis et al., 2018). This high nutrient content potentially contributes to eutrophication risk, which will also contribute to bad smell and possible detriments to human health. Water in the Breede river is also quite saline because of shale soils and municipal waste deposition in the river (high nutrient and pathogen content) (Department of Water and Sanitation, 2011). However, salinity in rivers is regulated by release of water from the Greater Brandvlei Dam (Department of Water and Sanitation, 2011; Cullis et al., 2018;). Other potential issues in this catchment

area include increased turbidity because of sand mining in Ashton, Barrydale and Suurbraak area. There is also a lot of organic materials deposited into the river because of cattle/dairy farming. Pressure on water resources will increase because of irrigation need and climate change (Cullis et al., 2018).

The tributaries of the Breede river, namely the Buffeljags and Grootvadersbosch Rivers in the Western Cape, support diverse riparian and freshwater habitats. These tributaries flow through both conservation areas and agricultural landscapes, providing an ideal setting to study how habitat structure and water quality influence riparian mammal distributions. Because these systems reflect both local habitat conditions and broader ecosystem health, riparian mammals here can serve as sentinel species, providing early indicators of changes in ecological condition.

2.4 Sentinel species

Sentinel species are organisms that reflect negative impacts on the environment such as pollution, disease or other human impacts (National Research Council, 1991; Stahl, 1997). This is possible because they are sensitive to any changes in the environment and respond faster to the presence of certain factors, e.g., the presence of toxins, than humans do, or need lower exposure to react accordingly (National Research Council, 1991; Bossart, 2006). Hazen et al. (2019), describes an ecosystem sentinel as ‘a species that responds to ecosystem variability and/or change in a timely, measurable, and interpretable way, and can indicate an otherwise unobserved change in ecosystem structure or function’. This makes them well suited to be indicators of changes in an environment and for human health (National Research Council, 1991; Van der Schalie et al., 1999; Hazen et al., 2019) before its effects become widespread or can cause harm (Milnes and Guillette, 2008).

In modern ecological monitoring, sentinel species are used in both terrestrial and aquatic systems to track the presence of heavy metals, pesticides, and other pollutants (Jiang et al., 2017; Baos et al., 2022). For example, changes in the population of the red kite (*Milvus milvus*) in certain areas of Spain were investigated because of wildlife poisoning (poisoned baiting) (Mateo-Tomás et al., 2020). Red kites are sentinels because they hunt or scavenge a variety of prey animals

including birds and rodents (Mateo-Tomás et al., 2020). The effect of poisons on their population was directly assessed investigating the number of poisoned individuals and the population decline over a period of ten years.

A few important criteria for sentinel species include ease of sampling their tissue; a higher order trophic position similar to humans and a clear response to changes in environment, (Schwacke et al., 2013; Hazen et al., 2019). A key example that fulfils these requirements is the Humpback whale (*Megaptera novaeangliae)* (Fleming et al., 2016). Adverse changes in behaviour and health of higher order species can affect the entire ecosystem (trophic cascades etc.) (Peterson and Schulte, 2016). Other main criteria for sentinels include abundance in presence, the ability to concentrate metals and potential pollutants as well as larger size to allow easy analysis (Beeby et al., 2001). Marine mammals such as dolphins and manatees are popularly used to monitor conditions in marine environments (Bossart, 1999; Zacharias and Roff, 2001; Schwacke et al., 2013; Hazen et al., 2019). These marine mammal sentinels are ideal because of their higher position in the food chain and their fat storage of toxins (Bossart, 2006). However, semi-aquatic predators are also important as sentinel species in freshwater habitats, as they fulfil many of the above requirements such as their position in the food chain as secondary or tertiary consumers. For example, in different studies of a mine spill in Spain using Eurasian otter *(Lutra lutra*) as a sentinel, the research clearly showed higher concentrations of metal in the scat after the spill (Baos et al., 2022). This data was used to monitor changes as the ecosystem recovered.

The main argument against the sentinel species concept includes the question whether one species can accurately reflect changes in an environment on behalf of all other species, as it only reflects its own niche (Simberloff, 1998). However, it is not possible to easily assess all components of an ecosystem to see the effects of an environmental threat. So, despite these arguments, sentinel species do function well as a warning signal of change in ecosystems (Baos et al., 2022) and thus also play an essential role in the conservation of biodiversity (Marneweck et al., 2022).

Some sentinels can double as flagship species as well (Zacharias and Roff, 2001). Flagship species are animal species that are used as ambassadors for conservation of the species or the habitat (Caro and O'Doherty, 1999). Both sentinels and flagships play a role in conservation of natural areas, but flagships

are more socio-economic in nature. They are often charismatic, larger mammal species that appeal to the public through their appearance and 'personality' e.g., dolphins (Milner-Gulland and Woodroffe, 2001; Stevens et al., 2011). The reason this works is that people allocate more worth to these charismatic species (Bossart, 1999). Not only do they capture the attention of the public, but this interest is transformed into generation of funding and support (Jepson and Barua, 2015).

Famous examples include the giant panda (*Ailuropoda melanoleuca)* (Jepson and Barua, 2015). However, flagship species are often less favourably regarded by communities that are in close proximity to the species than by global communities (Røskraft et al., 2007). This is seen in otters in fishing communities where they are regarded as competition or a nuisance for the yield of fish (Kloskowski, 2005; Ergete et al., 2018). Despite this, otter species are a key example of a species that act as both sentinel and flagship species. This quote by Syse (2013) sums up the important perceived role of otters in freshwater areas:

'Because otters required unpolluted, pure nature, having otters around became almost like a quality control confirming that this part of Britain was a good, pure and clean place to live and work in.'

The different species are used as sentinels of water quality and flagships globally in both marine and freshwater areas (Stevens, et al., 2011), especially the sea otter (*Enhydra lutris*) (Jessup et al., 2004) and the Eurasian otter (Foster-Turley, 1990; Butler and Hillman, 1995, Jessup et al., 2004; Brand et al., 2020; Baos et al., 2022). The sea otter is commonly used as sentinel species in marine habitats (Jessup et al., 2004). Shellfish concentrate chemicals in their tissues and form a large part of sea otters' diet. Sea otters can concentrate organochemicals up to 240 times the original concentration (Kannan et al., 2004) making it much easier to detect and unfortunately causing the detrimental effects to be more pronounced in these species. They are thus quite vulnerable to pollution. Bioaccumulation of pollutants can cause reduced reproduction and death (Foster-Turley, 1990). Additionally, their charismatic nature and public appeal make them good flagship species (Stevens et al., 2011). As flagship species, they raise awareness of issues affecting marine habitats because of the specific interest people have in them. Eurasian otters function as sentinels and flagships in several freshwater areas in Europe (Bifolchi and Lode, 2005; Stevens et al., 2011). Due to overhunting, water pollution and habitat destruction in the 1980s, the populations of Eurasian otters in

Europe experienced heavy declines and populations were highly fragmented (Mason and Macdonald, 1986; Foster-Turley, 1990).

Contamination of otter populations because of pollutants such as PCBs, metals and pesticides can occur through direct poisoning of otters via ingestion of prey or indirectly through prey population decline because of pollutants affecting those populations (Lemarchand et al., 2011). The concentration of metals in otter tissues can be used to investigate the effect of environmental disturbances on otters, such as research using otters to monitor the effect of industrialization on river systems in England (Brand et al., 2020) and pollution because of mine pollution in the Guadimar River (Cianfrani et al., 2011; Baos et al., 2022). Due to their wide diversity of prey, Eurasian otters are used as sentinels for pollutants such as PCBs in France in the Loire River Catchment (Lemarchand et al., 2011). North American river otters are considered as sentinels in Chesapeake Bay (Dunlap, 2008; Parker, 2021).

In Zimbabwe, African clawless otters have been used to monitor the effects of agricultural pesticides (PCBs and heavy metals) on the aquatic ecosystem by looking at the presence of these chemicals in the species (Butler and Hilman, 1995). Traces of PCB's were also found in the species in Kwazulu-Natal (Mason and Rowe-Rowe, 1992), and more recently in Cape Town (Okes, 2017).

Bioaccumulation of pollutants such as polychlorinated biphenyls (PCBs) and DDTs has been linked to reduced reproduction and mortality in otters (Foster-Turley, 1990; Roos et al., 2012). However, conservation efforts for otters have improved globally since the 1990s, especially in Europe (Estes, 1994). And it was found that when water quality and riparian vegetation improved, and pollutants were controlled, populations recovered well. This recovery of Eurasian otters (*Lutra lutra*) in parts of Europe demonstrates their resilience (Fox, 1999; Roos et al., 2012).

African clawless otters (*Aonyx capensis*) and spotted-necked otters (*Hydrictis maculicollis*) can potentially be used as indicators of health in riparian zones in South Africa by analysing their distribution in terms of water quality and the levels of glucocorticoid concentrations in their spraints (Majelantle, 2019; Majelantle, et al., 2020).

Whether African clawless other can truly be seen as sentinel species, is uncertain, since they do survive well in urban areas with considerable pollution (Okes et al., 2016; Okes and O'Riain, 2017). However, in addition to the concern for

long term effects of pollutants such as PCB's (Montano et al., 2022; Kean et al., 2021), there is also high stress and long-term consequences in urban environments for this species (Majelantle et al. 2020). Majelantle et al. (2020) found that African clawless otters had higher glucocorticoid concentrations in urban areas than those in undisturbed areas because of them experiencing more stress. The high concentrations of glucocorticoids have negative impacts in the long run (Boyle et al., 2021). Some impacts on vertebrates include the suppression of immune function (Kaiser et al., 2015); impaired learning ability and accelerated aging of the brain (MacKenzie, and Sapolsky, 2007); muscle atrophy and osteoporosis (Watson et al., 2012) and stunting of growth and reproduction (Sapolsky et al., 2000).

Spotted necked otters are more dependent on clean, pristine riverine areas than African clawless otters (Reed-Smith et al., 2015). Therefore, spotted necked otters' distribution can potentially be used to make assumptions about water and habitat quality. With African clawless otters, their distribution and abundance could not necessarily be correlated with high water quality, however their requirement for shelter, and dense vegetation means that the quality of riparian vegetation can play a role in their occurrence along riverine areas (Somers and Nel, 2004; Kubekha et al., 2013). Extending this idea, many other riparian mammals occupy a similar ecological niche, relying on both terrestrial and aquatic habitats for survival. Because riparian mammals are influenced by both aquatic and terrestrial ecosystems (Hamilton et al., 2015; Girmay et al., 2025), they provide valuable insights into how habitat condition and water quality influence their occurrence and behaviour.

In South Africa, generally, macroinvertebrates and fish are preferred as indicator/sentinel species (Dickens and Graham, 2002; Agboola et al., 2019; Levin, et al., 2019). For example, the carp was used as a bioindicator for microplastic pollution in the Vaal river (Saad et al., 2022). This makes sense because they are more directly affected by changes in stream ecology (Ballash et al., 2022).

Nevertheless, while fish and macroinvertebrates remain established indicators of aquatic ecosystem integrity, riparian mammals offer a complementary perspective that integrates both terrestrial and aquatic ecosystem processes. Their distribution patterns may thus serve as broader indicators of catchment health. In this study, occupancy models were used to explore how environmental variables such as water quality and vegetation structure influence riparian mammal site use,

thereby assessing their potential as sentinel species of riparian ecosystem condition.

2.5 Habitat covariates that influence riparian mammal distribution/site use

2.5.1 Riparian mammals

In South Africa, the mammals that are often associated with riparian zones are otters (*Aonyx capensis* and *Hydrictis maculicollis*), riverine rabbit (*Bunolagus monticularis*); water mongoose (*Atilax paludinosis*) (Crous et al., 2011; Okes et al., 2016). The hippopotamus (*Hippopotamus amphibicus*) and the water rat (*Dasymus incomtus*) are two other species closely associated with rivers (Lloyd, 2000). Species like bushbuck (*Tragelaphus scriptus*), nyala (*Tragelaphus angasii*) and kudu (*Tragelaphus strepsiceros*) as well as bushpigs (*Potamochoerus larvatus*) and Cape genets (*Genetta tigrina*) like wooded areas or areas with thick vegetation, meaning riparian zones are ideal habitat for them and they are often found there (see Hanekom and Randall, 2015). Most mammals that are found in riparian habitats in the Western Cape are riparian-dependent terrestrial species (Catterall et al., 2007). Riparian-dependant terrestrial species are defined by Catterall et al. (2007) as species that use riparian zones for shelter or food but are not necessarily exclusive to riparian zones. This means they do occur and utilize other habitats in the landscape as well. Riparian species often overlap with those commonly found in indigenous forest areas. The way mammals use riparian zones differ, for example, bushbuck use thick cover and vegetation as shelter and food. Otters often have their holts in the riparian zones and hunt in the rivers. Mammals commonly associated with riparian zones in the Western Cape are African clawless otters, water mongoose, riverine rabbits, Cape genets, Cape bushbuck and bushpigs. Due to the cover provided by the surrounding vegetation and resources in riparian zones other animals that can be expected to be found there are honey badgers (*Melivora capensis*) and Cape porcupines (*Hystrix africaustralis*). Caracal (*Caracal caracal*) like thick vegetation for hunting (Veals et al., 2020) and can thus reasonably be expected to occur in riparian zones on occasion. As indicators of ecosystem integrity, riparian mammals respond to variation in habitat structure and water

quality. The following section outlines key environmental drivers that influence their site use.

2.5.2 Habitat complexity and riparian mammals

The species richness of riparian zones could be attributed to habitat complexity (Maestas et al., 2023). More complex vegetation structure can lead to higher animal biodiversity (Hamilton et al., 2015). Habitat complexity is influenced by vegetation type and structure (Coverdale and Davies, 2023). Riparian vegetation is often taller, thicker and more complex than surrounding areas (Catterall et al., 2007). The abundance of trees, shrubs and reeds in riparian zones provide a variety of shelter, food and nesting places (Martin, 2002; Catterall et al., 2007). Such structural diversity in vegetation contributes to habitat heterogeneity (Swan et al., 2020). The heterogeneity of a habitat is an important factor in animal occurrence, since it affects the number of niches available and therefore the biodiversity of species that occur there (Bazzaz, 1975; Tews et al., 2003). For example, it has been found in rivers that an increase in woody debris and roots increases habitat for fish and crustaceans (Crook and Robertson, 1999; Talmage et al., 2002). Canopy cover and vegetation abundance also play an important role in determining species richness and occurrence in habitats. More vegetation cover contributes to higher food resources, higher insect population (Peng et al., 2020) and thus, more available prey, as well as more cover and shelter depending on vegetation height (Wolff et al., 2019). Vegetation type, canopy cover and vegetation cover can be seen as a broad scale measure of habitat heterogeneity however it might overlook some factors that influence occupancy/site use such as type and height of plants (e.g. shrubs vs small trees vs tall trees); depth of leaf litter and presence of debris such as logs, etc., as well as distance from habitat/plant community edge (Stirnemann et al., 2015; Swan et al., 2020). Habitat fragmentation leads to a decrease in species biodiversity (Basile et al., 2021). An increase in canopy cover such as in tree rich sections of riparian zones means an increase in shelter and covering from potential predators or better ambush of prey in the case of predators (Monroy-Vilchis et al., 2009; Regmi et al., 2023). Canopy cover is an environmental covariate that can attract various species (Oladimeji et al., 2024). Regmi et al (2023) found in their study of mammal occupancy in a hunting reserve in Nepal that an increase in

canopy cover led to a direct increase in occupancy in all the species they studied. There was a similar finding on their study of carnivores in Brazil, finding a positive correlation between occupancy and forest cover with occupancy decreasing when the habitat is more fragmented (Regolin et al., 2017). As seen in these examples, animal abundance is generally encouraged by an increase in vegetation, however if shrub density is too thick or impenetrable it can limit movement and distribution of animals (Patterson et al., 2008). It is also important to distinguish between the cover invasive vegetation and indigenous plants. In general, an increase in invasive plants leads to a decrease in animal abundance and species richness as found in a study by Clusella-Trullas and Garcia (2017). Disturbance, such as the removal of riparian vegetation for agriculture, reduces structural complexity and heterogeneity and favours generalist or invasive species, while negatively affecting indigenous and specialist species (Naiman et al., 1993; Tews et al., 2003; Clusella-Trullas and Garcia, 2017). Even when species richness appears unchanged, the loss of specialists often reflects reduced habitat quality and ecological function (Knapp et al., 2016; Hessen, 2024). In addition to the effect vegetation has on mammal occupancy, the restricted area of riparian zones also affect occupancy/site use. The fact that riparian zones are ‘linear’ and smaller in comparison with surrounding habitat, means that there is a higher chance of interaction between different species, which will also influence movement and therefore occupancy of the given area (Bonesi and Macdonald, 2004; Holland et al., 2019a). They are also more vulnerable to fragmentation (Shrestha et al., 2012) which decreases habitat quality and therefore influences occupancy/population density (Osman et al., 2022)

As outlined above, canopy cover, vegetation type, and habitat heterogeneity play key roles in determining species occurrence and biodiversity. In conclusion, it would be expected that higher vegetation density and canopy cover would be more conducive to an increase in animal occupancy (Laurance et al., 2008; Regmi et al., 2023), since as previously mentioned, these are key components of riparian zones that encourage high mammal biodiversity. There is a difference in how much they influence individual species however, so the next section examines how these structural characteristics of riparian zones influence the occupancy, habitat use, and ecological roles of individual riparian mammals in riparian zones of the Western Cape. High riparian vegetation cover positively influences benthic macro invertebrates and water quality (Gu et al., 2025).

2.6 The effect of water quality on riparian mammals

Water quality does not always directly affect the distribution of terrestrial riparian mammals, except at extreme levels of pollution, when it can cause illness when they drink the water. At lower levels of pollution, it can affect the aquatic fuana (i.e. fish, crustaceans etc) populations e.g. decline in populations or change in freshwater community structure (Singh et al., 2025). This directly affect mammals that depend on aquatic organisms for prey, such as otters and water mongoose. For primarily terrestrial riparian mammals, poor water quality may not directly limit occupancy, but it often coincides with degraded riparian vegetation and increased human disturbance, both of which reduce habitat suitability (Naiman and Décamps, 1997; Regmi et al., 2023; Chua et al., 2019). Removal of riparian vegetation leads to an increase in erosion, and therefore turbidity in the river, which affects dissolved oxygen and other water quality parameters, therefore decreasing water quality. It also means less filtering of pollutants from the river (Naiman and Decamps, 1997; Talmage et al., 2002). For example, Southern River otters (*Lontra provocax)* in Chile were found to prefer habitats with intact riparian vegetation (Medina-Vogel et al., 2003). This is specifically linked to their need for shelter and the effect that riparian degradation has on their prey populations.

In South Africa, SASS scores reflect the sensitivity of macroinvertebrate communities to pollution, flow alteration, and habitat degradation, and can therefore serve as indirect indicators of riparian habitat quality (Dickens and Graham, 2002). Higher SASS and ASPT scores are indicative of more pristine rivers (Dickens and Graham, 2002). Since riparian mammals depend on the integrity of both terrestrial vegetation and aquatic prey communities, lower SASS scores (indicating degraded water quality) could reflect reduced prey availability or habitat suitability for semi-aquatic predators such as otters and water mongooses (Medina-Vogel et al., 2003; Chua et al., 2019). In addition to SASS, physicochemical parameters such as dissolved oxygen, pH, water temperature, and conductivity provided direct measures of water quality that can influence the presence of riparian mammals. Including both biological indices and physicochemical covariates alongside vegetation covariates allows for a more holistic assessment of the environmental factors influencing riparian mammal occupancy (Villar-Rúa et al., 2024).

2.7 Riparian mammals in the Western Cape

The African clawless otter (*Aonyx capensis*) is strongly associated with riparian habitats (Kubekha et al., 2013). They prefer river sections with dense vegetation, reeds, boulders, and shallow pools, which provide shelter, holts, and protection from predators and human disturbance (Somers and Nel, 2004; Okes et al., 2016; Haring et al., 2023). River width and depth can also influence occurrence. Water level and prey availability, particularly of freshwater crabs, directly influence otter abundance in freshwater, highlighting the importance of well-structured riparian zones (Mason, 1995; Perrin and Carugati, 2000). Otters' reliance on dense vegetation (Somers and Nel, 2004) and abundant prey, especially crustaceans (Somers and Nel, 2007), is one of the reasons why they are often used as indicators of riparian condition (Bedford, 2009; Holland et al., 2019b; Moun et al., 2024).

Despite the wide distribution of African clawless otters, the species is classified as Near Threatened because of declines in freshwater habitat quality and removal of surrounding vegetation (Somers and Nel, 2013; Jacques et al., 2015; Haring et al., 2023). This emphasizes the need for monitoring and protection of riparian corridors (Okes et al., 2016). Water quality affects the distribution of this species mainly through its influence on prey populations. Okes and O'Riain (2017) found that the probability of otter occurrence decreases near urbanized areas, reflecting the effects of pollution and habitat alteration.

In South Africa, African clawless otters show some adaptability to urban environments (Okes and O'Riain, 2017), but still depend strongly on more pristine aquatic habitats, including rivers outside urban centres and marine protected areas (Kubheka et al., 2012; Okes et al., 2016; Ponsonby et al., 2019). Reliable estimates of population density are essential for monitoring these changes, as population density reflects both habitat suitability and ecosystem health (Connor et al., 2000; Majelantle et al., 2021). However, given the challenges of estimating density for secretive species such as African clawless otters, occupancy modelling offers a valuable alternative for understanding spatial distribution (MacKenzie et al., 2002).

Water mongooses are common in riparian areas and have some overlap with the niche of African clawless otters (Do Linh San et al., 2020; Baker et al., 2016; Haring, Weier and Lindin, 2023). They are 'generalist opportunistic foragers' but do have a preference for crustaceans (Baker et al., 2016; Do Linh San et al., 2020).

Water mongooses like rocky habitats as resting places (Baker et al., 2016). They require dense vegetation and prefer sandy/rocky shores with reeds (Haring et al., 2023). They also prefer narrower and shallower streams with copious amounts of leaf litter (Haring et al., 2023). They typically forage along riverbanks rather than in water (Streicher et al., 2021). They are common and are often captured on camera traps (De Luca and Mpunga, 2005; Haring et al., 2023). Their dependence on riparian zones means they are possibly good indicator species, but they are quite adaptable to pollution and generally not seen as sensitive enough to act as sentinels (Baker et al., 2016). However, pesticide usage did have a negative effect on water mongoose occupancy in the Drakensberg midlands, showing some potential sensitivity to pollution (Ramesh and Downs, 2015). They can occur in more urban areas but tend to show reduced home-range sizes in these landscapes (Streicher et al., 2021).

Cape genets (*Genetta tigrina*) are often found in riparian areas, as they prefer habitat close to permanent water and are thought to use riparian areas often as corridors in agricultural areas (Widdows et al., 2015). They are not directly dependant on water for food (Roberts et al., 2007) but have been found to prefer the riparian zones compared with other sections of the forest (Fuller et al., 1990). They are generalist predators and eat insects and small mammals (Plaatjie et al., 2024). The trees in riparian zones provide shelter and can allow safety from predators (Ramesh and Downs, 2014; Ehlers Smith et al. 2017). They are well adapted to urban areas as well (Widdows et al., 2015). In a study by Oladimeji et al. (2024), they found that there is a positive relationship between their distribution and canopy cover which is consistent with previous studies about their habitat preferences (Ramesh and Downs 2014). Their reliance on thick vegetation cover seemed to decrease in urban areas in studies by Widdows et al. (2015) in KwaZulu-Natal. This is because of buildings being available as shelter and the availabity of abundant food being a good motivator for venturing out of the safety of thick vegetation (Widdows and Downs 2015; Widdows et al., 2015).

Bushbucks (*Tragelaphus scriptus*) prefer deeply forested areas (Brock et al., 2003; Crous et al., 2011). They are therefore commonly found in riparian zones (Coates and Downs, 2007). Some earlier studies found that they are more likely to occupy riparian zones in winter than in other seasons for reasons unknown (Jacobsen, 1974; Seydack et al., 1998; Coates and Downs, 2007).

Bushpigs (*Potamochoerus larvatus*) occur in areas with dense vegetation (Seydack, 2017) such as forest, riparian zones and plantations (Venter et al., 2016). They feed on a variety of roots, fruits and invertebrates (Skinner et al., 1976). Due to their foraging habits, they play an important role in seed dispersion in forest habitats, highlighting their ecological role in riparian zones (Venter et al., 2016; Hikel et al., 2024). They are thought to have low population densities and slow population growth rate (Venter et. al., 2016), therefore occupancy would be expected to be low overall.

Cape porcupines (*Hystrix africaeaustralis*) have a wide distribution in South Africa and are found in riparian zones as well (Bragg and Child, 2016). They are herbivorous and feed on a variety of roots, plants, and bulbs (Skinner and Smithers, 1990). Like bushpigs, they also play a role in seedling recruitment and soil aeration (Bragg et al., 2005). Cape porcupines were found to have a positive relationship with canopy cover in studies done in Table Mountain National Park (Oladimeji et al., 2024).

Chacma baboons (*Papio ursinus*) have a wide ecological range and can therefore occur in many different habitats, including riparian zones (Hoffman and O'Riain, 2012; Slater et al., 2018). They are omnivorous and feed on a variety of fruits, roots, leaves, insects and small mammals (Slater and Du Toit, 2002). Baboons play an important role in seed dispersal (Slater and Du Toit, 2002; Pamla et al., 2024). In the arid parts of their zones, they are often found close to permanent water, but in higher rainfall zones they are less tied to water which could be because of adequate moisture found in their food (Hamilton III, 1986). In the Western Cape their abundance was found to be higher at lower altitudes (Hoffman and O'Riain, 2012). They prefer alien vegetation above indigenous vegetation (Hoffman and O'Riain, 2012). Sleeping sites play a role in their distribution as well, they prefer trees, cliffs or caves as ideal sites for sleep (Davidge, 1978; Hoffman and O'Riain, 2012). This could potentially be a drawing factor for them to riparian zones which often has trees and other surrounding landscape that is ideal for sleeping. Riparian zones also have an abundance of food (plants, fruits, etc.) which is another drawing factor.

Caracal (*Caracal caracal*) have a wide distribution, but they do require cover (Skinner and Chibamba, 2005; Avenant et al., 2016). There is no specific affinity recorded for riparian zones, however their preference for cover and the presence of

prey species, makes it highly likely that they occur there. They also use trees to store prey in (Avenant et al., 2016). They are higher order predators, thus playing a role in top down prey regulation in the habitats they occupy (Jooste, 2020; Crookes, 2023).

2.8 Occupancy modelling

2.8.1 Population size and conservation

Population size is an essential parameter in determining the conservation status of species (IUCN, 2024). Knowing accurate numbers of a population is important in conservation management planning and strategies (Sinclair et al., 2006; Jones, 2011; Marques et al., 2013). It is important that there is consistent monitoring of population size to identify and monitor the effect of threats and changes in ecosystems (Yoccoz, 2012).

There are different measures of population size, including population density (the number of individuals per unit area).Population density is useful in measuring population status because it can be compared between locations and times and considers survival and recruitment (Rogan et al., 2019). However, it is time consuming, expensive, and complex to gain accurate population density measures (Marques et al., 2013). This is especially true for cryptic species that have low population densities and/or large territories (Powell et al., 2012). Overestimation can lead to flawed conservation decisions (Quaglietta et al., 2015).

When direct counts are challenging because of low detections, occupancy modelling provides a viable alternative (Delisle et al., 2021; Wearn et al., 2022). Relative abundance indices (RAIs) can also give an approximate measure of abundance from camera trap data (Palmer et al., 2018).

2.8.2 Camera trapping for estimating abundance and occurrence

Camera trapping is a ‘core tool’ in ecology and conservation (Burton et al., 2015). Camera traps gather data that has previously been too expensive or complex to approach. They allow insight into various facets of ecology, and allow important information regarding the behaviour, movement patterns, and abundance of species to be determined (Silveira et al., 2003; Burton et al., 2015; Caravaggi et al., 2017). Camera trapping can be used to estimate different parameters of demography

including population density (Rowcliffe et al., 2008; Palencia et al., 2021); species richness (Tobler et al., 2008; Rowcliffe et al., 2014; Tanwar et al., 2021); occupancy (Cove et al., 2013); distribution (Sollman, 2018), and activity patterns (Edwards et al., 2020). This makes camera trapping a versatile, non-invasive tool for assessing both occupancy and abundance.

Camera trapping is a useful non-invasive method for estimating abundance in shy species, such as leopards (*Panthera pardus*) and caracal (*Caracal caracal*) (Claridge et al., 2004; Rowcliffe, 2017). They are seen as more accurate than indirect methods in giving accurate abundance/population density estimates (Day et al., 2016). Camera traps rely on infra-red sensors to pick up animals moving past the camera within specific diameters. Population abundance can thus be calculated because camera traps record the individuals' 'patterns of abundance through space and time' (Silveira et al., 2003). Mammals are often the focus of camera trapping research (Burton et al. 2015) but it has also been used for other animals such as birds (O'Brien and Kinnaird, 2008) and insects (Naqvi et al., 2022).

Camera traps allow accurate species identification even at night (Sollmann et al., 2013; Hanekom and Randall, 2015). The large number of photos that camera traps can take, allows a large amount of data to be collected (Burton et al., 2015) in relatively short periods of time and with low effort (Hanekom and Randall, 2015) over big spaces (Sollman et al., 2013). Although camera traps can require initial high costs to purchase and setup (Silveira et al., 2003; Schaus et al., 2020), it is worth it because of the advantages and time saved in labour.

A key assumption underlying all camera trap methods is that behaviour and movement are independent of camera trap placement (Caravaggi et al., 2020). This is not always the case since individuals can respond to camera traps and this can affect movement and behaviour (Caravaggi et al., 2020; Broekhuis et al., 2025). The placement of camera traps is also important with random placement versus specific placement often being debated. Random placement in relation to movement of the animals can lead to less detection in comparison with placing it where there is a good chance that an animal will be detected, e.g., along wildlife trails (Fonteyn et al., 2020). However, camera trap placement is not necessarily as important when a species is abundant and when there is high sampling effort (Cusack et. al., 2015; Fonteyn et al., 2020). Bias remains a concern when camera traps are placed along obvious areas such as latrines or trails (Kolowski and

Forrester, 2017). Therefore, random placement is important in population density estimation and occupancy in order to avoid bias while maximizing detection probability (Cusack et. al., 2015; Hofmeester et al., 2021).

There are different models/methods used in camera trap population density or distribution estimates. Common methods of calculating population density through camera traps are 'capture-recapture methods' (Rowcliffe et al., 2008; Xiao et al., 2019) which relies on recognition of individuals. In cases where individual recognition is not possible, camera trap rates can be used to estimate population density e.g. random encounter models (REM), random encounter and staying time models (REST) and camera trap distance sampling models (CT-DS) (Rowcliffe et al., 2008; Palencia et al., 2021). Camera trap rates itself cannot give reliable and accurate population density estimates since it does not consider variation in detection; territory size and movement (Rowcliffe et al. 2008; Rogan et al., 2019). However, camera trap rates can still provide a basic index of relative abundance or activity, allowing for preliminary comparisons across sites or times, even if they do not yield precise population density estimates (O'Brien et al., 2003; Sollmann et al., 2013). And example of camera trapping rate indices includes relative abundance indices (RAI's) (Sollmann et al., 2013). Another more robust method for areas where detections are sparse or species are elusive are occupancy models, a feasible alternative to abundance which estimates the probability of site use rather than population density (MacKenzie et al., 2002; Royle and Dorazio, 2008). Both RAIs and occupancy will be discussed in the next section.

2.8.3 Relative abundance indices

As mentioned before, photographic capture rates from camera trapping allow a rudimentary indication of animal abundance and presence (Sollmann, 2018) and can be used to develop indices of species numbers (Palmer et al., 2018). Camera trapping uses capture-recapture indices in the case of individually recognized animals, but when individuals are not recognizable then abundance indices are used (Sollmann et al., 2013). A common example of abundance indices is relative abundance indices (RAIs). There is a positive correlation between relative abundance and population size (Anderson, 2003; Martin-Garcia et al., 2022). Traditionally, RAIs are the count of animal signs directly related to the population

size (Martin-Garcia et al., 2022). However, it is more common to calculate it from photographic capture rates.

Relative abundance indices can be simply described as the number of independent photographs of the focal species per camera trap day (O'Brien and Kinnaird, 2011; Sollmann et al., 2013). This is based on the camera trap's capture frequency, which is calculated by multiplying the number of cameras with the number of days the cameras were operational (Carbone et al., 2001; Palmer et al., 2018).

Different variations of RAI exist, including:

- the number of days on which a species was photographed,
- the proportion of photographs of a species relative to all species detected,
- the number of photographs of a species per 100 camera-trap days (Nelson and Clark, 1973; Lijun et al., 2019; Gronwald and Russell, 2021).

Among these, the most widely used measure is the number of photographs per 100 camera-trap days, seen as a standardized RAI (O'Brien and Kinnaird, 2011; Sollmann et al., 2013; Palmer et al., 2018). This standardization allows comparisons between studies with differing sampling effort, as it expresses detection rates relative to a consistent measure of effort.

Relative abundance indices are influenced by camera placement, home ranges/territories and movement of species (Sollmann et al., 2013). They are often more precise in more open habitats (Palmer et al., 2018) and as camera trap effort increases (Palmer et al., 2018).

Limitations to relative abundance indices are that photographic capture rates tend to overestimate abundance (Silveira et al., 2003) and imperfect detection is unaccounted for (Sollmann et al., 2013). Relative abundance indices have variable encounter probabilities and therefore had bias toward species that are more easily detected (Sollmann et al., 2013). This means differences in the index amount does not necessarily correlate with the difference in the actual abundance/detection probabilities (Palmer et al., 2018). Other factors that can bias relative abundance indices are body mass (Anile and Devillard, 2015), behaviour (Harmsen et al., 2010), and camera trap survey design (Tobler and Powell, 2013).

Relative abundance indices therefore cannot be compared between studies and species without caution (O'Brien and Kinnaird, 2011). Relative abundance

indices are essentially not calibrated and not 'universally applicable' (Sollmann et al., 2013). Therefore, they are not seen as reliable indications of abundance, especially for conservation purposes (Anderson, 2003; Ulrich and Ollik, 2005; O'Brien and Kinnaird, 2011; Sollmann et al., 2013). Despite these limitations, relative abundance indices can still give useful comparative indications of abundance of mammals across sites and seasons when interpreted carefully (Palmer et al., 2018; Millward et al., 2022; D'Ammando et al., 2022; Huerta-Rodríguez et al., 2022).

All ecological survey methods, including camera trapping, are subject to imperfect detection (Kéry and Royle, 2008; Kéry and Schmidt, 2008). This is because of factors such as animal movement, vegetation density, and camera placement (MacKenzie et al., 2002; Sollmann et al., 2013; Burton et al., 2015; Rovero and Zimmermann, 2016). Also, most survey areas are too large to be surveyed effectively in a given time (Nichols et al., 2008). Additionally, some species are rare and therefore has a lower likelihood of being detected (Kéry and Royle, 2008).

2.8.4 Occupancy models

Unlike relative abundance indices, occupancy models account for imperfect detection, recognizing that species may be missed even when present (MacKenzie, 2002; Kéry and Schmidt, 2008; Royle and Dorazio, 2008). This makes them less biased and more statistically grounded, given that camera traps are set up in a way that minimizes bias(O'Connell et al., 2011). They are thus seen as a statistically sound alternative to relative abundance indices (MacKenzie et al., 2002; Sollmann et al., 2013). They do not require marking or identification of species and therefore allow the distribution patterns of elusive species to be determined (Bailey and Adams, 2005; MacKenzie and Royle, 2005; Sollmann, 2018). Occupancy is generally easier to obtain than population density estimates (Burton et al., 2015; Sollmann, 2018). It is important to note that these models focus on the probability of a site being occupied rather than abundance (MacKenzie et al., 2002; Kéry and Schmidt, 2008; Gaston and He, 2010; Sollmann et al., 2013).

Apart from presence, they are also used to study species co-occurrence, effects of habitat fragmentation, and to compare detection methods (Cooch and

White, 2022). Occupancy is sometimes seen as a surrogate for abundance (MacKenzie and Nichols, 2004; Sollmann et al., 2013), but the true relationship between occupancy and abundance is not fully known (Efford and Dawson, 2012). While occupancy and abundance are correlated (higher occupancy can indicate higher abundance) this relationship depends on species aggregation (Holt et al., 2002; MacKenzie, 2006). At the most, greater occupancy may indicate more locally adapted populations (Kéry and Royle, 2008).

Occupancy is specifically defined as the proportion of sites occupied by the species of interest (MacKenzie et al., 2002; MacKenzie and Royle, 2005). MacKenzie first developed it as a single-species model by looking at the presence or absence of a focal species in different sections of an area within a given season (MacKenzie et al., 2002). Occupancy models are hierarchical meaning they separate the ecological processes (e.g., occupancy) from observation processes (e.g., imperfect detection) (Iknayan et al., 2014; Donovan et al., 2024). Therefore, occupancy models have two main components, occupancy and detection probability:

Occupancy (Ψ) is the probability that a site is occupied by the target species (MacKenzie et al., 2002; Bailey and Adams, 2005).

Detection probability (pj) is the probability of detecting the species during the jth survey, given it is present (detectability) (Bailey and Adams, 2005).

Repeated sampling across time and space is necessary to account for imperfect detection (MacKenzie et al., 2002). During sampling, there is an observation for each occasion/visit (Sollman, 2018). Camera trapping can be used to capture this data (Sollmann, 2018). Even though camera trapping captures continuous data, the data can be made discrete by dividing it into specific intervals e.g. ‘occasions’ (Sollmann, 2018). Detection/non-detection data are then organized into a site by occasion matrix with binary values (MacKenzie et al., 2002; Sollmann, 2018). A one is assigned if there was an observation, while a zero is assigned if there was no observation (MacKenzie et al., 2002; Sollmann, 2018). This detection/non-detection data allows true absence to be separated from non-detection, which allows the detection probability to be estimated. To illustrate why detection probability is important, it is important to remember that it can be assumed that a species is present when it is detected on at least one occasion even if it is not detected on other occasions (Sollmann, 2018).

Occupancy also allows the relationship between wildlife and habitat to be investigated (Gould et al., 2019) and to see what specific environmental factors play a role in the occupancy of a species (Sollmann, 2018). Heterogeneity in the environment that is not modelled can cause negative bias in the occupancy estimates (MacKenzie and Bailey, 2004; MacKenzie et al., 2017). This is done by incorporating site and survey covariates (Cooch and White, 2022), which are incorporated using the logistic equation or logit-link function (Bailey and Adams, 2005). Site covariates refer to covariates that describe the environmental characteristics of the site such as habitat type, vegetation cover, and river width (Sollmann et al., 2018). It is used to describe the variation in occupancy along environmental gradients and make deductions about species distribution patterns (Zurell et al., 2023). In contrast, survey covariates show how detection probability varies across environmental gradients (MacKenzie et al., 2002; Zurell et al., 2023). Survey covariates represent the conditions when the data was collected such as ambient temperature, time of day, observer, and effort (MacKenzie et al., 2002).

Single-season occupancy studies assume a closed population, independent sites, and correctly identified species, with heterogeneity in occupancy and detection probability explained by covariates (Bailey and Adams, 2005; Miller et al., 2011; Ivan and Parks, 2018; Skelly et al., 2018).

The closure assumption requires that the ‘occupancy status’ does not change while the survey is taking place (Bailey and Adams, 2005). This is done by constraining sampling to a season where births, emigration and immigration is limited. If closure is assumed without the population truly being closed, then occupancy will change as emigration and immigration takes place during the survey and the occupancy estimation will be biased (Rota et al., 2009). The closure assumption, although important for single season models, can be limiting if you want to look at occupancy patterns over many months/years and what drives it (MacKenzie et al., 2003). Single-species occupancy models can be modified and expanded to incorporate dynamics over time (multi-seasons) (MacKenzie et al., 2003), false positives (such as with misidentification of animal species), relaxing of closure assumption (staggered entry models or multi-season models) (Dail and Madsen, 2010) and multiple species (Royle and Nichols, 2003; MacKenzie et al., 2004; Miller et al., 2011).

Multi-season or dynamic models account for colonization (γ) and extinction (ε) probabilities, major factors that influence occupancy dynamics (MacKenzie et al., 2003). Extinction probability refers to the probability that a species does not occur at a given site that was previously occupied, while colonization probability is the likelihood that a species now occurs in a site that it has not occupied before (MacKenzie et al., 2003; Hines et al., 2010). So multi-season models do not just look at whether a habitat is presently occupied but also looks at the processes driving changes in occupancy over time.

Multi-species occupancy models incorporate more than onespecies detection/non-detection data and allow estimation of both individual and community-level responses to environmental conditions (MacKenzie et al., 2003; Devarajan et al., 2020; Niedballa, 2024). They improve estimates for rare species by borrowing strength from the community (Dorazio et al., 2006; Devarajan et al., 2020; Niedballa, 2024). This is because these models can handle missing data or low detections better than single-species occupancy models. (Dorazio et al., 2006; Yamaura et al., 2010).

Thus, to summarize, there are four broad groups of models that are used for occupancy (MacKenzie et al., 2018).

1. Single-species, single-season: this is the original occupancy model accounting for imperfect detection. MacKenzie et al., 2002).

2. Single-species, multiple-season (also called dynamic occupancy models): this models colonization and extinction across seasons (MacKenzie et al. 2003; Hines et al., 2010)

3. Multiple-species, single-season: models multiple species, and borrows strength across species (MacKenzie et al., 2004). Has community level distribution outputs as well as species level output.

4. Multiple-species, multiple-season: models dynamic occupancy over time (colonization and extinction) combined with a multispecies structure (Dorazio et al., 2006; Zipkin et al., 2013; Tobler et al., 2015).

Occupancy models can be fitted using Presence (Hines, 2006) or run in R using the packages unmarked (Fiske and Chandler, 2011), RPresence (MacKenzie and Hines, 2018), or spOccupancy (Doser et al., 2022). In occupancy modelling, both frequentist (maximum likelihood) and Bayesian inference are used to fit models (MacKenzie et al., 2004; Royle and Dorazio, 2008). spOccupancy uses Bayesian

inference via MCMC sampling (Doser et al., 2022), while the other programs and packages typically use frequentist inference (MacKenzie et al., 2002).

Both approaches are useful in data analysis, but they present parameters and uncertainty differently (McElreath, 2020). In simple terms, frequentist inference defines the probability of occupancy in light of a specific hypothesis, whereas Bayesian inference evaluates the probability that a hypothesis is true given the data (Ellison, 2004). Frequentist inference is based on the concept of infinite repetition, while Bayesian inference reflects the degree to which one believes an event is likely to occur (Ellison, 2004; McElreath, 2020). In the frequentist framework, model parameters such as occupancy are viewed as fixed and true estimates, whereas in the Bayesian framework, parameters are treated as random variables (Ellison, 2004; McElreath, 2020).

Frequentist models express uncertainty using standard errors (SEs), confidence intervals (CIs), and p-values (Burnham and Anderson, 2002; McElreath, 2020). In contrast, Bayesian inference incorporates prior information about parameters and updates this with observed data to produce posterior probability distributions (MacKenzie et al., 2004; Gelman et al., 2013b). Instead of standard errors, Bayesian models use credible intervals, which represent the probability that a parameter lies within a certain range given the data and prior information (Ellison, 2004; Hooten and Hobbs, 2015; McElreath, 2020). Frequentist approaches are often faster computationally, whereas Bayesian inference is generally more flexible and intuitive (McElreath, 2020).

Single-species occupancy models can be fitted using either frequentist or Bayesian frameworks (Doser, 2022), although the frequentist approach is more common. Multi-species models, however, are usually fitted using Bayesian inference (MacKenzie et al., 2004), since the frequentist framework can become overly complex for multi-species data. Although there is no single study explicitly showing that Bayesian single-species models outperform frequentist ones under rare-species conditions, several studies have demonstrated that Bayesian occupancy models produce less biased and more reliable uncertainty estimates for species with low detection probabilities or few observations (Pacifici et al., 2015; Specht et al., 2017). This is because prior distributions help stabilize estimates when detections are sparse (Royle and Dorazio, 2008).

2.8.5 Fitting of models

'Model fit' refers to how well a model explains the observed data. Both overfitting and underfitting of models mean that prediction is not being done well (McElreath, 2020). With overfitting, too much noise is captured, instead of true data, and with underfitting, you are not learning enough from data and important relationships are missed (McElreath, 2020). The correct model structure and covariate selection are therefore crucial to balance capturing the necessary information and patterns without overcomplicating it. Model fit is assessed using Akaike's Information Criterion (AIC) values (in frequentist approaches) and WAIC or Deviance Information Criterion (DIC) in Bayesian approaches (Burnham and Anderson, 2002; Gelman et al., 2013a). These allow for comparison of different models based on goodness-of-fit and model complexity (MacKenzie and Bailey, 2004). Other tests for model fit are goodness-of-fit tests and posterior predictive checks (Lynch and Western, 2004; MacKenzie and Bailey, 2004; Gelman et al., 2013b). Tests of convergence such as R-hat values or trace plots in Bayesian analysis, are also essential to ensure that models have stabilized and that parameter estimates are reliable. Model fit is not the only factor that should guide model selection (Burnham and Anderson, 2002). For example, covariates should be ecologically justified, and collinearity among site covariates should be checked to avoid inflated variance and misinterpretation of results (Dormann et al., 2012).

2.8.6 Limitations to occupancy models

Occupancy models have certain limitations, despite its wide use and many benefits (Bailey et al., 2013; Delisle et al., 2021). As emphasized before, it is not a measure of abundance/population density and should not be confused as such (Efford and Dawson, 2012; Bowler et al., 2016). Although it is seen an alternative to population density/abundance estimates (MacKenzie and Nichols, 2004), even this should be done carefully since the factors that influence abundance and occupancy are different (Dibner et al., 2017).

Another concern is that because occupancy is designed for a single focal species, it might be assumed that they are not appropriate for additional non-target species (Nichols, 2010; Steenweg et al., 2019). This challenges the concept of

multi-species occupancy models. Occupancy models also do not always consider the movement of animals (Burton et al., 2015; Neilson et al., 2018), which makes the closure assumption difficult (Wright et al., 2016). Occupancy models are also difficult to fit (Welsh et al., 2013). An example is when there are low detections, occupancy modelling becomes unstable, and models struggle to converge (MacKenzie et al., 2002; MacKenzie., 2006; Specht et al., 2017). Part of the reason for this is that if the detection probabilities and number of sampling occasions are low, it is hard for the model to distinguish between true absence and non-detection (MacKenzie et al., 2002).

However, despite these disadvantages, occupancy has proven useful in many studies regarding occurrence of animal species (MacKenzie et al., 2018) and it allows species diversity and distribution to be compared between areas and to see how different environmental factors influence the species occurrence (Tobler et al., 2015).

CHAPTER 3: METHODOLOGY

3.1 Study area

3.1.1 Overview

The study was conducted in two river systems in the Western Cape, within the Breede-Gouritz Water Management Area (FBIS, 2023). The rivers (Buffeljags River and Grootvadersbosch River) of the study areas are tributaries of the lower reaches of the Breede River in the Overberg area of the Western Cape (Dallas et al., 2017; Pool-Stanvliet et al., 2017). See Figure 3.1. for a map of the location of the two study areas within the Western Cape, South Africa.

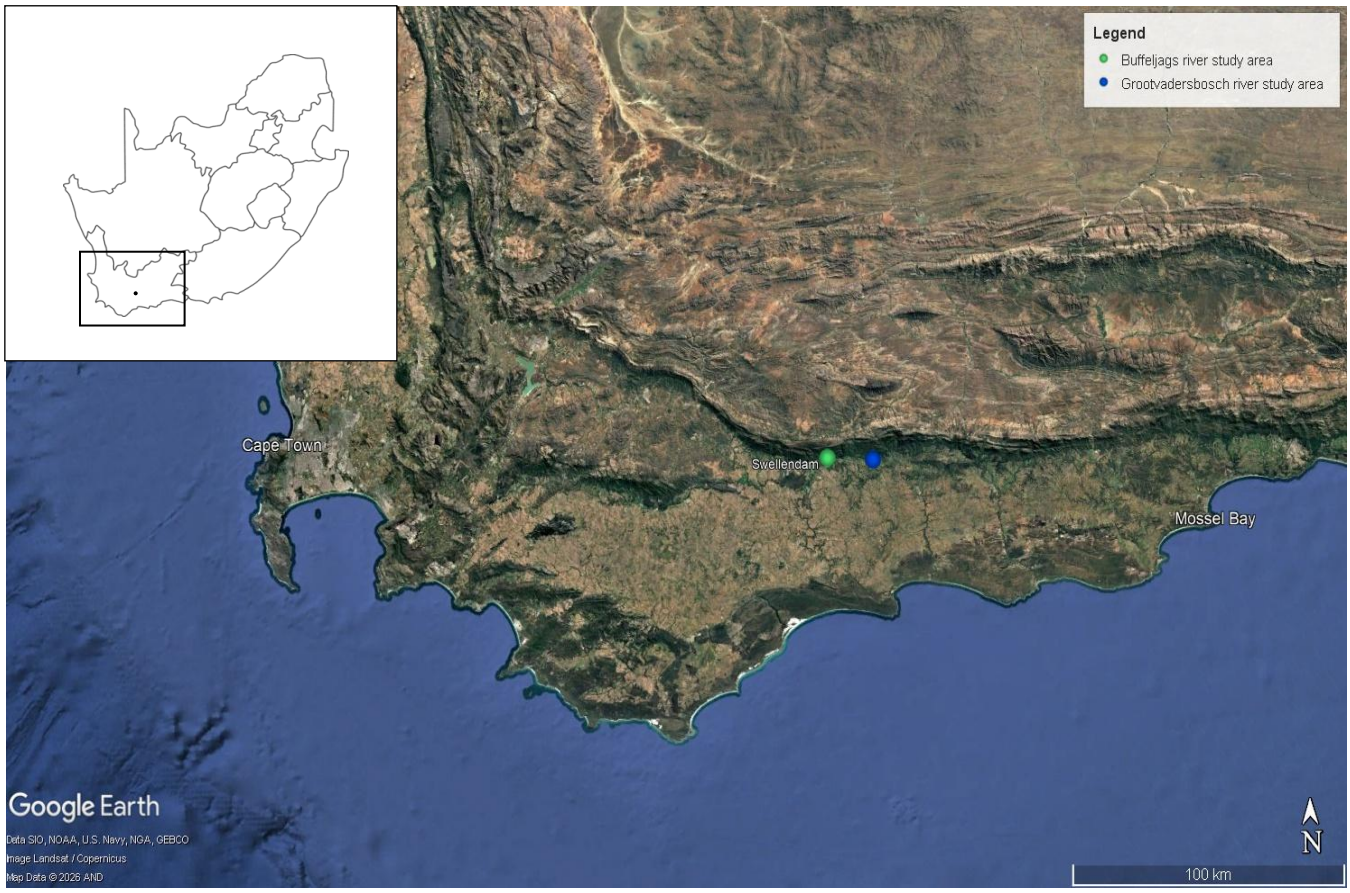

Figure 3.1 Map of the two study areas' location in the Western Cape, South Africa. Base map from Google Earth Pro (Google LLC, 2026). The approximate location of Buffeljags River sites is indicated in green, while the approximate location of Grootvadersbosch River sites is indicated in blue.

The Buffeljags River sites are classified by the FBIS as unprotected while Grootvadersbosch River sites are poorly protected (FBIS, 2023). The rivers of both sites are largely modified and critically endangered (FBIS, 2023), while the riparian zones, classified as Cape Alluvial Vegetation, are classified as endangered

(Creecy, 2022). The surrounding area is primarily a mixture of natural vegetation, guest lodgings and farm areas.

Both study areas are broadly situated in the Fynbos biome (Rebelo et al., 2006) and thus fall within the Cape Floral Region (CFR) which is a World Heritage Site (Pool-Stanvliet et. al., 2017; United Nations Educational, Scientific and Cultural, 2025). The CFR has extremely high endemism and biodiversity of plants and animals, especially invertebrates. There are patches of Southern Afrotemperate Forest in both study areas, but it is quite sparse. The sites specifically occur in the riparian zones which is classified as Cape Lowland alluvial vegetation (Mucina et al., 2006), which will be further discussed in section 3.3. The rivers of the study areas fall within the Breede River Catchment. The Breede River Catchment forms part of the Breede-Gouritz Water-Management area, along with the Gouritz Water Management Area. Both is managed by the Breede-Gouritz Catchment Management Agency (BGCMA) (Breede-Overberg Catchment Management Agency, 2011). Both fall within the freshwater region that are part of the Cape Fold Ecoregion (Dallas et al., 2017). The Cape Fold Ecoregion is one of five aquatic ecoregions in southern Africa (Shelton et al., 2017).

This ecoregion consists of the streams and rivers that flow from the Cape Fold Mountains (Abell et al., 2008). The Cape Fold Ecoregion is bound by the Orange River and the Karoo to the North-West and the Indian Ocean to the East (Jones, 2019). The main rivers in the area are the Berg and Breede, Olifants and Gourtiz rivers (Pool-Stanvliet et al., 2017). The two rivers of the study areas (Buffeljags River and Grootvadersbosch River) are both tributaries of the Breede river. The Breede river originates in the Cape Fold Mountains near Ceres (Pool-Stanvliet et al., 2017; Cullis et al., 2018) while its mouth is at Cape Infanta around 90 km from the most southern point of the Western Cape (Flemming and Martin, 2021). The Grootvadersbosch River originates in the Langeberg Mountains on the second study site, while the Buffeljags River is a bigger river that originates where the Grootvadersbosch River and Tradouw river converge just east of the little town of Suurbaak. The Buffeljags River flows through Suurbraak in the direction of Swellendam joining the Breede River. Although the Buffeljags River is downstream from Grootvadersbosch River and thus potentially have lower water quality (Dodds and Oakes, 2007), there is one factor that changes that pattern: The proximity of the Buffeljags River to the Langeberg Mountain (within 50–100 m), means there is

a lot of inflow from mountain streams and runoff. This potentially affects the water quality in a way making it potentially higher than it would have been without the fresh water contributing to the river.

The two rivers have the typical characteristics of rivers in the Cape Fold Ecoregion, in that they are oligotrophic (nutrient poor), peat-stained and acidic (Dallas and Day, 2007; Jones, 2019). This is because of the high concentration of humic substances that are deposited into the water from the Fynbos areas. There is high biodiversity of aquatic organisms in the rivers of the CFE, including 42 unique fish species (Ellender et al., 2017). Some of the important indigenous fish species are Breede River redfin (*Pseudobarbus burchelli*), Cape galaxias (*Galaxias zebratus*) and the Cape kurper (*Sandelia capensis*) and Berg-Breede whitefish (*Pseudobarbus capensis*) (Ellender et al., 2017; Jordaan et al., 2020).

3.1.2 Buffeljags River sites

The Buffeljags River study area includes three adjacent farms that were treated as a single study site: Frog Mountain Getaway (34.00029°S, 20.60194°E), Somerset Gift Farm (34.00182°S, 20.58753°E) and Franken Farm (34.00329°S, 20.57734°E). Figure 3.2 shows how the general landscape looks next to the Buffeljags River. These three adjacent farms are situated next to the Buffeljags River at the foot of the Langeberg mountains. Frog Mountain Farm and Somerset Gift Farm are dairy and guest farms and fall within the Grootvadersbosch Conservancy. The Grootvadersbosch Conservancy focuses on partnering conservation with agricultural areas (Botha, 2019).

There is regular human presence on both Frog Mountain Getaway Guest Farm and Somerset Gift Farm, with hiking paths next to sections of the river and river activities such as canoeing and ziplining. There are alien trees and plants, especially black wattle (*Acacia mearnsii*), beefwood (*Casuarina* spp.) and blackberry (*Rubus fruticosis*) on both farms. Alien vegetation clearing activities are ongoing in the area, and the indigenous forest and fynbos species are increasing. The Somerset gift farm has mainly pasture next to the river, but reeds and natural vegetation remain in the 20-metre zone next to the river. Cattle roam in sections of both these farms. The Franken farm is a non-inhabited dairy farm with no housing and little human presence on the farm itself. There are cattle and horses grazing

close to the river. The land consists mostly of pastures with thick vegetation next to the river, mainly invasive vegetation such as black wattle (*Casuarina cunninghamiana*), beefwood (*Casserina* species) and bugweed (*Solanum mauritianum).* There are also thick stands of reeds. However, there are large patches of Fynbos and Southern Afrotemperate Forest on the northern bank of the river. The incline increases sharply (within 50m) on the northern side of the river (see Figure 3.2).

Figure 3.2 Photos showing the landscape of the Buffeljags River sites.

3.1.3 Grootvadersbosch River sites

The sites next to the Grootvadersbosch River are two adjacent dairy farms, Grootvadersbosch Farm (34.01017°S, 20.78320°E) and Glen Etive Farm (34.00663°S, 20.76361°E), that falls within the Grootvadersbosch Conservancy and the Overberg Renosterveld Conservation Trust. The general landscape looks next to the Grootvadersbosch River is shown in Figure 3.3. The farms are close to the Grootvadersbosch Nature Reserve and mostly well conserved. Grootvadersbosch Farm has large areas of natural veld. The eastern section of the Grootvadersbosch Farm runs through Southern Afrotemperate forest (Mucina et al., 2006)-see photo in the middle of Figure 3.3. There are some pasture areas. The water quality is almost pristine according to SASS done here in the previous two years (FBIS, 2023). Glen Etive Farm has slightly lower water quality than Grootvadersbosch farm. As with Frogmountain Getaway Farm, Grootvadersbosch Farm also forms part of the Grootvadersbosch Conservancy. This means there is more active removal of alien vegetation on these sites and conservation of natural sections of the farm (Botha, 2019).

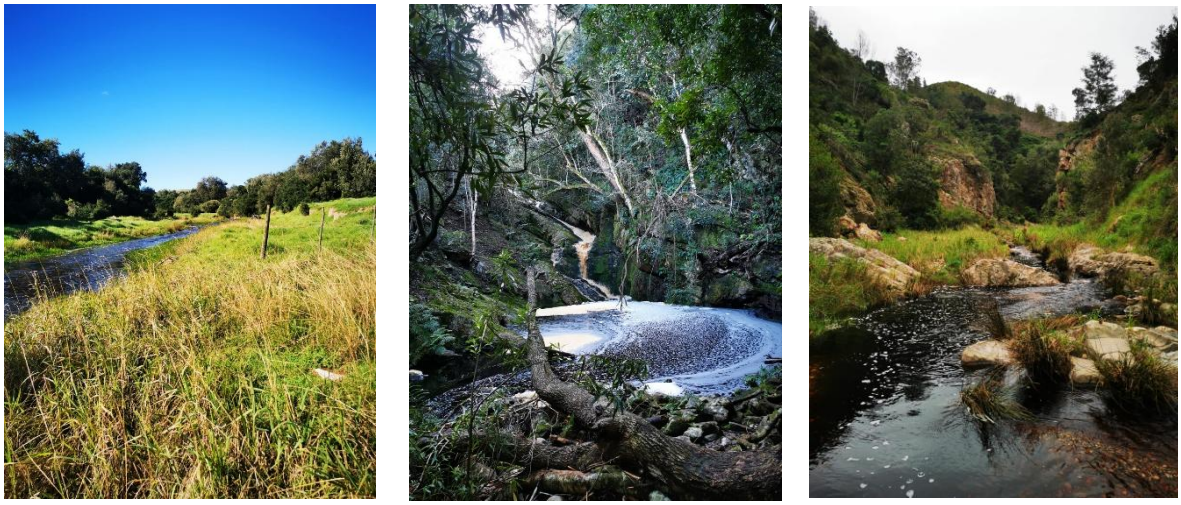

Figure 3.3 Photos showing the landscape of the Grootvaderbosch river study site

3.1.4 Ecological state of the rivers on the study areas (FBIS, 2023)

Information presented in Table 3.1 was extracted from FBIS (2023) and summarizes the ecological and administrative characteristics of the rivers at the study areas. It includes site codes, vegetation bioregions, ecoregions, water management areas, and climate zones. The table also presents the rivers' ecosystem threat status, present ecological state, and protection level, highlighting that both the Buffeljags and Grootvadersbosch Rivers are classified as critically endangered, largely modified, and either not protected or poorly protected. Additional environmental information such as rainfall intensity and mean annual precipitation zones is also provided. Although the rainfall intensity zone dataset classifies Buffeljags as higher (Zone 3) and Grootvadersbosch as lower (Zone 2), other climate accounts of the area indicate that Grootvadersbosch generally receives more rainfall to Buffeljags (Pool-Stanvliet et al., 2017; Scott-Shaw et al., 2017).

Table 3.1 Ecological state of the rivers of the study areas (FBIS, 2023)

	Site 1: Buffeljags River sites	**Site 2: Grootvadersbos River sites**
Site codes	**H7BUFF-LORIE and H7BUFF-FROGM**	**H7GROO-GLENE, H7GROO-USGVB, H7GROO-FOHUT**
SA Veg Bioregion	Alluvial Vegetation	Southern Fynbos Bioregion
SA Province and SADC boundaries	Western Cape	Western Cape
SA Ecoregion Level 1 **SA Ecoregion Level 2**	22 SOUTHERN COASTAL BELT 22.02	22 SOUTHERN COASTAL BELT 22.02
Strategic Water Source Areas – Surface	Langeberg	Langeberg
Freshwater Ecoregion of the World	Cape Fold	Cape Fold
River Ecosystem Threat Status	Critically endangered	Critically endangered
River Present Ecological State 2018	Largely Modified (D)	Largely Modified (D)
River Ecosystem Protection Level	Not protected	Poorly Protected
Water Management Area	Breede_Gouritz	Breede_Gouritz
Sub Water Management Area	Lower Breede	Lower Breede
Rainfall Intensity Zones	Zone 3 (high)	Zone 2 (medium)
Mean Annual Precipitation Zones (mm per year)	1000	800
Historical KG Climate Zones (1979 - 2008)	Temperate without dry season, hot summer	Arid steppe cold

3.2 Climate and weather

The Western Cape is situated at the south-westernmost point of South Africa (Stanvliet et al., 2017). The Indian Ocean lies to the south while the Atlantic Ocean lies to the west and south-west (Pool-Stanvliet et al., 2017). The Western Cape has a Mediterranean climate with cool, wet winters and hot, dry summers (Pool-Stanvliet et al., 2017). The winter rainfall is caused by 'mid latitude cyclones'/cold fronts coming from the South Atlantic on westerly winds and some orographic rain in the spring and autumn (Rebelo et al., 2006). In the summer there are dry South-east winds. Both study areas primarily experience winter rainfall but with some summer-winter transitional rain (Mucina et. al., 2006; Pool-Stanvliet et al., 2017).

Grootvadersbosch Nature Reserve, which is close to Grootvadersbosch farm, receives an average of around 900 mm to 1050 mm of rainfall a year (Jordaan et al., 2020). Rain is common throughout the year at this site, but spring months (October and November) normally have the highest rainfall (Jordaan, Gouws and Anderson, 2020). The long term Mean Annual Precipitation (MAP) for Buffeljags River is 636 mm (Scott-Shaw et al., 2017).

It should be considered that in the time of the study period, especially the winter, there was an above-average amount of winter rainfall and considerable flooding. At Buffeljags River, sections of the bank were washed away during the winter season which influenced camera trap placement in the summer season.

3.3 Vegetation and landscape

3.3.1 Vegetation

The study areas are a combination of natural vegetation and farmland (mainly pastures). The study areas occur in the Fynbos biome. On the mountain slopes there is primarily South Langeberg Sandstone Fynbos (South African National Biodiversity Institute, 2018) which is montane fynbos rich in *Protea* and *Erica* species (Mucina et al., 2006). In sheltered areas on both study areas, Southern Afrotemperate forest occur (McDonald, 1993; Mucina et al., 2006). In the Western Cape, Southern Afrotemperate forest is often confined to kloofs or fire safe areas given that it is wet enough (Mucina et al., 2006). It consists of tall forest, often with a relatively dense shrub layer (Mucina et al., 2006). Southern Afrotemperate forest

is often dominated by yellowwoods (*Podocarpus* spp.); black ironwood (*Olea capensis*) and stinkwood (Mucina et al., 2006; Cape Nature, 2023a).

The main vegetation surrounding the Buffeljags River region is Swellendam Silcrete Fynbos, which has a mixture of Fynbos and Renosterveld vegetation (Mucina et al., 2006; Rebelo et al., 2006; SANBI, 2018) and mainly consists of shrubland that is rich in Asteraceous plants, restios and grasses. In the area surrounding Grootvadersbosch River there is a mixture of Eastern Rûens Shale Renosterveld and Swellendam Silcrete Fynbos (SANBI, 2018). Eastern Rûens Shale Renosterveld is grassy shrubland dominated by Renosterbos (*Dicerothamnus rhinocerotis)* and also features *Aloe* species and other succulents; *Searsia* spp. trees, and various grass species (Mucina et al., 2006; Rebelo et al., 2006).

The riparian habitat falls within the Cape Lowland Alluvial Vegetation (SANBI, 2018), which is a classification of riparian vegetation (Mucina et al., 2006). Cape Lowland Alluvial Vegetation is a type of azonal vegetation. This is distinguished from zonal vegetation types, which is based on climate zones, while azonal vegetation mean that other habitat characteristics have an overriding effect on the vegetation such as a unique soil type or waterlogged soil (Mucina et al., 2006). Therefore, Cape Lowland Alluvial Vegetation is distinct from the surrounding Fynbos vegetation (Mucina et al., 2006). Species typically expected to be found in Cape Alluvial Lowland vegetation are trees like Cape willow (*Salix mucronata)* and willow karee (*Searsia angustifolia*), as well as keurboom (*Virgilla divarcata*) and Breede river yellowwood (*Podocarpus elongatus*). Shrubs like ricebush (*Cliffortia* species); *Senecio halimifolius* and honeybell bush (*Freylinia lanceolata*) are also common. Reed species include palmiet (*Prionium serratum*) and Cape rush (*Juncus capensis*), restios like *Calopsis paniculata* and sedges like *Cyperus thunbergii* and *Cyperus congestus)*. Grass species are also common, and typically include *Cynodon dactylon* and *Cynodon denudatus*, *Eragrostis sarmentosa*.

3.3.2 Landscape

The geology of the area consists mainly of sandstone, quartzite, and shale of the Cape Supergroup (Rebelo et al., 2006). Prominent features of the broader study area are the Cape fold mountains. The main mountain range in the study area, as previously mentioned, is the Langeberg mountains. The Langeberg mountains are primarily made up of Table Mountain Group (Cape Supergroup) sandstone and of pre-Cape Malmesbury Group sandstone and quartzites (McDonald, 1993; Tham et al., 2006) and is approximately 322 km in length from origin to estuary (Lamberth et. al, 2008). It consists of hard, nutrient-poor sandstone and quartzite stones (Rebelo et. al., 2006). The folds of the Cape Fold Mountains verge Northward and 'trends' from east to west north-south axis Cederberg, Groot Winterhoek, and Witzenberg mountain ranges) and east-west axis including the Langeberg, Riviersonderend, Outeniqua and Swartberg Mountain ranges (Pool-Stanvliet et al., 2017). The soil varies between fine sandy, silty or clayey soils mostly derived from Table Mountain sandstone or Cape granite (Mucina et al., 2006).

Buffeljags River sites consist of flat pastures on the south bank of the river, with a stretch on the Franken farm having steep banks. There is a shale bank on the south side of the river. The north bank of the river has steep inclines and hills as it is the foothills of the mountains. The sections next to the river is quite rocky with soils mainly consisting of alluvial sands (Scott-Shaw et al., 2017). The river is quite broad in most places and vary between fast flowing at Frog Mountain Getaway and slower flowing as it flows towards Buffeljags dam close to the Franken Farm.

The landscape next to Grootvadersbosch River is quite similar, however the river is not as wide nor deep as Buffeljags River, and there are more rocks and boulders in the riverbed. Upstream the river tends to have steep surrounding cliffs and more natural vegetation. The river broadens slightly as it moves downstream. There are also more pastures and disturbed areas downstream. Soil is also mostly sandy, with clay soil in some areas.

3.4 Data collection

Twenty camera traps were set up in each study area (Buffeljags River and Grootvadersbosch River, respectively). The detection/non-detection data collected by the camera traps were used to create detection histories for each species detected. This would allow the probability of site use of each species to be determined. Camera trap and survey level covariates were collected to explain any heterogeneity in probability of site use and detection probability. Camera trap level covariates collected were physicochemical water quality data, SASS5 survey data and vegetation cover. The physicochemical data and SASS5 survey data is related to water quality (Dickens and Graham, 2002). I collected these two covariates since it would hypothetically affect the probability of site use of African clawless otter and water mongoose by affecting their prey populations. It potentially affected other riparian mammal species more indirectly by being an indicator of habitat quality and riparian vegetation integrity (Dosskey et al., 2022; Bulto et al., 2026).

Vegetation cover such as canopy cover and ground vegetation cover affects riparian mammal probability of site use in that plants provide food for herbivores such as the Cape bushbuck (*Tragelaphus scriptus*) and shelter/cover for all species (Zungu et al., 2020). Ground vegetation cover and canopy cover also affect microclimates which affects insects and small mammals (Li et al., 2025), which are part of the prey base of some riparian mammals such as genets (Roberts et al., 2007) and Cape grey mongoose (Mbatyoti et al., 2010).

The physicochemical data, SASS5 data were collected by Grootvadersbosch conservancy at two or three different representative points along the river, while other habitat covariates such as vegetation cover and canopy cover were collected at each camera trap location using line transects and quadrats. Survey effort, the number of days that camera traps were active, was the survey covariate included in the occupancy modelling.

3.4.1 Methods to investigate water quality of site

3.4.1.1 Field data collection

In the present study, river condition and water quality was assessed using South African Scoring System 5 (SASS5) (Dickens and Graham, 2002) as well as physicochemical parameters. SASS data for this study was collected by Grootvadersbosch Conservancy in April and November 2023 at the study areas

where the camera traps were set up and uploaded on the Freshwater Biodiversity Information System (FBIS, 2023). This data was used with Grootvadersbosch Conservancy's permission (see Appendix E). The Freshwater Biodiversity Information System (FBIS) is an open-access platform developed by the Freshwater Research Centre. It assembles both historical and contemporary freshwater biodiversity data (including SASS data of rivers across South Africa) into a standardized format (Dallas and Shelton, 2024; Kajee et al., 2023). It is an important tool to help inform and support national conservation decisions (Kajee et al., 2023).

The SASS and physicochemical data were collected at two points at Buffeljags River and three different points at Grootvadersbosch River in 2023 and 2024 (see Figures 3.6 and 3.7 for location of SASS survey points).

Physico-chemical parameters (such as dissolved oxygen, conductivity, pH, and temperature) were measured in June 2024 at both rivers at the same survey points to verify Grootvadersbosch conservancy's data. Measurements included dissolved oxygen, pH, conductivity, salinity, and water temperature, recorded using an AZ86031 Water Quality Meter (AZ Instrument Corp).

3.4.2 Methods to assess vegetation cover of study site

3.4.2.1 Field data collection

Vegetation cover data

Eight different habitat covariates of the camera traps sites of each study area were incorporated into the occupancy model.

- Bare ground
- Debris (defined as any loose plant material such as twigs or leaves or reeds)
- Forbs
- Grass
- Trees
- Reeds
- Rocks
- Shrubs

The habitat covariates cover data was collected by using 1 m by 1 m quadrats placed at two metre intervals in a direct line from the water's edge (see Figure 3.4). Habitat covariates were surveyed for the approximately 15 m length from the camera to the river. Quadrats were placed on alternating sides of a measuring tape that was extended. This was repeated between five and seven times depending on how far the camera was placed from the riverside. A photo was taken of each quadrat and filed according to camera trap site. Afterwards a grid was superimposed on the photos. This allowed the estimation of the percentage cover.

The habitat covariate cover assessment was done after surveying was finished for both seasons. It was done once, so seasonal variation was not accounted for. During the surveys, I tried to keep the camera trap position the same for both seasons, but because of flooding and changes in river level this was not always possible. If the camera traps changed position, habitat covariate assessments were done for both locations.

The percentage cover for each habitat covariates at each camera trap site were estimated by using 1 m by 1 m grids superimposed on the photos in Microsoft Publisher. Canopy cover was estimated using Google maps and Google earth. The 10m scale on map was used to make a 10 m-by-10 m quadrat (see Figure 3.5). This was placed across the camera location with the GPS location of the camera trap being placed in the centre of the grid. The photos of Google Earth were dated 28 November 2022. Canopy cover (%) was then estimated using the satellite image.

The distribution and variability of habitat covariates in each area across all camera trap sites were visualised using boxplots in R (R Core Team, 2025), showing how values were distributed around the mean for each covariate type.

Figure 3.4 Example of quadrat used to assess vegetation cover

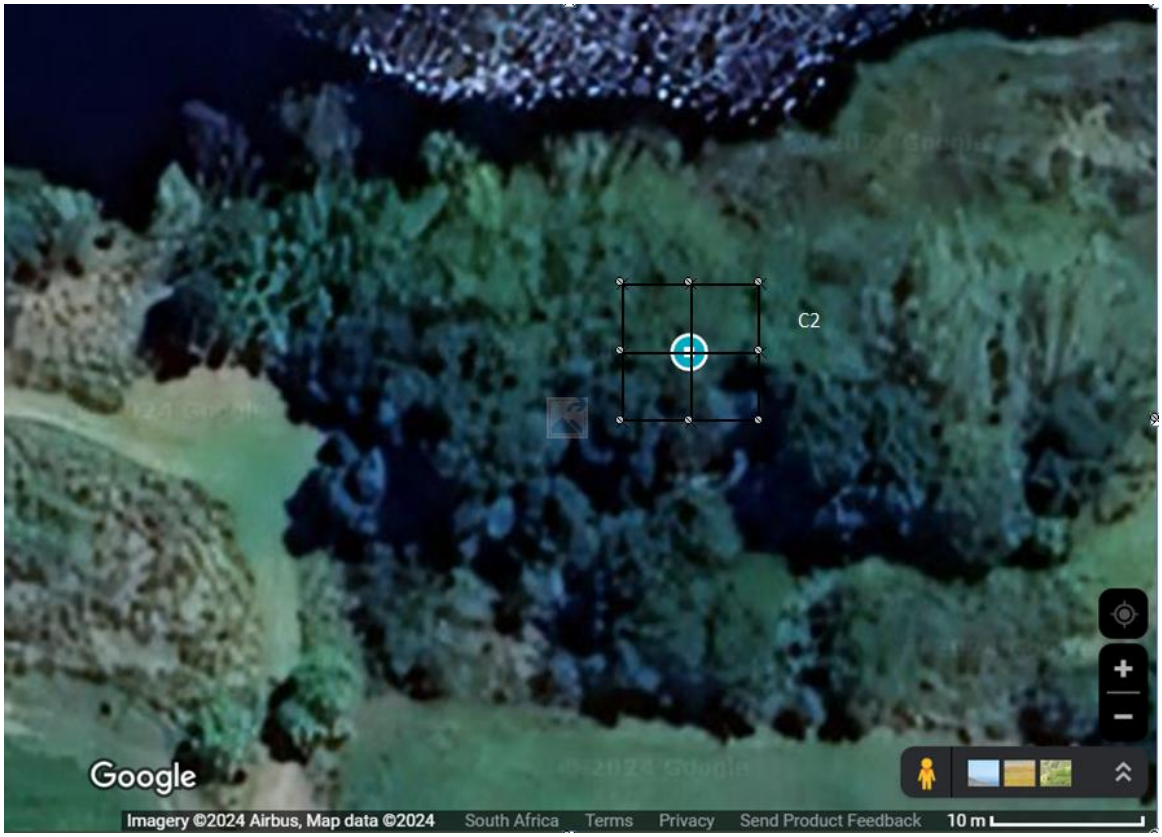

Figure 3.5 Example of quadrat used to calculate canopy cover at camera trap sites using Google Earth Pro. (Google LLC, 2024)

3.4.3 Methods to determine mammal site use

3.4.3.1 Field data collection

Occupancy models were run but due to the violation of the closure assumption of occupancy models and camera traps therefore not being independent (Rota, 2009) occupancy models were instead used to investigate probability of site use (DiRenzo et al., 2022). To gather detection/non-detection data for the probability of site use by riparian mammal species, camera traps (Primos Proof Cam 02, Primos Hunting, USA and Spypoint Force-Dark, SPYPOINT Inc., Montreal, Quebec, Canada) were placed at both study areas for three to four weeks. See Figures 3.6 and 3.7 for location of camera traps at each study site.

Figure 3.6 Map of Buffeljags River sites and SASS survey points. Base imagery from Google Earth Pro (Google LLC, 2023).

Figure 3.7 Map of Grootvadersboschriver sites and SASS survey points. Base imagery from Google Earth Pro (Google LLC, 2023).

Two surveys were conducted at each study site: firstly in late autumn/early to middle winter and secondly in early to late summer. The late autumn/early to middle winter survey is defined as winter, and the early to late summer surveys are defined as summer from here on forth. This allowed the comparison of site use and relative abundance of riparian mammals between two areas in the Breede river catchment, which are broadly similar in land use and water quality.

At each study area, 20 camera traps were placed along a 2 km stretch of river at approximately 200 m intervals. Most of the species has home ranges of more than two kilometres e.g. African clawless otters have minimum home ranges of between 14.3 km for females and 19.6 km for males in Tsitsikamma area (Arden-Clarke, 1986) and overall ranges of between 4.9 km and 54 km in the Western Cape according to Somers and Nel, (2004). Therefore, the close proximity of the camera traps to each other means that the closure assumption was violated and the occupancy estimations were most likely overestimated (Rota et al., 2009). Therefore, as mentioned earlier, all occupancy results were interpreted as probability of site use instead of occupancy (Grant, 2015; DiRenzo et al., 2022).

Camera traps were placed around 10 m to 15 m from the water. This rationale was based on the observation that African clawless otter, one of the focal species, rarely roam more than 15 m from shore (Larivière, 2001). Camera traps were placed approximately 50 cm above the ground based on guidelines that this is a good standard height to capture a variety of medium sized to larger mammals and minimize accidental captures because of plant movement (Tobler et al. 2008; Wearn and Glover-Kapfer, 2017; Bernard et al., 2023). It must be noted however that camera traps set up at a lower height, e.g. 30 cm, would potentially be more efficient at capturing the smaller mammals such as water mongoose and Cape genets and some species could be missed by setting the camera up at a height of 50 cm (Sereno-Cadierno et al., 2026). This is according to recommendations by Meek et al. (2012) that camera traps be set up at the shoulder height of the focal species. However, for multispecies studies, such as this, 50 cm is more appropriate to capture a variety of species (Sereno-Cadierno et al., 2026).

Passive infrared camera traps were used (Primos Proof Cam 02, Primos Hunting, USA and Spypoint Force-dark, SPYPOINT Inc., Montreal, Quebec, Canada). Each camera trap was set to take four photos when triggered with a ten second delay after being triggered. Care was taken to place camera traps in places partially hidden from view, if possible. Leaves and branches that could move and potentially lead to false triggers, was cleared away from the field of view of the camera trap. The location of each camera trap was recorded using a handheld GPS (Garmin etrex 30, Garmin Ltd., Olathe, USA). Each camera trap was already marked with a specific number (e.g. TM06). During the summer surveys, which took place after the winter surveys, it was attempted for camera traps to be placed at the same point as in winter. However, if this was not possible because of difficulty in access or habitat changes because of flooding, then camera traps were placed at a point as close as possible to the previous point.

The following information was noted during the setup of the camera traps:

- Study area
- GPS point
- Specific camera trap site name (e.g. C1) and camera trap number (e.g. TM35)
- The SD card number

- Date
- Distance from the water
- Distance in between camera traps

3.4.3.2 Processing the camera trap images

Camera traps were removed after approximately three weeks at the specific site. The following steps were taken to organize and process the photos.

- Images were transferred from SD card to computer and hard drive and sorted according to site, season and camera trap number.
- Images containing any mammal species were identified and classified according to species. Smithers' Mammals of Southern Africa: A Field Guide (Apps and Smithers, 2012) and the descriptions and photos on the IUCN Red List (IUCN, 2025) were used to help identify the species.

3.4.3.3 Creating detection histories and site and observation covariate tables

The camera trap data was used to set up a detection history for each species identified on the camera trap. To ensure that detections were independent, consecutive photos of the same species within a 24-hour period were treated as a single detection event (Goldstein et al., 2024). This approach avoids inflating detection probabilities by counting repeated captures of the same individual or group within a short timeframe (Rovero and Marshall, 2009; O'Brien and Kinnaird, 2011). Although this reduced the total number of detections, it ensured temporal independence between detection occasions, which is an important assumption of occupancy modelling (MacKenzie et al., 2002). So, if the species was detected on that day, then a one was assigned. If there were no photographs taken, then it was seen as the species being not detected for that occurrence and a zero is assigned (MacKenzie et al., 2002; Bailey and Adams, 2005).

Detection histories were organised as camera trap site by day matrices for each species. Both sites and seasons were collected in one table. Winter data covered days 1–30, and summer data, days 31–60. While camera trap numbers 1 to 20 referred to Buffeljags River camera traps and 21 to 40 referred to Grootvadersbosch River camera traps.

Camera trap-level covariates (see section 3.4.1 and 3.4.2) were compiled into a camera-trap site by covariate matrix, with each row representing a camera site and each column representing a habitat or environmental variable. Site level covariates are used to explain heterogeneity in occupancy probability (Bailey and Adams, 2005). All continuous covariates were z-standardised (mean = 0, SD = 1) in R (R Core Team, 2025).

Effort, defined as the number of days that cameras were active, was included as the only detection/survey covariate. Effort is a survey covariate, meaning it was primarily incorporated to explain differences in detection probability (Bailey and Adams, 2005). A value of one was assigned when a camera trap was active on a given day, and zero when it was inactive, allowing detection probability to be modelled as a function of camera activity.

3.4.3.4 Calculating relative abundance indices

The Relative Abundance Index (RAI) provides a standardized measure of riparian mammal activity per 100 camera trap days across sites and seasons. The RAI was computed by calculating the number of independent camera trap photos divided by the total amount of camera days and multiplying it with 100. Calculations were performed in Microsoft Excel.

Formula: 100 x (number of independent pictures / total monitoring days)

3.4.3.5 Use of artificial intelligence (AI)

ChatGPT was used as a reference to help with the coding in R for the occupancy modelling and plotting of data. It was also used to help format the R scripts and check for syntax errors. Please note however that the words in this thesis are my own.

3.4.3.6 Species richness species accumulation curves

Temporal and spatial accumulation curves were created from the detection histories for all eleven species using the vegan package (Oksanen et al., 2024) in R (R Core Team, 2025). Species accumulation curves give an indication of species richness and show how thorough sampling effort was in terms of the number of sampling days and camera traps (Ugland et al., 2003; Deng et al., 2015). Temporal

accumulation curves show how species increase as survey days increase, at which rate new species are added over thirty days. Spatial accumulation curves show how species richness increase as more camera traps are added.

The temporal species accumulation curves were created using the data in the form of matrices of species (columns) by days (rows), where each entry indicated whether a species was detected on a given day. Curves were created using the specaccum() function with the collector method. This method keeps the chronological order of detections so that each step represents the first appearance of a new species. The spatial accumulation curves are smoothed curves that are created by randomly assigning the order of camera trap added with 100 permutations. The 95% confidence intervals (CI) were represented by shaded areas.

3.4.3.7 Fitting of occupancy models

The accuracy of the occupancy results is sensitive to low detections (Steenweg et al., 2019), therefore low detections can lead to poor model convergence (Welsh et al., 2013). Overall detection probabilities above 0.2 is necessary to avoid poor convergence and to allow accurate model estimation (Guillera-Arroita et al., 2010; Kéry and Royle, 2015; Steenweg et al., 2019). After testing the detection histories of all species in Presence version 2.15.18 (Hines et al., 2006), it was clear that species with four or less detections had very high standard errors, and model estimates could not be interpreted accurately. This aligns with recommendations from species distribution modelling, which suggest reliable inference typically requires between five and 200 occurrences, depending on data complexity (Santini et al., 2021; Erickson and Smith, 2023). Although these studies refer to species distribution models, the same logic applies to occupancy models: very sparse detection data can lead to unstable parameter estimates (Welsh et al., 2013).

Multispecies and single-species occupancy models were fitted using the spOccupancy package (version 0.8.0) in R (Doser et al., 2022; R Core Team, 2025). spOccupancy was used because Bayesian inference is better at handling rare species and low detections, even in single-species occupancy models (Pacifici et al., 2015). However, multispecies models handle rare species better than single-

species models, and therefore this was the main model that was fitted for data analysis.

There were a few test runs done with occupancy models with the unmarked package (version 1.5.0) (Fiske and Chandler, 2011) and spOccupancy (Doser et al., 2022) in R (R Core Team, 2025). As mentioned before, the overall low detections meant occupancy models failed to converge even with multispecies modelling. Even after excluding species with fewer detections than five (Welsh et al., 2013), models still did not converge. Therefore, there was a limit to how many covariates could be included (Guillera-Arroita et al., 2010). The following site covariates were collected: ground vegetation cover (the percentage of ground covered by plants); canopy cover (how much of ground is covered by tree canopy from satellite image); SASS, ASPT, number of taxa; conductivity; dissolved oxygen; and pH (see sections 3.4.2 and 3.4.3). Correlations among site-level covariates were assessed per season before modelling to avoid multicollinearity (see Table A1 and A2). This was done in R (R Core Team, 2025). Pearson's correlation coefficients were calculated, and where pairs of covariates were highly correlated ($r > 0.7$), only one was retained for modelling to ensure model stability and interpretability i.e. SASS and ASPT had a correlation coefficient of -0.01. Site-level physiochemical covariates used in the occupancy models were measured directly during the water quality surveys (Table 4.3). Additional water quality data were obtained from the Freshwater Biodiversity Information System (FBIS, 2023) and the Grootvadersbosch Conservancy. Where possible, units and measurement protocols were standardized, as some FBIS variables were recorded using inconsistent units across sites. The following covariates were chosen:

'Area', 'Dissolved Oxygen', 'Canopy Cover', 'Ground Vegetation Cover', 'SASS', and 'Conductivity'.

Detection and observation covariate matrices were cleaned to remove inconsistent or missing values. NA patterns were synchronized across species to ensure that each site by day combination matched between the detection histories and observation covariates. If, for example, day 1 of camera trap 15 for one species accidentally had a NA while the other species had a one or a zero, because of an error in transferring data to R, this could lead to incorrect inferences. This did not change any of the data but aligned any potentially missing values.

Models were run using Markov Chain Monte Carlo (MCMC) sampling, and posterior distributions were obtained for all parameters. The multispecies models were run using msPGOcc. Two multispecies occupancy models were fitted per season (winter and summer) using spOccupancy with a Polya-Gamma latent variable formulation. MCMC sampling was run for 3,000 iterations per chain, with a burn-in of 500 iterations, thinning rate of five, and two chains, resulting in 1,000 posterior samples per parameter. Parallel computation with four threads was used. This produced posterior means and credible intervals for each species. Posterior means and 95% credible intervals (CrI) were used to summarise parameter estimates, with the 95% CrI showing the range within which the true parameter value lies with 95% probability, given the data and model. The species-level and community-level estimates of occupancy (ψ) and detection probability (p) were derived from these posterior samples. The console results for both multispecies models are available in Appendix D.

Convergence of the multispecies occupancy models was assessed using the potential scale reduction factor (Rhat) and effective sample size (ESS). Most parameters had Rhat values close to one and sufficiently high effective sample sizes, which showed the models converged relatively well. A few species-specific parameters had higher Rhat and lower effective sample sizes, which can be expected in multispecies models with sparse data for certain species. All Rhat and effective sample size values can be found in the console results (see Appendices D1 and D2).

Single-species occupancy models were also run with spOccupancy for the seven species using the same six covariates. These models were used to confirm that multispecies results were robust: similar mean occupancy and detection probability estimates, along with overlapping credible intervals, showed that the multispecies models were reliable (see Table A6). Single-species models were run using the PGOcc function (Doser et al., 2022). Model assessment in spOccupancy is possible using either ppcOcc (posterior predictive check using Bayesian p-values) or waicOcc (Watanabe Applicable Information Criterion) (Doser et al., 2022). WAIC was calculated to compare relative fit, but the primary inference was based on the multispecies models (Doser et al., 2022; Watanabe, 2010). Single-species models were used primarily to check consistency with multispecies estimates. Cape

bushbuck and Chacma baboon were selected for comparison because they had the highest detection rates (see Table 4.5).

3.4.3.8 Extraction of mean species-level occupancy and detection summaries

Species-level mean occupancy (ψ) and detection (p) probabilities were calculated for each riparian mammal species from both models. Posterior samples were extracted from the fitted models, and occupancy and detection parameters were converted from the logit scale to the probability scale using the logistic transformation. For each species, the average posterior means and 95% credible intervals were computed. This allows the overall mean occupancy and detection probability to be seen. These estimates provide an indication of general species presence..

3.4.3.9 Extraction of community level results

Posterior samples of community-level occupancy (ψ) and detection probability (p) parameters were extracted from each season's multispecies occupancy models. For each community-level parameter, the mean and 95% credible interval were calculated on the logit scale (see Table A7) and then transformed to the probability scale using the logistic function. Occupancy parameters included the intercept and effects of site-level covariates, while detection parameters included the intercept and the effect of survey effort. These summaries provided overall community-level patterns in occupancy and detectability, rather than species-specific responses.

3.4.3.10 Extraction of species level covariate effects

Species-specific posterior estimates of occupancy (ψ) covariates were extracted from the winter and summer multispecies occupancy models. For each species–covariate combination, regression coefficients were transformed from the logit to the probability scale, and the mean probability and 95% credible intervals were calculated. These tables (see Tables A8 and A9) were used to visualize species-level responses to environmental covariates. The logit covariate effects per seasons (see Figures 4.7 and 4.8) were displayed in results and the probability scale covariate effects were displayed in Table A8 and A9. Logit-scale coefficients indicate the direction and relative magnitude of species–covariate relationships

(Figure 4.8), while back-transformed probabilities provided an interpretable estimate of expected occupancy for each species and season (Table A8 and A9). This allows the specific effect that each covariate has on individual species to be seen. The covariate-specific effects highlight which environmental variables are most strongly associated with changes in occupancy probability across sites in contrast to average species occupancy probabilities which highlight general presence of each species for each season.

Area (Buffeljags River versus Grootvadersbosch River) was included as a categorical site covariate rather than running separate models per site. This allowed area-specific differences to be estimated directly within a single modelling framework while maintaining a consistent baseline for detection probability and other covariates (Doser, 2023). Buffeljags River was treated as the reference category, which means that positive coefficients for 'Area Grootvadersbosch' indicate higher occupancy at Grootvadersbosch River relative to Buffeljags River, and negative coefficients indicate the opposite.

3.4.3.11 Visualisation of species-specific covariate effects on occupancy and detection probability

The effects of environmental covariates on species occupancy were generated from the fitted multispecies models in R (R Core Team, 2025) using the packages spOccupancy (Doser et al., 2022), ggplot2 (Wickham, 2016), dplyr (Wickham et al., 2026) and patchwork (Pedersen, 2024). The covariates were standardised using the mean and standard deviation of all winter survey sites prior to prediction, consistent with the scaling applied during model fitting. Predictions were generated on the logit scale using posterior samples of species-specific regression coefficients obtained via Markov chain Monte Carlo (MCMC) sampling, including intercept, covariate effects, and area effects where applicable, and were then back-transformed to the probability scale using the inverse logit function. Mean predicted occupancy and 95% credible intervals were calculated across posterior samples for each covariate value. The predictions were plotted separately for each study area and season (winter and summer). The predicted occupancy values across the covariate gradients were plotted as species-specific occupancy response curves to visualise how occupancy probability changed along each environmental gradient.

To visualise the relationship between detection probability and sampling effort, posterior predictions were generated separately for winter and summer using the final fitted multispecies models in R (R Core Team, 2025). These predictions were obtained by varying the effort covariate across its observed range (0–1) while holding all other detection covariates at their mean values and excluding random effects to obtain population-level responses. Predicted detection probabilities were calculated from posterior samples and summarised as posterior means. Detection–effort response curves were plotted on the same standardised effort scale (0–1) for both seasons to allow direct comparison between seasons and species. All visualisations were produced using the ggplot2 package (Wickham, 2016)

3.4.3.12 Plotting the occupancy results

Species-level covariates and effort logit effects were visualized as dot plots (see Figures 4.7 and 4.8), while the probability covariate effects were not plotted but are displayed in table (see Tables A8 and A9) (R Core Team, 2025). Area differences were plotted as bar graphs with error bars in R (R Core Team, 2025). The bar graphs displaying the RAI values were created in Excel.

3.4.3.13 Calculating area level occupancy and detection probabilities

The mean probabilities of occupancy (ψ) and detection probability (p) were calculated for each species within each study area (Buffeljags RiverBuffeljags River and Grootvadersbosch River) for both seasons. This allowed species-level summaries of occupancy and detectability for both study areas. This is not to be confused with the overall mean occupancy and detection per species (see section 3.4.3.7).

Area level occupancy and detection probabilities were calculated in the following way: Posterior samples of site-level occupancy (psi.samples) were extracted from the models. For each species, occupancy draws across sites were averaged within each area (Buffeljags RiverBuffeljags River and Grootvadersbosch River). The mean occupancy per area was calculated as the posterior mean of these site averages, and 95% credible intervals were computed from the 2.5th and 97.5th percentiles of the posterior draws. The detection probabilities were computed in a

similar way by converting posterior samples on the logit scale (alpha.samples) to probabilities using the logistic function, then averaging across draws. In R, this was implemented by extracting the relevant MCMC samples for each species and area, computing the *rowMeans* across sites, and summarising with *mean()* and *quantile()* functions. This approach incorporates both the estimated effect of area and variation across other site-level covariates. Results are available in Table A10 and A11.

3.4.3.14 Interpreting probability of site use and detection probability results

Area was incorporated as a site covariate in the occupancy models. In all models, the ‘Area (Grootvadersbosch)’ covariate represents the difference in probability of site use relative to the Buffeljags River sites. At logit scale positive coefficients for 'Area Grootvadersbosch’, indicate higher probability of site use at Grootvadersbosch River relative to Buffeljags River, while negative coefficients indicate the opposite. On the probability scale, higher values indicate greater probability of site use at Grootvadersbosch River, while lower values indicate higher probability of site use at Buffeljags River. The intercept represents the baseline probability of site use on the logit scale for the reference category (Buffeljags River) when all continuous covariates are held at their mean values (if standardised.. Although the outputs of the occupancy model results were in logit scale it was back transformed into probability scale to ease interpretation. The sign of the logit coefficients was used to see the direction of the relationship between covariate and mean estimates. As stated in section 3.4.3.1, because of camera traps being too close to each other and therefore not independent, all occupancy results were interpreted as probability of site use instead of occupancy (DiRenzo et al., 2022).

CHAPTER 4: RESULTS

4.1 Water quality results

4.1.1 SASS5 results

SASS data is reported in Table 4.2 for 2022-2024 to give an indication of variation around the 10 months (April 2023–February 2024) that the surveys took place in. The codes (e.g. H7BUFF-FROGM) for the sites are the survey points where the Grootvadersbosch Conservancy team did the SASS surveys, and it is named as such on the FBIS database (FBIS, 2023). SASS surveys were done only once in 2022 to 2024 at Buffeljags River while Grootvadersbosch RiverGrootvadersbosch River had SASS surveys done each year between 2022 and 2024 for that period.

As mentioned previously, sites with higher SASS or ASPT scores indicated higher water quality (Dickens and Graham, 2002). As seen in Table 4.1, these scores are used to assign a code (A to F) to each survey point which reflects the ecological condition of the river (as done by Dallas in her assignment of ecological bands) (Dallas, 2007; Dallas and Day, 2007).

Table 4.1. Ecological categories for interpreting SASS data (based on Dallas, 2007)

Ecological category	Ecological category name	Description
A	Natural	Natural without modification
B	Good	Mostly natural with some modification
C	Fair	Moderate modification
D	Poor	Largely modified
E	Seriously modified	Seriously modified
F	Critically modified	Critically/extremely modified

Grootvadersbosch River sites had higher ASPT and SASS scores than Buffeljags River sites overall, indicating slightly better water quality in general (see Table 4.2). However, the water quality of Grootvadersbosch River was also more variable between points than the water quality of the Buffeljags River sites. The survey points in the middle of the Grootvadersbosch study area, H7GROO_USGVB, and the downstream site, H7GROO-GLENE were designated

a code C and D, respectively, in early 2023 (before surveys started) and April 2024 (after surveys were finished) respectively. All three Grootvadersbosch sites were classified as A (pristine) in October-December 2023. At Buffeljags River there was lower water quality upstream at SASS point (H7BUFF-LORI) based on the collected SASS data (see Table 4.2). The upstream SASS point (H7BUFF-LORI) had an ecological category of B compared with the downstream SASS point (H7BUFF-FROGM) which had an assigned ecological category of A.

Table 4.2. SASS 5 Surveys done by Grootvadersbosch Conservancy

SASS site	Date of latest SASS	SASS5 /SASS 4 Score	Number of Taxa	ASPT	Code (SASS SCORES, No of taxa and ASPT)	GPS
Site 1: Buffeljags River Sites						
H7BUFF-LORI	2023/04/13	111	18	6.17	B	-34.000, 20.606
H7BUFF-FROGM	2022/07/06	92	14	6.57	A	-34.001, 20.581
Site 2: Grootvadersbosch River Sites						
H7GROO-FOHUT	2023/04/09	149	18	8.28	A	-33.997, 20.799
	2023/12/04	136	16	8.50	A	
	2024/04/02	149	20	7.45	B	
H7GROO_USGVB	2023/04/13	136	19	7.16	C	-34.004, 20.795
	2023/10/25	138	16	8.63	A	
	2024/04/13	139	20	6.95	C	
H7GROO-GLENE	2023/04/04	109	18	6.06	D	-34.008, 20.777
	2023/10/26	90	10	9.00	A	
	2024/04/30	113	17	6.65	D	

4.1.2 Physicochemical data

Physicochemical data was only collected at one of the survey points at Buffeljags River (H7BUFF-LORI) (see Table 4.3). At this survey point, the summer months had higher conductivity indicating higher dissolved solids which is possibly because of generally lower rainfall in summer, as well as higher evaporation. Water temperature increased from autumn to summer as expected. Dissolved oxygen increased from 37% to 60%. pH decreased showing more acidic conditions in winter compared with autumn. River discharge, defined as the volume of water flowing through a river (European Space Agency, n.d), increased from autumn to winter.

Conductivity showed the same pattern for Grootvadersbosch survey sites, again showing that there are generally higher dissolved solutes in water in the summer. The water of Grootvadersbosch was generally much more acidic than Buffeljags River varying between a pH of 4 and 6. The H7GROO-FOHUT site had the lowest pH of all Grootvadersbosch sites, possibly because it is in a section of Afrotemperate forest. Rivers and streams in Afrotemperate forests are often acidic because of dissolved organic acids leaching from the leaf litter and the peat soils (Day et al., 1998).

Water temperature was higher in summer as can be expected because of seasonal changes. There was a bigger difference between water temperature in autumn and summer in Grootvadersbosch River, compared to Buffeljags River.

River discharge was not measured for every single SASS survey point at Grootvadersbosch River. The data that is available for the one SASS point at Grootvadersbosch River (H7GROO-GLEN), is much lower (20.68 m^3/s) than discharge at Buffeljags River at both SASS survey points (82.86 m^3/s and 368 m^3/s at H7BUFF-LORIE and 41.8 m^3/s at H7BUFF-FROGM).

Physicochemical data was measured in March 2023 after camera trap surveys were completed to confirm the Grootvadersbosch conservancy data. Overall, Grootvadersbosch River sites appear to have higher water quality, based on higher SASS scores and ASPT values, higher dissolved oxygen and lower conductivity and total dissolved solids.

Table 4.3 Physochemical data surveyed at SASS5 points during and after survey period

Survey site code	Date	Conductivity (uS/cm)	Dissolved Oxygen (% unless indicated as mg/L)	pH	Water temperature (°C)	Discharge(m3/s)	Total Dissolved Solids
Site 1: Buffeljags River Sites							
Data collected by Grootvadersbosch Conservancy							
H7BUFF-FROGM	2022/07/06	N/A	N/A	N/A	N/A	41.8	N/A
	2023/11/01	N/A	N/A	N/A	N/A	N/A	N/A
H7BUFF-LORIE	2023/04/13	158	37	7.11	15.7	82.86	N/A
	2023/11/01	277	60.1	6.54	19.6	368.44	N/A
Data collected by myself							
H7BUFF-FROGM	2024/06/04	199.9	71.7	6.40	14.2	N/A	N/A
Second attempt	2024/06/12	478	78.6	6.51	15.1	N/A	238
H7BUFF-LORIE	2024/06/04	237	59.0	6.08	13.8	N/A	119
Site 2: Grootvadersbosch River Sites							
Data collected by Grootvadersbosch Conservancy							
H7GROO-FOHUT	2023/04/09	46	6.3 mg/L	4.6	14.9	N/A	N/A
	2023/12/04	144	-	4.46	20.36	N/A	N/A
	2024/04/02	85	6.04 mg/L	4.26	17.57	N/A	N/A
H7GROO_USGVB	2023/04/13	98	4.05 mg/L	5.13	16.7	N/A	N/A
	2023/10/25	200	5.62 mg/L	6.06	20.91	N/A	N/A
	2024/04/13	151	3.75 mg/L	4.04	15.45	N/A	N/A
H7GROO-GLENE	2023/04/04	97	59.9 mg/L	5.72	16.2	20.68	N/A
	2023/10/26	N/A	-	-	-	N/A	N/A
	2024/04/30	171	7.7 mg/L	5.7	17.74	N/A	N/A
H7GROO-FOHUT	2024/06/06	could not reach the point because of high water levels					
Second attempt	2024/06/12	77.1	87.8	3.78	13.4	N/A	37.8
H7GROO_USGVB	2024/06/06	159.5	96.3	5.19	13.3	N/A	79.7
Second attempt	2024/06/12	184.5	81.3	5.32	14.8	N/A	92.0
H7GROO-GLENE	2024/06/06	132	95.5	6.4	13.9	N/A	86.2
Second attempt	2024/06/12	67.9	75.7	6.33	14.7	N/A	33.9

4.2 Habitat covariates results

Habitat covariates were surveyed for the approximately 15m length from the camera to the river and canopy cover was 10m² surrounding the camera trap. It was done once, so seasonal variation was not accounted for. These differences in habitat structure provide important context for interpreting species probability of site use and relative abundance patterns, as species often respond to both vegetation density and canopy cover when selecting sites.

Buffeljags River

The vegetation survey for Buffeljags River (see Figure 4.1 and Table A4) showed that the environment where the camera traps were placed varied from bare and rocky to quite grassy. Open areas dominated but there were areas that were quite shady and had high canopy cover. Reeds and debris were patchy, showing diversity in microhabitat along the river.

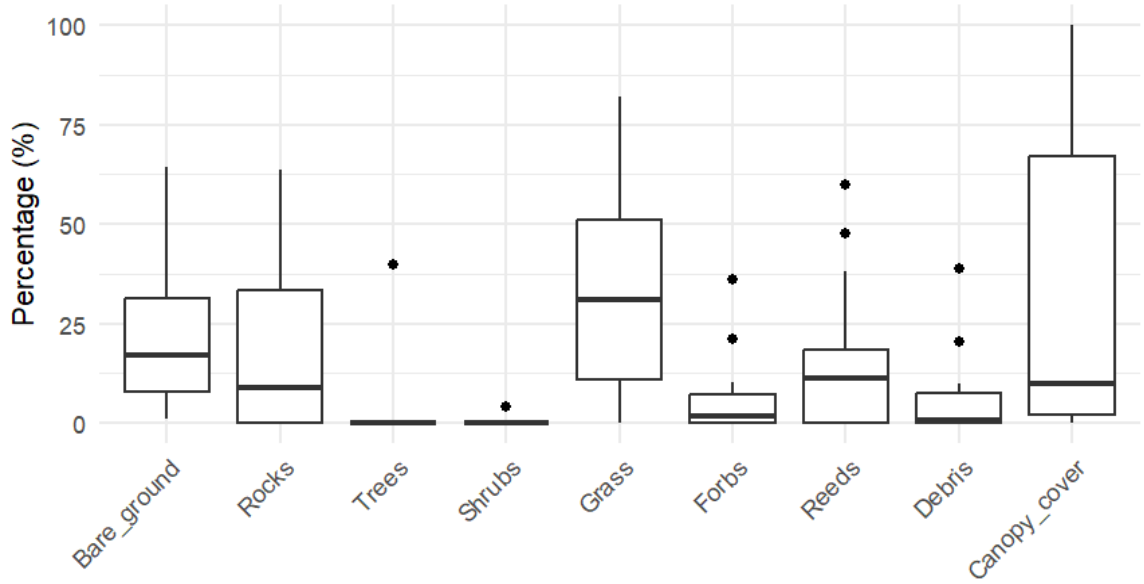

Figure 4.1 Habitat covariates at Buffeljags River. Boxplots show the median, interquartile range, and outliers for each covariate measured within ~15 m of the camera to the river.

Grootvadersbosch River

Grootvadersbosch River had many camera trap sites with dense grass cover with few shrubs or forbs (see Figure 4.2 and Table A5). There was similar rock cover as at Buffeljags River. As with Buffeljags River, canopy cover was somewhat variable

with a lot of camera trap sites being quite open, but canopy cover was generally higher and more continuous at stretches there. There was not a lot of shrubs or smaller plants. Reeds and debris were not evenly spread, only occurring at some camera trap sites, according to survey notes, often at the wetter areas as can be expected.

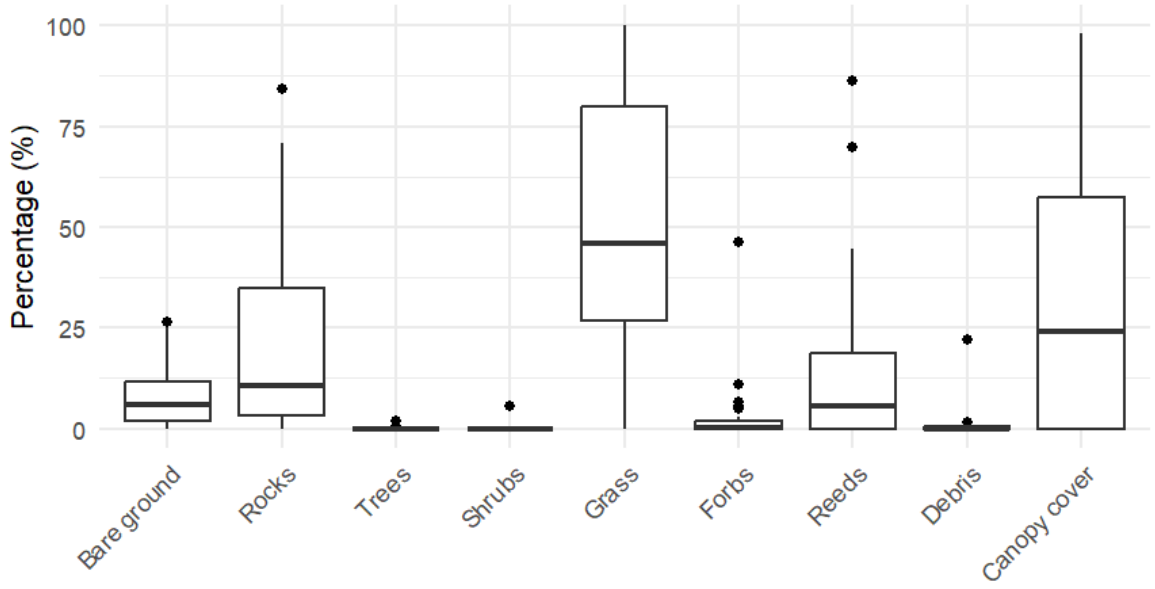

Figure 4.2 Habitat covariates at Grootvadersbosch River. Boxplots show the median, interquartile range, and outliers for each covariate measured within ~15 m of the camera to the river.

4.3 Camera trap surveys

During the winter survey at Grootvadersbosch River, four cameras were lost to flooding, and one camera was stolen at Buffeljags River during the summer survey (see Table 4.4). Therefore, fewer cameras were available for the summer survey at Grootvadersbosch River. This resulted in some differences in survey effort between sites and seasons. A total of 1,807 camera trap days across both study areas yielded 330 detections of eleven mammal species (see Table 4.4. and Table 4.5).

Table 4.4 Summary of camera trap surveys

Survey area and season	Buffeljags River Winter survey	Buffeljags River Summer Survey	Grootvadersbosch River Winter Survey	Grootvadersbosch River Summer Survey
Duration	12 April to 9 May 2023	11 November to 7 December 2023	23 May 2023 to 18 June 2023	19 February 2024 to 4 March 2024
Camera trap days (per camera)	28	27	27	18
Number of cameras set up	20	20	20	13
Camera trap days (total days per study area per season)	516	543	533	215

A summary of independent detections per species and season is provided in Table 4.5. The Chacma baboon (*Papio ursinus*) had the most independent photographs followed closely by Cape bushbuck (*Tragelaphus sylvaticus*) and Cape genet (*Genetta tigrina*). Caracal (*Caracal caracal*), vervet monkey (*Chlorocebus pygerythrus*) and honey badger (*Mellivora capensis)* were each only detected on one occasion. Seven of the eleven species met the criterion of having five or more detections to be suitable for occupancy modelling (see section 3.4.1.): African clawless otter (*Aonyx capensis*), bushpig (*Potamochoerus larvatus*), Cape bushbuck (*Tragelaphus sylvaticus)*, Cape porcupine (*Hystrix africaeaustralis*), Cape genet (*Genetta tigrina*), Chacma baboon (*Papio ursinus*) and water mongoose *(Atilax paludinosis*).

Table 4.5 Number of detections per species per survey

	Buffeljags Winter	Grootvaders-bosch Winter	Buffeljags Summer	Grootvaders-bosch Summer	Total
Bushpig (*Potamochoerus larvatus*)	2	8	1	0	11
Cape bushbuck (*Tragelaphus sylvaticus*)	26	12	36	17	91
African clawless otter (*Aonyx capensis*)	5	4	2	0	11
Cape genet (*Genetta tigrina*)	21	2	8	0	31
Cape gray mongoose (*Galerella pulverulenta*)	2	0	2	0	4
Cape porcupine (*Hystrix africaeaustralis*)	1	7	0	0	8
Caracal (*Caracal caracal*)	1	0	0	0	1
Chacma baboon (*Papio ursinus*)	86	17	35	9	147
Honey badger (*Mellivora capensis*)	0	1	0	0	1
Vervet monkey (*Chlorocebus pygerythrus*)	0	0	1	0	1
Water mongoose (*Atilax paludinosus*)	5	6	1	12	24
Total	149	57	86	38	330

4.4 Species richness

In addition to the eleven mammal species identified on the photos, several bird species, such as Brown-hooded kingfisher (*Halcyon albiventris*), and Black-headed heron *(Ardea melanocephala*), were identified. However, birds are outside the scope of the study. Domesticated animals such as cattle, horses, cats and dogs were often recorded but were not considered in the count. The mammal species that were recorded can be seen in Figure 4.3. Seven of the species were recorded at both Buffeljags River and Grootvadersbosch River, while caracal *(Caracal caracal),* Cape grey mongoose *(Galerella pulverulenta),* and vervet monkey *(Chlorocebus pygerythrus),* were only recorded at Buffeljags River, and honey badger (*Mellivora capensis*), was only recorded at Grootvadersbosch River.

Overall Buffeljags River had higher species richness than Grootvadersbosch River. At Buffeljags River, nine species were detected in the winter, while there were eight species detected during the summer.

*Figure 4.3 Riparian mammals detected during survey period. **A** Bushpig (Potamochoerus larvatus), **B** Cape bushbuck (Tragelaphus sylvaticus), **C** African clawless otter (Aonyx capensis), **D** Cape genet (Genetta tigrina), **E** Caracal (Caracal caracal), **F** Cape grey mongoose (Galerella pulverulenta), **G** Cape porcupine (Hystrix africaeaustralis), **H** Chacma baboon (Papio ursinus), **I** Honey badger (Mellivora capensis), **J** Vervet monkey (Chlorocebus pygerythrus), **K** Water mongoose (Atilax paludinosis)*

4.4.1 Species accumulation curves

Both species accumulation and temporal accumulation curves show the differences in species accumulation between seasons and sites. Additionally, both accumulation curves showed that overall, more species were detected in the winter season compared with the summer season at both study areas.

At Grootvadersbosch River eight species were detected in the winter, while only three species were detected during the summer. With the temporal accumulation curves, there were no new species after ten days-showing that the sampling period was likely enough to detect most species. The spatial accumulation curves did not level off by twenty camera traps, showing that possibly more camera trap effort was needed to detect all species present in the two study areas.

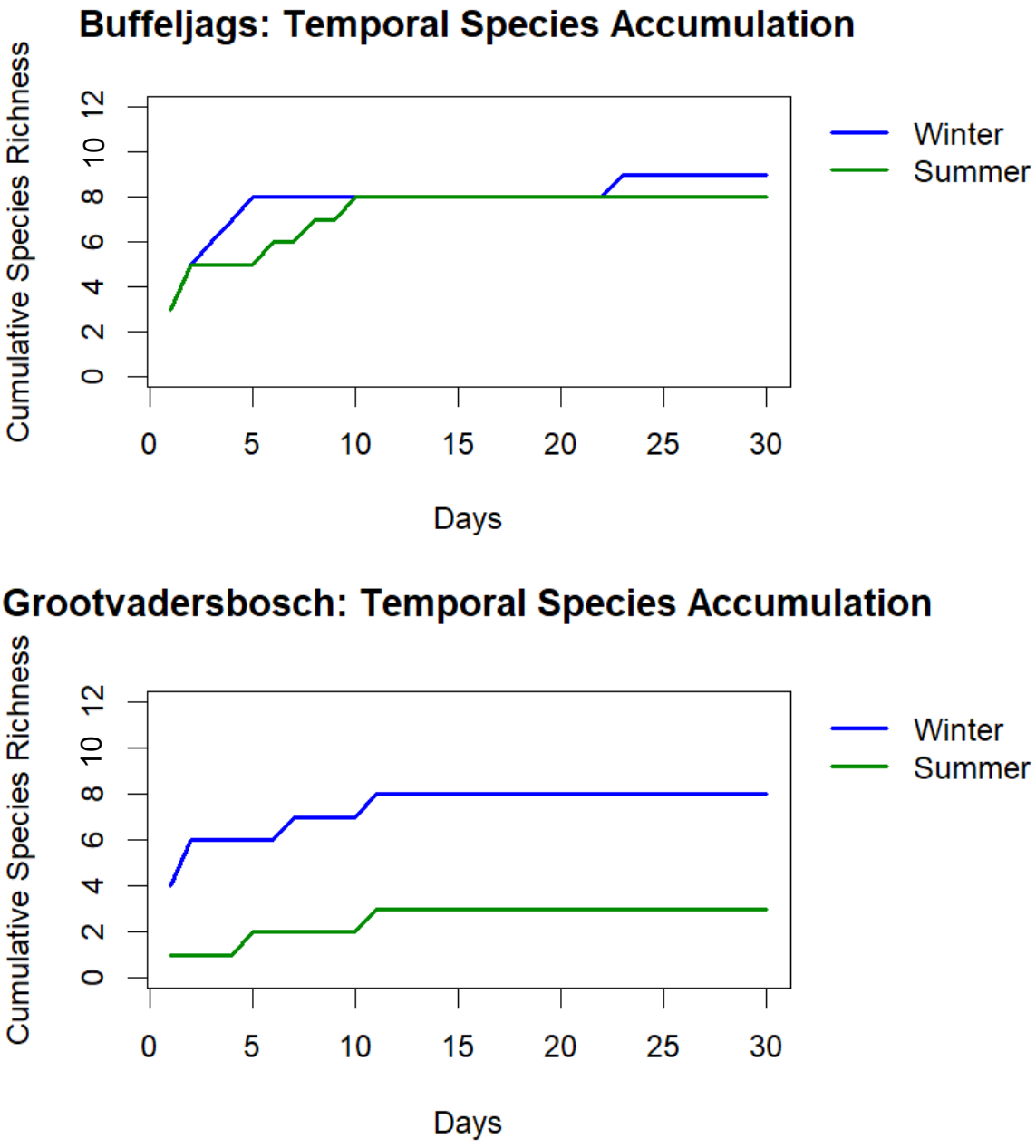

Figure 4.4: Temporal species accumulation curves for both sites and seasons over thirty days. The curve shows how species richness increased with sampling duration (days).

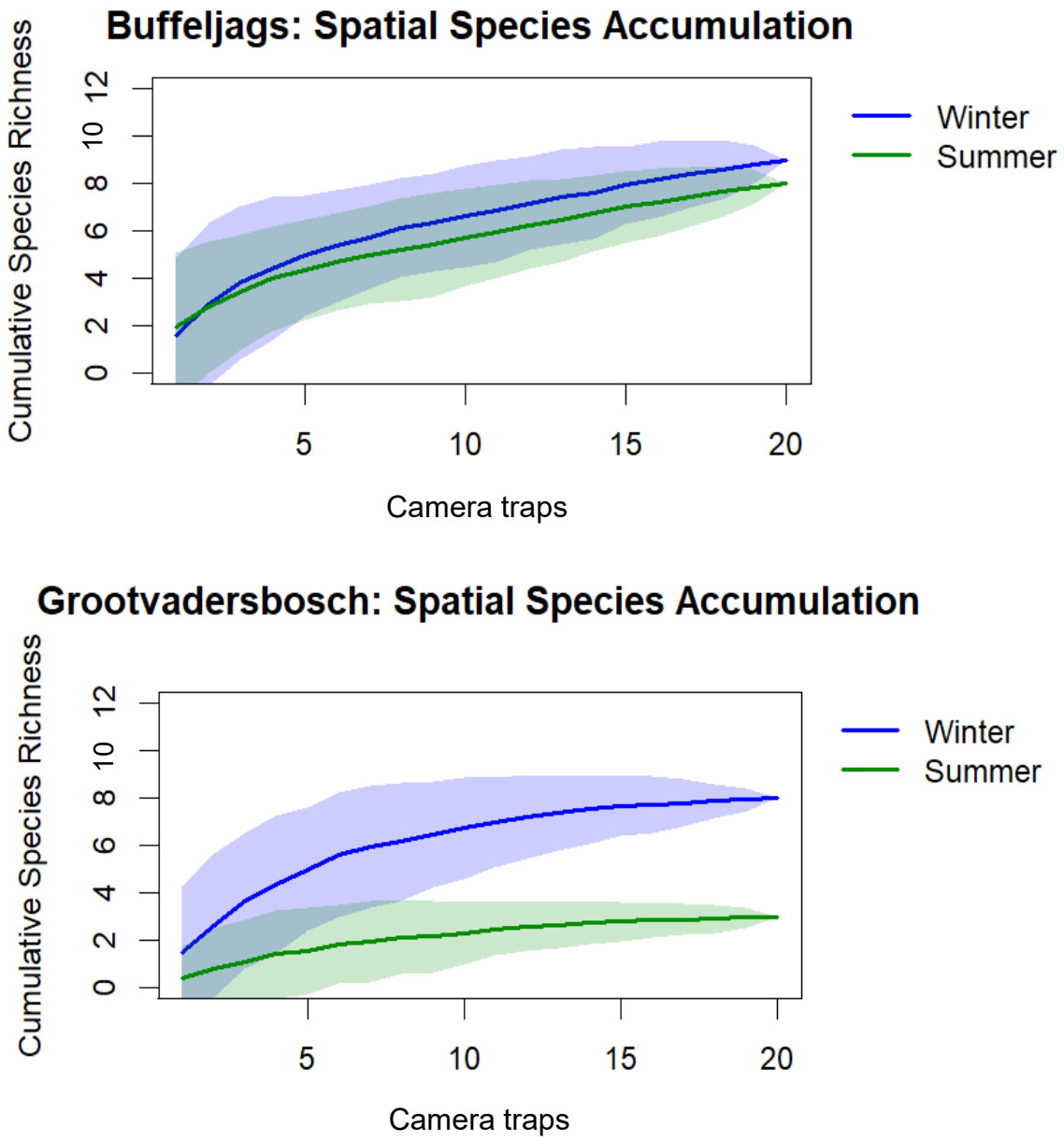

Figure 4.5 Spatial species accumulation curves for both survey areas, showing the cumulative number of species detected as additional cameras were added (twenty in total per survey area). The curves show how species richness increases with the number of camera traps added. The shading around the lines indicated the 95% confidence intervals.

4.5 Number of detections and relative abundance indices

Specific relative abundance index values mentioned are derived from Table A3. Relative abundance index values were used descriptively and not statistically tested because there were low detections overall and many zero detections. Across both rivers, Chacma baboon and Cape bushbuck had the highest RAIs relative to this study, with Chacma baboons RAIs ranging between 4.1 and 16.7, and total RAI being 30.5 and Cape bushbuck ranging between 2.3 and 7.9, with the total RAI

being 21.8 (Figure 4.6). This indicated that they were relatively more abundant or more frequently detected. African clawless otter, bushpig, Cape genet, and water mongoose had comparatively moderate RAIs from between 0.3 and 5.6. while Cape porcupine, Cape grey mongoose, caracal, honey badger, and vervet monkey had low RAIs. Their RAIs ranged between 0.18 to 1.5 across sites and seasons.

Relative abundance index values also varied seasonally and between rivers. Most species were more frequently detected at Buffeljags River, except for porcupine and water mongoose, which were more common at Grootvadersbosch River. Seasonal patterns were also evident: relative abundance indices were generally higher in winter than summer, with bushpig, otter, and genet not detected during the summer survey at Grootvadersbosch River, likely reflecting the lower camera trap effort for that survey. Notably, bushbuck and water mongoose showed higher RAIs in the summer: Bushbuck had RAIs of 6.63 and 7.91 in summer compared with 5.04 and 2.25 in winter, while water mongoose had RAIs of 0.18 and 5.58 in summer compared with 0.97 and 1.13 in winter, suggesting their continued occurrence along riparian zones during hotter months

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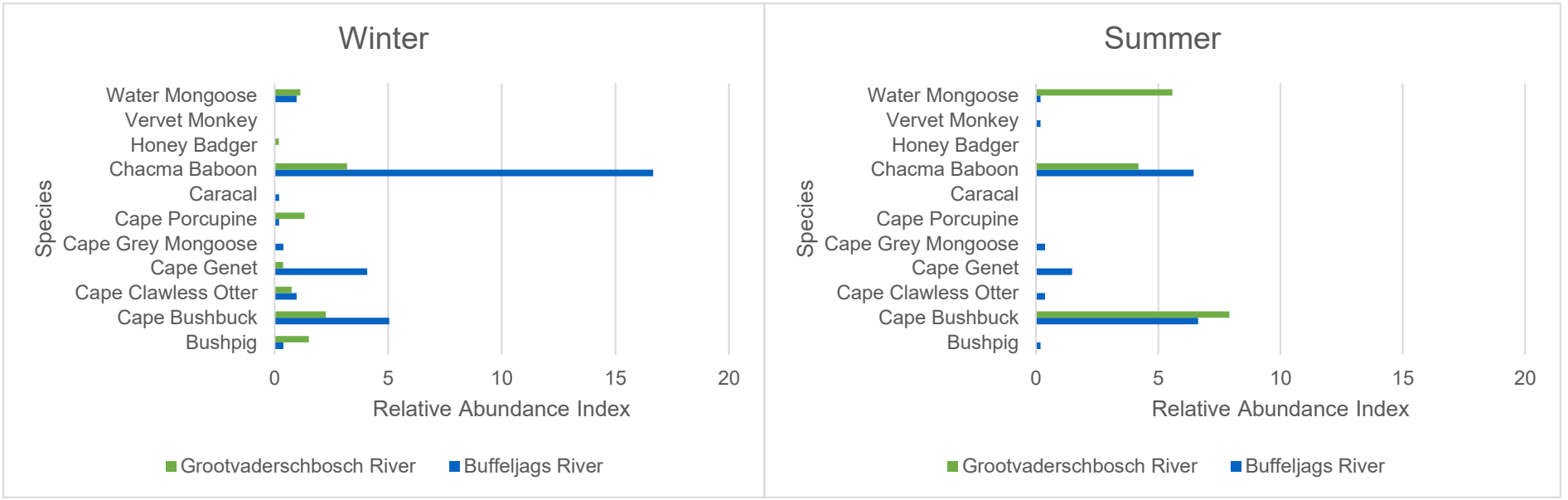


Figure 4.6 Relative Abundance Index (RAI) of riparian mammals per 100 camera trap days across sites and seasons The figure on the left-hand side displays relative abundance index values for the winter season, while the figure on the right-hand side displayed relative abundance index values for the summer surveys.

4.6 Community-level responses

All credible intervals were broad and any patterns should therefore be treated as tentative. Seasonal variation in probability of site use was evident (Table 4.6 and Tables A7, A8 and A9): baseline community probability ofsite use was higher in summer in Buffeljags river (mean ψ = 0.81, 95% CrI: 0.48–0.97) than in winter (mean ψ = 0.47, 95% CrI: 0.23–0.81), with less uncertainty in summer values. Probability of site use in Buffeljags River, under average conditions, was higher in summer than in winter.

Detection probabilities were generally low in both seasons (mean p = 0.01 in winter; 0.03 in summer), but increased with camera trap effort, which was higher in winter (mean detection probability with effort = 0.85) than in summer (mean p = 0.41), highlighting the importance of sampling effort.

Table 4.6 Community-level covariate effects on site use and detection probability compared between seasons

	Winter			Summer		
Detection probability parameter	**Mean predicted detection probability (p)**	**95% Credible Interval**	**Logit effect direction**	**Mean predicted detection probability (p)**	**95% Credible Interval**	**Logit effect direction**
Intercept	0.01	0–0.03	_	0.03	0.01–0.25	-
Effort	0.85	0.62–0.95	+	0.41	0.23–0.6	-
Occupancy parameter	**Mean predicted occupancy probability (ψ)**	**95% CrI**		**Mean predicted occupancy probability (ψ)**	**95% CrI**	
Intercept	0.47	0.23–0.81	–	0.81	0.48–0.97	+
Area Grootvadersbosch	0.53	0.22–0.86	+	0.36	0.03–0.90	–
Canopy cover	0.50	0.26–0.79	–	0.59	0.30–0.84	+
Conductivity	0.42	0.20–0.64	–	0.41	0.05–0.93	–
Dissolved oxygen	0.60	0.29–0.83	+	0.60	0.10–0.96	+
Ground vegetation cover	0.40	0.22–0.63	–	0.67	0.37–0.92	+
SASS	0.47	0.23–0.71	–	0.31	0.05–0.79	–

The effect of Area varied seasonally (see Table 4.6): Grootvadersbosch River had higher overall predicted site use under average conditions than Buffeljags River (the

reference site) in winter (ψ = 0.53, 95% CrI: 0.22 – 0.86) but lower overall predicted site use in summer (ψ = 0.36, 95% CrI: 0.03 – 0.90).

Among site covariates, dissolved oxygen was consistently associated with higher probability of predictedsite use under average conditions, in both seasons (corresponding to ψ = 0.60, 95% CrI 0.29 to 0.83) (although there was less uncertainty in winter) suggesting it is possibly a positive driver of community probability of site use under average conditions. Under average conditions, canopy cover had similareffects in bith seasons: with a probability of site use of 0.50 (95% CrI: 0.26–0.79) in winter and a slightly positive logit effect in summer corresponding to a probability of site use (ψ) of 0.59 (95% CrI: 0.3 to 0.84). Ground vegetation cover was associated with slightly lower probability of site use in winter (ψ 0.40, 95% CrI: 0.22–0.63), but moderately positive in summer, corresponding to a probability of site useψ) of 0.67 (95% CrI: 0.37–0.92) under average conditions. Conductivity and SASS had weak negative effects in both seasons (corresponding to a probability of site use less than 0.50), with wide credible intervals, indicating uncertainty in their influence. Overall, these results showed that predicted community-level occupancy responses to environmental covariates under average conditions were generally weak and uncertain, with only minor seasonal differences. Detection probability was strongly influenced by survey effort and varied by season, being overall higher in winter, reflecting the other results.

4.7 Seasonal comparison of probability of site useand detection probability per species

Multi-species probability of site use estimates were largely consistent with single-species model results, supporting the robustness of the multispecies approach (see Table A6). See Table 4.7 for species level probability of site use and detection probability results. Across both seasons, species-level mean occupancy estimates showed that most riparian mammals are moderately to widely distributed, with their average probability of site use ranging from 0.39 to 0.65. Porcupine (ψ = 0.59, 95% CrI 0.16–0.98) and African clawless otter (ψ = 0.55, 95% CrI: 0.13–0.96) had the highest probability of site use in winter, whereas Chacma baboon (ψ = 0.65, 95% CrI: 0.16–0.88), Cape bushbuck (ψ = 0.59, 95% CrI 0.05–0.99), bushpig (ψ = 0.56, 95% CrI: 0.05–0.88), and water mongoose (ψ = 0.57, 95% CrI: 0.01–1) were most

common in summer. In contrast, Cape genet (ψ = 0.39, 95% CrI: 0.09–0.78) and bushpig (ψ = 0.41, 95% CrI: 0.05–0.88) had the lowest probability of site use in winter, and African clawless otter (ψ = 0.41, 95% CrI: 0.0–0.98), Cape porcupine (ψ = 0.40, 95% CrI: 0.0–0.98), and Cape genet (ψ = 0.43, 95% CrI: 0.0–0.98) were least detected in summer.

The 95% credible intervals were relatively wide for mean probability of site use, particularly in summer, which reflects high uncertainty, There was slightly less uncertainty in winter, possibly because of increased camera trap effort in winter. Detection probabilities were generally low across species and seasons, ranging from near zero to 0.06. Baboon and bushbuck had the highest detection probabilities (p = 0.04–0.06), consistent with their higher amount of detections/RAI values (see Table 4.5 and Figure 4.6). The near 0 detection probabilities for bushpig and porcupine in summer does not mean it is truly 0, it just means it is a very low number (< 0.005).

Table 4.7 Species level mean occupancy (ψ) and detection probability (p) estimates with 95% credible intervals

Winter				
Species	**Mean occupancy (ψ)**	**95% CrI**	**Mean detection probability (p)**	**95% CrI**
Bushpig	0.41	0.05–0.88	0.02	0–0.05
Cape Bushbuck	0.48	0.17–0.87	0.03	0–0.08
African clawless Otter	0.55	0.13–0.96	0.01	0–0.04
Cape Genet	0.39	0.09–0.78	0.02	0–0.08
Cape Porcupine	0.59	0.16–0.98	0.01	0–0.04
Chacma Baboon	0.48	0.16–0.88	0.04	0–0.15
Water Mongoose	0.51	0.13–0.93	0.02	0–0.05
Summer				
Species	**Mean occupancy (ψ)**	**95% CrI (ψ)**	**Mean detection probability (p)**	**95% CrI (p)**
Bushpig	0.56	0.01–0.99	0.00	0–0.02
Cape Bushbuck	0.59	0.05–0.99	0.06	0.03–0.10
African clawless Otter	0.41	0–0.98	0.01	0–0.03
Cape Genet	0.43	0–0.98	0.02	0.01–0.04
Cape Porcupine	0.40	0–0.98	0.00	0–0.01
Chacma Baboon	0.65	0.06–0.99	0.06	0.04–0.09
Water Mongoose	0.57	0.01–1	0.01	0–0.02

4.8 Probability of site use species specific habitat associations

4.8.1 Detection probability patterns

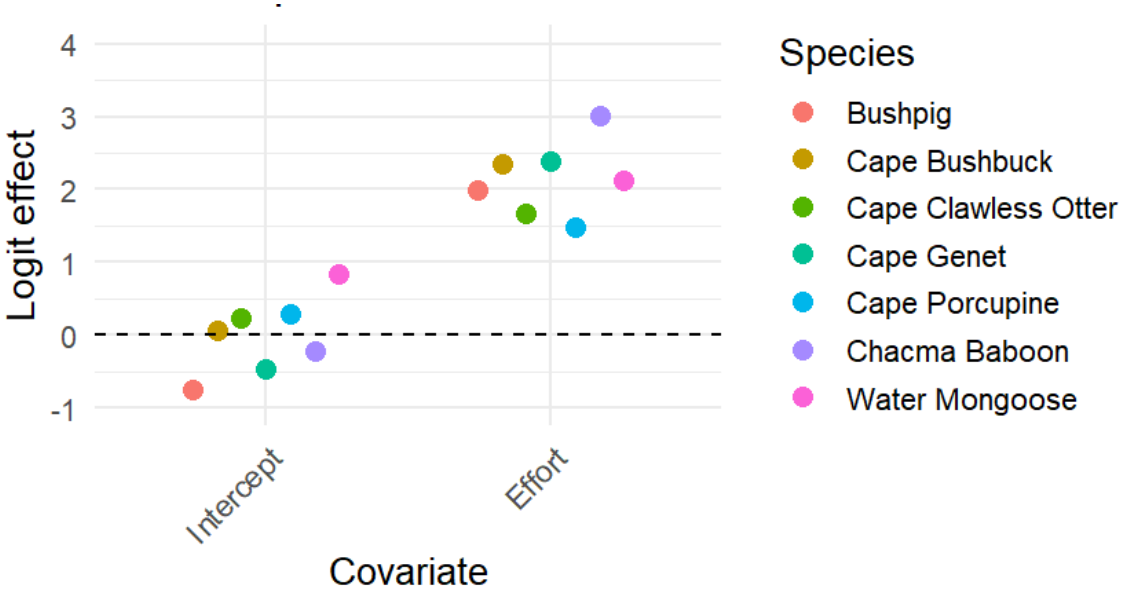
Detection probability were low in general (see Table 4.7 and Figure 4.7). Camera trap effort i.e. the number of days that camera trap days were active, had a strong positive effect across all species (see Figure 4.7). This indicates that detection increased substantially with additional camera effort in winter Chacma baboon and Cape genet showed the highest effort-related detectability while Cape porcupine had the lowest.

In summer, baseline detectability was very low for all species (see Table 4.7 and Figure 4.7). Animals were therefore unlikely to be detected without increased effort. However, there are potentially other factors that can influence detection

probability that was not modelled for, such as distance from road etc. Effort effects were weaker and mostly negative compared with that of winter (see Figure 4.7). African clawless otter and Cape bushbuck had the highest effort-related detectability while Cape porcupine and Cape genet had the lowest. Thus, additional camera effort had a limited effect on improving detection during the summer survey.

Overall, species showed higher baseline detection probabilities and stronger responses to effort in winter, whereas summer detections were low and less responsive to additional sampling effort.

Winter



Summer

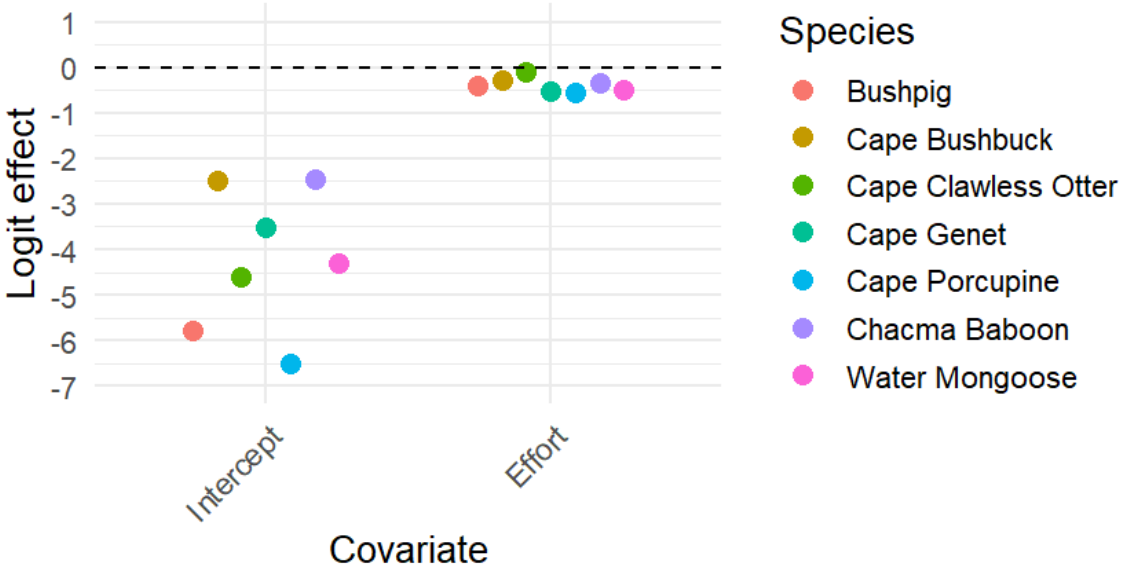


Figure 4.7 Winter and summer species level detection probability logit effect

4.8.2 Probability of site use patterns

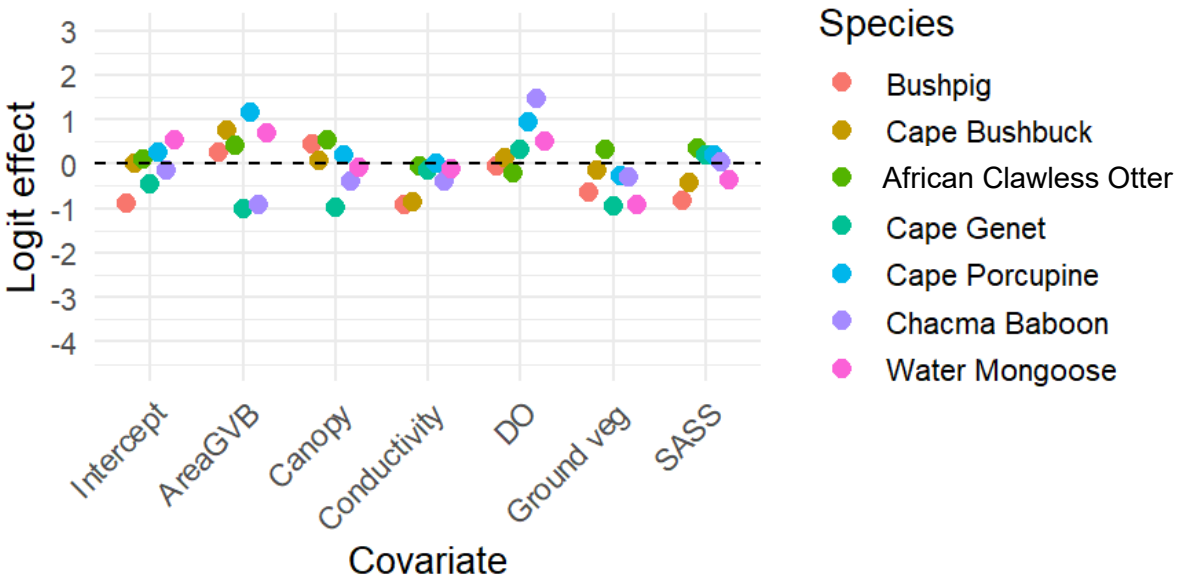
Covariate effects on the probability of site use of each species are summarized in Figure 4.8. The direction and strength of associations differed among species and covariates, with some species showing positive responses to specific environmental variables while others showed weak or inconsistent effects. These results provide an overview of the modelled probability of site use relationships, which are shown in more detail using the response curves presented in Figures 4.9–4.18. Predicted probability of site use are provided in Table B1.

During winter, species-level probability of site use responses was generally more variable across covariates compared with summer (Figure 4.8 and Table A8; specific probability of site use probability numbers extracted from Table A8). Area effects showed that most species had higher probability of site use in Grootvadersbosch sites (ψ = 0.54–0.64), except for Cape genet (ψ = 0.30, 95% CrI: 0.038–0.70) and Chacma baboon (ψ = 0.31, 95% CrI: 0.0644–0.639), which had higher probability of site use at Buffeljags River. This correlates with the community level results (see Table 4.6). Canopy cover were positively correlated with higher predicted probability of site use for bushpig (ψ = 0.59, 95% CrI: 0.27-0.93) and otter (ψ = 0.60, 95% CrI: 0.24-0.95), while baboon (ψ = 0.41, 95% CrI: 0.22-0.61) and genet (ψ = 0.29, 95% CrI: 0.06-0.54) showed negative associations, and bushbuck, porcupine, and water mongoose responses were weak or neutral. High ground vegetation cover was negatively correlated with bushpig, genet, and water mongoose's predicted probability of site use (ψ ≈ 0.30–0.37) but positively correlated with otter's predicted probability of site use (ψ = 0.55, 95% CrI: 0.21-0.96). High dissolved oxygen was positively related to baboon probability of site use (ψ = 0.79, 95% CrI 0.59-0.95), porcupine (ψ = 0.67, 95% CrI: 0.24-0.99), genet (ψ = 0.58, 95% CrI: 0.30-0.83), and water mongoose (ψ = 0.61, 95% CrI 0.19-0.99), while bushpig (ψ = 0.50, 95% CrI: 0.07-0.84), bushbuck (ψ = 0.53, 95% CrI: 0.23-0.82), and otter (ψ =0.47, 95% CrI: 0.05-0.86) showed weaker associations. Higher conductivity was generally associated with lower predicted probability of site use for most species (ψ = 0.32–0.42), except for otter, porcupine, and water mongoose which were weakly negative/neutral (correspondting to probability of site use between 0.48–0.50). SASS scores showed mixed associations: with higher SASS scores showing slightly higher predicted probability of site use for genet, otter, and porcupine (ψ = 0.54-0.57), but lower predicted probability of site use for bushpig,

bushbuck, and water mongoose (ψ = 0.34-0.43), while Chacma baboon was near neutral (ψ = 0.51, 95% CrI 0.29-0.77). Overall, winter probability of site use was moderate across species with the mean probability of site use probability ranging between 0.30–0.64, with wider credible intervals than in summer, indicating greater uncertainty in winter species distributions.

During the summer season, probability of site use probabilities and covariate relationships across species showed a more grouped response compared with winter (Figure 4.8). The Grootvadersbosch area covariate was associated with lower predicted probability of site use for otter, genet, and porcupine (ψ = 0.22–0.30) and a weak lower predicted probability of site use for bushpig (ψ = 0.44, 95% CrI: 0.03-1), indicating lower probability of site use for these species at Grootvadersbosch River compared with Buffeljags River. In contrast, bushbuck, baboon, and water mongoose showed higher predicted probability of site use (ψ = 0.51–0.54), indicating greater probability of site use at Buffeljags River sites in summer. Most species had higher predicted probability of site use under higher canopy cover (ψ = 0.55–0.60), - ground vegetation cover (ψ = 0.60–0.68), and - dissolved oxygen (ψ = 0.53–0.62). Higher conductivity was associated with lower predicted probability of site use for nearly all species (ψ = 0.35–0.46), except bushpig and otter, on which it had a more neutral effect (ψ = 0.45–0.46). Associations with SASS were more variable, with high SASS values being associated with lower predicted probability of site use for otter and water mongoose (ψ = 0.24–0.28) and a bit higher for baboon (ψ = 0.50, 95% CrI: 0.07-0.98). Overall, in summer higher predicted probability of site use was generally associated with sites with dense vegetation, high canopy cover, and moderate water quality, with mean species probability of site use probabilities (ψ) ranging from 0.65 to 0.82.

Winter



Summer

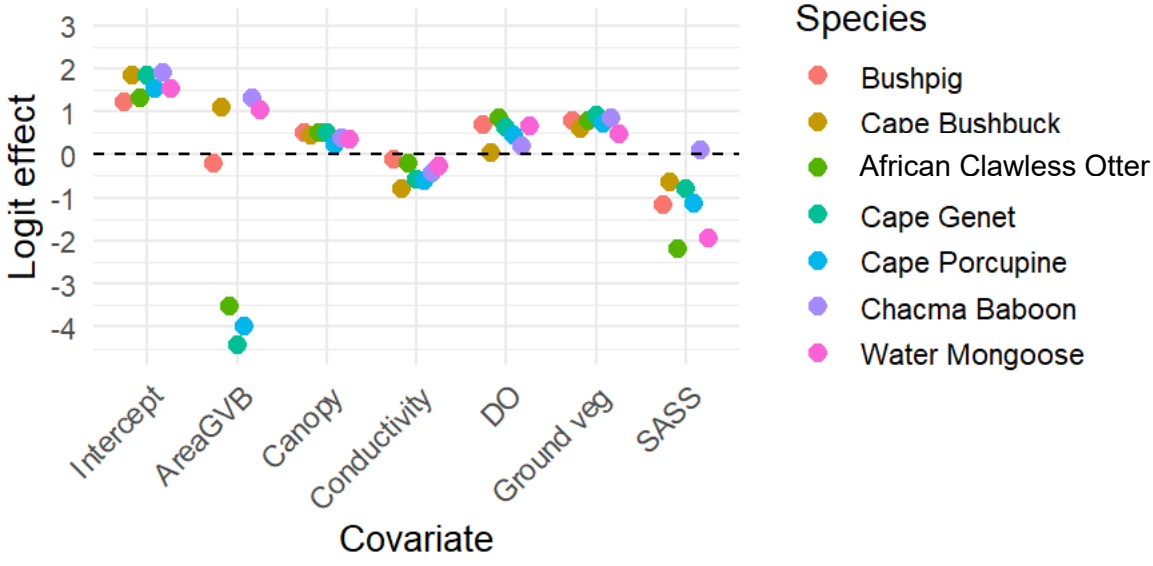


Figure 4.8 Species-level covariate effects (logit scale) for both winter and summer multispecies probability of site use models. The logit-scale coefficients show the direction and magnitude of species responses to changes in site covariates.

4.9 Species-specific probability of site use and detection probability responses to environmental and survey covariates

4.9.1 Detection probability

Mean detection probabilities were low across species in both seasons (Table 4.6 and Table 4.7), reflecting generally low detectability. This is reflected in detection probability-effort relationships, provided in Table B1. During winter, increased sampling effort was associated with slightly higher detection probabilities for all species, although the extent of this effect varied among species. In contrast, detection probability during summer showed little or no response to increased sampling effort

4.9.2 Probability of site use

Species-specific probability of site use–covariate response curves (Figures 4.9 to 4.18) show the relationships between environmental variables and probability of site use probability, and the differences in these relationships between areas and seasons. Covariate response curves represent conditional probability of site use probabilities and therefore do not reflect overall mean probability of site use, which is summarised separately in Table 4.7.

4.9.2.1 Winter response curves

During winter, the relationships between probability of site use probability and environmental covariates were generally weak, linear, and characterised by relatively high uncertainty in both rivers. Credible intervals around the probability of site use response curves were generally wider in winter than in summer. Although overall probability of site use differed between rivers (see Section 4.10), the shape and direction of covariate responses were broadly similar between Buffeljags and Grootvadersbosch Rivers.

Water quality covariates showed limited influence in winter. Conductivity exhibited minimal effects on probability of site use at both rivers, with shallow declines observed for a few species such as bushpig, bushbuck and baboon and flat responses (no increase nor decrease) for the other species (see Figure 4.9). Porcupine had slightly higher probability of site use at Grootvadersbosch River.

Dissolved oxygen had positive associations with most species at both areas (see Figure 4.11). However, responses were gradual and lacked the pronounced threshold patterns observed in summer (see Figure 4.11 and 4.12). Otter probability of site use showed a gradual increase associated with higher dissolved oxygen and then declined at higher dissolved oxygen values-giving a sigmoidal shaped curve. The bushpig response curves had a similar shape but an even weaker response.

SASS scores showed weak effects generally, with largely flat responses (slightly positive for most species) at both rivers (see Figure 4.13). Some species at Buffeljags River, such as bushpig and bushbuck, showed slight declines in probability of site use as SASS increased, while bushbuck, bushpig and water mongoose showed slightly stronger declines at Grootvadersbosch River.

The two vegetation covariates showed gradual effects, with no evidence of threshold responses. Threshold responses in this context refers to a sudden or non-linear change in a species probability of site use response to an ecosystem driver/covariate, as can be seen in most of the species' response to dissolved oxygen in Figure 4.11 (Groffman et al., 2006; Bestelmeyer et al., 2011).

Canopy cover had weak and variable associations with probability of site use in general, except for African clawless otter and bushpig that displayed slight positive relationships with it (see Figure 4.15). The probability of site use of genet and baboon declined as canopy cover increased. Bushbuck and porcupine had flat responses, not showing much increase nor decline in probability of site use in relationship with canopy cover. Ground vegetation had consistent negative effects in both areas, for most species, especially for bushpig, baboon, genet, and water mongoose (see Figure 4.17). Bushbuck and porcupine had a more neutral, slightly negative relationship with it. Otter was the only species that had a positive relationship with ground vegetation cover in both areas.

4.9.2.2 Summer response curves

In contrast to winter covariate response curves, summer probability of site use responses was stronger and showed more structured relationships with environmental covariates in both areas, especially the aquatic variables.

Conductivity showed different patterns between rivers (see Figure 4.10). In Buffeljags River, probability of site use generally declined with increasing

conductivity across most species, with the steepest negative relationships evident for bushpig and bushbuck and weaker declines for the other species. In Grootvadersbosch River, the conductivity responses were mostly flat, indicating little change in probability of site use across the observed conductivity range, although Cape genet and chacma baboon showed slight positive trends. Overall, conductivity effects were weak relative to dissolved oxygen, and uncertainty was greater at the ends of the gradients.

Dissolved oxygen had a strong positive effect on probability of site use in summer (see Figure 4.12). During summer, probability of site use showed a strong positive relationship with dissolved oxygen at both Buffeljags River and Grootvadersbosch River, with most species exhibiting clear threshold responses. The overall shape of the response curves was highly consistent between areas, indicating similar ecological responses across sites. In both areas, dissolved oxygen acted as a limiting factor, with probability of site use constrained at lower oxygen levels and increasing rapidly above higher concentrations. Although uncertainty increased toward the extremes of the gradient because of data sparsity, the steep increases in probability of site use occurred within the range of observed values.

Species-specific probability of site use were generally slightly higher in Buffeljags River than in Grootvadersbosch River across much of the gradient. Otters showed one of the strongest responses in both areas, with probability of site use remaining low below approximately 65–70% dissolved oxygen before increasing rapidly and reaching high probabilities (>0.75) at higher oxygen levels. Water mongooses displayed a similar, though slightly less pronounced threshold response in both areas. Baboons also exhibited a strong threshold pattern, with sharp increases in probability of site use at higher oxygen concentrations. The remaining species showed weaker but still positive responses that were broadly consistent between areas. Genets showed marked increases in probability of site use above approximately 70% dissolved oxygen, while bushpig and bushbuck exhibited more gradual increases and broad plateaus. Cape porcupines showed only shallow increases in probability of site use across the gradient. Overall, while the strength of dissolved oxygen effects was comparable between areas, there were differences in baseline probability of site use levels.

SASS scores response curves showed contrasting patterns between rivers (see Figure 4.14). In Buffeljags River, probability of site use responses to SASS

were generally flat across species, indicating little sensitivity to variation in macroinvertebrate-based water quality within the observed range. In contrast, in Grootvadersbosch River, most species showed declining probability of site use with increasing SASS values, with more clear negative trends for otters, bushpig, bushbuck, and water mongoose.

The vegetation covariates (canopy cover and ground vegetation cover) had similar influences during summer at both areas. During summer, canopy cover showed consistent but generally weak positive relationships with probability of site use across species and both rivers (see Figure 4.16). Most species exhibited gradual increases in probability of site use with increasing canopy cover, with no clear threshold responses. Otter, bushpig, bushbuck, and water mongoose showed the clearest positive trends, with probability of site use increasing steadily from low to moderate canopy cover and levelling off at higher values. Genet and baboon showed similar but shallower responses, while porcupine showed little change across the gradient. The patterns in canopy cover were similar at both rivers.

Ground vegetation cover showed stronger and more consistent positive effects on probability of site use during summer than canopy cover, although the increase was mostly gradual (see Figure 4.18). All species displayed showed clear increases in probability of site use with increasing ground vegetation cover at both rivers, indicating a widespread association with denser understory/grassy conditions. These responses were smooth and similar across species, lacking threshold behaviour, and were similar between rivers. Otter, bushpig, bushbuck, and water mongoose showed especially strong increases across the gradient in both areas, with probability of site use rising steadily from low cover and approaching high probabilities at moderate to high ground vegetation cover. Genet, porcupine, and baboon showed similar patterns, although the strength of the increase varied among species

Overall, summer covariate effects were stronger and more clearly defined than winter effects, while in winter probability of site use changed more gradually across environmental gradients.

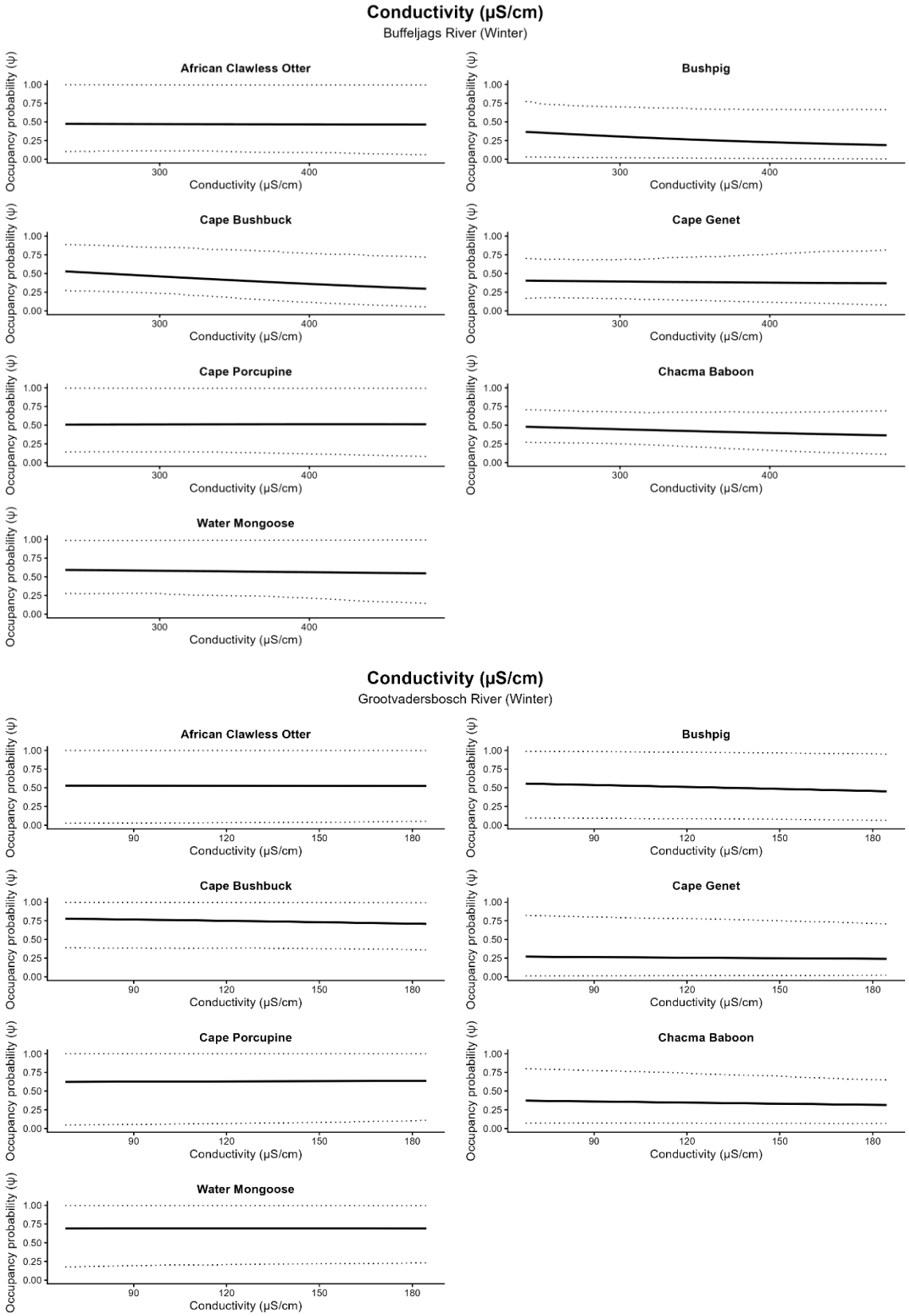


Figure 4.9 Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of **conductivity (µS/cm)** for **winter** surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.

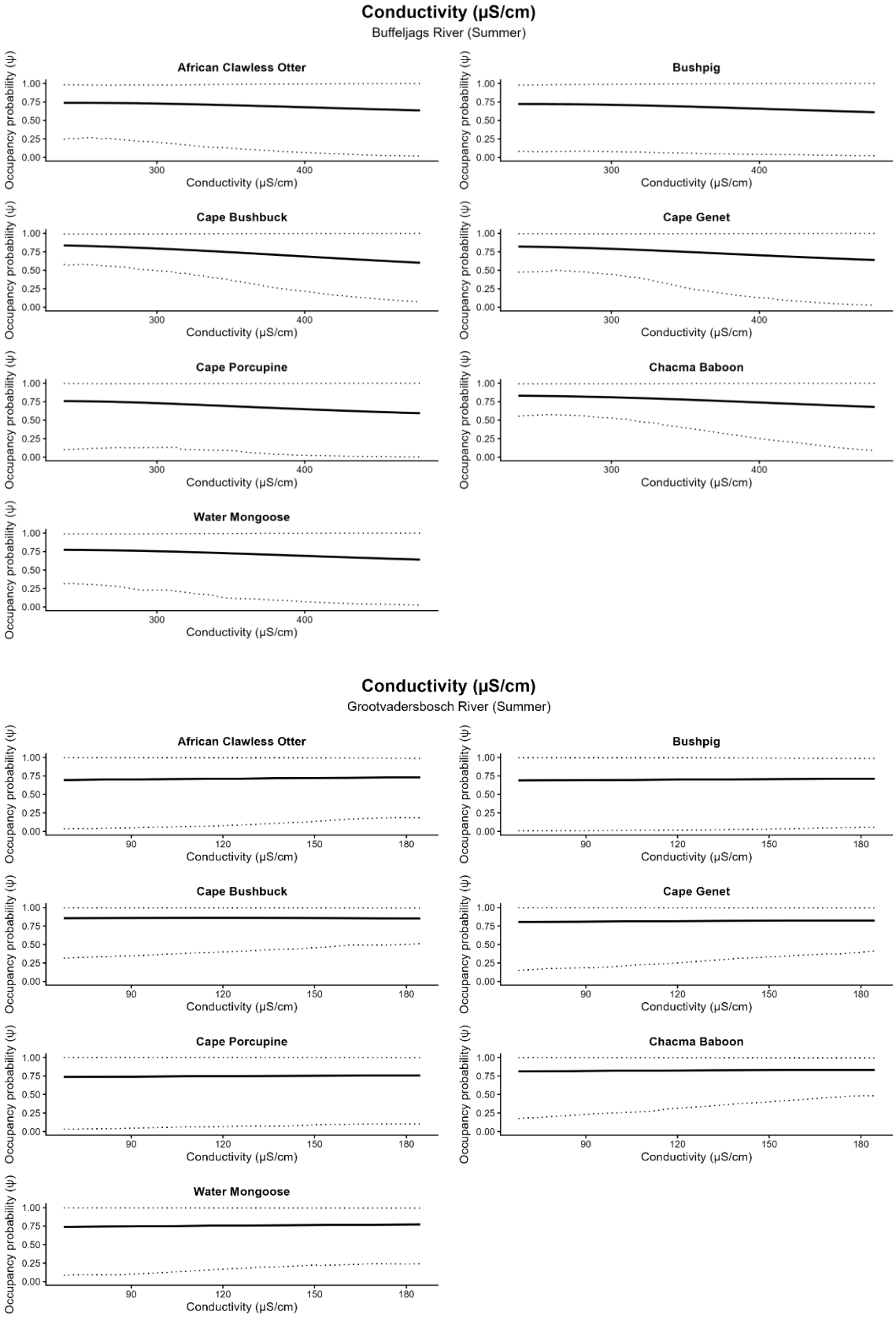


Figure 4.10 *Species-specific occupancy response curves showing predicted probability of site use (ψ) across a gradient of* ***conductivity (µS/cm)*** *for* ***summer*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

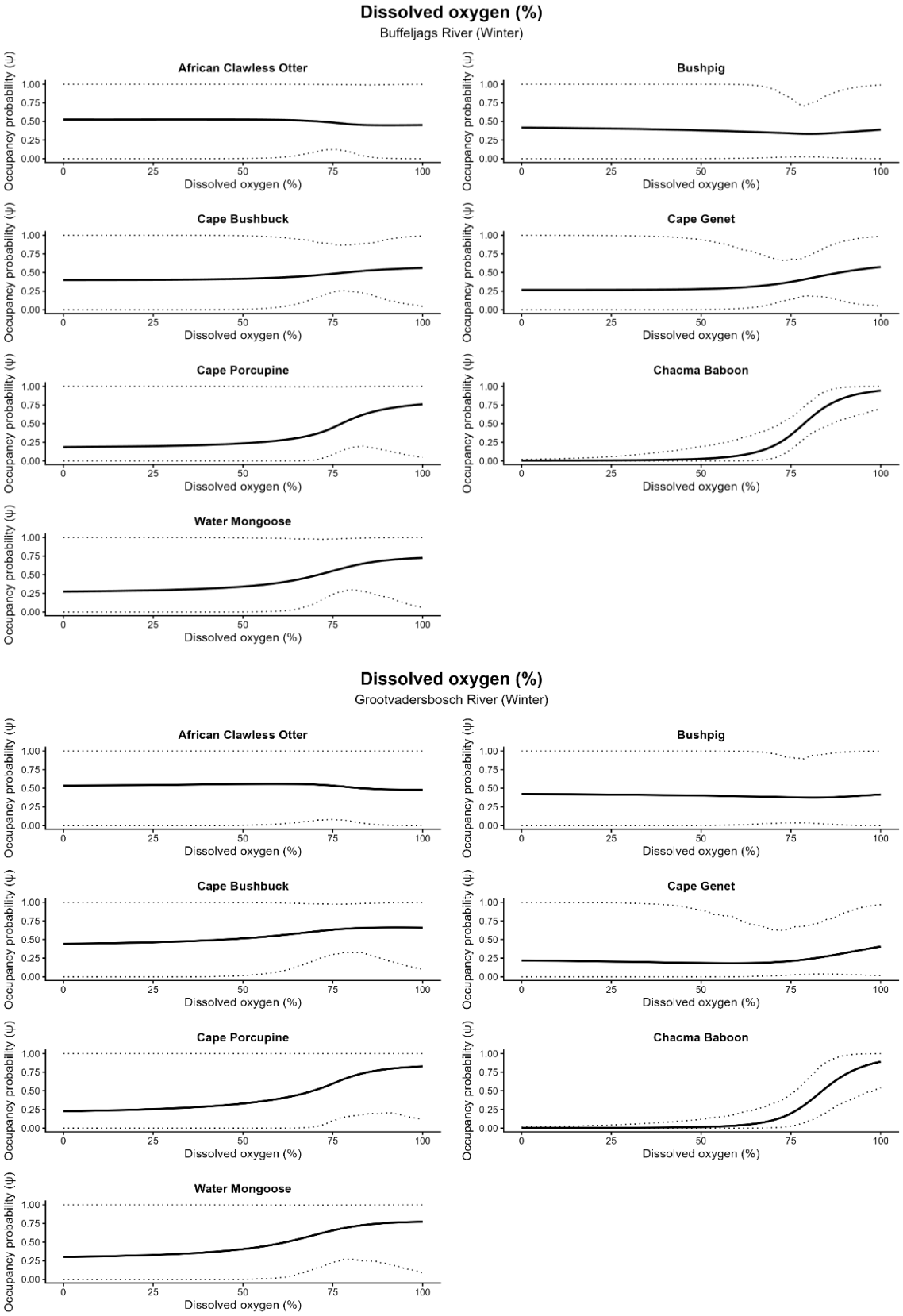


Figure 4.11 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of* ***dissolved oxygen (%)*** *for the* ***winter*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

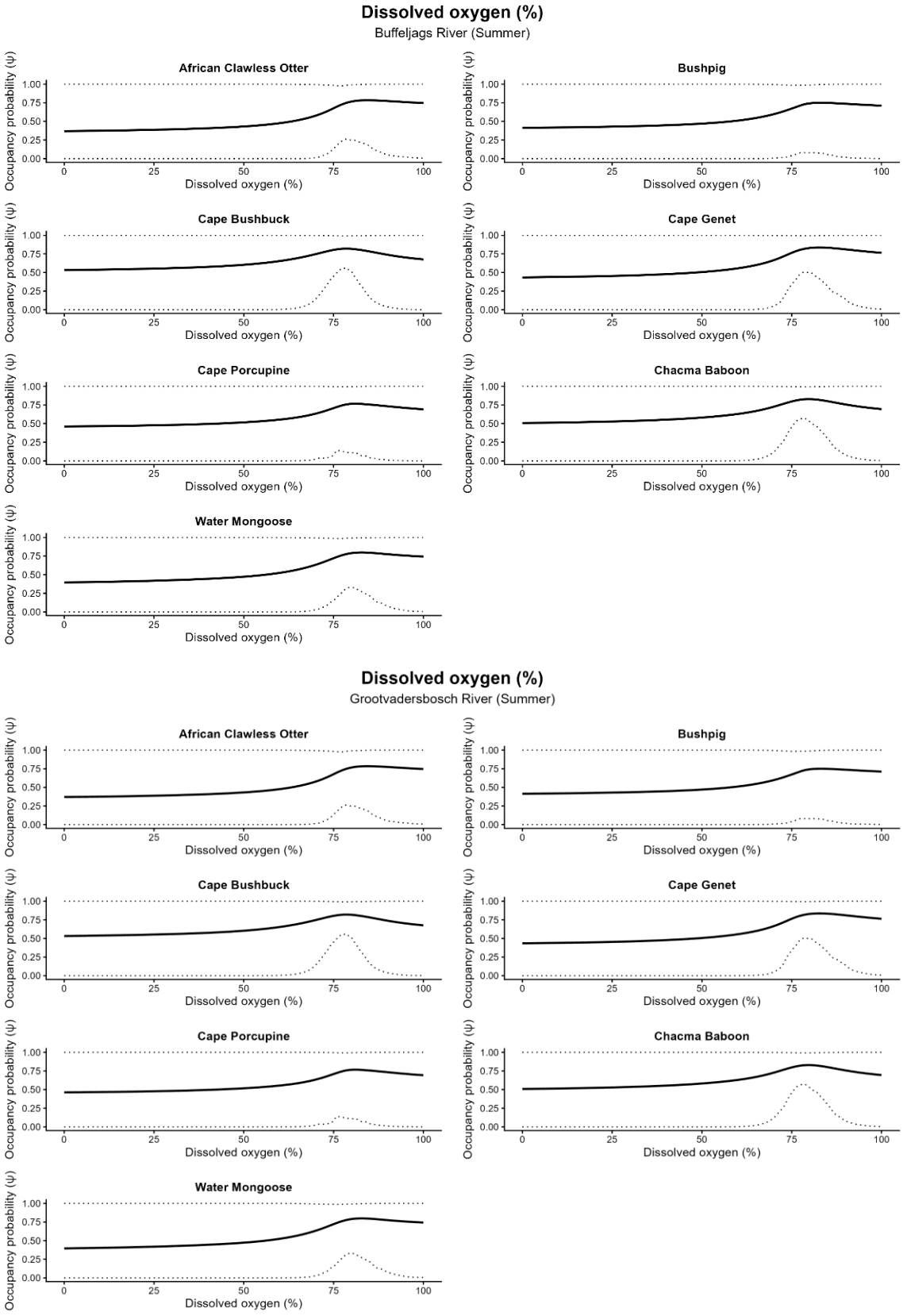


Figure 4.12 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of* ***dissolved oxygen (%)*** *for the* ***winter*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

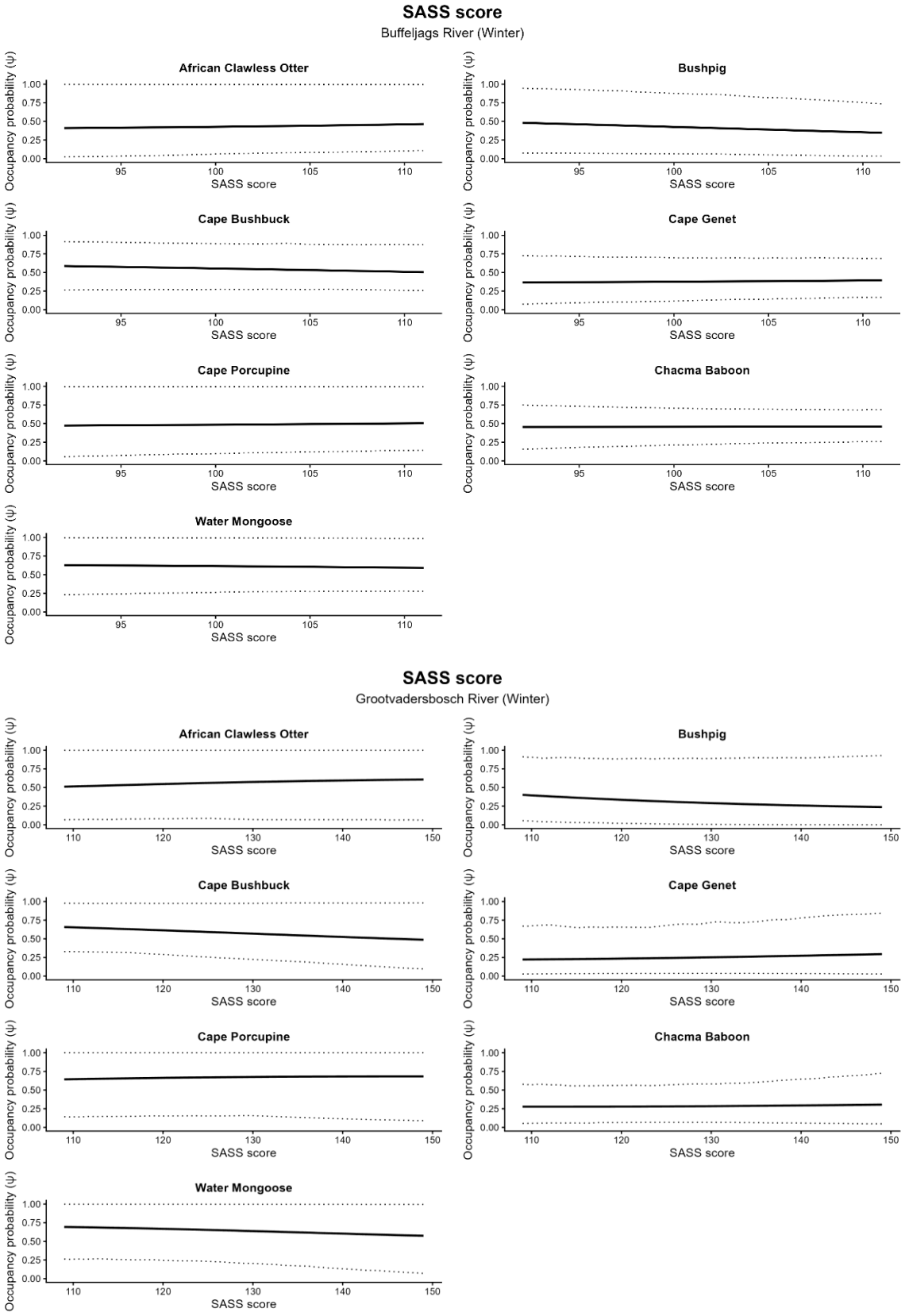


Figure 4.13 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***SASS scores*** *for the* ***winter*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

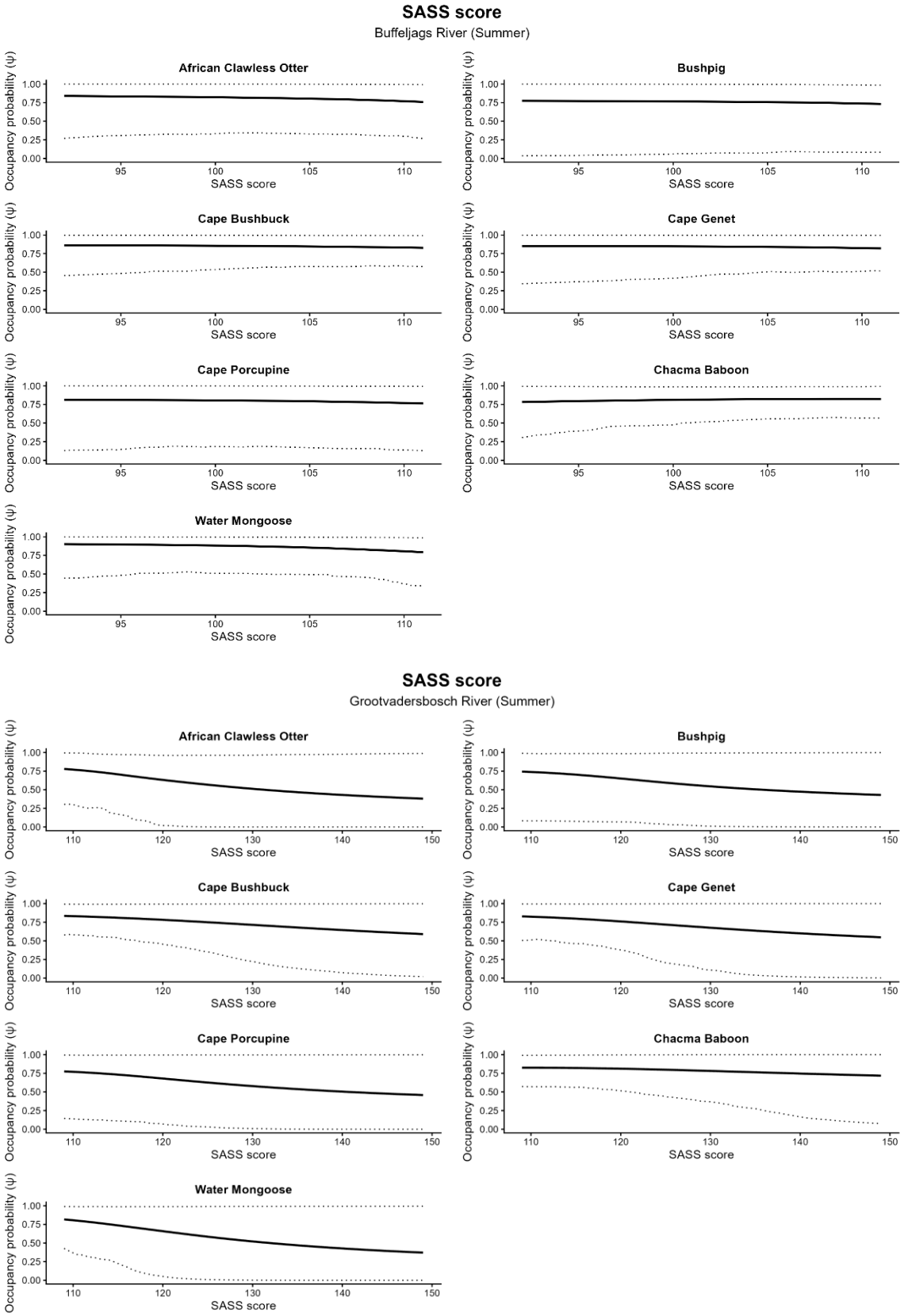


Figure 4.14 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***SASS scores*** *for the* ***summer*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

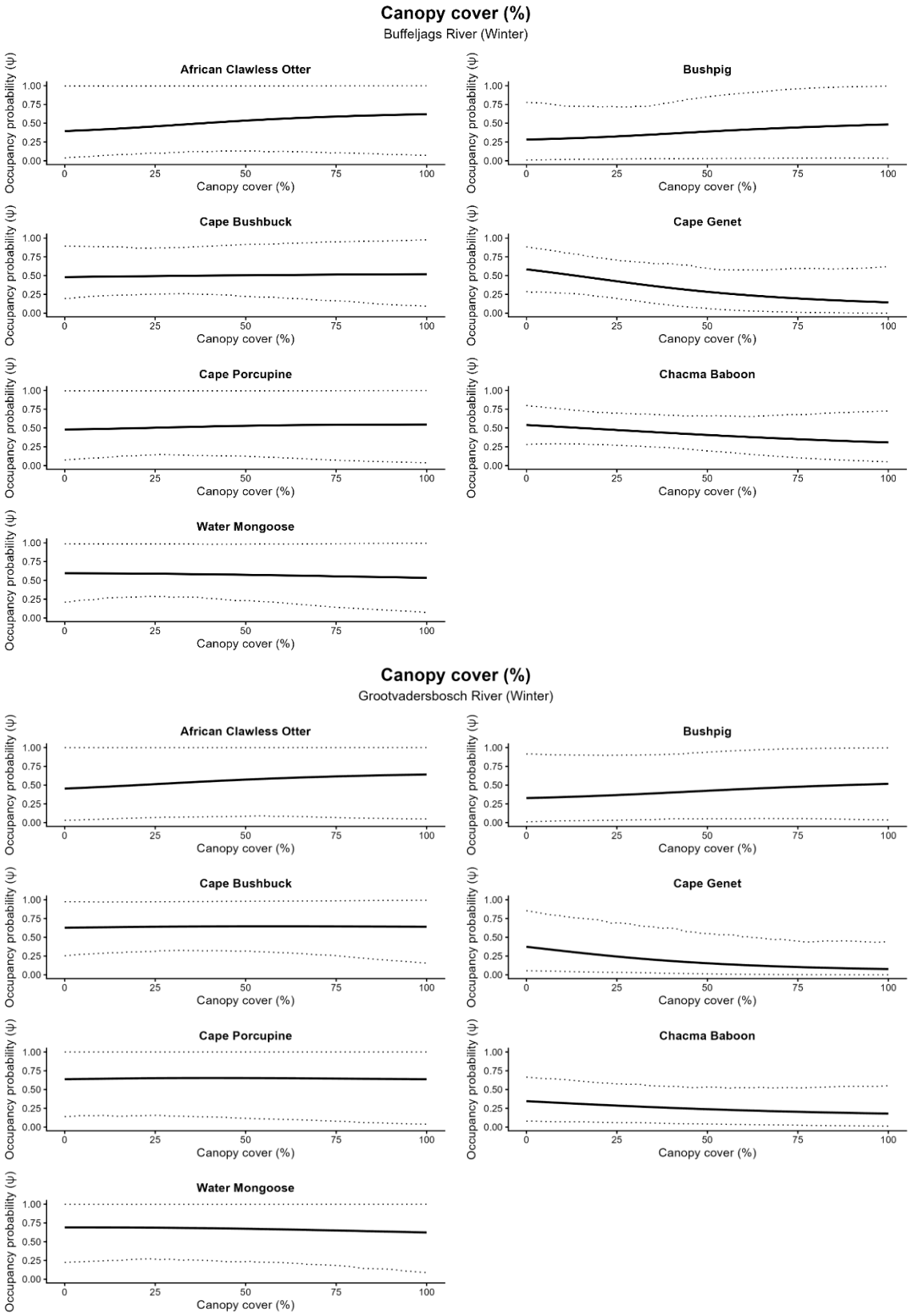


Figure 4.15 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***canopy cover (%)*** *for the* ***winter*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

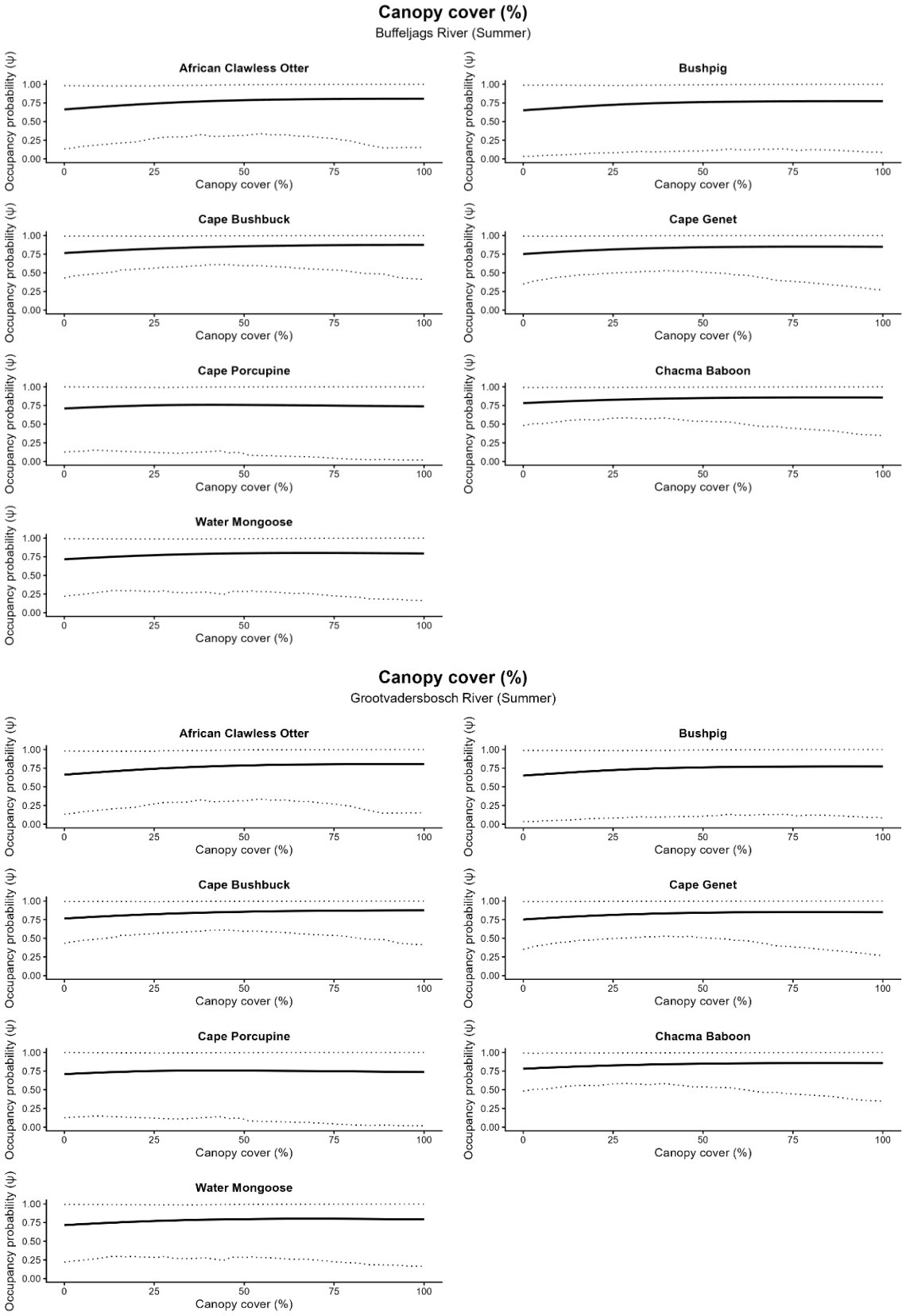


Figure 4.16 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***canopy cover (%)*** *for the* ***summer*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

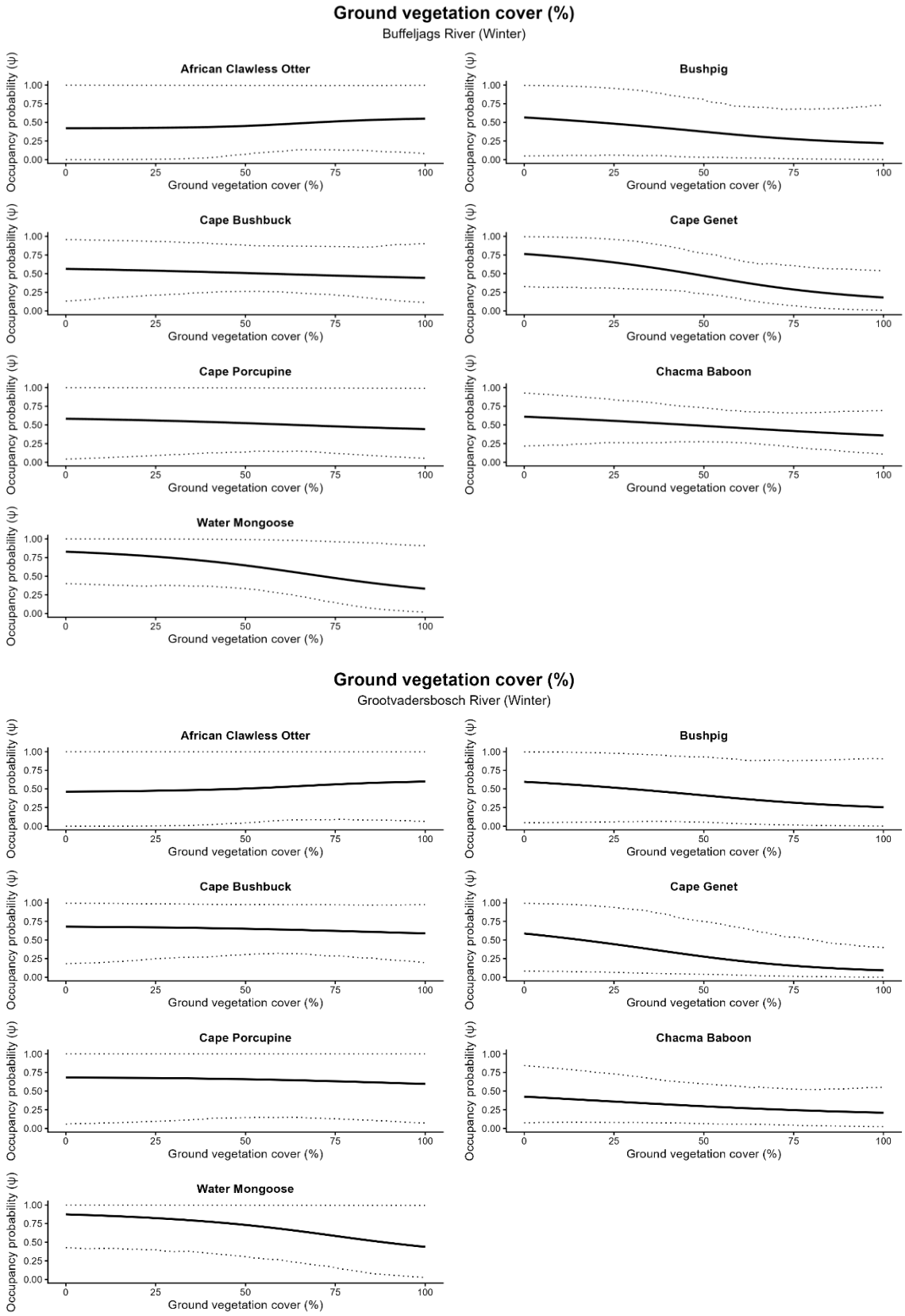


Figure 4.17 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***ground vegetation cover (%)*** *for the* ***winter*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

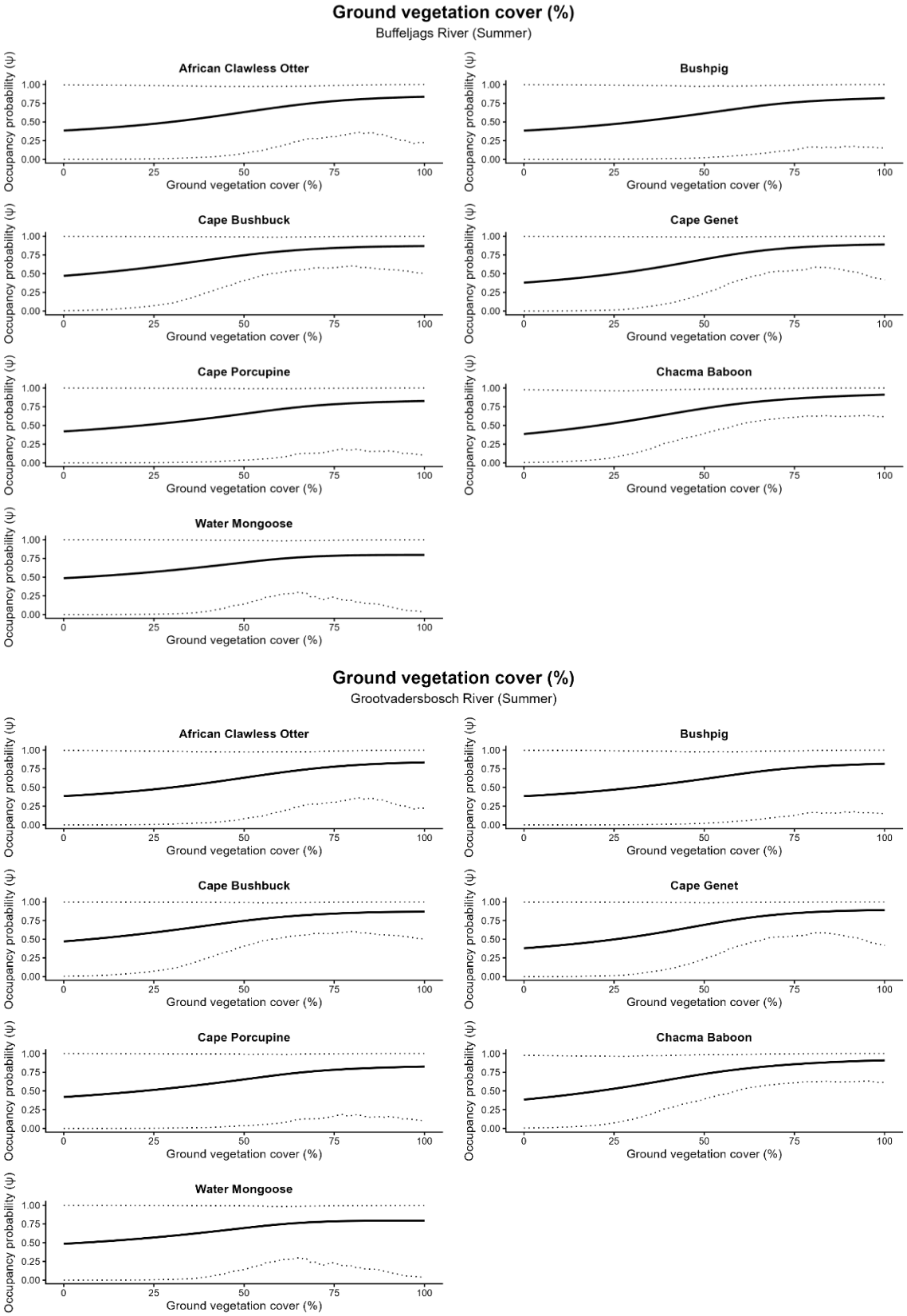


Figure 4.18 *Species-specific occupancy response curves showing predicted occupancy probability (ψ) across a gradient of the* ***ground vegetation cover (%)*** *for the* ***summer*** *surveys at Buffeljags River (top panel) and Grootvadersbosch River (bottom panel). Solid lines represent posterior mean estimates, and dotted lines indicate 95% credible intervals derived from the multispecies occupancy model. Each subplot corresponds to an individual species.*

4.10 Comparison of probability of site use and detection probabilities between areas

Detection probabilities were generally low across species, and in both seasons (see Table 4.6). Therefore, probability of site use (ψ) are used as the primary component of probability of site use for comparing Buffeljags River and Grootvadersbosch River across seasons (see Figures 4.19 and 4.20).

Probability of site use were relatively similar between the two study areas in winter, with overlapping credible intervals which showed similar site use across the community (see Figure 4.19 and Table A10). However, some species exhibited clear area-specific differences Bushpig, bushbuck, otter, porcupine, and water mongoose showed slightly higher probability of site use in Grootvadersbosch, while baboon and genet were more common at Buffeljags. Probability of site use ranged from 0.36 (95% CrI: 0.08-0.64) (bushpig) to 0.56 (95% CrI: 0.32-0.86) (water mongoose) at Buffeljags River, with bushbuck, baboon, and water mongoose showing the highest probability of site use..

Probability of site use ranged from 0.28 (95% CrI: 0.08-0.57) for genet to 0.63 (95% CrI: 0.30-0.99) for water mongoose at Grootvadersbosch, with porcupine, bushbuck, and water mongoose showing relatively higher values. Genet and baboon had notably lower probability of site use in Grootvadersbosch River compared with Buffeljags River, whereas porcupine and bushbuck were more frequent there. Detection probabilities were substantially higher in winter than in summer (ranging between 0.73 and 0.93 across species) and showed minimal differences between areas.Probability of site use patterns shifted in summer (see Figure 4.20 and Table A11), with Buffeljags River supporting higher probability of site use (ψ = 0.65–0.76) than Grootvadersbosch River (ψ = 0.31–0.66) in general (see Figure 4.19). Bushbuck, baboon, genet, and water mongoose showed the highest probability of site use at Buffeljags, while African clawless otter, genet, and porcupine had the lowest probability of site use at Grootvadersbosch.

Detection probabilities were uniformly poor at both areas, indicating strong seasonal effects on species activity. Bushbuck and baboon showed slightly higher detectability ($p \approx 0.08$) compared with other species, but overall detection remained low.

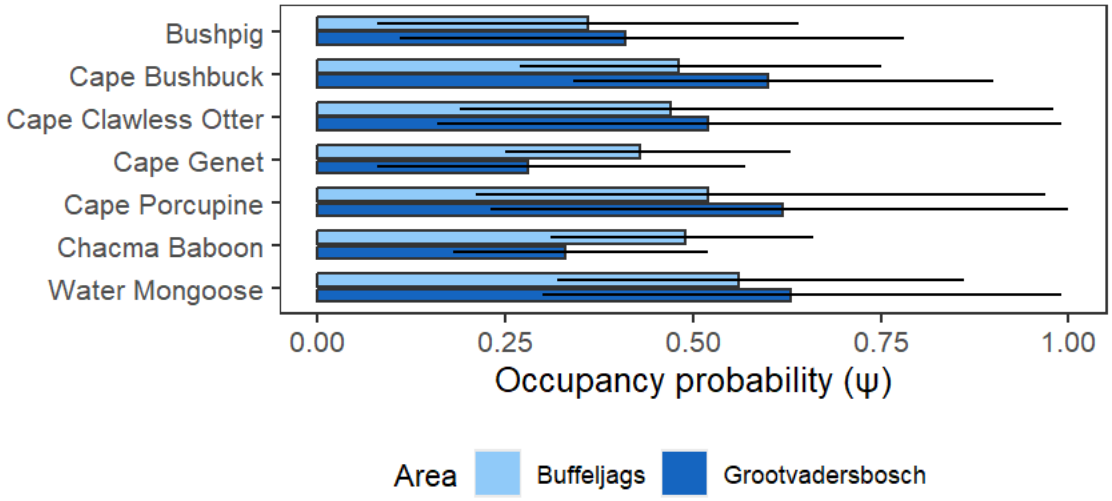


Figure 4.19 ***Winter****: Species level occupancy probability comparison between areas.*

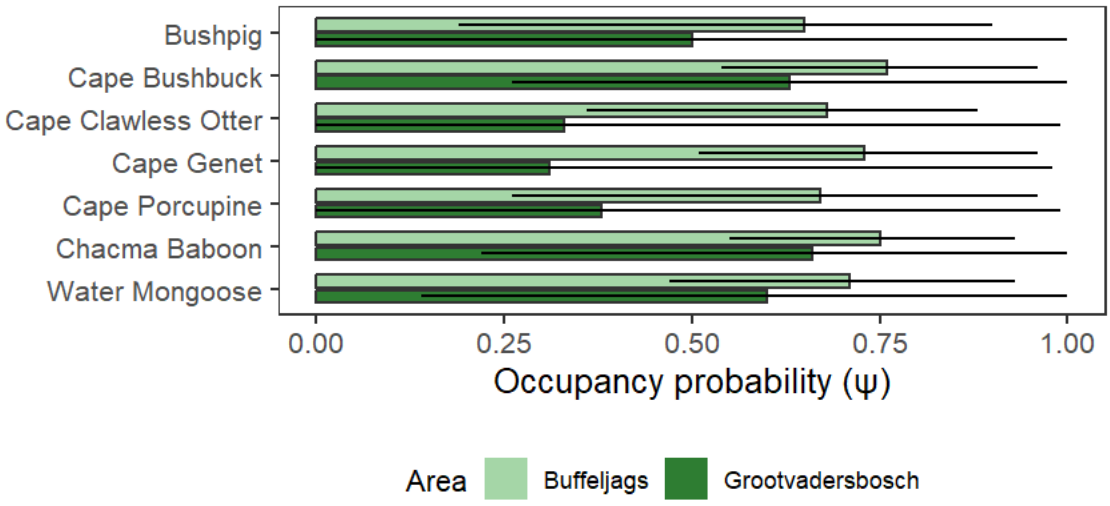


Figure 4.20 ***Summer****: Species level occupancy probability comparison between areas*

CHAPTER 5: DISCUSSION

5.1 Overview of aims

This study aimed to investigate the probability of site use and detection probabilities of riparian mammals across two rivers in the Breede–Gouritz water management area, the Buffeljags and Grootvadersbosch Rivers. Specifically, the objectives were to: (1) estimate species-specific probability of site use and detection probabilities from camera trap data and compare these with relative abundance indices (RAIs); (2) evaluate how water quality, as indicated by SASS5 scores and physicochemical parameters, relates to mammal probability of site use; and (3) assess the influence of vegetation structure, including canopy and ground vegetation cover, on patterns of probability of site use.

5.2 Species richness

Species richness helps provides additional insight into the overall ecological condition of the landscape (Zipkin et al., 2009).

Species accumulation curves help determine whether the amount of camera traps used (i.e. spatial species accumulation curves) within the survey and the amount of survey days (i.e. temporal species accumulation curves) is enough to adequately capture all the species present in an area (Thompson and Thompson, 2007).

Temporal species accumulation curves in general followed a stepwise pattern (Silveira et al., 2003), while spatial species accumulation curves were smoothed. The temporal curves showed that most species were detected early on in the survey period, followed by a plateau as fewer new species were recorded. Extending the sampling duration beyond ten days did not lead to a significant increase in species detections. Therefore, survey length was adequate, although longer survey length could have yielded more species detections since it was still increasing.

None of the spatial species accumulation curves reached a clear asymptote, except for the Grootvadersbosch summer curve which started to level off around eighteen camera traps. Increasing the amount of camera traps does increase the detection of species even after all the common species were detected. This

indicates that more camera traps are required to capture all the species present in both study areas.

It should be also noted that in general species accumulation curves tend to underestimate species richness, since it does not account for “spatial heterogeneity” (Bevilacqua et al., 2017).

The detection of eleven mammal species across both areas indicates moderately high species richness, even though as shown on the temporal and species accumulation curves, it is potentially not all the species present in the area. This indicates that riparian zones within mixed-use landscapes, continue to support diverse mammal communities despite adjacent agricultural activity and human presence. A mixed-use landscape or multifunctional landscape is one that supports several purposes at once (Stürck and Verburg, 2016), such as conservation, farming and recreation e.g. farms that retain patches of natural vegetation. Both study areas can be classified as multifunctional landscapes. These multifunctional landscapes can help sustain species richness (Beukes et al., 2025). Research on occupancy in multifunctional landscapes by Beukes et al. (2025) found that agricultural areas supported higher species richness, while natural or protected areas showed higher relative abundance. Similarly, moderately developed areas can also support higher occupancy and relative abundance when green corridors are maintained (Parsons et al., 2018). All this emphasizes that even though abundance might be higher in natural areas, farm or transitional areas can have more species given that patches of natural areas are maintained. Chacma baboons, bushpigs, and Cape genets are often found in these transitional zones (Beukes et al., 2025), while African clawless otters can persist even in urban settings when adequate prey and refuge are available (Okes and O’Riain, 2017). This is reflected in the results.

5.3 Field observations of riparian mammals

African clawless otter was often recorded in areas with deeper water, further from human habitation (personal observation). They were most often seen in groups of two, but individuals were recorded as well. They were recorded once in daytime (early morning) but the other times they were recorded at night. They were often seen on the camera trap photos in the water, with only their eyes visible. Water

mongoose does not swim/hunt in the river, so that made these photos easy to identify. I observed two adult and two juvenile African clawless otters at Grootvadersbosch in February 2024. They were also often seen in pairs and common at both sites. They seemed to be most common at Grootvadersbosch River, as was reflected in the number of detections and relative abundance indices (see Table 4.5 and Figure 4.6).

Bushbuck was common at both sites. Both males and females were recorded on the camera traps, and they were sometimes seen in groups of two or three. They seemed to favour thick vegetation and were often photographed in thick grass or shrub. This is reflected in the probability of site use at the Buffeljags River sites. Chacma baboons were the most common of all species and were often seen on camera trap photos in large troops at both study areas. They were the one species that showed interest in the camera traps, often pulling them down and tampering with them.

There are three species that were only recorded once: caracal, vervet monkey, and honey badger. The caracal that was photographed, was recorded at the Buffeljags River in the first survey. The camera trap where it was seen was at an uninhabited wilder section of the Frog Mountain Guest farm where there is not much human activity. The honey badger was recorded during the first survey at the Grootvadersbosch farm in the Afromontane Forest section of the study site. The vervet monkey was photographed at Buffeljags River, during the second survey. According to the farm owners, they are not common in the area. During the removal of camera traps at Grootvadersbosch farm, there were vervet monkeys close to the river, however they were never recorded on the camera traps there. Cape porcupines were often recorded alone, but they were sometimes recorded in pairs, especially at Grootvadersbosch.

Observed behaviour of species were consistent with known behaviour: Cape genets, porcupines, and bushpigs were detected only at night, whereas Chacma baboons were only recorded during the day. African clawless otters, water mongoose and bushbuck were seen both during the day and at night. Honey badger was recorded late afternoon, and caracal early morning, while vervet monkeys were recorded during the day.

5.4 Habitat and water quality covariates

Differences in habitat structure between the two areas provided important context for interpreting species probability of site use and relative abundance patterns. The surveyal of habitat covariates was done once after the camera trap surveys, so seasonal variation in habitat covariates was not accounted for.

5.4.1 Differences in vegetation covariates between sites.

As mentioned in section 4.4, the habitat next to Buffeljags River and Grootvadersbosch River was quite similar with both sites having areas of open grass interspersed with area of thicker shrubs and trees/canopy cover and some rocky areas. Reeds were also common in some sections of the river in both areas. There were some differences between the two with Grootvadersbosch River generally having more grassy areas and higher canopy cover. This may have influenced probability of site use differences between the two areas, since more grass potentially support herbivores, while higher canopy cover provides more shelter which might be more attractive to certain species such as bushbuck (Brock et al., 2003). Bushbuck did have higher probability of site use at Grootvadersbosch in summer but not in winter, potentially because canopy cover is not the only factor influencing the species probability of site use. Specific species responses to habitat covariates will be discussed further later on in this chapter.

5.4.2 Patterns observed in SASS5 data

All three Grootvadersbosch sites were rated as pristine according to SASS data for December 2023. This was after a very wet winter season and the pristine rating could possibly be linked to high flow rates. High flows generally enhance water quality by increasing dilution (lower total dissolved solutes) and oxygenation, while also flushing accumulated organic matter, debris and pollutants from the system. These conditions favour sensitive macroinvertebrate taxa, which contribute to higher SASS and ASPT scores and therefore better ecological classification (Dallas, 2007). The SASS5 survey points upstream at Grootvadersbosch River in the forest section (H7GROO_FOHUT), were a bit more isolated and had less human and farming activity. This site had higher water quality overall than the two sites further downstream.

At Buffeljags River there was lower water quality at the SASS survey point that was upstream (H7BUFF-LORI). This is unexpected because this is the area where invasive plants were removed five years before the survey and where care was taken to prevent pollution (Riviera, pers.comm.). Downstream (H7BUFF_FROGM) seemed to have more visible pollution with a lot of litter like plastic bags present and oil visible on the water (pers. obs.). This is also the section that had more invasive plants in general. The difference in water quality is slight and could be because of the runoff from the mountain which could increase water quality despite conditions not being pristine. It should also be noted that the data available for these two sites were limited therefore the SASS5 data for H7BUFF_LORI was from April 2023 while it was for July 2022 for H7BUFF_FROGM.

5.4.3 Patterns observed in physicochemical parameters

The increase in measured dissolved oxygen observed from autumn to summer (see Table 4.3) is possibly because of higher flow or increased photosynthetic activity, despite warmer water temperature generally reducing oxygen solubility. pH decreased showing more acidic conditions in winter compared with autumn. The increase in river discharge from autumn to winter is possibly due to autumn being just after the dry summer season of 2022/2023, while November 2023 was after the wet winter season. The increase in conductivity from autumn to summer for Grootvadersbosch River survey sites, showed that there are generally higher dissolved solutes in water in the summer. As mentioned before in section 4.5.2, the water of Grootvadersbosch River was more acidic compared with Buffeljags River. The H7GROO-FOHUT site had the lowest pH of all Grootvadersbosch River sites, possibly because it is in a section of Afrotemperate forest. Due to high tannin content in the rivers and decomposition of leaves in rivers, rivers in forests are often acidic.

5.5 Community-level probability of site use and detection probability

Community-level probability of site use (intercept) of riparian mammals was higher in summer (ψ = 0.81, 95% CrI: 0.48-0.97) than in winter (ψ = 0.47, 95% CrI: 0.23-0.81) at Buffeljags River under average conditions showing possible preference of

riparian zones in the warmer, drier season in Buffeljags river (Royle and Dorazio, 2008). However, community level probability of site use was higher in winter in Grootvadersbosch River in winter (ψ = 0.53, 95% CrI: 0.22-0.86) compared with summer (ψ = 0.36, 95% CrI: 0.03-0.90).

Community-level detection probabilities were generally low (p = 0.01 and 0.03) without accounting for effort, particularly in winter. However, it improved substantially with survey effort in both seasons (p = 0.85 in winter; p = 0.41 in summer; see Table 4.6). This highlights the inherent difficulty of detecting riparian mammals and reinforces that probability of site use estimates represents the probability of site use under imperfect detection rather than absolute presence (MacKenzie et al., 2002; Hochachka et al., 2023).

Seasonal reversals in probability of site use between rivers—higher in Grootvadersbosch in winter but lower in summer, with the opposite pattern at Buffeljags—may reflect differences in vegetation cover, weather conditions, or effort-related detection biases rather than true changes in abundance. This will be discussed in more detail later.

Vegetation structure—particularly canopy and ground cover—was generally more influential in summer, especially for bushbuck, baboon, and water mongoose. Covariate effects in winter were weaker and more variable, possibly reflecting reduced mobility and cooler, wetter conditions.

Community mean probability of site use was negatively associated with SASS scores, particularly in summer (community mean ψ = 0.47 in winter and 0.31 in summer), suggesting that the detection of mammals may not be as greatly influenced by healthier river ecology as initially expected (Dallas and Day, 2004; De Villiers et al., 2019). However, negative SASS effects can also reflect stream accessibility and low camera trap effort rather than avoidance of high-quality habitats.

Dissolved oxygen had a generally positive effect on probability of site use across seasons (community mean ψ = 0.60 in both winter and summer; winter 95% CrI: 0.29–0.83, summer 95% CrI: 0.10–0.96), indicating that areas with higher oxygen levels were somewhat more favourable for riparian mammals.

Community-level conductivity effects on probability of site use were similar across seasons (negative), although uncertainty increased in summer (community

mean ψ = 0.42; 95% CrI: 0.20–0.64 in winter; ψ = 0.41; 95% CrI: 0.05–0.93 in summer).

In contrast to the community's response to SASS metrics, which indicated they were not greatly influenced by more pristine stretches of river, the positive association with high dissolved oxygen and lower conductivity indicates that higher water quality may have increase presence after all.

5.6 Overall probability of site use

It should be noted that any seasonal trends and differences in probability of site use should be treated as tentative given the broad credible intervals and subsequent uncertainty. Semi-aquatic species such as African clawless otters showed potentially lower summer probability of site use (ψ = 0.41, 95% CrI: 0.0-0.98) compared with winter (ψ = 0.55, 95% CrI: 0.13–0.96). A spraint-based occupancy study in the Cape peninsula (Okes and O'Riain (2017) found that the species had an overall higher average detection probably (p = 0.29 compared with 0.01 for this study), with the null model having a detection probability of 0.56. However, the null model displayed an occupancy value of 0.26 which is lower than for the current study (Okes and O'Riain, 2017). They surveyed 600m sections of 10 rivers three times each in three months (Okes and O'Riain, 2017). Although there is a correlation between tracks/sign rate and photographic rates (Silveira et al., 2003), occupancy results from sign and camera trap studies are not technically directly comparable (Day et al., 2016). Sign and camera trap surveys use different detection methods (i.e. tracks and spraints vs camera traps) and time scales, since sign surveys reflect presence over longer time periods while camera traps survey for discrete, often shorter time periods (Lyra-Jorge et al., 2008; Roberts, 2011). In general sign surveys tend to yield higher occupancy and detection probabilities than camera trap surveys in occupancy modelling (Seidlitz et al., 2021). However, they can complement one another and give some insight into how occupancy compares between areas (Lyra-Jorge et al., 2008; Day et al., 2016).

In contrast, the semi-aquatic water mongoose showed the opposite pattern, with slightly higher probability of site use in summer (ψ = 0.57, 95% CrI: 0.01–1 in summer and ψ = 0.51, 95% CrI: 0.13–0.93 in winter). This pattern was similar to more generalist and terrestrial species such as Cape bushbuck (ψ = 0.48 → 0.59)

and Chacma baboon (ψ = 0.48 → 0.65) that maintained or possibly slightly increased their probability of site use in summer.

Probability of site use estimates of Cape genet in this study, which was 0.39; (95% CrI: 0.09–0.78) and 0.43 (95% CrI: 0–0.98) in winter and summer respectively was similar to the average estimated occupancy of Cape genet in farmlands in Kwazulu-Natal in Ramesh and Downs' (2014) study, which was 0.42 ± 0.10. It must be noted though that uncertainty was much higher for the present study, especially in summer. Ramesh and Downs' (2014) the number of camera traps were higher with 43 camera traps set up over 21 days. In another occupancy study in urban areas, Ramesh and Downs (2015) had similar number of camera traps (fifteen camera traps set up for thirty days) as the current study, however the estimated occupancy was higher, 0.62 ± 0.14 with a detection probability of 0.19 ± 0.03. The higher occupancy values could be because the study took place in an urban area. The abundance of food available in urban areas potentially increased their overall presence in the area (Ramesh and Downs, 2015).

5.7 Probability of site use compared with relative abundance index values

Building on the overall species-level probability of site patterns described above, I compared these estimates with relative abundance indices to assess how probability of site use aligns with observed species detection across sites and seasons.

In winter, most species such as porcupine, bushbuck, and water mongoose showed higher probability of site use in Grootvadersbosch, whereas genet and baboon were more common in Buffeljags. In summer, Buffeljags supported higher probability of site use for bushbuck, baboon, and water mongoose, while otter, genet, and porcupine were lower in Grootvadersbosch. Therefore, season influenced probability of site use.

Relative abundance indices generally mirrored probability of site use patterns, showing similar spatial and seasonal patterns, indicating that camera-trap detections provided a reliable proxy for species habitat use. However, some mismatches were observed: Cape porcupine and African clawless otter had low RAIs but relatively high site use, highlighting the ability of occupancy models to

account for imperfect detection. Seasonal patterns in RAI and probability of site use also differed among species; for example, bushbuck and water mongoose had higher values in summer, whereas genet, porcupine, and otter were more abundant in winter. Bushpig and Chacma baboon showed higher probability of site use in summer but higher RAIs in winter, suggesting these discrepancies likely reflect seasonal differences in detectability or movement rather than true absence.

The high RAIs of baboon and bushbuck reflects the high captures of the same species in the Garden Route by Hanekom and Randall (2015), where they were the most frequently recorded species.

These seasonal patterns in detectability provide context for examining how environmental covariates influence probability of site use on a species level.

5.8 Site covariate effects and species-specific patterns

The relatively wide 95% credible intervals for mean probability of site use overall reflect high uncertainty because of low detections and partially missing camera data for Grootvadersbosch sites.

Although winter surveys produced higher detection rates and narrower credible intervals for predicted probability of site use (Tables A8 and A9), response curves (see Figure 4.9 to 4.18) showed greater uncertainty in covariate relationships. This shows that winter probability of site use could be estimated more precisely, but habitat associations were weak. In contrast, summer probability of site use estimates were less precise, likely because of lower effort and missing cameras (especially at Grootvadersbosch), yet covariate responses were more pronounced, suggesting habitat use was more strongly influence by environmental covariates. Thus, the winter uncertainty in the response curves mainly reflected variation in habitat use, while summer uncertainty was driven more by the survey limitations.

Covariate-specific probability of site use helped identify the main environmental gradients which influenced distributions across seasons. While overall probability of site use remained moderate to high across most species, species–habitat associations varied by both covariate and season. These patterns reflect how different ecological factors—vegetation structure, water quality, and location—all influence habitat use across the mammal community.

5.8.1 The effect of habitat covariates on species probability of site use

As emphasized before, structurally complex habitats are thought to support higher biodiversity (Tews et al., 2003; Báldi and Sadler, 2008). Higher canopy cover and denser vegetation often contribute to habitat complexity through the creation of microhabitats such as debris and logs and trees providing a variety of hiding spots and potential nesting spaces. In support of this, studies often find that canopy cover increases species richness and presence, especially in areas of higher human disturbance (Turshak et al., 2011; Boron et al., 2019; Ferreira et al., 2020). However, this is not always the case, and some studies have found higher mammal diversity in areas with lower canopy cover or more open understory. For example, Sullivan and Sullivan (2001) reported that ground-dwelling mammal diversity was lowest in uncut forests with dense canopy and high structural complexity, but higher in harvested areas with reduced canopy and greater herb and grass abundance. This suggests that some riparian mammals are more likely to occupy areas with lower canopy or more accessible ground cover, possibly because ofincreased food availability or ease of movement. So, the preference for ground cover and canopy cover is not always cohesive across species in general. Preferences may vary depending on each species' foraging strategy, movement behaviour, and sensitivity to cover density.

In addition to contributing to habitat structure, riparian vegetation also influences both water quality and habitat structure, filtering pollutants and regulating flow (Crous et al., 2011; Ramião et al., 2020). Structurally complex, tree-rich riparian zones enhance stream integrity and support healthier macroinvertebrate communities (Popescu et al., 2021; Lyons et al., 2000). Therefore, land-use changes are often reflected in water quality and macroinvertebrate communities (Palt et al., 2022). Consequently, semi-aquatic mammals such as otters are frequently associated with well-vegetated, high-quality riparian habitats (Mengistu et al., 2022). This aligns with findings in winter, where species such as water mongoose showed higher predicted probability of site use in sites with lower canopy cover and lower ground vegetation cover, suggesting that more open riparian zones may provide more favourable foraging conditions or movement corridors for them in this season. This is supported by general findings that the species seem to prefer

more open water sources compared with more densely vegetated marshlands (River Health Programme, 2004; Streicher et al., 2021). However, they often tunnel and move through reeds and shrubs next to the river indicating some dependence on thick vegetation (River Health Programme, 2004). Water mongoose can adapt to urban areas but will choose the habitat patches with higher bush/wood cover and lower human disturbance (Streicher et al., 2021).

Surprisingly, bushbuck was not strongly correlated with canopy cover in the winter season despite their general preference for forested areas (Brock et al., 2003). This is consistent with the response curve which showed that overall probability of site use was relatively high but an increase in canopy cover did not affect their probability of site use much (flat curve). They were often sighted in more open areas on camera trap photos. There were however always bushes and forests in proximity (pers. obs). This could correlate with changes in foraging patterns in winter. Unlike many other ungulates, bushbuck population density is primarily a function of food supply rather than spatial or social requirements (Furstenburg, 2010). Since bushbuck are browsers (MacLeod et al., 1996), ground vegetation cover should be a major driver in their distribution, which reflects summer results but in winter it only had a marginal effect. This could be because they move further inland during the colder months, when food is more abundant overall and water is less of a limiting factor compared with summer. This pattern is supported by the mean probability of site use results, which show a lower probability of site use in winter (ψ = 0.48, 95% CrI) compared with summer (ψ = 0.59) (MacKenzie et al., 2002).

Cape genets also had a negative association with canopy cover and ground vegetation cover in both seasons, shown with their predicted probability of site use generally decreasing with an increase in these two covariates. This is consistent with Widdows et al. (2015) who reported lower probability of site use with higher canopy cover in farm areas but inconsistent with Ramesh and Downs (2014), who reported increased detection probability with denser habitat as well as increased human presence in urban areas in Kwazulu-Natal. However, this is possibly because of difference in land use, with buildings in urban areas providing shelter for the species lessening the need for thick vegetation and canopy cover (Widdows and Downs, 2015; Widdows et al., 2015).

Porcupine and bushpig, in contrast to genet and baboon, preferred slightly higher canopy cover in the winter, as illustrated with their (albeit weak) increase as canopy cover increased compared with the other species. This could be because of a need for refuge from cold, wet conditions or it might coincide with their preferred food resources, such as fallen fruits, seeds, or roots (Ghiglieri et al., 1982; Ngcobo et al., 2019). Bushpig's association with higher canopy cover coincides with findings by Ramesh and Downs (2015), who also found that the species had higher probability of site use in areas with more canopy cover.

Interestingly, the probability of site use of all the species included in the probability of site use model (except for otters), declined with an increase in ground vegetation cover in winter, possibly reflecting greater movement or detectability in more open areas when dense vegetation could restrict mobility. Bushbuck and porcupine had a weaker negative association than the other species (see Figure 4.8). Otters had higher probability of site use associated with moderate vegetation density and canopy cover. Their positive association with ground vegetation cover during winter reflects the species' general preference for thick vegetation (Arden-Clarke, 1986; Perrin and Carugati, 2000; Somers and Nel, 2007). Dense cover provides shelter, resting sites, and concealment from predators (Perrin and Carugati, 2000).

Trees are important for baboons as sleeping spots (Hamilton III, 1986; Hoffman and O'Riain, 2012). However, for baboons in the winter, canopy cover and vegetation density did not play a big role in their occurrence (shown in the declines in probability of site use as canopy cover and ground vegetation cover increases in Figure 4.15 and 4.17). However, these covariates were more important for probability of site use in summer, reflecting the patterns seen in the other species. Their omnivorous diet and tendency to have large home ranges (Slater et al., 2018), also possibly explain why they use a wider variety of habitats in winter especially.

In summer, all species were correlated with both high canopy cover and high ground vegetation cover, although the general increase in probability of site use with an increase in canopy cover was weaker than with an increase in ground vegetation cover. This pattern is consistent with the broader summer trend described earlier. This is possibly because of hot and dry conditions, meaning there is greater need for shade and cover. Additionally, proximity to the river provides shade, water, lush vegetation for herbivores and refuge in comparison with drier surrounding areas

(Xiang et al., 2017; Weinberger et al., 2019). A study regarding otters in Portugal showed that carnivore occurrence in riparian zones was closely linked to water channel type and standing water availability, supporting the idea that water resources facilitate persistence during hot, dry periods (Santos et al., 2011). This also aligns with the finding that several species (like bushbuck, bushpig and otter) maintained or increased probability of site use in summer despite lower detection probabilities, supporting the idea that riparian areas is a refuge in hotter, drier seasons through access to water and prey (Moun et al., 2024). Winter being wetter and cooler, means that limitation is removed and species are more likely to have a more variable response to covariates.

5.8.2 The effect of water quality covariates on species probability of site use

As mentioned before, semi-aquatic carnivores because of their trophic role are known to be sensitive to disturbances in freshwater quality (Kubheka et al., 2012; Holland et al., 2019a; Brand et al., 2020; Baos et al., 2022; Whipkey et al., 2025). Otters, as higher-order predators, and water mongooses, which feed primarily on aquatic prey, depend on healthy riparian systems, make water-quality covariates (dissolved oxygen, conductivity, and SASS) particularly relevant to their probability of site use. As discussed in section 5.2, water quality covariates showed mixed effects across species and seasons. SASS generally had a slight negative effect on probability of site use, especially in summer.

African clawless otter showed a slightly increase in predicted probability of site use when SASS scores increased in winter at both areas, but had a weaker association with SASS in summer with predicted probability of site use decreasing slightly with an increase in SASS scores, especially at Grootvadesrbosch river, echoing the patterns seen in the other species.

This shows that in this area otters do not necessarily prefer more pristine rivers. Otters also showed inconsistent patterns in their relationship with physicochemical parameters: they were weakly negatively associated with dissolved oxygen and conductivity in winter but showed a strong positive association with dissolved oxygen, although predicted probability of site use decreased at the uppermost values, which can be expected in a threshold response.

Otters also showed a slight negative association with conductivity in summer in Buffeljags River.

This suggests that otters may have a slightly stronger response to short-term physicochemical conditions than macroinvertebrate communities. This aligns with Holland (2016), who found no clear relationship between river otter presence and macroinvertebrate communities, suggesting that prey availability and structural habitat features might be more influential than biological water-quality indices.

In contrast to African clawless otters, the probability of site use of water mongooses showed clearer, more consistent patterns with water quality covariates. They were positively associated with dissolved oxygen and negatively with conductivity in both seasons (Figure 4.8) which was also supported by response curves which indicated a flat response in winter and a slight decrease in probability of site use in summer as conductivity increased and vice versa for dissolved oxygen. This indicates possible greater sensitivity to good water quality (based on physicochemical covariates), than otters. However, like otters, probability of site use was negatively associated with SASS in both seasons, with the strongest negative effect in summer, especially at Grootvadersbosch River, with their probability of site use declining strongly with an increase in SASS scores. This is possibly because of the fact that higher order predators (such as African clawless otters) have a delayed response to changes in the lower trophic levels (Sherley et al., 2022). Therefore, changes in environmental conditions do not reflect in the populations of higher order predators immediately.

Although water-quality covariates were included primarily to assess their influence on otters and water mongooses, healthier water conditions may indirectly benefit all riparian mammals because they reflect intact riparian zones (Maestas et al., 2023; Gu and Li, 2024). Therefore, some species, strong associations with dissolved oxygen, conductivity, or SASS likely reflect indirect habitat proxy effects rather than direct dependence on river water quality (Gu and Li, 2024).

Among the more generalist species, water-quality covariates played a weaker or more variable role in shaping habitat use. Bushbuck and baboons showed only mild responses to dissolved oxygen and SASS, consistent with their broad ecological tolerance and weaker dependence on riparian quality (Brock et al., 2003; Grebe et al., 2024). For baboons in particular, water availability influences distribution mainly in drier systems (Hamilton III, 1986), so it is unsurprising that

physicochemical parameters had only modest effects in these relatively wet catchments. Bushpigs, by contrast, showed consistently strong negative associations with high conductivity and low SASS in both seasons, suggesting some sensitivity to degraded riparian conditions, while their response to dissolved oxygen shifted from neutral in winter to positive in summer. Cape genets displayed a mixed pattern: generally positive associations with dissolved oxygen, weak negative or flat associations with conductivity and weakly negative or flat responses to SASS, indicating that they may benefit from well-oxygenated sites but are not tightly linked to macroinvertebrate community condition. Cape porcupines, another generalist species, showed strong positive associations with dissolved oxygen in both seasons, while SASS and conductivity effects varied between seasons. Porcupines did show a decline with higher SASS scores in Grootvaderbosch river in summer but flat response to SASS at this site in winter and both rivers in other seasons and similarly had a weak negative response to conductivity in summer at Grootvadersbosch but showed more flat curves at both rivers in winter and in summer at Buffeljags River. Their wide habitat tolerance and flexible space use (Skinner and Chimimba, 2005; Ngcobo et al., 2019), likely explain these weaker and inconsistent water-quality relationships. Overall, the generalists tended to respond less strongly or less consistently to water-quality metrics than otters and water mongooses, reflecting their ability to occupy a broader range of riparian conditions.

5.8.3 Relationships among water-quality variables

The contrasting responses of the species in this study to dissolved oxygen, conductivity, and SASS are possibly because these measures reflect conditions over different time periods. Dissolved oxygen and conductivity fluctuate over short time scales (days to weeks) with flow, rainfall, and water temperature (Schäfer et al., 2021) whereas SASS reflects longer-term biological conditions and is more indicative of river condition overall (Dickens and Graham, 2002). Positive associations with dissolved oxygen and low conductivity therefore likely reflect favourable short-term conditions (e.g. higher flow, cooler water, flushing of pollutants). Seasonal water dynamics reinforce this: higher winter rainfall increases dissolved oxygen, reduces conductivity, and improves short-term water quality,

which aligns with the stronger winter effect of dissolved oxygen across several species.

Physicochemical parameters influence macroinvertebrate communities and are thus linked to SASS scores (Gordon et al., 2015; Makumbe et al., 2022; Fentaw et al., 2024). Conductivity strongly influences SASS scores. Gordon et al. (2015) showed that conductivity is a key driver of variation in SASS scores and taxa richness, while turbidity is more closely associated with ASPT. High conductivity often reflects nutrient enrichment from catchment runoff, which can affect macroinvertebrate communities (Gafner and Robinson, 2007).

However, in the current study, the expected negative relationship between conductivity and dissolved oxygen was not consistently observed (see Table 4.3). As a result, sites with high conductivity did not always exhibit low dissolved oxygen levels, reflecting the variable responses across sites and species.

Another reason why some sites showed good short-term water chemistry (high dissolved oxygen and low conductivity) but poor condition of macroinvertebrate communities (low SASS), can be because physicochemical conditions can be expected to recover faster than macroinvertebrate communities after disturbance. Physicochemical conditions can shift rapidly after river flow restoration (Foley et al., 2015) and although macroinvertebrates are resilient to a certain extent to disturbances (Jesus and Monteiro, 2022; Marino et al., 2024), these communities may lag behind as it takes time for species to return and for the community to re-establish its balance (Muehlbauer et al., 2011). In fact, studies show that macroinvertebrate community structure may take years to return to pre-disturbance states, particularly in river systems without many refuges (Asmamaw et al., 2021; Marino et al., 2024).

5.8.4 The influence of riparian vegetation on macroinvertebrate communities

As mentioned earlier, vegetation structure also influences macroinvertebrate communities. In the current study, vegetation covariates (canopy and ground cover) were only weakly correlated with SASS (see Tables A1 and A2). Overall, these findings suggest that macroinvertebrate-based water-quality indices may not fully reflect the structural habitat conditions relevant to mammal distribution in this system. indicating these structural measures do not fully capture biological water-

quality conditions—a pattern consistent with Holland (2016), who found that canopy cover was not a major driver of macroinvertebrate communities. High SASS did not consistently correspond to structurally complex riparian vegetation, which can contribute to mismatches between riparian mammal occupancy/probability of site use and macroinvertebrate metrics. This is also supported by findings that local riparian vegetation effects can be overridden by larger-scale catchment stressors (Palt et al., 2022).

5.8.5 African Clawless otters as sentinel species

Regarding African clawless otters as a potential sentinel species, there was no significant correlation between water quality and otter probability of site use in this study. In fact, otters were more abundant in areas with slightly lower water quality (Buffeljags River sites). This highlights the species' adaptability, consistent with observations by Okes and O'Riain (2017). Research showed that riparian habitat heterogeneity and forest composition strongly affect otter persistence, suggesting that even in areas with moderate water quality, well-structured riparian vegetation can support otter activity (Moun et al. (2024). However, other research indicates that otter occupancy/probability of site use is not always well-predicted by habitat-based models. For example, Holland et al. (2019b) reported weak and uncertain relationships between otters and environmental gradients, with prey availability playing a stronger role. Similar findings from Europe show that canopy cover and disturbance often shape otter habitat suitability (Martin-Collado et al., 2020), yet otters sometimes occupy degraded habitats because of population expansion or territorial constraints (Delibes et al., 2009; Couturier et al., 2023). These patterns may also reflect a 'buffer effect', where individuals use suboptimal sites when prime habitat is saturated. Together, these nuances suggest that African clawless otters, like Eurasian otters (*Lutra lutra),* should be used cautiously as indicators of freshwater ecosystem health (Madsen and Prang, 2001; Delibes et al., 2009; Reid et al., 2014). Their distribution reflects a balance between prey availability, refuge from human disturbance, and habitat connectivity, and they may occupy marginal or urban areas depending on whether they are moving through corridors or seeking foraging sites (Okes and O'Riain, 2017; Ponsonby et al., 2019; Martin-Collado et al., 2020).

5.8.6 Summary of vegetation and water quality influences on riparian mammals

Overall, these findings suggest that vegetation structure and water quality jointly influence the spatial and seasonal distribution of riparian mammals. The strong responses to canopy and ground vegetation cover highlight the importance of maintaining riparian vegetation buffers (Graziano et al., 2022).

The role of riparian habitats as refuges during hotter months—providing cover and access to water and food resources—was specifically clear in this study. These patterns are consistent with other studies in semi-arid riparian systems, where species adjust habitat use seasonally to balance finding food and thermal regulation (Santos et al., 2011; Moun et al., 2024).

5.9 Differences in probability of site use between areas

Having established the influence of water-quality and structural habitat covariates on species probability of site use, this section examines how these patterns translate into differences in probability of site use probabilities between the Buffeljags and Grootvadersbosch Rivers.

In winter, most species showed higher probability of site use at Grootvadersbosch River, while in summer this trend reversed for several species, including bushbuck, baboon, and water mongoose. Overall, probability of site use in summer were higher at Buffeljags River. This seasonal difference may reflect increased activity or resource availability during warmer months or possibly result from lower camera effort at Grootvadersbosch in summer. In winter, probability of site use was generally higher at Grootvadersbosch for most species, except for Cape genet and Chacma baboon, which were more common at Buffeljags.

These area-specific differences likely relate to variation in riparian structure and local habitat conditions between the two river systems. Grootvadersbosch tends to have denser vegetation and more continuous forest cover, which may provide suitable shelter and foraging opportunities for species such as water mongoose, Cape porcupine, and Cape bushbuck. In contrast, Buffeljags River includes more open and heterogeneous riparian stretches, which may better suit generalist species like baboon and genet. In summer, baboon, water mongoose, and bushbuck maintained the highest probability of site use in both areas, while

otter and genet showed lower probability of site use at Grootvadersbosch, possibly because of local habitat or prey availability. Otter and porcupine also had the lowest probability of site use at Buffeljags River. Bushbuck, baboon, and genet showed lower uncertainty in probability of site use estimates, suggesting these species were consistently detected and more adaptable across riparian habitats.

Detection probabilities were substantially higher in winter than in summer, likely because of increased species activity in cooler conditions and the influence of higher camera effort. This indicates that apparent differences in site use during winter are more reliable, as imperfect detection had less influence on probability of site use estimates. In contrast, during summer, detection probabilities were uniformly low across species and areas, suggesting either reduced activity in warmer months or decreased detectability because ofseasonal behaviour (e.g., reduced movement in response to heat) (Bonebrake et al., 2020).

5.10 Limitations

Higher probability of site use estimates in summer may partly reflect low detection probabilities rather than true increases in site use (Emmet et al., 2021). Low detection probabilities, particularly for small or nocturnal carnivores, limit the precision of probability of site use estimates. The relatively short sampling period per season and unequal camera effort across sites may also have influenced detectability (Burton et al., 2015). When detections are sparse, the model accounts for possible missed detections, which can inflate probability of site use values (Neilson et al., 2018). Consequently, observed differences in probability of site use between winter and summer should be interpreted cautiously, as they may reflect detectability patterns rather than true changes in site use.

In this study two cameras were stolen, this reflects a general problem in camera trap studies (Oladimeji et al., 2024). Additionally, a few cameras were lost or damaged because of flooding, emphasizing the importance of preparing for adverse weather conditions. The loss of these cameras resulted in the loss of valuable sampling data, which may have affected detection estimates.

Missing data from undeployed cameras in the Grootvadersbosch summer survey reduced sampling effort across sites for that season. Combined with low detection rates for several species, this limited the feasibility of fitting a dynamic

multispecies occupancy model (Osinga et al., 2025). The data were therefore analysed separately by season, allowing descriptive but not formal statistical comparisons of seasonal patterns.

Camera trap sites were located approximately 200 m apart, which reduced spatial independence between sites as required by occupancy models and potentially inflated occupancy estimates (MacKenzie et al., 2002; 2004, Rota et al., 2009). This is why occupancy results was discussed as probability of site use rather than occupancy (DiRenzo et al., 2022).

Some short-term fluctuations in survey and site conditions, such as rainfall and river flow, were not explicitly modelled, and habitat and water-quality covariates were collected after the camera trap surveys (i.e. SASS data were collected either before or after), which could influence occupancy modelling results since it is unmodelled heterogeneity (MacKenzie et al., 2002). Use of survey effort as the sole detection covariate limited the amount of variation in detection probability captured in the occupancy models (MacKenzie and Bailey, 2004). Additional detection covariates such as distance from road; distance from houses/buildings; rainfall and distance from river could have helped to account for differences in detection probability (Niedballa et al., 2015).

Although the models identified specific patterns in probability of site use and detection probability, the results should be viewed as indicative rather than definitive of population status. Monitoring over multiple years would be needed to confirm long-term trends (Lindenmayer et al., 2022).

Future research should investigate additional drivers of occupancy/probability of site use, such as prey availability and diet composition, especially for semi-aquatic species such as the African clawless otter and water mongoose. These factors may exert stronger influences on habitat use than water-quality variables (Perrin and Carugati, 2000; Somers and Nel, 2007; Do Linh San et al., 2020; Streicher et al., 2021).

Although riparian mammals have been proposed as potential sentinels of freshwater ecosystem health, the present study found limited evidence for consistent responses to water-quality variables across species. This suggests that, while certain semi-aquatic mammals (e.g., the African clawless otter and water mongoose) may reflect river health to some extent, most species are likely influenced more strongly by vegetation structure than by water quality. Riparian

mammals may therefore rather act as complementary indicators of the integrity of riparian zones rather than direct sentinels of water quality.

Chapter 6: Conclusions

This study aimed to improve understanding of how environmental variables influence the probability of site use and detection probabilities of riparian mammals and to assess their potential as sentinels of riparian ecosystem condition. It focused on two rivers in the Western Cape (the Buffeljags and Grootvadersbosch Rivers) examining the effects of habitat structure, water quality, and seasonal variation on species occurrence.

Overall, probability of site use patterns were species- and season-specific, reflecting the combined effects of vegetation structure, water quality, and variation in habitat and site characteristics between rivers. Semi-aquatic species such as African clawless otters and water mongooses showed contrasting seasonal patterns, with otters exhibiting lower summer probability of site use and water mongooses slightly higher, while all the other generalist species, except for Cape porcupine tended to maintain or increase probability of site use in summer.

Community-level probability of site use was higher in summer, highlighting the role of riparian zones as refuges during hotter, drier periods. Structural habitat features—particularly canopy and ground vegetation cover—emerged as important drivers of probability of site use, especially in summer, providing shelter, foraging opportunities, and thermal protection. Water-quality covariates (dissolved oxygen, conductivity, and SASS scores) showed mixed effects, most species responded weakly or inconsistently. However, even semi-aquatic mammals, especially African clawless otters, did not consistently respond positively to water chemistry or SASS scores, suggesting that riparian vegetation and structural habitat features were more important drivers of probability of site use. In addition, prey availability—an important determinant of habitat use—may have played a stronger role, although it was not assessed in this study.

Area-specific differences between the two rivers modestly influenced probability of site use patterns. Both areas were farms with fragments of natural area. Grootvadersbosch River sites were slightly more forested and continuous, while Buffeljags River sites were more heterogeneous and fragmented. These subtle differences in habitat structure may have contributed to slightly higher probability of site use for some species in each river, although seasonal variation also played a role in shaping these patterns.

Limitations of the study included low detection probabilities, uneven camera effort, short sampling periods, missing data from undeployed or lost cameras, and habitat or water-quality covariates collected after the camera trap surveys. Short-term environmental fluctuations such as rainfall and river flow were not directly measured, which may have influenced species activity and probability of site use estimates. Consequently, the results should be interpreted as indicative of habitat use rather than absolute population status.

In conclusion, riparian mammals exhibit nuanced, species-specific responses to habitat and water-quality conditions, with vegetation cover playing a particularly important role during the dry season. While certain semi-aquatic mammals may reflect short-term changes in water quality, most species appear more influenced by riparian habitat integrity, suggesting that these mammals are better considered complementary indicators of riparian ecosystem condition rather than direct sentinels of freshwater quality. Future studies incorporating monitoring over multiple years, better fine scale measurement of environmental covariates, as well as the addition of prey availability assessments will help clarify the drivers of habitat use and refine the use of riparian mammals as bioindicators in riparian zones.

References

Abdullah, S., Mohd Sidek, L., Sherjeel, M., Mustafa, M. and Kurniawan, T.A. (2025). Seasonal variations in water quality and hydrological dynamics in a tropical reservoir driven by rainfall, runoff, and anthropogenic activities. *Scientific Reports*, 15(1). https://doi.org/10.1038/s41598-025-09808-z.

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S.C., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E. and Higgins, J.V. (2008). Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. *BioScience*, 58(5), pp.403–414. https://doi.org/10.1641/b580507.

Agboola, O.A., Downs, C.T. and O'Brien, G. (2019). Macroinvertebrates as indicators of ecological conditions in the rivers of KwaZulu-Natal, South Africa. *Ecological Indicators*, 106, 105465. https://doi.org/10.1016/j.ecolind.2019.105465

Agboola, O.A., Downs, C.T. and O'Brien, G. (2020). A multivariate approach to the selection and validation of reference conditions in KwaZulu-Natal rivers, South Africa. *Frontiers in Environmental Science*, 8, 584923. https://doi.org/10.3389/fenvs.2020.584923

Alemu, T., Bahrndorff, S., Hundera, K., Alemayehu, E. and Ambelu, A. (2017). Effect of riparian land use on environmental conditions and riparian vegetation in the east African highland streams. *Limnologica*, 66, pp.1–11. https://doi.org/10.1016/j.limno.2017.07.001

Anderson, D. (2003). *Response to engeman: index values rarely constitute reliable information on jstor*. Jstor.org. [Online]. Available at: https://www.jstor.org/stable/3784387 [Accessed 14 September 2024].

Anile, S. and Devillard, S. (2015). Study design and body mass influence RAIs from camera trap studies: evidence from the Felidae. *Animal Conservation*, 19(1), pp.35–45. https://doi.org/10.1111/acv.12214

Anthony, R.G., O'Connell, M.A., Pollock, M.M. and Hallett, J.G. (2003). Associations of mammals with riparian ecosystems in Pacific Northwest forests. In: C.J. Zabel and R.G. Anthony (eds.), *Mammal Community Dynamics: Management and Conservation in the Coniferous Forests of Western North America*. Cambridge:

Cambridge University Press, pp. 510–563. https://doi.org/10.1017/CBO9780511615757.016

Arden-Clarke, C.H.G. (1986). Population density, home range size and spatial organization of the African clawless otter, *Aonyx capensis,* in a marine habitat. *Journal of Zoology*, 209(2), pp.201–211. https://doi.org/10.1111/j.1469-7998.1986.tb03576.x

Apps, P. and Smithers, R.H.N. (2012). *Smithers' Mammals of Southern Africa: A Field Guide*. 4th ed. Cape Town: Penguin Random House South Africa

Ashton, P.J. (2007). The role of good governance in sustainable development: implications for integrated water resource management in southern Africa. *Water Resources Development and Management*, pp.77–100. https://doi.org/10.1007/978-3-540-46266-8_5

Asmamaw, M., Tiku Mereta, S. and Ambelu, A. (2021). Response of macroinvertebrates to changes in stream flow and habitat conditions in Dinki watershed, central highlands of Ethiopia. *Ecological Indicators*, 133, 108448. https://doi.org/10.1016/j.ecolind.2021.108448

Avenant, N.L., Drouilly, M., Power, R.J., Thorn, M., Martins, Q., Neils, A., du Plessis, J. and Do Linh San, E. (2016). *A conservation assessment of Caracal caracal*. In M.F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo and H.T. Davies-Mostert (eds.), *The 2016 Red List of Mammals of South Africa, Swaziland and Lesotho*. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. [Online]. Available: https://capeleopard.org.za/images/docs/publications/2016_Mammal_Red_List_Caracal%20caracal_LC.pdf [Accessed 20 September 2025].

Bailey, L.L., Simons, T.R. and Pollock, K.H. (2004). Estimating site occupancy and species detection probability parameters for terrestrial salamanders. *Ecological Applications*, 14(3), pp.692–702. https://doi.org/10.1890/03-5012

Bailey, L. and Adams, M.J. (2005). Occupancy models to study wildlife. *Fact Sheet*. https://doi.org/10.3133/fs20053096

Bailey, L.L., MacKenzie, D.I. and Nichols, J.D. (2013). Advances and applications of occupancy models. *Methods in Ecology and Evolution*, 5(12), pp.1269–1279. https://doi.org/10.1111/2041-210x.12100

Baker, C., Stuart, C., Stuart, M., Nqinana, A. and Do Linh San, E. (2016). A conservation assessment of *Atilax paludinosus*. In: Child, M.F., Roxburgh, L., Do Linh San, E.,

Raimondo, D. and Davies-Mostert, H.T. (eds.) *The Red List of Mammals of South Africa, Swaziland and Lesotho*. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. [Online]. Available at: https://www.researchgate.net/publication/321875851_A_conservation_assessment_of_Atilax_paludinosus [Accessed 23 April 2025].

Báldi, A. and Sadler, J. (2008). Habitat heterogeneity overrides the species-area relationship. *Journal of Biogeography*, 35(4), pp.675–681. [Online]. Available at: https://www.jstor.org/stable/30054691 [Accessed 7 June 2026].

Ballash, G.A., Baesu, A., Lee, S., Mills, M.C., Mollenkopf, D.F., Sullivan, S.M.P., Lee, J., Bayen, S. and Wittum, T.E. (2022). Fish as sentinels of antimicrobial resistant bacteria, epidemic carbapenemase genes, and antibiotics in surface water. *PLOS ONE*, 17(9), e0272806. https://doi.org/10.1371/journal.pone.0272806

Baos, R., Cabezas, S., González, M.J., Jiménez, B. and Delibes, M. (2022). *Eurasian otter (Lutra lutra)* as sentinel species for the long-term biomonitoring of the Guadiamar River after the Aznalcóllar mine spill. *Science of The Total Environment*, 802, p.149669. https://doi.org/10.1016/j.scitotenv.2021.149669

Bashir, I., Lone, F.A., Bhat, R.A., Mir, S.A., Dar, Z.A. and Dar, S.A. (2020). Concerns and threats of contamination on aquatic ecosystems. *Bioremediation and Biotechnology*, 1(1), pp.1–26. https://doi.org/10.1007/978-3-030-35691-0_1

Basile, M., Storch, I. and Mikusiński, G. (2021). Abundance, species richness and diversity of forest bird assemblages – The relative importance of habitat structures and landscape context. *Ecological Indicators*, 133, 108402. https://doi.org/10.1016/j.ecolind.2021.108402

Bates, B., Wu, S., Zbigniew Kundzewicz, A.W. and Palutikof, J. (2008). *Climate change and water intergovernmental panel on climate change IPCC technical paper VI*. [Online] Available at: https://archive.ipcc.ch/pdf/technical-papers/climate-change-water-en.pdf [Accessed 20 September 2025].

Bazzaz, F.A. (1975). Plant species diversity in old field successional ecosystems in southern Illinois. *Ecology*, 56(2), pp.485–488. https://doi.org/10.2307/1934981

Bedford, S.J. (2009). The effects of riparian habitat quality and biological water quality on the European Otter *(Lutra lutra*) in Devon. *Bioscience Horizons*, 2(2), pp.125–133. https://doi.org/10.1093/biohorizons/hzp015

Beeby, A. (2001). What do sentinels stand for? *Environmental Pollution*, 112(2), pp.285–298. https://doi.org/10.1016/s0269-7491(00)00038-5

Bega, S. (2024). *SA's rivers and dams can no longer recover from pollution, say water scientists*. [online] The Mail & Guardian. Available at: https://mg.co.za/the-green-guardian/2024-03-27-sas-rivers-and-dams-can-no-longer-recover-from-pollution-say-water-scientists/ [Accessed 5 September 2025].

Bernard, A., Moolman, L., De, M.A., Guerbois, C., Venter, J.A. and Fritz, H. (2023). Height-related detection bias in camera trap surveys: Insights for combining data sets. *Koedoe*, 65(1). https://doi.org/10.4102/koedoe.v65i1.1734

Bestelmeyer, B.T., Ellison, A.M., Fraser, W.D., Gorman, K.B., Holbrook, S.J., Laney, C., Ohman, M.D., Debra, Pillsbury, F.C., Rassweiler, A., Schmitt, R.J. and Sharma, S. (2011). Analysis of abrupt transitions in ecological systems. *Ecosphere*, 2(12), art129. https://doi.org/10.1890/es11-00216.1

Beukes, M., Perry, T., Parker, D., Beukes, O. and Mgqatsa, N. (2025). Biodiversity beyond protected areas: Mammal responses to land-use and vegetation in a semi-arid mixed-use landscape. *Biological Conservation*, 313, 111556. https://doi.org/10.1016/j.biocon.2025.111556

Birmachu, A.B., Feyissa, T.A. and Tukura, N.G. (2024). Assessment of effects of turbidity variation on water temperature and evaporation of Gilgel Gibe I Reservoir, Omo-Gibe River Basin, Ethiopia. *Journal of Engineering*, 2024, e8891990. https://doi.org/10.1155/2024/8891990

Blanchard, R. and Holmes, P.M. (2008). Riparian vegetation recovery after invasive alien tree clearance in the Fynbos Biome. *South African Journal of Botany*, 74(3), pp.421–431. https://doi.org/10.1016/j.sajb.2008.01.178

Bonebrake, T.C., Rezende, E.L. and Bozinovic, F. (2020). Climate change and thermoregulatory consequences of activity time in mammals. *The American Naturalist*, 196(1), pp.45–56. https://doi.org/10.1086/709010

Bonesi, L. and Macdonald, D.W. (2004). Impact of released Eurasian otters on a population of American mink: a test using an experimental approach. *Oikos*, 106(1), pp.9–18. https://doi.org/10.1111/j.0030-1299.2004.13138.x

Boron, V., Deere, N.J., Xofis, P., Link, A., Quiñones-Guerrero, A., Payan, E. and Tzanopoulos, J. (2019). Richness, diversity, and factors influencing occupancy of mammal communities across human-modified landscapes in Colombia. *Biological Conservation*, 232, pp.108–116. https://doi.org/10.1016/j.biocon.2019.01.030

Bossart, G.D. (1999). The Florida manatee: on the verge of extinction? *Journal of the American Veterinary Medical Association*, 214(8), pp.1178–1183. [Online]. Available at: https://pubmed.ncbi.nlm.nih.gov/10212678/ [Accessed 22 May 2024].

Bossart, G.D. (2006). Marine mammals as sentinel species for oceans and human health. *Oceanography*, 19(2), pp.134–137. https://doi.org/10.5670/oceanog.2006.77

Botha, N. (2019). Agriculture vs. conservation: how Grootvadersbosch Conservancy finds the common ground. *South African Geographical Journal*, 102(3), pp.1–17. https://doi.org/10.1080/03736245.2019.1694970

Bowler, M.T., Tobler, M.W., Endress, B.A., Gilmore, M.P. and Anderson, M.J. (2016). Estimating mammalian species richness and occupancy in tropical forest canopies with arboreal camera traps. *Remote Sensing in Ecology and Conservation*, 3(3), pp.146–157. https://doi.org/10.1002/rse2.35

Boyle, S.A., de la Sancha, N.U., Pérez, P. and Kabelik, D. (2021). Small mammal glucocorticoid concentrations vary with forest fragment size, trap type, and mammal taxa in the Interior Atlantic Forest. *Scientific Reports*, 11(1), 2068. https://doi.org/10.1038/s41598-021-81073-2

Brancalion, P.H., Melo, F.P., Tabarelli, M. and Rodrigues, R.R. (2013). Biodiversity persistence in highly human-modified tropical landscapes depends on ecological restoration. *Tropical Conservation Science*, 6(6), pp.705–710. [Online] Available at: https://journals.sagepub.com/doi/pdf/10.1177/194008291300600601 [Accessed 24 April 2024].

Bragg, C.J. and Child, M.F. (2016). A conservation assessment of *Hystrix africaeaustralis*. In: Child, M.F., Roxburgh, L., Do Linh San, E., Raimondo, D. and Davies-Mostert, H.T. (eds.) *The 2016 Red List of Mammals of South Africa, Swaziland and Lesotho.* South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. [Online]. Available at: https://ewt.org/wp-content/uploads/2022/11/29.-Cape-Porcupine-Hystrix-africaeaustralis_LC.pdf [Accessed 22 January 2026].

Bragg, C.J., Donaldson, J.D. and Ryan, P.G. (2005). Density of Cape porcupines in a semi-arid environment and their impact on soil turnover and related ecosystem processes. *Journal of Arid Environments*, 61(2), pp.261–275. https://doi.org/10.1016/j.jaridenv.2004.09.007 \\

Brand, A.F., Hynes, J., Walker, L.A., Pereira, M.G., Lawlor, A.J., Williams, R.J., Shore, R.F. and Chadwick, E.A. (2020). Biological and anthropogenic predictors of metal concentration in the Eurasian otter, a sentinel of freshwater ecosystems. *Environmental Pollution*, 266, 115280. https://doi.org/10.1016/j.envpol.2020.115280

Breede-Overberg Catchment Management Agency. (n.d.). *Breede-Overberg Catchment Management Strategy*. Breede-Gouritz Catchment Management Agency. [Online]. Available: https://www.breedegouritzcma.co.za/pdfs/governance/bgcma_gov_catchment-management-strategy-1.pdf [Accessed 25 November 2025].

Brock, A.P., Nortje, E. and Gaigher, I. (2003). Habitat preference of a local population of bushbuck in the Soutpansberg Mountains, South Africa. *South African Journal of Wildlife Research*, 33(2), pp.131–135. [Online]. Available at: https://www.researchgate.net/publication/281538564_Habitat_preference_of_a_local_population_of_bushbuck_in_the_Southpansberg_mountains_South_Africa [Accessed 2 June 2025].

Broekhuis, F., Mwiu, S.N., Liefting, Y., Sankale, P., Meiponyi, P., Tenke, M. and Western, G. (2025). Location, location, location: Habitat-specific differences of camera trap placement on species detections and capture rates. *Global Ecology and Conservation*, 62, e03819. https://doi.org/10.1016/j.gecco.2025.e03819.Brouwer, N. (2021). 9.2: Population Ecology Research Methods. *Biology LibreTexts*, [Online]. Available at: https://bio.libretexts.org/Courses/Gettysburg_College/01%3A_Ecology_for_All/09%3A_The_Ecology_of_Populations/9.02%3A_Population_Ecology_Research_Methods [Accessed 24 May 2026].

Bulto, T.W., Alemu, T., Werku, B.C. and Ambelu, A. (2026). Effects of riparian vegetation on water quality and macro invertebrates in the Gilgel Gibe Tributaries, Southwest Ethiopia. *Applied Water Science*, 16(112). https://doi.org/10.1007/s13201-026-02794-1

Bunn, S.E. and Arthington, A.H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management*, [online] 30(4), pp.492–507. https://doi.org/10.1007/s00267-002-2737-0

Burnham, K.P. and Anderson, D.R. (2002). *Model selection and multimodel inference*. New York: Springer. https://doi.org/10.1007/b97636

Burton, A.C., Neilson, E., Moreira, D., Ladle, A., Steenweg, R., Fisher, J.T., Bayne, E. and Boutin, S. (2015). Review: Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. *Journal of Applied Ecology*, 52(3), pp.675–685. https://doi.org/10.1111/1365-2664.12432

Butler, J.R.A. and Hillman, C. (1995). African clawless otters (*Aonyx capensis*) as indicators of water pollution by heavy metals and pesticides in Zimbabwe. *IUCN Otter Specialist Group Bulletin*, 11, pp. 30–31. [Online]. Available at: https://www.iucnosgbull.org/Volume11/Butler_Hillman_1995.html [Accessed 20 October 2024].

Butynski, T.M. and De Jong, Y.A. (2019). *Chlorocebus pygerythrus*. *The IUCN Red List of Threatened Species*.

Bevilacqua, S., Ugland, K.I., Plicanti, A., Scuderi, D. and Terlizzi, A. (2017). An approach based on the total-species accumulation curve and higher taxon richness to estimate realistic upper limits in regional species richness. *Ecology and Evolution*, 8(1), pp.405–415. https://doi.org/10.1002/ece3.3570

CapeNature. (2020). *Biodiversity Capabilities: 2020 Western Cape State of Conservation Report*. CapeNature. https://www.capenature.co.za/uploads/files/CapeNature-2020-State-of-Conservation-Report.pdf [Accessed 26 August 2024].

Cape Nature (2023a). *Indigenous forests of the Western Cape - Capenature*. Capenature.co.za. [Online]. Available at: https://www.capenature.co.za/news/2023/indigenous-forests-of-the-western-cape. [Accessed 22 August 2024].

Cape Nature (2023b). *Western Cape 2023 State of Biodiversity Report*. [Online]. Available at: https://www.capenature.co.za/uploads/files/2023-CapeNature-Western-Cape-State-of-Biodiversity-Report_2024-06-19-094619_bkzs.pdf [Accessed 22 August 2024].

Cape Nature (2025). Western Cape State of Biodiversity 2025. [Online] Available at: https://www.capenature.co.za/uploads/files/2025-Western-Cape-State-of-Conservation-Report.pdf [Accessed 1 October 2025].

Capon, S.J. and Dowe, J.L. (2007). Diversity and dynamics of riparian vegetation. In: S. Lovett and P. Price, eds. *Principles for riparian lands management*, pp.15–320. Canberra: Land and Water Australia. [Online]. Available at: https://www.researchgate.net/publication/265567903_CHAPTER_2_Diversity_and_dynamics_of_riparian_vegetation [Accessed 11 March 2025].

Caravaggi, A., Banks, P.B., Burton, A.C., Finlay, C.M.V., Haswell, P.M., Hayward, M.W., Rowcliffe, M.J. and Wood, M.D. (2017). A review of camera trapping for conservation behaviour research. *Remote Sensing in Ecology and Conservation*, 3(3), pp.109–122. https://doi.org/10.1002/rse2.48

Caravaggi, A., Burton, A.C., Clark, D.A., Fisher, J.T., Grass, A., Green, S., Hobaiter, C., Hofmeester, T.R., Kalan, A.K., Rabaiotti, D. and Rivet, D. (2020). A review of factors to consider when using camera traps to study animal behavior to inform wildlife ecology and conservation. *Conservation Science and Practice*, 2(8), e239. https://doi.org/10.1111/csp2.239

Carbone, C., Christie, S., Conforti, K., Coulson, T., Franklin, N., Ginsberg, J.R., Griffiths, M., Holden, J., Kawanishi, K., Kinnaird, M., Laidlaw, R., Lynam, A., Macdonald, D.W., Martyr, D., McDougal, C., Nath, L., O'Brien, T., Seidensticker, J., Smith, D.J.L. and Sunquist, M. (2001). The use of photographic rates to estimate densities of tigers and other cryptic mammals. *Animal Conservation*, 4(1), pp.75–79. https://doi.org/10.1017/s1367943001001081

Carnie, T. (2023). *Official reports reveal massive scale of the South African water crisis*. Daily Maverick. [Online]. Available at: https://www.dailymaverick.co.za/article/2023-12-05-official-reports-reveal-massive-scale-of-the-waste-pollution-and-poorly-treated-water-crisis/.

Caro, T.M. and O'Doherty, G. (1999). On the use of surrogate species in conservation biology. *Conservation Biology*, 13(4), pp. 805–814. https://doi.org/10.1046/j.1523-1739.1999.98338.x

Catterall, C.P., Lynch, R.J. and Jansen, A. (2007). Riparian wildlife and habitats. In: S. Lovett and P. Price, eds. *Principles for riparian lands management*, pp.141–158. Canberra: Land and Water Australia. [Online] Available at: https://www.researchgate.net/publication/29466142_CHAPTER_8_Riparian_wildlife_and_habitats [Accessed 20 October 2024].

Cavaleri, M.A., Ostertag, R., Cordell, S. and Sack, L. (2014). Native trees show conservative water use relative to invasive trees: Results from a removal experiment in a Hawaiian wet forest. *Conservation Physiology*, 2(1), cou016. https://doi.org/10.1093/conphys/cou016

Cesarini, G. and Scalici, M. (2022). Riparian vegetation as a trap for plastic litter. *Environmental Pollution*, 292, 118410. https://doi.org/10.1016/j.envpol.2021.118410

Chamier, J., Schachtschneider, K., le Maitre, D.C., Ashton, P.J. and van Wilgen, B.W. (2012). Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. *Water SA*, 38(2), pp.345–356. [Online] Available at: https://www.scielo.org.za/scielo.php?script=sci_arttext&pid=S1816-79502012000200019 [Accessed 5 November 2024].

Chapman, D. (1996). *Water quality assessments: A guide to the use of biota, sediments and water in environmental monitoring*. 2nd ed. London: E & FN Spon. https://doi.org/10.1201/9781003062103

Chaves, M.L., Costa, J.L., Chainho, P., Costa, M.J. and Prat, N. (2006). Selection and validation of reference sites in small river basins. *Hydrobiologia*, 573(1), pp.133–154. https://doi.org/10.1007/s10750-006-0270-5

Chua, E.M., Wilson, S.P., Vink, S. and Flint, N. (2019). The influence of riparian vegetation on water quality in a mixed land use river basin. *River Research and Applications*, 35(3), pp.259–267. https://doi.org/10.1002/rra.3410

Cianfrani, C., Lay, G.L., Maiorano, L., Satizábal, H.F., Loy, A. and Guisan, A. (2011). Adapting global conservation strategies to climate change at the European scale: The otter as a flagship species. *Biological Conservation*, 144(8), pp.2068–2080. https://doi.org/10.1016/j.biocon.2011.03.027

Claridge, A.W., Mifsud, G., Dawson, J. and Saxon, M.J. (2004). Use of infrared digital cameras to investigate aspects of the social behaviour of cryptic species. *Wildlife Research*, 31(6), p.645-650. https://doi.org/10.1071/wr03072

Clark-Wolf, T.J., Holt, K.A., Johansson, E., Nisi, A.C., Rafiq, K., West, L., Boersma, P.D., Hazen, E.L., Moore, S.E. and Abrahms, B. (2024). The capacity of sentinel species to detect changes in environmental conditions and ecosystem structure. *Journal of Applied Ecology*. https://doi.org/10.1111/1365-2664.14660

Clusella-Trullas, S. and Garcia, R.A. (2017). Impacts of invasive plants on animal diversity in South Africa: A synthesis. *Bothalia*, 47(2). https://doi.org/10.4102/abc.v47i2.2166

Coates, G.D. and Downs, C.T. (2007). Population estimates of bushbuck (*Tragelaphus scriptus*) in valley thicket and coastal bushveld–grassland habitat. *South African Journal of Wildlife Research*, 37(1), pp.91–95. https://doi.org/10.3957/0379-4369-37.1.91

Coetzee, J.A. and Hill, M.P. (2011). The role of eutrophication in the biological control of water hyacinth, *Eichhornia crassipes,* in South Africa. *BioControl*, 57(2), pp.247–261. https://doi.org/10.1007/s10526-011-9426-y

Collins, K. and Du Toit, J.T. (2016). Population status and distribution modelling of the critically endangered riverine rabbit (*Bunolagus monticularis*). *African Journal of Ecology*, 54(2), pp.195–206. https://doi.org/10.1111/aje.12285

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C. and Likens, G.E. (2009). Controlling eutrophication: Nitrogen and phosphorus. *Science*, 323(5917), pp.1014–1015. https://doi.org/10.1126/science.1167755

Cooch, E.G. and White, G.C. (eds.) (2022). *Program MARK: A Gentle Introduction* (22nd ed.). [Online]. Available: http://www.phidot.org/software/mark/docs/book/ [Accessed 19 September 2025]

Couturier, T., Steinmetz, J., Defos, P., Marc, D., Trichet, E., Gomes, R. and Aurélien Besnard (2023). Intensive agriculture as the main limiting factor of the otter's return in southwest France. *Biological Conservation*, 279, 109927. https://doi.org/10.1016/j.biocon.2023.109927

Cove, M.V., Spínola, R.M., Jackson, V.L., Sàenz, J.C. and Chassot, O. (2013). Integrating occupancy modeling and camera-trap data to estimate medium and large mammal detection and richness in a Central American biological corridor. *Tropical Conservation Science*, 6(6), pp.781–795. https://doi.org/10.1177/194008291300600606

Coverdale, T.C. and Davies, A.B. (2023). Unravelling the relationship between plant diversity and vegetation structural complexity: A review and theoretical framework. *Journal of Ecology*, 111(7), pp.1378–1395. https://doi.org/10.1111/1365-2745.14068

Creecy, B. (2022). *Department of Forestry, Fisheries and the Environment*. [Online] Available at: https://www.gov.za/sites/default/files/gcis_document/202211/47526gon2747.pdf [Accessed 25 August 2024].

Crook, D.A. and Robertson, A.I. (1999). Relationships between riverine fish and woody debris: implications for lowland rivers. *Marine and Freshwater Research*, 50(8), p.941. https://doi.org/10.1071/mf99072

Crookes, D.J. (2023). Ecology, opportunity or threat? Drivers of the caracal (*Caracal caracal*) population decline in South Africa. *African Journal of Ecology*, 61(4). https://doi.org/10.1111/aje.13150

Crous, C.J., Jacobs, S.M. and Esler, K.J. (2011). Conserving our rivers: A meander along riparian zones. *Quest*, 7(4), pp.10–13. [Online]. Available: https://www.fabinet.up.ac.za/people/1124/Quest_Crous%20et%20al%202011.pdf [Accessed 25 April 2024]

Crowley, S.M., Hodder, D.P., Johnson, C.J. and Yates, D. (2018). Wildlife health indicators and mercury exposure: A case study of river otters (*Lontra canadensis*) in central British Columbia, Canada. *Ecological Indicators*, 89, pp.63–73. https://doi.org/10.1016/j.ecolind.2018.01.061

Cusack, J.J., Dickman, A.J., Rowcliffe, J.M., Carbone, C., Macdonald, D.W. and Coulson, T. (2015). Random versus game trail-based camera trap placement strategy for monitoring terrestrial mammal communities. *PLOS ONE*, 10(5), e0126373. https://doi.org/10.1371/journal.pone.0126373

D'Ammando, G., Caro, T., Oelze, V.M., Phillips, S., Sime, P., Stewart, F.A. and Piel, A.K. (2022). Ecological drivers of habitat use by meso mammals in a Miombo ecosystem in the Issa Valley, Tanzania. *Frontiers in Ecology and Evolution*, 10, 773568. https://doi.org/10.3389/fevo.2022.773568

Dail, D. and Madsen, L. (2010). Models for estimating abundance from repeated counts of an open metapopulation. *Biometrics*, 67(2), pp.577–587. https://doi.org/10.1111/j.1541-0420.2010.01465.x

Dallas, H. (2007). *River Health Programme: South African Scoring System (SASS) data interpretation guidelines prepared for: Institute of natural resources and Department of Water Affairs and Forestry*. [Online] Available at: http://safrass.com/reports/SASS%20Interpretation%20Guidelines.pdf

Dallas, H.F. and Day, J.A. (2004). The effect of water quality variables on aquatic ecosystems. *Water Research Commission, Freshwater Research Unit. Cape Town: University of Cape Town*.

Dallas, H.F. and Day, J.A. (2007). Natural variation in macroinvertebrate assemblages and the development of a biological banding system for interpreting bioassessment data—a preliminary evaluation using data from upland sites in the south-western Cape, South Africa. *Hydrobiologia*, 575(1), pp.231–244. https://doi.org/10.1007/s10750-006-0374-y.

Dallas, H.F. and Rivers-Moore, N. (2014). Ecological consequences of global climate change for freshwater ecosystems in South Africa. *South African Journal of Science*, 110(5/6), pp.01-11. https://doi.org/10.1590/sajs.2014/20130274

Dallas, H. and Rivers-Moore, N. (2019). *Environmental water temperature guidelines for perennial rivers in South Africa. Volume 2: A technical manual for setting water temperature targets.* Report No. TT 799/2/19. Water Research Commission, Pretoria. [Online]. Available at: https://frcsa.org.za/wp-content/uploads/2020/05/Dallas-Rivers_Moore-2019-TT-799-2-Technical-Manual.pdf [Accessed 19 August 2024].

Dallas, H. and Shelton, J. (2024). Biodiversity data at your fingertips: The Freshwater Biodiversity Information System. *South African Journal of Science*, 120(7/8), 18705. https://doi.org/10.17159/sajs.2024/18705

Dallas, H.F., Shelton, J.M., Paxton, B.R. and Weyl, O.L.F. (2017). *Freshwater fishes of the Cape Fold Ecoregion and climate change: Volume 1: research synthesis*. [Online]. Available at: https://www.researchgate.net/publication/320799494_Freshwater_Fishes_of_the_Cape_Fold_Ecoregion_and_Climate_Change_Volume_1_Synthesis_of_Research_Findings [Accessed 19 August 2024].

DiRenzo, G.V., Miller, D.A.W. and Grant, E.H.C. (2022). Ignoring species availability biases occupancy estimates in single-scale occupancy models. *Methods in Ecology and Evolution*, 13(8), pp.1790–1804. https://doi.org/10.1111/2041-210x.13881

Davidge, C. (1978). Ecology of baboons (*Papio ursinus*) at Cape Point. *African Zoology*, 13(2), pp.329–350. [Online] Available at: https://journals.co.za/doi/pdf/10.10520/AJA00445096_1996 [Accessed 24 November 2025].

Day, C.C., Westover, M.D., Hall, L.K., Larsen, R.T. and McMillan, B.R. (2016). Comparing direct and indirect methods to estimate detection rates and site use of a cryptic semi-aquatic carnivore. *Ecological Indicators*, 66, pp.230–234. https://doi.org/10.1016/j.ecolind.2016.01.039

Dayaram, A., Skowno, A.L., Driver, A., Sink, K., Van Deventer, H., Smith-Adao, L., Van Niekerk, L., Harris, L.R., Job, N. and Nel, J.L. (2021). *The South African national ecosystem classification system Handbook: First Edition.* [Online] Available at:

https://www.sanbi.org/wp-content/uploads/2024/06/2021_NECS-Handbook.pdf [Accessed 30 January 2025].

Decamps, H., Fortune, M., Gazelle, F. and Pautou, G. (1988). Historical influence of man on the riparian dynamics of a fluvial landscape. *Landscape Ecology*, **1**(3), pp.163–173. [Online]. Available: https://d1wqtxts1xzle7.cloudfront.net/39199929/54410cda0cf2a6a049a4872b-libre.pdf [Accessed 25 August 2024].

De Luca, D. and Mpunga, N. (2005). *Carnivores of the Udzungwa Mountains Presence, distributions and threats*. [Online]. Available at: https://www.easternarc.or.tz/wp-content/uploads/2020/08/Udz_Carnivores_DDL.pdf [Accessed 3 December 2024].

Deng, C., Daley, T. and Smith, A. (2015). Applications of species accumulation curves in large-scale biological data analysis. *Quantitative Biology*, 3(3), pp.135–144. https://doi.org/10.1007/s40484-015-0049-7

De Villiers, C., Driver, A., Clark, B., Euston-Brown, D., Day, L., Job, N., Helme, N., Holmes, P., Brownlie, S. and Rebelo, T. (2005). *Ecosystem guidelines for environmental assessment in the Western Cape*. [Online] Available at: https://www.greenagri.org.za/assets/documents-/Tips-Tools/Useful-Tips/Plants/ECOSYSTEM-GUIDELINES-FOR-ENVIRONMENTAL-ASSESSMENT-IN-THE-WESTERN-CAPE.pdf [Accessed 22 September 2025].

Delibes, M., Cabezas, S., Jiménez, B. and González, M.J. (2009). Animal decisions and conservation: the recolonization of a severely polluted river by the Eurasian otter. *Animal Conservation*, 12(5), pp.400–407. https://doi.org/10.1111/j.1469-1795.2009.00263.x

Delisle, Z.J., Flaherty, E.A., Nobbe, M.R., Wzientek, C.M. and Swihart, R.K. (2021). Next-generation camera trapping: systematic review of historic trends suggests keys to expanded research applications in ecology and conservation. *Frontiers in Ecology and Evolution*, 9, 617996. https://doi.org/10.3389/fevo.2021.617996

Department of Forestry, Fisheries and the Environment (2022). *National Environmental Management: Biodiversity Act, 2004. (Act no. 10 of 2004). The revised national list of ecosystems that are threatened and in need of protection*. [Online]. Available at: https://www.gov.za/sites/default/files/gcis_document/202211/47526gon2747.pdf.

Department of Water Affairs and Forestry, South Africa (2004). *Breede Water Management Area: Internal strategic perspective*. DWAF Report No P WMA 18/000/00/0304. [Online]. Available at:

https://www.dws.gov.za/Documents/Other/WMA/18/BreedeISPOct04full.pdf [Accessed 15 March 2024]

Department of Water and Sanitation (2011). *Rivers of the Breede Water Management Area: State of Rivers Report*. River Health Programme. [Online]. Available at: https://www.dws.gov.za/iwqs/rhp/state_of_rivers/WCape/Breede2011.pdf [Accessed 5 May 2024]

Department of Water and Sanitation (2016). *REMP (River Eco-status Monitoring Programme) history - formerly RHP (River Health Programme), RQIS (Resource Quality Information Services) - Department of Water and Sanitation - South Africa* -.www.dws.gov.za. [Online]. Available at: https://www.dws.gov.za/iwqs/rhp/rhp_background.aspx [Accessed 5 May 2024]

Department of Water and Sanitation. (2023). *Blue Drop Report 2023*. [Online]. Available: https://ws.dws.gov.za/IRIS/releases/BDWR.pdf [Accessed 14 August 2025).

Devarajan, K., Morelli, T.L. and Tenan, S. (2020). Multi-species occupancy models: review, roadmap, and recommendations. *Ecography*, 43(11), pp.1612–1624. https://doi.org/10.1111/ecog.04957

Dibner, R.R., Doak, D.F. and Murphy, M. (2017). Discrepancies in occupancy and abundance approaches to identifying and protecting habitat for an at-risk species. *Ecology and Evolution*, 7(15), pp.5692–5702. https://doi.org/10.1002/ece3.3131

Dickie, M., McNay, S.R., Sutherland, G.D., Cody, M. and Avgar, T. (2019). Corridors or risk? Movement along, and use of, linear features varies predictably among large mammal predator and prey species. *Journal of Animal Ecology*, 89(2), pp.623–634. https://doi.org/10.1111/1365-2656.13130

Dickens, C.W. and Graham, P.M. (2002). The South African Scoring System (SASS) Version 5 rapid bioassessment method for rivers. *African Journal of Aquatic Science*, 27(1), pp.1–10. https://doi.org/10.2989/16085914.2002.9626569

Dodds, W.K. and Oakes, R.M. (2007). Headwater influences on downstream water quality. *Environmental Management*, 41(3), pp.367–377. https://doi.org/10.1007/s00267-007-9033-y.

Dodds, W.K. and Whiles, M.R. (2010). *Freshwater Ecology: Concepts and Environmental Applications of Limnology*. 2nd ed. Amsterdam: Elsevier, pp.330–333.

Do Linh San, E., Nqinana, A., Madikiza, Z.J.K. and Somers, M.J., 2020. Diet of the marsh mongoose around a non-permanent reservoir: Response of a generalist

opportunist forager to the absence of crabs. *African Zoology*, *55*(3), pp.240-244. https://doi.org/10.1080/15627020.2020.1768145

Donnenfeld, Z., Crookes, C. and Hedden, S. (2018). A delicate balance: water scarcity in South Africa. *Southern Africa Report*, 13, pp. 1–20. [Online] Available at: https://journals.co.za/doi/10.10520/EJC-1486c3180a [Accessed 20 April 2024].

Donovan, T., Hines, J. and MacKenzie, D. (2024). OccupancyTuts: Occupancy modelling tutorials with RPresence. *Methods in Ecology and Evolution*, 15(3), pp.477–483. https://doi.org/10.1111/2041-210x.14285

Dorazio, R.M., Royle, J.A., Söderström, B. and Glimskär, A. (2006). Estimating species richness and accumulation by modeling species occurrence and detectability. *Ecology*, 87(4), pp.842–854. https://doi.org/10.1890/0012-9658(2006)87[842:esraab]2.0.co;2.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., McClean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell, D. and Lautenbach, S. (2012). Collinearity: A review of methods to deal with it and a simulation study evaluating their performance. *Ecography*, 36(1), pp.27–46. https://doi.org/10.1111/j.1600-0587.2012.07348.x

Doser, J.W., Finley, A.O., Kéry, M. and Zipkin, E.F. (2022). spOccupancy: An R package for single-species, multi-species, and integrated spatial occupancy models. *Methods in Ecology and Evolution*, 13(8), pp.1670–1678. https://doi.org/10.1111/2041-210x.13897

Doser, J.W. (2023). *Convergence diagnostics and other considerations when fitting spatial occupancy models*. Doserlab.com. [Online]. Available at: https://doserlab.com/files/spoccupancy-web/articles/modelconsiderations. [Accessed 20 May 2025].

Dosskey, M.G., Vidon, P., Gurwick, N.P., Allan, C.J., Duval, T.P. and Lowrance, R. (2022). The Role of Riparian Vegetation in Protecting and Improving Chemical Water Quality in Streams. *Journal of the American Water Resources Association 1-18*, 46(2), pp.1–17. [Online]. Available at: https://research.fs.usda.gov/treesearch/36859 [Accessed 4 June 2026].

Dube, R., Maphosa, B., Malan, A., Fayemiwo, D., Ramulondi, D. and Zuma, T. (2017). *Response of urban and peri-urban aquatic ecosystems to riparian zones, land uses and human settlements: A study of the rivers Jukskei, Kuils and Pienaars.*

Report to the Water Research Commission. [Online]. Available at: https://www.wrc.org.za/wp-content/uploads/mdocs/2339-1-17.pdf [Accessed 23 February 2026].

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J. and Sullivan, C.A. (2006). Freshwater biodiversity: Importance, threats, status and conservation challenges. *Biological Reviews*, 81(02), pp.163–182. https://doi.org/10.1017/s1464793105006950

Dugdale, S.J., Malcolm, I.A., Kantola, K. and Hannah, D.M. (2018). Stream temperature under contrasting riparian forest cover: Understanding thermal dynamics and heat exchange processes. *Science of The Total Environment*, 610-611, pp.1375–1389. https://doi.org/10.1016/j.scitotenv.2017.08.198

Dunlap, J. (2008). Aquatic ambassadors: River otters are poster pups for conservation. *The River Otter Journal*, XVII(II). [Online] Available at: https://www.riverotterjournal.org/publications/28_ROJ_Issue_Fall_2008.pdf [Accessed 20 May 2024].

Du Plessis, N.S., Rebelo, A.J., Richardson, D.M. and Esler, K.J. (2021). Guiding restoration of riparian ecosystems degraded by plant invasions: Insights from a complex social-ecological system in the Global South. *Ambio*, 51(6), pp.1552–1568. https://doi.org/10.1007/s13280-021-01691-y

Du Toit, M.J., Kotze, D.J. and Cilliers, S.S. (2020). Quantifying long-term urban grassland dynamics: Biotic homogenization and extinction debts. *Sustainability*, 12(5), p.1989. https://doi.org/10.3390/su12051989

Dye, P. and Jarmain, C. (2004). Water use by black wattle (*Acacia mearnsii*): Implications for the link between removal of invading trees and catchment streamflow response. *South African Journal of Science*, 100(1), pp.40–44. [Online] Available at: https://journals.co.za/doi/pdf/10.10520/EJC96215 [Accessed 20 May 2024].

Edwards, S., Noack, J., Heyns, L. and Rodenwoldt, D. (2020). Are camera traps a reliable method for estimating activity patterns? A case study comparing technologies for estimating brown hyaena activity curves. *Remote Sensing in Ecology and Conservation*, 7(2), pp.175–185. https://doi.org/10.1002/rse2.175

Efford, M.G. and Dawson, D.K. (2012). Occupancy in continuous habitat. *Ecosphere*, 3(4), art32. https://doi.org/10.1890/es11-00308.1

Ellender, B.R., Wasserman, R.J., Chakona, A., Skelton, P.H. and Weyl, O.L.F. (2017). A review of the biology and status of Cape Fold Ecoregion freshwater fishes. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27(4), pp.867–879. https://doi.org/10.1002/aqc.2730

Ellison, A.M. (2004). Bayesian inference in ecology. *Ecology Letters*, 7(6), pp.509–520. https://doi.org/10.1111/j.1461-0248.2004.00603.x

Emmet, R.L., Long, R.A. and Gardner, B. (2021). Modeling multi-scale occupancy for monitoring rare and highly mobile species. *Ecosphere*, 12(7), e03637. https://doi.org/10.1002/ecs2.3637

Ergete, E.A., Hailemariam, T.W., Balakrishnan, M. and Serfass, T.L. (2018). Fishermen knowledge and conflict with African clawless otters in and around Lake Tana, Ethiopia. *African Journal of Ecology*, 56(2), pp.409–413. https://doi.org/10.1111/aje.12447

Erickson, K.D. and Smith, A.B. (2023). Modeling the rarest of the rare: A comparison between multi-species distribution models, ensembles of small models, and single-species models at extremely low sample sizes. *Ecography*, 2023(6), pp. 1682–1694. https://doi.org/10.1111/ecog.06500

Erwin, R.M. (1982) Observer variability in estimating numbers: an experiment, *Journal of Field Ornithology*, 53(2), pp.159–167. [Online]. Available at: https://sora.unm.edu/sites/default/files/journals/jfo/v053n02/p0159-p0167.pdf [Accessed: 5 March 2025].

European Space Agency (ESA). (n.d.). *River discharge climate change initiative*. Available at: https://climate.esa.int/en/projects/river-discharge/ (Accessed: 25 February 2026).

Fathi, A. (2022) Role of nitrogen (N) in plant growth, photosynthetic pigments, and N use efficiency: A review, *Agrosistemas*, 28, pp. 1–8. https://doi.org/10.5281/zenodo.7143588.

Fentaw, G., Beneberu, G., Wondie, A. and Getnet, B. (2024). Composition and dynamics of macroinvertebrate communities in relation to physicochemical parameters of hydrogeologically connected wetlands in Abbay River Basin, Ethiopia. *PLoS ONE*, 19(12), e0314969. https://doi.org/10.1371/journal.pone.0314969

Ferreira, A.S., Peres, C.A., Dodonov, P. and Cassano, C.R. (2020). Multi-scale mammal responses to agroforestry landscapes in the Brazilian Atlantic Forest: the

conservation value of forest and traditional shade plantations. *Agroforestry Systems*, 94(6), pp.2331–2341. https://doi.org/10.1007/s10457-020-00553-y

Fiske, I. and Chandler, R. (2011). unmarked: An*R*Package for Fitting Hierarchical Models of Wildlife Occurrence and Abundance. *Journal of Statistical Software*, 43(10), pp.1–23. https://doi.org/10.18637/jss.v043.i10

Fleming, A.H., Clark, C.T., Calambokidis, J. and Barlow, J. (2016). Humpback whale diets respond to variance in ocean climate and ecosystem conditions in the California Current. *Global Change Biology*, 22(3), pp.1214–1224. https://doi.org/10.1111/gcb.13171.

Flemming, B. and Martin, K. (2021). The Breede River estuary (Cape Province, South Africa): A historical perspective on hydrology, geomorphology, and sedimentology. *Geo-Marine Letters*, 41(1), pp.53–66. https://doi.org/10.1007/s00367-021-00686-8

Foley, M.M., Duda, J.J., Beirne, M.M., Paradis, R., Ritchie, A. and Warrick, J.A. (2015). Rapid water quality change in the Elwha River estuary complex during dam removal. *Limnology and Oceanography*, 60(5), pp.1719–1732. https://doi.org/10.1002/lno.10129

Fonteyn, D., Vermeulen, C., Deflandre, N., Cornelis, D., Lhoest, S., Houngbégnon, F.G.A., Doucet, J. and Fayolle, A. (2020). Wildlife trail or systematic? Camera trap placement has little effect on estimates of mammal diversity in a tropical forest in Gabon. *Remote Sensing in Ecology and Conservation*, 7(2), pp.321–336. https://doi.org/10.1002/rse2.191

Foster-Turley, P., Macdonald, S.M. and Mason, C.F. (1990). *Otters: An action plan for their conservation*. Gland, Switzerland: IUCN. https://doi.org/10.2305/IUCN.CH.1990.SSC-AP.3.en

Freshwater Biodiversity Information System (FBIS). (2023). Data downloaded from https://freshwaterbiodiversity.org in May 2023.

Fuller, T.K., Biknevicius, A.R. and Kat, P.W. (1990). Movements and behavior of large spotted genets (*Genetta maculata* -Gray 1830) near Elmenteita Kenya (Mammalia Viverridae). *Tropical Zoology*, 3(1), pp.13–19. https://doi.org/10.1080/03946975.1990.10539446

Furstenburg, D. (2010). Focus on the bushbuck (*Tragelaphus scriptus*). *SA Hunter*, 02037. [Online]. Available at: https://www.researchgate.net/publication/316165634_Focus_on_the_Bushbuck_Tragelaphus_scriptus. [Accessed 4 May 2025].

Gafner, K. and Robinson, C.T., 2007. Nutrient enrichment influences the responses of stream macroinvertebrates to disturbance. *Journal of the North American Benthological Society*, *26*(1), pp.92-102. https://doi.org/10.1899/0887-3593(2007)26[92:NEITRO]2.0.CO;2

Gaston, K.J. and Blackburn, T.M. (2003). Dispersal and the interspecific abundance-occupancy relationship in British birds. *Global Ecology and Biogeography*, 12(5), pp.373–379. https://doi.org/10.1046/j.1466-822x.2003.00054.x

Gaston, K. and He, F. (2010). *Species occurrence and occupancy*. [Online] Available at: https://sites.ualberta.ca/~fhe/He-publications/Gaston_He.book2010.pdf. [Accessed 20 September 2025]

Gelman, A., Carlin, J.B., Stern, H.S., Dunson, D.B., Vehtari, A. and Rubin, D.B., (2013a). *Bayesian data analysis.* 3rd ed. Boca Raton, FL: Chapman and Hall/CRC. https://doi.org/10.1201/b16018

Gelman, A., Hwang, J. and Vehtari, A. (2013b). Understanding predictive information criteria for Bayesian models. *Statistics and Computing*, 24(6), pp.997–1016. https://doi.org/10.1007/s11222-013-9416-2

George, G., Hurley, M. and Hewitt, D. (2007). The impact of climate change on the physical characteristics of the larger lakes in the English Lake District. *Freshwater Biology*, 52(9), pp.1647–1666. https://doi.org/10.1111/j.1365-2427.2007.01773.x

Ghiglieri, M.P., Butynski, T.M., Struhsaker, T.T., Leland, L., Wallis, S.J. and Waser, P. (1982). Bush pig (*Potamochoerus porcus*) polychromatism and ecology in Kibale Forest, Uganda. *African journal of ecology*, 20(4), pp.231–236. https://doi.org/10.1111/j.1365-2028.1982.tb00298.x

Giriat, D., Gorczyca, E. and Sobucki, M. (2016). Beaver ponds' impact on fluvial processes (Beskid Niski Mts., SE Poland). *Science of The Total Environment*, 544, pp.339–353. https://doi.org/10.1016/j.scitotenv.2015.11.103

Girmay, M., Birhane, E., Gebrehiwot, K., Warkineh, B., Alemu, S., Negussie, A., Berhe, D.H., Gidey, T. and Gebretsadik, Y. (2025). Riparian flora and fauna (emphasizing vertebrates) studies in the Hirmi dryland ecosystem, Tigray, Ethiopia. *Discover Environment*, 3(1), p.241. https://doi.org/10.1007/s44274-025-00460-y

Goldstein, B.R., Jensen, A.J., Kays, R., Cove, M.V., McShea, W.J., Rooney, B., Kierepka, E.M. and Pacifici, K. (2024). Guidelines for estimating occupancy from autocorrelated camera trap detections. *Methods in Ecology and Evolution*, 15(7), pp.1177–1191. https://doi.org/10.1111/2041-210X.14359

Google LLC. (2023). *Google Earth Pro* (Version 7.3). Available at: https://earth.google.com [Accessed: 30 October 2023].

Google LLC. (2024). *Google Earth Pro* (Version 7.3). Available at: https://earth.google.com [Accessed: 12 August 2024].

Google LLC. (2026). *Google Earth Pro* (Version 7.3). Available at: https://earth.google.com [Accessed: 7 February 2026].

González, P.S., Stehr, A. and Barra, R.O. (2024). Assessment of water quality trends through the application of an aggregated water quality index with historical monitored data in a Mediterranean Andean basin. *Ecological Indicators*, [online] 166, 112373. https://doi.org/10.1016/j.ecolind.2024.112373

Gordon, A.K., Griffin, N.J. and Palmer, C.G. (2015). The relationship between concurrently measured SASS (South African Scoring System) and turbidity data archived in the South African River Health Programme's rivers database. *Water SA*, 41(1), pp.21–26. https://doi.org/10.4314/wsa.v4lil4

Gould, M.J., Gould, W.R., Cain, J.W. and Roemer, G.W. (2019). Validating the performance of occupancy models for estimating habitat use and predicting the distribution of highly-mobile species: A case study using the American black bear. *Biological Conservation*, 234, pp.28–36. https://doi.org/10.1016/j.biocon.2019.03.010

Grant, E.H.C. (2014). Please don't misuse the museum: 'declines' may be statistical. *Global Change Biology*, 21(3), pp.1018–1024. https://doi.org/10.1111/gcb.12702.

Graziano, M.P., Deguire, A.K. and Surasinghe, T.D. (2022). Riparian buffers as a critical landscape feature: insights for riverscape conservation and policy renovations. *Diversity*, 14(3), p.172. https://doi.org/10.3390/d14030172

Grebe, C.D., Mathewson, P.D., Porter, W.P. and McFarland, R. (2024). Evaluating the physiological benefits of behavioral flexibility in chacma baboons (*Papio ursinus)* using a biophysical model. *Journal of Thermal Biology*, 127, p.104042. https://doi.org/10.1016/j.jtherbio.2024.104042

Green, D. (2007). *Understanding Arizona's Riparian Areas*. [Online]. Available at: https://rangelandsgateway.org/sites/default/files/2021-01/az1432.pdf#page=61 [Accessed 9 September 2024].

Gregory, S.V., Swanson, F.J., McKee, W.A. and Cummins, K.W. (1991). An ecosystem perspective of riparian zones. *BioScience*, 41(8), pp.540–551. https://doi.org/10.2307/1311607

Griffin, N., Palmer, C. and Scherman, P. (2014). *Critical analysis of environmental water quality in South Africa: historic and current trends report to the WATER RESEARCH COMMISSION*. [Online]. Available at: https://www.wrc.org.za/wp-content/uploads/mdocs/2184-1-14.pdf. [Accessed 5 March 2024].

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C. and Wiens, J. (2006). Ecological thresholds: the key to successful environmental management or an important concept with no practical application? *Ecosystems*, 9(1), pp.1–13. https://doi.org/10.1007/s10021-003-0142-z

Gronwald, M. and Russell, J. (2021). Measuring rat relative abundance using camera traps and digital strike counters for Goodnature A24 self-resetting traps. *New Zealand Journal of Ecology*, 45(1), 7. https://doi.org/10.20417/nzjecol.45.7

Gu, J.Y., Lee, J.-W., Lee, S.W., Park, Y. and Park, S.R. (2025). Enhancing stream ecosystems through riparian vegetation management. *Land*, 14(6), pp.1248–1248. https://doi.org/10.3390/land14061248

Gu, S. and Li, S. (2024). Riparian habitat quality as an indicator of land use/land cover effects on riverine water quality. *Geography and Sustainability*, 5(1), pp.135–143. https://doi.org/10.1016/j.geosus.2023.11.005

Guillera-Arroita, G., Ridout, M.S. and Morgan, B.J.T. (2010). Design of occupancy studies with imperfect detection. *Methods in Ecology and Evolution*, 1(2), pp.131–139. https://doi.org/10.1111/j.2041-210x.2010.00017.x

Hamilton III, W. J., 1986. Namib Desert Chacma baboon (*Papio ursinus*) use of food and water resources during a food shortage. *Madoqua*, *1986*(4), pp.397-407.

Hamilton, B.T., Roeder, B.L., Hatch, K.A., Eggett, D.L. and Tingey, D. (2015). Why is small mammal diversity higher in riparian areas than in uplands? *Journal of Arid Environments*, 119, pp.41–50. https://doi.org/10.1016/j.jaridenv.2015.03.007.

Hancock, N. (2016). *TDS and pH*. Safe drinking water foundation. [Online]. Available at: https://www.safewater.org/fact-sheets-1/2017/1/23/tds-and-ph. [Accessed 20 March 2024].

Hanekom, N. and Randall, R.M. (2015). Assessments of occurrence and distribution of mammals in forests of the Garden Route National Park based on camera trapping. *Koedoe*, 57(1), pp.1-7. https://doi.org/10.4102/koedoe.v57i1.1257

Haring, C., Weier, S. and Linden, B. (2023) Distribution and habitat preference of African clawless otters (*Aonyx capensis*) and water mongooses (*Atilax paludinosus*) in the Soutpansberg, South Africa. *IUCN Otter Specialist Group Bulletin*, 40(1), pp.26–38. Available at: https://www.iucnosgbull.org/Volume40/Haring_et_al_2023.pdf (Accessed 20 August 2025).

Harmsen, B.J., Foster, R.J., Silver, S., Ostro, L. and Patrick, D.C. (2010). Differential use of trails by forest mammals and the implications for cameratrap studies: A case study from Belize. *Biotropica*, 42(1), pp.126–133. https://doi.org/10.2307/27742872

Haycock, N.E. and Pinay, G. (1993). Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *Journal of Environment Quality*, 22(2), p.273. https://doi.org/10.2134/jeq1993.00472425002200020007x

Hazen, E.L., Abrahms, B., Brodie, S., Carroll, G., Jacox, M.G., Savoca, M.S., Scales, K.L., Sydeman, W.J. and Bograd, S.J. (2019). Marine top predators as climate and ecosystem sentinels. *Frontiers in Ecology and the Environment*, 17(10), pp.565–574. https://doi.org/10.1002/fee.2125

Hedden, S. and Cilliers, J. (2014). *Parched prospects – The emerging water crisis in South Africa*. Pretoria: Institute for Security Studies. [Online]. Available at: https://www.issafrica.org/research/papers/parched-prospects-the-emerging-water-crisis-in-south-africa [Accessed 20 October 2024].

Hessen, D.O. (2024). Biotic deterioration and homogenization: Why it matters. *International Journal of Politics Culture and Society*. https://doi.org/10.1007/s10767-024-09498-x

Hikel, K.B., Peters, M.K., Linden, J. and Linden, B. (2024). The role of bushpigs (*Potamochoerus larvatus*) as seed dispersers in indigenous forests in the Soutpansberg Mountain range, South Africa. *Animal Biology*, 74(1), pp.37–58. https://doi.org/10.1163/15707563-bja10124.

Hines, J.E. (2006). *PRESENCE v2.2 – Software to estimate patch occupancy and related parameters*. [Online]. Available: http://www.mbr-pwrc.usgs.gov/software/PRESENCE.html

Hines, J.E., Nichols, J.D., Royle, J.A., MacKenzie, D.I., Gopalaswamy, A.M., Kumar, N.S. and Karanth, K.U. (2010). Tigers on trails: occupancy modeling for cluster sampling. *Ecological Applications*, 20(5), pp.1456–1466. https://doi.org/10.1890/09-0321.1

Hoffman, T.S. and O'Riain, M.J. (2012). Landscape requirements of a primate population in a human-dominated environment. *Frontiers in Zoology*, 9(1), pp.1. https://doi.org/10.1186/1742-9994-9-1

Hofmeester, T.R., Thorsen, N.H., Cromsigt, J.P.G.M., Kindberg, J., Andrén, H., Linnell, J.D.C. and Odden, J. (2021). Effects of camera-trap placement and number on detection of members of a mammalian assemblage. *Ecosphere*, 12(7), e03662. https://doi.org/10.1002/ecs2.3662

Holland, A. (2016). *Bridging aquatic and terrestrial ecosystems: Ecology of semi-aquatic mammals in southern Illinois.* PhD thesis, Southern Illinois University Carbondale. Available at: https://opensiuc.lib.siu.edu/dissertations/1269/ [Accessed 14 November 2025].

Holland, A.M., Schauber, E.M., Nielsen, C.K. and Hellgren, E.C. (2019a). Occupancy dynamics of semi-aquatic herbivores in riparian systems in Illinois, USA. *Ecosphere*, 10(3), e02614. https://doi.org/10.1002/ecs2.2614

Holland, A.M., Schauber, E.M., Nielsen, C.K. and Hellgren, E.C. (2019b). River otter and mink occupancy dynamics in riparian systems. *The Journal of Wildlife Management*, 83(7), pp.1552–1564. https://doi.org/10.1002/jwmg.21745

Holmes, P.M., Richardson, D.M., Esler, K.J., Witkowski, E.T.F. and Fourie, S. (2005). A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. *South African Journal of Science*, 101(11/12), pp.553–564. [Online] Available at: https://scholar.sun.ac.za/items/a551ee6a-ae97-4ecb-8014-cf979b79e6b1 [Accessed 20 April 2024].

Holmes, P.M., Esler, K.J., Richardson, D.M. and Witkowski, E.T.F. (2008). Guidelines for improved management of riparian zones invaded by alien plants in South Africa. *South African Journal of Botany*, 74(3), pp.538–552. https://doi.org/10.1016/j.sajb.2008.01.182

Holt, A.R., Gaston, K.J. and He, F. (2002). Occupancy-abundance relationships and spatial distribution: A review. *Basic and Applied Ecology*, 3(1), pp.1–13. https://doi.org/10.1078/1439-1791-00083

Hood, G.A. (2020). Accessing and mobilizing 'new' data to evaluate emerging environmental impacts on semi-aquatic mammals. *Land*, 9(10), p.345. https://doi.org/10.3390/land9100345.

Hood, W.G. and Naiman, R.J. (2000). Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant Ecology*, 148(1), pp.105–114. https://doi.org/10.1023/a:1009800327334

Hooten, M.B. and Hobbs, N.T. (2015). A guide to Bayesian model selection for ecologists. *Ecological Monographs*, 85(1), pp.3–28. https://doi.org/10.1890/14-0661.1

Huerta-Rodríguez, J.O., Rosas-Rosas, O.C., López-Mata, L., Alcántara-Carbajal, J.L. and Tarango-Arámbula, L.A. (2022). Camera traps reveal the natural corridors used by mammalian species in eastern Mexico. *Ecological Processes*, 11(1). https://doi.org/10.1186/s13717-022-00394-w.

Hunt, S.D., Guzy, J.C., Price, S.J., Halstead, B.J., Eskew, E.A. and Dorcas, M.E. (2013). Responses of riparian reptile communities to damming and urbanization. *Biological Conservation*, 157, pp.277–284. https://doi.org/10.1016/j.biocon.2012.08.035

Hwang, S.J. (2020). Eutrophication and the ecological health risk. *International Journal of Environmental Research and Public Health*, 17(17), 6332. https://doi.org/10.3390/ijerph17176332

Iknayan, K.J., Tingley, M.W., Furnas, B.J. and Beissinger, S.R. (2014). Detecting diversity: emerging methods to estimate species diversity. *Trends in Ecology and Evolution*, 29(2), pp.97–106. https://doi.org/10.1016/j.tree.2013.10.012

Ivan, J. and Parks, C. (2018). *Density estimation*. [Online]. Available at: http://www.phidot.org/software/mark/docs/book/pdf/chap20.pdf [Accessed 5 August 2024].

IUCN (2024). *Guidelines for using the IUCN red list categories and criteria*. Version 16. Prepared by the Standards and Petitions Committee. [Online]. Available at: http://www.iucnredlist.org/documents/RedListGuidelines.pdf. [Accessed 20 August 2024].

IUCN, 2025. *The IUCN Red List of Threatened Species. Version 2025-2.* [Online] Available at: https://www.iucnredlist.org [Accessed 15 December 2025].

Jacobsen, N.H.G. (1974). Distribution, home range and behaviour patterns of bushbuck in the Lutope and Sengwa Valleys, Rhodesia. *South African Journal of Wildlife Research-24-month delayed open access*, 4(2), pp.75-93. [Online]. Available at: https://www.semanticscholar.org/paper/Distribution%2C-home-range-and-

behaviour-patterns-of-Jacobsen/d728c32a9c6ad6af27fdb2dbfe134dd6c668fd63 [Accessed 10 October 2025].

Jacques, H., Reed-Smith, J. and Somers, M.J. (2015). *Aonyx capensis*. *The IUCN Red List of Threatened Species 2015*, e.T1793A21938767. [Online] Available at: https://www.iucnredlist.org/species/1793/21938767 [Accessed 25 November 2025].

Janecke, B.B. and Bolton, J.G. (2020). Variation in mammal diversity and habitat affect heterogeneity and processes of a granite catena. *KOEDOE - African Protected Area Conservation and Science*, 62(2), pp.1-12. https://doi.org/10.4102/koedoe.v62i2.1592

Jepson, P. and Barua, M. (2015). A theory of flagship species action. *Conservation and Society*, 13(1), pp.95–104. [Online]. Available at: https://journals.lww.com/coas/fulltext/2015/13010/a_theory_of_flagship_species_action.9.aspx [Accessed 25 August 2024].

Jessup, D.A., Miller, M., Ames, J., Harris, M., Kreuder, C., Conrad, P.A. and Mazet, J.A. (2004). Southern sea otter as a sentinel of marine ecosystem health. *EcoHealth*, 1(3), pp.239–245. https://doi.org/10.1007/s10393-004-0093-7

Jesus, T. and Monteiro, A. (2022). Effects on macroinvertebrate communities a year after the rehabilitation of an urban river (Tinto River, Portugal). *Limnetica*, 41(2), p.1. https://doi.org/10.23818/limn.41.22

Jiang, C., Geng, J., Hu, H., Ma, H., Gao, X. and Ren, H. (2017). Impact of selected non-steroidal anti-inflammatory pharmaceuticals on microbial community assembly and activity in sequencing batch reactors. *PLoS ONE*, 12(6), e0179236. https://doi.org/10.1371/journal.pone.0179236

Jones, J.P.G. (2011). Monitoring species abundance and distribution at the landscape scale. *Journal of Applied Ecology*, 48(1), pp.9–13. https://doi.org/10.1111/j.1365-2664.2010.01917.x.

Jones, G. (2019). *Cape Fold (Ecoregion 578)*. Freshwater ecoregions of the world (FEOW). University of Cape Town, Freshwater Research Unit, South Africa. [Online]. Available: https://www.feow.org/ecoregions/details/578 [Accessed 26 November 2025].

Jooste, E.C., 2020. *Ecology and diet of the caracal (Caracal caracal) on lethal and non-lethal control farms in the Karoo*. Master's thesis, University of the Western Cape.

Available at: https://uwcscholar.uwc.ac.za/items/59736fb6-4365-4e85-9cf8-8bd13c5c95f8 [Accessed 4 February 2026].

Jordaan, M., Gouws, J. and Anderson, A. (2020). *Field report aquatic surveys for Grootvadersbosch Conservancy*. [online] Available at: https://www.gvbconservancy.co.za/uploads/2/6/3/0/26303016/gvb_conservancy_field_report_07.05.2020_final.pdf [Accessed 19 September 2025].

Kaiser, K., Devito, J., Jones, C.G., Marentes, A., Perez, R., Umeh, L., Weickum, R.M., McGovern, K.E., Wilson, E.H. and Saltzman, W. (2015). Reproductive and immune effects of chronic corticosterone treatment in male White's treefrogs, *Litoria caerulea*. *Conservation Physiology*, 3(1), cov022. https://doi.org/10.1093/conphys/cov022

Kajee, M., Dallas, H.F., Griffiths, C.L., Kleynhans, C.J. and Shelton, J.M. (2023). The status of South Africa's freshwater fish fauna: A spatial analysis of diversity, threat, invasion, and protection. *Fishes*, 8(12), p.571. https://doi.org/10.3390/fishes8120571.

Kannan, K., Kajiwara, N., Watanabe, M., Nakata, H., Thomas, N.J., Stephenson, M., Jessup, D.A. and Tanabe, S. (2004). Profiles of polychlorinated biphenyl congeners, organochlorine pesticides, and butyltins in southern sea otters and their prey. *Environmental Toxicology and Chemistry*, 23(1), p.49. https://doi.org/10.1897/03-53

Karr, J.R. and Schlosser, I.J. (1978). Water resources and the land–water interface: Water resources in agricultural watersheds can be improved by effective multidisciplinary planning. *Science*, 201(4352), pp.229–234. [Online]. Available at: https://www.researchgate.net/publication/279558106_Water_resources_and_the_land-water_interface_Water_resources_in_agricultural_watersheds_can_be_improved_by_effective_multidisciplinary_planning [Accessed 20 November 2024].

Kean, E.F., Shore, R.F., Scholey, G., Strachan, R. and Chadwick, E.A. (2021). Persistent pollutants exceed toxic thresholds in a freshwater top predator decades after legislative control. *Environmental Pollution*, 272, 116415. https://doi.org/10.1016/j.envpol.2020.116415

Kéry, M. and Royle, J.A. (2008). Hierarchical Bayes estimation of species richness and occupancy in spatially replicated surveys. *Journal of Applied Ecology*, 45(2), pp.589–598. https://doi.org/10.1111/j.1365-2664.2007.01441.x

Kéry, M. and Royle, J.A. (2015) *Applied hierarchical modeling in ecology: Analysis of distribution, abundance and species richness in R and BUGS*. Amsterdam: Academic Press

Kéry, M. and Schmidt, B. (2008). Imperfect detection and its consequences for monitoring for conservation. *Community Ecology*, 9(2), pp.207–216. https://doi.org/10.1556/comec.9.2008.2.10

Kinnoumè, S.M.D., Gouwakinnou, G.N., Noulèkoun, F., Balagueman, R.O., Houehanou, T.D. and Natta, A.K. (2024). Trees diversity explains variations in biodiversity-ecosystem function relationships across environmental gradients and conservation status in riparian corridors. *Frontiers in Forests and Global Change*, 7. https://doi.org/10.3389/ffgc.2024.1291252

Kleynhans, C., Thirion, C. and Rowntree, K. (2007). *River ecoclassification manual for ecostatus determination (version 1) including detailed methods for - - fish response assessment index - macroinvertebrates response assessment index - physico-chemical driver assessment index*. [Online]. Available at: https://www.dws.gov.za/iwqs/rhp/publications/ecostatus_manual.pdf [Accessed 28 September 2025].

Knapp, S., Winter, M. and Klotz, S. (2016). Increasing species richness but decreasing phylogenetic richness and divergence over a 320-year period of urbanization. *Journal of Applied Ecology*, 54(4), pp.1152–1160. https://doi.org/10.1111/1365-2664.12826

Knight, J. (2018). Water Resources in South Africa. *World Regional Geography Book Series*, pp.91–101. https://doi.org/10.1007/978-3-319-94974-1_10

Kloskowski, J. (2005). Otter *Lutra lutra* damage at farmed fisheries in southeastern Poland, II: Exploitation of common carp *Cyprinus carpio*. *Wildlife Biology*, 11(3), pp.257–261. https://doi.org/10.2981/0909-6396(2005)11[257:OLLDAF]2.0.CO;2

Kolowski, J.M. and Forrester, T.D. (2017). Camera trap placement and the potential for bias due to trails and other features. *PLOS ONE*, 12(10), e0186679. https://doi.org/10.1371/journal.pone.0186679

Kubheka, S.P., Rowe-Rowe, D.T., Alletson, J.D. and Perrin, M.R. (2012). Possible influence of increased riparian activity (stream modification and agricultural intensification) on abundance of South African otters. *African Journal of Ecology*, 51(2), pp.288–294. https://doi.org/10.1111/aje.12033

Lamberth, S., van Niekerk, L. and Hutchings, K. (2008). Comparison of, and the effects of altered freshwater inflow on, fish assemblages of two contrasting South African estuaries: the cool-temperate Olifants and the warm-temperate Breede. *African Journal of Marine Science*, 30(2), pp.311–336. https://doi.org/10.2989/ajms.2008.30.2.9.559.

Larivière, S. (2001). *Aonyx capensis*. *Mammalian Species*, 2001(671), pp. 1–6. [Online]. Available at: https://www.science.smith.edu/departments/biology/VHAYSSEN/msi/pdf/671_Aonyx_capensis.pdf [Accessed 15 May 2024].

Laurance, W.F., Croes, B.M., Guissouegou, N., Buij, R., Dethier, M. and Alonso, A. (2008). Impacts of roads, hunting, and habitat alteration on nocturnal mammals in African rainforests. *Conservation Biology*, 22(3), pp.721–732. https://doi.org/10.1111/j.1523-1739.2008.00917.x

Lemarchand, C., Rosoux, R. and Berny, P. (2011). Ecotoxicology of the Eurasian otter (*Lutra lutra*) along Loire River (France) and predictable trends due to global change. *Proceedings of XIth International Otter Colloquium*, IUCN Otter Specialist Group Bulletin, 28B, pp.5–14. [Online]. Available at: https://www.iucnosgbull.org/Volume28B/Lemarchand_et_al_2011.html [Accessed 25 April 2024].

Le Maitre, D.C., Blignaut, J.N., Clulow, A., Dzikiti, S., Everson, C.S., Görgens, A.H.M. and Gush, M.B., 2020. *Impacts of plant invasions on terrestrial water flows in South Africa.* In: B.W. van Wilgen, J. Measey, D.M. Richardson, J.R. Wilson and T.A. Zengeya, eds. *Biological invasions in South Africa.* Cham: Springer, pp. 431–457. https://doi.org/10.1007/978-3-030-32394-3_15

Levin, J.C., Woodford, D.J. and Snow, G.C. (2019). Evaluating the effectiveness of freshwater fishes as bio-indicators for urban impacts in the Crocodile (West) catchment, South Africa. *Water SA*, 45(3), pp.477–486. https://doi.org/10.17159/wsa/2019.v45.i3.6745

Lijun, C., Wenhong, X. and Zhishu, X. (2019). Limitations of relative abundance indices calculated from camera-trapping data. *Biodiversity Science*, 27(3), pp.243–248. https://doi.org/10.17520/biods.2018327

Linden, D.W., Fuller, A.K., Royle, J.A. and Hare, M.P. (2017). Examining the occupancy-density relationship for a low-density carnivore. *Journal of Applied Ecology*, 54(6), pp.2043–2052. https://doi.org/10.1111/1365-2664.12883

Lindenmayer, D.B., Lavery, T. and Scheele, B.C. (2022). Why we need to invest in large-scale, long-term monitoring programs in landscape ecology and conservation biology. *Current Landscape Ecology Reports*, 7(4), pp.137-146. https://doi.org/10.1007/s40823-022-00079-2

Lloyd, P.H. (2000). *State of Biodiversity: Western Cape Province, South Africa — Mammals*. Stellenbosch: Scientific Services, Western Cape Nature Conservation Board. [Online]. Available at: https://the-eis.com/elibrary/sites/default/files/downloads/State%20of%20Biodiversity_%20Western%20Cape%20Province_South%20Africa%20Mammals.pdf [Accessed 20 April 2024].

Lowrance, R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. (1984). Riparian forests as nutrient filters in agricultural watersheds. *BioScience*, 34(6), pp.374–377. https://doi.org/10.2307/1309729

Lukhele, T. and Alfred, T. (2024). Eutrophication of inland surface waters in South Africa: an overview. *International journal of environmental research*, 18(2), 27. https://doi.org/10.1007/s41742-024-00568-8

Lynch, S.M. and Western, B. (2004). Bayesian posterior predictive checks for complex models. *Sociological Methods and Research*, 32(3), pp.301–335. https://doi.org/10.1177/0049124103257303

Lyons, J., Thimble, S.W. and Paine, L.K., 2000. Grass versus trees: Managing riparian areas to benefit streams of Central North America 1. *JAWRA Journal of the American Water Resources Association*, *36*(4), pp.919-930. https://doi.org/10.1111/j.1752-1688.2000.tb04317.x

Lyra-Jorge, M.C., Ciocheti, G., Pivello, V.R. and Meirelles, S.T., 2008. Comparing methods for sampling large- and medium-sized mammals: camera traps and track plots. *European Journal of Wildlife Research*, 54(4), pp.739–744. https://doi.org/10.1007/s10344-008-0205-8

Macdonald, S.M. and Mason, C.F. (1983). Some factors influencing the distribution of otters (*Lutra lutra*). *Mammal Review*, 13(1), pp.1–10. https://doi.org/10.1111/j.1365-2907.1983.tb00259.x

McDonald, D.J. (1993). The vegetation of the southern Langeberg, Cape Province. 1. The plant communities of the Boosmansbos Wilderness Area. *Bothalia*, 23(1), pp.129–151. https://doi.org/10.4102/abc.v23i1.798

Machtans, C.S., Villard, M. and Hannon, S.J. (1996). Use of riparian buffer strips as movement corridors by forest birds. *Conservation Biology*, 10(5), pp.1366–1379. https://doi.org/10.1046/j.1523-1739.1996.10051366.x

MacKenzie, D.I. (2006). Modeling the probability of resource use: the effect of, and dealing with, detecting a species imperfectly. *The Journal of Wildlife Management*, 70(2), pp.367–374. https://doi.org/10.2193/0022-541X(2006)70[367:MTPROU]2.0.CO;2

MacKenzie, D.I. and Bailey, L.L. (2004). Assessing the fit of site-occupancy models. *Journal of Agricultural, Biological, and Environmental Statistics*, 9(3), pp.300–318. https://doi.org/10.1198/108571104x3361

MacKenzie, D.I., Bailey, L.L. and Nichols, J.D. (2004). Investigating species co-occurrence patterns when species are detected imperfectly. *Journal of Animal Ecology*, 73(3), pp.546–555. https://doi.org/10.1111/j.0021-8790.2004.00828.x.

MacKenzie, D.I. and Hines, J.E. (2018). *RPresence: R interface for Program PRESENCE. R package version 2.12. 32*

MacKenzie, D.I., Nichols, J.D., Hines, J.E., Knutson, M.G. and Franklin, A.B. (2003). Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. *Ecology*, 84(8), pp.2200–2207. https://doi.org/10.1890/02-3090

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A. and Langtimm, C.A. (2002). Estimating site occupancy rates when detection probabilities are less than one. *Ecology*, 83(8), p.2248. https://doi.org/10.2307/3072056

MacKenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L., and Hines, J.E. (2018). *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence* (2nd ed.). Academic Press.

MacKenzie, D.I. and Royle, J.A. (2005). Designing occupancy studies: general advice and allocating survey effort. *Journal of Applied Ecology*, 42(6), pp.1105–1114. https://doi.org/10.1111/j.1365-2664.2005.01098.x

MacLeod, S.B., Kerley, G.I.H. and Gaylard, A. (1996). Habitat and diet of bush buck (*Tragelaphus scriptus*) in the Woody Cape Nature Reserve: observations from faecal analysis. *South African Journal of Wildlife Research*, 26(1), pp.19–25. [Online]. Available at: https://journals.co.za/doi/pdf/10.10520/EJC116988 [Accessed 20 April 2025].

Maestas, J.D., Wheaton, J.M., Bouwes, N., Swanson, S. and Dickard, M. (2023). Water is life: Importance and management of riparian areas for rangeland wildlife. In: L.B. McNew, D.K. Dahlgren and J.L. Beck, eds., *Rangeland Wildlife Ecology and Conservation.* Springer Nature, pp.177–208. https://doi.org/10.1007/978-3-031-34037-6_7

Malhi, Y., Franklin, J., Seddon, N., Solan, M., Turner, M.G., Field, C.B. and Knowlton, N. (2020). Climate change and ecosystems: threats, Opportunities and Solutions. *Philosophical Transactions of the Royal Society B: Biological Sciences*, [online] 375(1794). https://doi.org/10.1098/rstb.2019.0104

Majelantle, T.L. (2019). Evaluation of anthropogenic disturbance on African clawless otter (*Aonyx capensis*) physiological stress, behaviour, and population density using non-invasive methods. M.Sc. (Zoology) thesis, University of Pretoria. [online] Available at: https://repository.up.ac.za/server/api/core/bitstreams/8cf4a97a-f146-4419-8fe8-b7e0fb26a5ec/content [Accessed 25 November 2024].

Majelantle, T.L., McIntyre, T. and Ganswindt, A. (2020). Monitoring the effects of land transformation on African clawless otters (*Aonyx capensis*) using fecal glucocorticoid metabolite concentrations as a measure of stress. *Integrative Zoology*, 15(4), pp. 293–306. https://doi.org/10.1111/1749-4877.12428

Majelantle, T.L., Ganswindt, A., Jordaan, R.K., Slip, D.J., Harcourt, R. and McIntyre, T. (2021). Increased population density and behavioural flexibility of African clawless otters (*Aonyx capensis*) in specific anthropogenic environments. *Urban Ecosystems*, 24(4), pp.691–699. https://doi.org/10.1007/s11252-020-01068-1

Makumbe, P., Kanda, A. and Chinjani, T. (2022). The Relationship between benthic macroinvertebrate assemblages and water quality parameters in the Sanyati Basin, Lake Kariba, Zimbabwe. *The Scientific World Journal*, 2022, e5800286. https://doi.org/10.1155/2022/5800286.

Malan, G. 2001. The avifauna of riparian-*Pinus* habitat edges at Mooiplaas forestry estate, KwaZulu-Natal, South Africa. *South African Journal of Wildlife Research* (24-month delayed open access), 31(3), pp.73–84. https://doi.org/10.10520/EJC117133

Malan, G., Meyer, E. and Panagos, M.D. 2007. Riparian-zone rehabilitation in pine plantations: Grassland vs woodland for plants and birds? *South African Journal of Wildlife Research* (24-month delayed open access), 37(2), pp.159–178. https://doi.org/10.10520/EJC117268

Malanson, G.P. (1993). *Riparian Landscapes*. Cambridge: Cambridge University Press.

Mangachena, J.R. and Geerts, S. 2017. Invasive alien trees reduce bird species richness and abundance of mutualistic frugivores and nectarivores: A bird's eye view on a conflict of interest species in riparian habitats. *Ecological Research*, 32(5), pp.667–676. https://doi.org/10.1007/s11284-017-1481-0

Marino, A., Fenoglio, S. and Bo, T. (2024). The impact of catastrophic floods on macroinvertebrate communities in low-order streams: A study from the Apennines (Northwest Italy). *Water*, 16(18), 2646. https://doi.org/10.3390/w16182646.

Maritz, B. and Alexander, G.J. (2007). Herpetofaunal utilisation of riparian buffer zones in an agricultural landscape near Mtunzini, South Africa. *African Journal of Herpetology*, 56(2), pp.163–169. https://doi.org/10.1080/21564574.2007.9635561

Marneweck, C.J., Allen, B.L., Butler, A.R., Do Linh San, E., Harris, S.N., Jensen, A.J., Saldo, E.A., Somers, M.J., Titus, K., Muthersbaugh, M. and Vanak, A. (2022). Middle-out ecology: small carnivores as sentinels of global change. *Mammal Review*, 52(4), pp.471–479. https://doi.org/10.1111/mam.12300

Marques, T.A., Thomas, L., Martin, S.W., Mellinger, D.K., Ward, J.A., Moretti, D.J., Harris, D. and Tyack, P.L. (2012). Estimating animal population density using passive acoustics. *Biological Reviews*, 88 (2), pp.287–309. https://doi.org/10.1111/brv.12001

Martin, C. (2002). *Riparian habitat management for mammals on corps of engineers projects*. [Online] Available at: https://apps.dtic.mil/sti/tr/pdf/ADA406006.pdf [Accessed 21 February 2025].

Martin-Collado, D., Jiménez, M.D., Rouco, C., Ciuffoli, L. and Torre, R. (2020). Potential of restored gravel pits to provide suitable habitats for Eurasian otters in anthropogenic landscapes. *Restoration Ecology*, 28(4), pp.995–1005. https://doi.org/10.1111/rec.13129.

Martin-Garcia, S., Rodríguez-Recio, M., Peragón, I., Bueno, I. and Virgós, E. (2022). Comparing relative abundance models from different indices, a study case on the red fox. *Ecological Indicators*, 137, 108778. https://doi.org/10.1016/j.ecolind.2022.108778

Mason, C.F. (1995). Habitat quality, water quality and otter distribution. *Hystrix – The Italian Journal of Mammalogy*, 7(1-2), pp.195–207. https://doi.org/10.4404/hystrix-7.1-2-4070

Mason, C.F., Last, N.I. and Macdonald, S.M. (1986). Mercury, cadmium, and lead in British otters. *Bulletin of Environmental Contamination and Toxicology*, 37(1), pp.844–849. https://doi.org/10.1007/BF01607848

Mason, C.F. and Rowe-Rowe, D.T. (1992). Organochlorine pesticide residues and PCBs in otter scats from Natal. *South African Journal of Wildlife Research*, 22(1), pp. 29–31. [online]. Available: https://journals.co.za/doi/pdf/10.10520/EJC116885 [Accessed 26 October 2024].

Mateo-Sagasta, J., Zadeh, S.M., Turral, H. and Burke, J. (2017). Water pollution from agriculture: A global review. Executive summary. Rome, Italy: FAO; Colombo, Sri Lanka: International Water Management Institute (IWMI). https://hdl.handle.net/10568/88070

Mateo-Tomás, P., Olea, P.P., Mínguez, E., Mateo, R. and Viñuela, J. (2020). Direct evidence of poison-driven widespread population decline in a wild vertebrate. *Proceedings of the National Academy of Sciences*, 117(28), pp.16418-16423. https://doi.org/10.1073/pnas.1922355117

Mbatyoti, O.A., Somers, M.J. and Do, E. (2010). The diet of the Cape grey mongoose (*Galerella pulverulenta*) in the Albany Thicket Biome (Eastern Cape). *2010 Symposium of the Southern African Wildlife Management Association*. [Online] Available at: https://www.researchgate.net/publication/266374789_The_diet_of_the_Cape_grey_mongoose_Galerella_pulverulenta_in_the_Albany_Thicket_Biome_Eastern_Cape. [Accessed: 4 June 2026].

McCaffrey, S. (n.d.). *Archived perspective notice: Water quality parameters & indicators*. [Online]. SSWM. Available at: https://sswm.info/sites/default/files/reference_attachments/MCCAFFREY%20ny%20Water%20Quality%20Parameters%20&%20Indicators.pdf [Accessed 25 August 2025].

McElreath, R. (2020). *Statistical Rethinking A Bayesian Course with Examples in R and Stan*. 2nd ed. Chapman and Hall. [Online]. Available at: https://civil.colorado.edu/~balajir/CVEN6833/bayes-resources/RM-StatRethink-Bayes.pdf. [Accessed 20 May 2025].

Medina-Vogel, G., Kaufman, V.S., Monsalve, R. and Gomez, V. (2003). The influence of riparian vegetation, woody debris, stream morphology and human activity on the

use of rivers by southern river otters in *Lontra provocax* in Chile. *Oryx*, 37(04). https://doi.org/10.1017/s0030605303000784.

Meek, C.S., Richardson, D.M. and Mucina, L. (2010). A river runs through it: Land-use and the composition of vegetation along a riparian corridor in the Cape Floristic Region, South Africa. *Biological Conservation*, 143(1), pp.156–164. https://doi.org/10.1016/j.biocon.2009.09.021.

Meek, P.D., Ballard, G. and Fleming, P. (2012). *An introduction to camera trapping for wildlife surveys in Australia*.PestSmart Toolkit Publication, Invasive Animals Cooperative Research Centre, Canberra, Australia. [Onine]. Available at: https://pestsmart.org.au/wp-content/uploads/sites/3/2020/06/CameraTrapManual_2012.pdf [Accessed 6 June 2026].

Mengistu, T.S., Garcias, B., Castellanos, G., Seminati, C., Molina-López, R.A. and Darwich, L. (2022). Occurrence of multidrug resistant Gram-negative bacteria and resistance genes in semi-aquatic wildlife - *Trachemys scripta, Neovison vison* and *Lutra lutra* - as sentinels of environmental health. *Science of The Total Environment*, 830, 154814. https://doi.org/10.1016/j.scitotenv.2022.154814.

Midgley, S.J.E. and Lötze, E. (2011). Climate change in the Western Cape of South Africa: trends, projections and implications for chill unit accumulation. *Acta Horticulturae*, (903), pp.1127–1134. https://doi.org/10.17660/actahortic.2011.903.157

Miller, D.A., Nichols, J.D., McClintock, B.T., Grant, E.H.C., Bailey, L.L. and Weir, L.A. (2011). Improving occupancy estimation when two types of observational error occur: non-detection and species misidentification. *Ecology*, 92(7), pp.1422–1428. https://doi.org/10.1890/10-1396.1

Millward, L.S., Wilson, T.M., Weldy, M.J., Rowland, M.M., Duarte, A., Lesmeister, D.B. and Ripple, W.J. (2022). Small mammal relative abundance within riparian ecosystems of the Blue Mountains. *Forest Ecology and Management*, 505, 119899. https://doi.org/10.1016/j.foreco.2021.119899

Milner-Gulland, E.J. and Woodroffe, R. (2001). Mammals, conservation efforts for. In: Levin, S.A. (ed.) *Encyclopedia of Biodiversity*, Volume 3. San Diego: Academic Press, pp.811–824. https://doi.org/10.1016/B0-12-226865-2/00181-4

Milnes, M.R. and Guillette, L.J. (2008). Alligator tales: New lessons about environmental contaminants from a sentinel species. *BioScience*, [online] 58(11), pp.1027–1036. https://doi.org/10.1641/b581106

Mpanyaro, Z., Kalumba, A.M., Zhou, L. and Afuye, G.A. (2024). Mapping and assessing riparian vegetation response to drought along the Buffalo river catchment in the Eastern Cape Province, South Africa. *Climate*, 12(1), 7. https://doi.org/10.3390/cli12010007.

Monroy-Vilchis, O., Urios, V., Zarco-González, M. and Rodríguez-Soto, C. (2009). Cougar and jaguar habitat use and activity patterns in central Mexico. *Animal Biology*, 59(2), pp.145–157. https://doi.org/10.1163/157075609x437673

Montano, L., Pironti, C., Pinto, G., Ricciardi, M., Buono, A., Brogna, C., Venier, M., Piscopo, M., Amoresano, A. and Motta, O. (2022). Polychlorinated Biphenyls (PCBs) in the environment: occupational and exposure events, effects on human health and fertility. *Toxics*, 10(7), p.365. https://doi.org/10.3390/toxics10070365

Moog, O., Schmutz, S. and Schwarzinger, I. (2018). *Biomonitoring and bioassessment*. In: S. Schmutz and J. Sendzimir, eds. *Riverine ecosystem management*. Aquatic Ecology Series, vol. 8. Cham: Springer, pp.371-390. https://doi.org/10.1007/978-3-319-73250-3_19

Moore, R.D., Spittlehouse, D.L. and Story, A. (2005). Riparian microclimate and stream temperature response to forest harvesting: A review. *Journal of the American Water Resources Association*, 41(4), pp.813–834. https://doi.org/10.1111/j.1752-1688.2005.tb03772.x.

Moun, A., Ramesh, T. and Kalle, R. (2024). Riparian habitat quality and seasonality drives spatiotemporal habitat ecology of sympatric Asian otters in southern Western Ghats. *Frontiers in Mammal Science*, 3, 1504722. https://doi.org/10.3389/fmamm.2024.1504722.

Mucina, L., Rutherford, M.C., Powrie, L.W., Gerber, J., Bezuidenhout, H., Sieben, E.J.J., Cilliers, S.S., du Preez, P.J., Manning, J.C., Hoare, D.B. and Boucher, C. (2006). Inland azonal vegetation. In: L. Mucina and M.C. Rutherford (eds), *The vegetation of South Africa, Lesotho and Swaziland* (Strelitzia 19), pp. 616–657. Pretoria: South African National Biodiversity Institute.

Muehlbauer, J.D., Doyle, M.W. and Bernhardt, E.S. (2011). Macroinvertebrate community responses to a dewatering disturbance gradient in a restored stream.

Hydrology and Earth System Sciences, 15(6), pp.1771–1783. https://doi.org/10.5194/hess-15-1771-2011.

Naiman, R.J., Decamps, H. and Pollock, M. (1993). The Role of Riparian Corridors in Maintaining Regional Biodiversity. *Ecological Applications*, 3(2), pp.209–212. https://doi.org/10.2307/1941822

Naiman, R.J. and Décamps, H. (1997). The ecology of interfaces: Riparian Zones. *Annual Review of Ecology and Systematics*, 28(1), pp.621–658. https://doi.org/10.1146/annurev.ecolsys.28.1.621

Naqvi, Q., Wolff, P.J., Molano-Flores, B. and Sperry, J.H. (2022). Camera traps are an effective tool for monitoring insect–plant interactions. *Ecology and Evolution*, 12(6), e8962. https://doi.org/10.1002/ece3.8962

National Institute of Environmental Health Sciences (2023). *Algal blooms*. National Institute of Environmental Health Sciences. [Online]. Available at: https://www.niehs.nih.gov/health/topics/agents/algal-blooms [Accessed 15 August 2025].

National Research Council (1991). *Animals as sentinels of environmental health hazards*. Washington, DC: National Academies Press. https://doi.org/10.17226/1351.

Neilson, E.W., Avgar, T., Burton, A.C., Broadley, K. and Boutin, S. (2018). Animal movement affects interpretation of occupancy models from camera-trap surveys of unmarked animals. *Ecosphere*, 9(1), e02092. https://doi.org/10.1002/ecs2.2092

Nel, J.L. and Driver, A. (2015). *National River Ecosystem Accounts for South Africa. Discussion document for Advancing SEEA Experimental Ecosystem Accounting Project, October 2015*. Pretoria: South African National Biodiversity Institute. [Online]. Available at: https://www.statssa.gov.za/wp-content/uploads/2016/08/National-River-Ecosystem-Accounts-Discussion-Document-FINAL.pdf [Accessed 7 June 2026].

Nel, J.L., Roux, D.J., Maree, G., Kleynhans, C.J., Moolman, J., Reyers, B., Rouget, M. and Cowling, R.M. (2007). Rivers in peril inside and outside protected areas: a systematic approach to conservation assessment of river ecosystems. *Diversity and Distributions*, 13(3), pp.341–352. https://doi.org/10.1111/j.1472-4642.2007.00308.x

Ngcobo, S.P., Wilson, A.-L. and Downs, C.T. (2019). Habitat selection of Cape porcupines in a farmland-suburban context in KwaZulu-Natal, South Africa.

Mammalian Biology, 98, pp.111–118. https://doi.org/10.1016/j.mambio.2019.08.004.

Nichols, J.D. (2010). The wildlife picture index, monitoring and conservation. *Animal Conservation*, 13(4), pp.344–346. https://doi.org/10.1111/j.1469-1795.2010.00382.x

Niedballa, J., Sollmann, R., Mohamed, A.B., Bender, J. and Wilting, A. (2015). Defining habitat covariates in camera-trap based occupancy studies. *Scientific Reports*, 5(1), 17041. https://doi.org/10.1038/srep17041

Niedballa, J. (2024). *5. Multi-species occupancy models*. R-project.org. [Online]. Available at: https://cran.r-project.org/web/packages/camtrapR/vignettes/camtrapr5.html [Accessed 29 September 2025].

Nilsson, C. and Berggren, K. (2000). Alterations of riparian ecosystems caused by river regulation. *BioScience*, 50(9), pp.783-792. https://doi.org/10.1641/0006-3568(2000)050[0783:aorecb]2.0.co;2

Nimma, D., Devi, O.R., Laishram, B., Ramesh, J.V.N., Boddupalli, S., Ayyasamy, R., Tirth, V. and Arabi, A. (2024). Implications of climate change on freshwater ecosystems and their biodiversity. *Desalination and Water Treatment*, 321, p.100889. https://doi.org/10.1016/j.dwt.2024.100889

Oberholster, P.J. and Botha, A. (2014). Importance of water quality to the food industry in South Africa. *Understanding the Food-Energy-Water Nexus*. WWF-SA, South Africa. [Online] Available at: https://awsassets.wwf.org.za/downloads/5__a16269_water_quality_online.pdf [Accessed 20 March 2024].

O'Brien, T.G., Kinnaird, M.F. and Wibisono, H.T. (2003). Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. *Animal Conservation*, 6(2), pp.131–139. https://doi.org/10.1017/s1367943003003172.

O'Brien, T.G. and Kinnaird, M.F. (2008). A picture is worth a thousand words: the application of camera trapping to the study of birds. *Bird Conservation International*, 18(S1), S144–S162. https://doi.org/10.1017/s0959270908000348

O'Brien, T.G. and Kinnaird, M.F. (2011). Density estimation of sympatric carnivores using spatially explicit capture–recapture methods and standard trapping grid. *Ecological Applications*, 21(8), pp.2908–2916. https://doi.org/10.1890/10-2284.1

O'Connell, A.F., Nichols, J.D. and Karanth, K.U. eds., (2011). *Camera traps in animal ecology*. Tokyo: Springer Japan, pp.163-190. https://doi.org/10.1007/978-4-431-99495-4

Okes, N., Ponsonby, D., Rowe-Rowe, D., Page-Nicholson, S., Selier, J., Child, M. and Do Linh San, E. (2016). *Aonyx capensis*. SANBI. [Online]. Available at: https://speciesstatus.sanbi.org/assessment/last-assessment/1943/ [Accessed 20 November 2024].

Okes, N.C. (2017). *Conservation ecology of the African clawless otter, Aonyx capensis, in an urban environment*. PhD thesis, University of Cape Town. [Online]. Available: https://open.uct.ac.za/bitstream/11427/27353/4/thesis_sci_2017_okes_nicola_catherine.pdf [Accessed 26 November 2025].

Okes, N.C. and O'Riain, M.J. (2017). Otter occupancy in the Cape Peninsula: Estimating the probability of river habitat use by African clawless otters, *Aonyx capensis,* across a gradient of human influence. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27(3), pp.706–716. https://doi.org/10.1002/aqc.2738

Oladimeji, A., Woodgate, Z. and O'Riain, M.J. (2024). Wildlife resilience in an urban landscape: understanding land-use impacts in Cape Town. *Urban Ecosystems,* 27(6), pp.2517–2530. https://doi.org/10.1007/s11252-024-01606-1

Ollis, D.J., Snaddon, C.D., Job, N.M. and Mbona, N. (2013). *Classification system for wetlands and other aquatic ecosystems in South Africa: User manual – inland systems (SANBI Biodiversity Series No. 22)*. [Online] Available at: https://www.sanbi.org/wp-content/uploads/2018/04/sanbi-biodiversity-series-wetlands-classification-no-22.pdf [Accessed 20 October 2024].

Ortega, J.C.G., Figueiredo, B.R.S., Graça, W.J., Agostinho, A.A. and Bini, L.M. (2020). Negative effect of turbidity on prey capture for both visual and non-visual aquatic predators. *Journal of Animal Ecology*, 89(11), pp.2427–2439. https://doi.org/10.1111/1365-2656.13329

Osinga, T., de Knegt, H.J., Frauendorf, M. and Hofmeester, T.R. (2025). Balancing detection probability and survey effort in multistate occupancy models: A camera trap simulation analysis. *Ecosphere*, 16(9). https://doi.org/10.1002/ecs2.70402

Pacifici, K., Reich, B.J., Dorazio, R.M. and Conroy, M.J. (2015). Occupancy estimation for rare species using a spatially-adaptive sampling design. *Methods in Ecology and Evolution*, 7(3), pp.285–293. https://doi.org/10.1111/2041-210x.12499.

Palencia, P., Rowcliffe, J.M., Vicente, J. and Acevedo, P. (2021). Assessing the camera trap methodologies used to estimate density of unmarked populations. *Journal of Applied Ecology*, 58(8), pp.1583-1592. https://doi.org/10.1111/1365-2664.13913

Palmer, M.S., Swanson, A., Kosmala, M., Arnold, T. and Packer, C. (2018). Evaluating relative abundance indices for terrestrial herbivores from large-scale camera trap surveys. *African Journal of Ecology*, 56(4), pp.791–803. https://doi.org/10.1111/aje.12566

Palt, M., Le Gall, M., Piffady, J., Hering, D. and Kail, J. (2022). A metric-based analysis on the effects of riparian and catchment landuse on macroinvertebrates. *Science of The Total Environment*, 816, 151590. https://doi.org/10.1016/j.scitotenv.2021.151590

Pamla, L., Vukeya, L.R. and Mokotjomela, T.M. (2024). The potential of foraging chacma baboons (*Papio ursinus*) to disperse seeds of alien and invasive plant species in the Amathole Forest in Hogsback in the Eastern Cape Province, South Africa. *Diversity*, 16(3), pp.168–168. https://doi.org/10.3390/d16030168

Parker, H.S. (2021). River otters: sentinel species for the Chesapeake Bay. *OutLook by the Bay*. [Online] Available at: https://outlookbythebay.com/outlook-for-the-bay/2021/05/river-otters-chesapeake-bay/ outlookbythebay.com [Accessed 20 May 2024].

Parsons, A.W., Forrester, T., Baker-Whatton, M.C., McShea, W.J., Rota, C.T., Schuttler, S.G., Millspaugh, J.J. and Kays, R. (2018). Mammal communities are larger and more diverse in moderately developed areas. *eLife*, 7, e38012. https://doi.org/10.7554/elife.38012.

Pedersen, T.L. (2024). *patchwork: The Composer of Plots*. R package version 1.3.2. [Online]. Available at: https://patchwork.data-imaginist.com/

Peng, M.H., Hung, Y.C., Liu, K.L. and Neoh, K.B. (2020). Landscape configuration and habitat complexity shape arthropod assemblage in urban parks. *Scientific Reports*, 10(1), 16043. https://doi.org/10.1038/s41598-020-73121-0

Perrin, M.R. and Carugati, C. (2000). Habitat use by the African clawless otter and the spotted-necked otter in the KwaZulu-Natal Drakensberg, South Africa. *South African Journal of Wildlife Research*, 30(3), pp.103–113. [Online]. Available at: https://www.researchgate.net/publication/281981422_Habitat_use_by_the_Cape_clawless_otter_and_the_spotted-necked_otter_in_the_KwaZulu-Natal_Drakensberg_South_Africa. [Accessed 20 October 2025].

Perrin, M.R. and D'Inzillo Carranza, I. (2000) Habitat use by spotted-necked otters in the KwaZulu-Natal Drakensberg, South Africa. *South African Journal of Wildlife Research*, 30(1), pp.8–14. Available at: https://journals.co.za/doi/10.10520/EJC117092 [Accessed 5 March 2025].

Phillips, M.J., Swift, L.W. Jr. and Blinn, C.R. (2000). Best management practices for riparian areas. In: Verry, E.S., Hornbeck, J.W. and Dolloff, C.A., eds. *Riparian management in forests of the continental Eastern United States*. Boca Raton, FL: Lewis Publishers, CRC Press, pp.273–286. [Online] Available at: https://research.fs.usda.gov/treesearch/1383 [Accessed 20 August 2025].

Plaatjie, A., Carvalho, F., Rabelani Marikhele, Zimkitha JK Madikiza and Do, E. (2024). Cape genet *Genetta tigrina* diet in the Albany Thicket Biome, South Africa. *African Zoology*, pp.1–7. https://doi.org/10.1080/15627020.2024.2370789

Poff, N.L. and Ward, J.V. (1989). Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. *Canadian Journal of Fisheries and Aquatic Sciences*, 46(10), pp.1805–1818. https://doi.org/10.1139/f89-228

Poff, N.L., Brinson, M.M. and Day, J.W. Jr. (2002). *Aquatic ecosystems and global climate change: Potential impacts on inland freshwater and coastal wetland ecosystems in the United States*. Arlington, VA: Pew Center on Global Climate Change. [Online]. Available at: https://pofflab.colostate.edu/wp-content/uploads/2019/08/Poff_2002_Ecologicalresponsetoandmanagment.pdf pofflab.colostate.edu [Accessed 2 November 2024].

Ponsonby, D.W., Madisha, M.T., Schwaibold, S. and Dalton, D.L. (2019). Genetic diversity of African clawless otters (*Aonyx capensis*) occurring in urbanised areas of Gauteng, South Africa. *South African Journal of Science*, 115(7/8), pp.1-8. https://doi.org/10.17159/sajs.2019/4889

Pool-Stanvliet, R., Duffell-Canham, A., Pence, G. and Smart, R. (2017). *Western Cape Biodiversity*. [Online]. Available at: https://www.westerncape.gov.za/eadp/sites/eadp.westerncape.gov.za/files/atoms/files/DEADP_CN_WCBSP_Handbook_2017.compressed-ilovepdf-compressed.pdf [Accessed 22 August 2024].

Popescu, C., Oprina-Pavelescu, M., Dinu, V., Cazacu, C., Burdon, F.J., Forio, M.A.E., Kupilas, B., Friberg, N., Goethals, P., McKie, B.G. and Rîșnoveanu, G. (2021).

Riparian vegetation structure influences terrestrial invertebrate communities in an agricultural landscape. *Water*, 13(2), 188. https://doi.org/10.3390/w13020188.

Quaglietta, L., Hájková, P., Mira, A. and Boitani, L. (2015). Eurasian otter (*Lutra lutra*) density estimate based on radio tracking and other data sources. *Mammal Research*, 60(2), pp.127–137. https://doi.org/10.1007/s13364-015-0216-2.

Ramesh, T. and Downs, C.T. (2014). Modelling large spotted genet (*Genetta tigrina)* and slender mongoose (*Galerella sanguinea*) occupancy in a heterogeneous landscape of South Africa. *Mammalian Biology*, 79(5), pp.331–337. https://doi.org/10.1016/j.mambio.2014.05.001

Ramesh, T. and Downs, C.T. (2015). Impact of land use on occupancy and abundance of terrestrial mammals in the Drakensberg Midlands, South Africa. *Journal for Nature Conservation*, 23, pp.9–18. https://doi.org/10.1016/j.jnc.2014.12.001

Ramey, T.L. and Richardson, J.S. (2017). Terrestrial invertebrates in the riparian zone: mechanisms underlying their unique diversity. *BioScience*, 67(9), pp.808–819. https://doi.org/10.1093/biosci/bix078

Ramião, J.P., Cássio, F. and Pascoal, C. (2020). Riparian land use and stream habitat regulate water quality. *Limnologica*, 82, 125762. https://doi.org/10.1016/j.limno.2020.125762

R Core Team (2025). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing, Vienna, Austria. Available at: https://www.R-project.org/

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L., Rutherford, M.C., Smit, W.J., Powrie, L.W., Ellis, F., Lambrechts, J.J.N., Scott, L., Radloff, F.G.T., Johnson, S.D., Richardson, D.M., Ward, R.A., Procheş, Ş., Oliver, E.G.H., Manning, J.C., Jürgens, N., McDonald, D.J., Janssen, J.A.M., Walton, B.A., Le Roux, A., Skowno, A.L., Todd, S.W. and Hoare, D.B. (2006). *Fynbos Biome*. In L. Mucina and M.C. Rutherford (eds), *The Vegetation of South Africa, Lesotho and Swaziland* (Strelitzia 19), pp.52–219. Pretoria: South African National Biodiversity Institute.

Reed-Smith, J., Jacques, H. and Somers, M.J. (2015). Hydrictis maculicollis. The IUCN Red List of Threatened Species 2015, e.T12420A21936042. http://dx.doi.org/10.2305/IUCN.UK.2015-%202.RLTS.T12420A21936042.en

Regmi, S., Belant, J.L., Pant, B. and Hari Prasad Sharma (2023). Factors influencing mammalian community occupancy in Dhorpatan Hunting Reserve, Nepal. *Ecology and Evolution*, 13(4), e9980. https://doi.org/10.1002/ece3.9980.

Reid, N., Lundy, M.G., Hayden, B., Waterman, T., Looney, D., Lynn, D., Marnell, F., McDonald, R.A. and Montgomery, W.I. (2014). Covering over the cracks in conservation assessments at EU interfaces: A cross-jurisdictional ecoregion scale approach using the Eurasian otter *(Lutra lutra*). *Ecological Indicators*, 45, pp.93–102. https://doi.org/10.1016/j.ecolind.2014.03.023.

Reinecke, M., Brown, C., Esler, K. and King, J. (2022). Using riparian plant species to locate the 1:2-year floodline on single channel rivers. *African Journal of Aquatic Science*, 47(3), pp.1–14. https://doi.org/10.2989/16085914.2021.2007044

Regolin, A.L., Cherem, J.J., Graipel, M.E., Bogoni, J.A., Ribeiro, J.W., Vancine, M.H., Tortato, M.A., Oliveira-Santos, L.G., Fantacini, F.M., Luiz, M.R., Castilho, P.V. de, Ribeiro, M.C. and Cáceres, N.C. (2017). Forest cover influences occurrence of mammalian carnivores within Brazilian Atlantic Forest. *Journal of Mammalogy*, 98(6), pp.1721–1731. https://doi.org/10.1093/jmammal/gyx103

River Health Programme (2004). State-of-Rivers Report: Berg River System. Department of Water Affairs and Forestry. Pretoria.

Rivers-Moore, N.A., Dallas, H.F., De Moor, F.C. and Barendse, J. (2018). Relationships of water temperature and aquatic macroinvertebrate community structure with non-native riparian plant densities in the southern Cape, South Africa. *African Journal of Aquatic Science*, 43(3), pp.215–227. https://doi.org/10.2989/16085914.2018.1491385

Roberts, P.D., Somers, M.J., White, R.M. and Nel, J.A.J. (2007). Diet of the South African large-spotted genet *Genetta tigrina* (Carnivora, Viverridae) in a coastal dune forest. *Acta Theriologica*, 52(1), pp.45–53. https://doi.org/10.1007/bf03194198.

Roberts, N.J. (2011). Investigation into survey techniques of large mammals: surveyor competence and camera-trapping vs. transect-sampling. *Bioscience Horizons: The International Journal of Student Research*, 4(1), pp.40–49. https://doi.org/10.1093/biohorizons/hzr006

Rogan, M.S., Balme, G.A., Distiller, G., Pitman, R.T., Broadfield, J., Mann, H., Whittington-Jones, G., Thomas, L.H. and O'Riain, M.J. (2019). The influence of movement on the occupancy–density relationship at small spatial scales. *Ecosphere*, 10(8), e02807. https://doi.org/10.1002/ecs2.2807.

Roessig, J.M., Woodley, C.M., Cech Jr, J.J. and Hansen, L.J. (2004). Effects of global climate change on marine and estuarine fishes and fisheries. *Reviews in Fish*

Biology and Fisheries, 14(2), pp.251–275. https://doi.org/10.1007/s11160-004-6749-0.

Roos, A.M., Bäcklin, B.M.V., Helander, B.O., Rigét, F.F. and Eriksson, U.C. (2012). Improved reproductive success in otters (*Lutra lutra*), grey seals (*Halichoerus grypus*) and sea eagles (*Haliaeetus albicilla*) from Sweden in relation to concentrations of organochlorine contaminants. *Environmental Pollution*, 170, pp. 268–275. https://doi.org/10.1016/j.envpol.2012.07.017

Røskraft, E., Händel, B., Bjerke, T. and Kaltenborn, B.P. (2007). Human attitudes towards large carnivores in Norway. *Wildlife Biology*, 13(2), pp.172–185. https://doi.org/10.2981/0909-6396(2007)13[172:HATLCI]2.0.CO;2

Rota, C.T., Fletcher Jr, R.J., Dorazio, R.M. and Betts, M.G. (2009). Occupancy estimation and the closure assumption. *Journal of Applied Ecology*, 46(6), pp.1173-1181. https://doi.org/10.1111/j.1365-2664.2009.01734.x

Rovero, F. and Marshall, A.R. (2009). Camera trapping photographic rate as an index of density in forest ungulates. *Journal of Applied Ecology*, 46(5), pp.1011–1017. https://doi.org/10.1111/j.1365-2664.2009.01705.x.

Rovero, F. and Zimmermann, F. (2016). Camera features related to specific ecological applications. In: Rovero, F. and Zimmermann, F. (eds.) *Camera Trapping for Wildlife Research*. Exeter, UK: Pelagic Publishing, pp.8–21.

Rowcliffe, J.M., Carbone, C., Kays, R., Kranstauber, B. and Jansen, P.A. (2008). Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45(4), pp.1228–1236. https://doi.org/10.1111/j.1365-2664.2008.01473.x

Rowcliffe, J.M., Kays, R., Kranstauber, B., Carbone, C. and Jansen, P.A. (2014). Quantifying levels of animal activity using camera trap data. *Methods in Ecology and Evolution*, 5(11), pp.1170–1179. https://doi.org/10.1111/2041-210x.12278.

Rowcliffe, J.M. (2017). Key frontiers in camera trapping research. *Remote Sensing in Ecology and Conservation*, 3(3), pp.107–108. https://doi.org/10.1002/rse2.65

Royle, J.A. and Dorazio, R.M. (2008). *Hierarchical modeling and inference in ecology: The analysis of data from populations, metapopulations and communities*. Amsterdam: Elsevier. [Online]. Available at: https://www.sciencedirect.com/book/monograph/9780123740977/hierarchical-modeling-and-inference-in-ecology [Accessed 7 October 2025].

Royle, J.A. and Nichols, J.D. (2003). Estimating abundance from repeated presence–absence data or point counts. *Ecology*, 84(3), pp.777–790. https://doi.org/10.1890/0012-9658(2003)084[0777:EAFRPA]2.0.CO;2

Saad, D., Chauke, P., Cukrowska, E., Richards, H., Nikiema, J., Chimuka, L. and Tutu, H. (2022). First biomonitoring of microplastic pollution in the Vaal river using Carp fish (*Cyprinus carpio*) 'as a bio-indicator'. *Science of The Total Environment*, 836, 155623. https://doi.org/10.1016/j.scitotenv.2022.155623

Sabo, J.L., Sponseller, R., Dixon, M., Gade, K., Harms, T., Heffernan, J., Jani, A., Katz, G., Soykan, C., Watts, J. and Welter, J. (2005). Riparian zones increase regional species richness by harboring different, not more, species. *Ecology*, 86(1), pp.56–62. https://doi.org/10.1890/04-0668

Saklaurs, M., Liepiņa, A.A., Elferts, D. and Jansons, Ā. (2022). Social perception of riparian forests. *Sustainability*, 14(15), p.9302. https://doi.org/10.3390/su14159302.

Santini, L., Benítez-López, A., Maiorano, L., Čengić, M. and Huijbregts, M.A.J. (2021). Assessing the reliability of species distribution projections in climate change research. *Diversity and Distributions*, 27(6), pp.1035–1050. https://doi.org/10.1111/ddi.13252

Santos, M.J., Matos, H.M., Palomares, F. and Santos-Reis, M. (2011). Factors affecting mammalian carnivore use of riparian ecosystems in Mediterranean climates. *Journal of Mammalogy*, 92(5), pp.1060–1069. https://doi.org/10.1644/10-mamm-a-009.1.

Sapolsky, R.M., Romero, L.M. and Munck, A.U. (2000). How do glucocorticoids influence stress responses? Integrating permissive, suppressive, stimulatory, and preparative actions*. *Endocrine Reviews*, 21(1), pp.55–89. http://dx.doi.org/10.1210/er.21.1.55

Schachtschneider, K. and Reinecke, K. (2014). Riparian trees as common denominators across the river flow spectrum: are ecophysiological methods useful tools in environmental flow assessments? *Water SA*, 40(2), pp.287–296. [Online]. Available at: https://www.scielo.org.za/scielo.php?script=sci_arttext&pid=S1816-79502014000200018 [Accessed 2 December 2024].

Schäfer, B., Heppell, C.M., Rhys, H. and Beck, C. (2021). Fluctuations of water quality time series in rivers follow superstatistics. *iScience*, 24(8), 102881. https://doi.org/10.1016/j.isci.2021.102881

Schaus, J., Uzal, A., Gentle, L.K., Baker, P.J., Bearman-Brown, L., Bullion, S., Gazzard, A., Lockwood, H., North, A., Reader, T., Scott, D.M., Sutherland, C.S. and Yarnell, R.W. (2020). Application of the random encounter model in citizen science projects to monitor animal densities. *Remote Sensing in Ecology and Conservation*, 6(4), pp.514–528. https://doi.org/10.1002/rse2.153

Schwacke, L.H., Gulland, F. and White, S.W. (2013). Sentinel species in oceans and human health. *Springer eBooks*, pp.503–528. https://doi.org/10.1007/978-1-4614-5764-0_18

Scott-Shaw, B.C., Everson, C.S. and Clulow, A.D. (2017). Water-use dynamics of an alien-invaded riparian forest within the Mediterranean climate zone of the Western Cape, South Africa. *Hydrology and Earth System Sciences*, 21(9), pp.4551–4562. https://doi.org/10.5194/hess-21-4551-2017.

Scott-Shaw, B.C. and Everson, C.S. (2019). Water-use dynamics of an alien-invaded riparian forest within the summer rainfall zone of South Africa. *Hydrology and Earth System Sciences*, 23(3), pp.1553–1565. https://doi.org/10.5194/hess-23-1553-2019

Seeliger, L., De Clercq, W.P., Hoffmann, W., Cullis, J.D., Horn, A.M. and De Witt, M. (2018). Applying the water-energy-food nexus to farm profitability in the middle Breede Catchment, South Africa. *South African Journal of Science*, 114(11–12), pp.1–10. https://doi.org/10.17159/sajs.2018/5062

Seidlitz, A., Bryant, K.A., Armstrong, N.J., Calver, M.C. and Wayne, A.F. (2021). Sign surveys can be more efficient and cost effective than driven transects and camera trapping: A comparison of detection methods for a small elusive mammal, the numbat (*Myrmecobius fasciatus*). *Wildlife Research*, 48(6), pp.491–500. https://doi.org/10.1071/WR20020.

Sereno-Cadierno, J., Hofmeester, T.R., Becker, M., Bernard, A., Moolman, L., Fritz, H. and Acevedo, P. (2026). Knee height is often right: evaluating device height effects on camera trapping rate. *Remote Sensing in Ecology and Conservation*, https://doi.org/10.1002/rse2.70053

Seydack, A.H., Huisamen, J. and Kok, R. (1998). Long-term antelope population monitoring in southern Cape forests. *Southern African Forestry Journal*, 182(1), pp.9–19. [Online]. Available at: https://www.tandfonline.com/doi/abs/10.1080/10295925.1998.9631185 [Accessed 24 November 2025].

Seydack, A.H.W. (2017). Bushpig *Potamochoerus larvatus* (F. Cuvier, 1822). In: Melletti, M. and Meijaard, E. (eds.), *Ecology, Conservation and Management of Wild Pigs and Peccaries*, pp.122–133. Cambridge University Press. https://doi.org/10.1017/9781316941232.014

Shelton, J.M., Weyl, O.L.F., Chakona, A., Ellender, B.R., Esler, K.J., Impson, N.D., Jordaan, M.S., Marr, S.M., Ngobela, T., Paxton, B.R., Van Der Walt, J.A. and Dallas, H.F. (2017). Vulnerability of Cape Fold Ecoregion freshwater fishes to climate change and other human impacts. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 28(1), pp.68–77. https://doi.org/10.1002/aqc.2849

Sherley, R.B., Ludynia, K., Dyer, B.M., Crawford, R.J.M., Makhado, A.B., Roux, J.-P., Waller, L.J. and Underhill, L.G. (2022). Delayed trophic response of a marine predator to ocean condition and prey availability during the past century. Global Change Biology, 28, pp.6350–6366. https://doi.org/10.1111/gcb.16338

Shrestha, M.K., York, A.M., Boone, C.G. and Zhang, S. (2012). Land fragmentation due to rapid urbanization in the Phoenix Metropolitan Area: Analyzing the spatiotemporal patterns and drivers. *Applied Geography*, 32(2), pp.522–531. https://doi.org/10.1016/j.apgeog.2011.04.004

Silveira, L., Jácomo, A.T.A. and Diniz-Filho, J.A.F. (2003). Camera trap, line transect census and track surveys: A comparative evaluation. *Biological Conservation*, 114(3), pp.351–355. https://doi.org/10.1016/s0006-3207(03)00063-6

Simberloff, D. (1998). Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? *Biological Conservation*, 83(3), pp.247–257. https://doi.org/10.1016/S0006-3207(97)00081-5.

Simelane, F.N., Mahlaba, T.A.M., Shapiro, J.T., MacFadyen, D. and Monadjem, A. (2018). Habitat associations of small mammals in the foothills of the Drakensberg Mountains, South Africa. Mammalia, 82(2), pp.144–152. https://doi.org/10.1515/mammalia-2016-0130

Sinclair, A.R.E., Fryxell, J.M. and Caughley, G. (2006). *Wildlife Ecology, Conservation, and Management*. 2nd ed. Blackwell Publishing, Oxford, UK.

Singh, A., Singh, S., Mehta, P., Jatav, S.K. and Pathak, K.N. (2025). Water pollution and their impact on aquatic animals. In: *Sustainable Aquaculture & Fisheries: From Innovation to Conservation*, pp. 95–113. [Online]. Available at: https://www.researchgate.net/publication/391248329_Water_Pollution_and_Their_Impact_on_Aquatic_Animals [Accessed 27 November 2025].

Skelly, B., Lewis, K., Tyl, R., Dimmig, G. and Rota, C. (2018). *Occupancy models – Multi-species*. In: Cooch, E. and White, G. (eds.). *Program MARK: A Gentle Introduction*. [Online]. Available at: http://www.phidot.org/software/mark/docs/book/pdf/chap22.pdf [Accessed 5 August 2024].

Skinner, J.D., Breytenbach, G.J. and CTA, M. (1976). Observations on the ecology and biology of the bushpig *Potamochoerus porcus* Linn. in the northern Transvaal. *South African Journal of Wildlife Research*, *6*(2), pp.123-128. Available at: https://journals.co.za/doi/pdf/10.10520/AJA03794369_3217 [Accessed 7 March 2024].

Skinner, J.D. and Smithers, R.H.N. (1990). *The mammals of the Southern African subregion*. 2nd ed. Pretoria: University of Pretoria.

Skinner, J.D. and Chimimba, C.T. (2005). *The Mammals of the Southern African Sub-region*. 3rd ed. *Cambridge University Press*. Cambridge: Cambridge University Press. [Online]. Available at: https://www.cambridge.org/core/books/mammals-of-the-southern-african-subregion/5C2E1D1A9BD464C8A0F3D5A550595AFA. [Accessed 12 April 2024].

Skowno, A.L., Poole, C.J., Raimondo, D.C., Sink, K.J., Van Deventer, H., Van Niekerk, L., Harris, L.R., Smith-Adao, L.B., Tolley, K.A., Zengeya, T.A., Foden, W.B., Midgley, G.F. and Driver, A. (2019). *National Biodiversity Assessment 2018: The status of South Africa's ecosystems and biodiversity. Synthesis Report*. Pretoria: South African National Biodiversity Institute. [Online]. Available at: https://www.sanbi.org/wp-content/uploads/2024/06/2019_NBA-Synthesis-report.pdf [Accessed 24 November 2025].

Slater, K. and Du Toit, J.T., 2002. Seed dispersal by Chacma baboons and syntopic ungulates in southern African savannas. *South African Journal of Wildlife Research-24-month delayed open access*, *32*(1), pp.75-79. Available at: https://hdl.handle.net/10520/EJC117136 [Accessed 20 May 2025].

Slater, K., Barrett, A. and Brown, L.R. (2018). Home range utilization by chacma baboon (Papio ursinus) troops on Suikerbosrand Nature Reserve, South Africa. *PLOS ONE*, 13(3), e0194717. https://doi.org/10.1371/journal.pone.0194717

Sollmann, R. (2018). A gentle introduction to camera-trap data analysis. *African Journal of Ecology*, 56(4), pp.740–749. https://doi.org/10.1111/aje.12557

Sollmann, R., Mohamed, A., Samejima, H. and Wilting, A. (2013). Risky business or simple solution – relative abundance indices from camera-trapping. *Biological Conservation*, 159, pp.405–412. https://doi.org/10.1016/j.biocon.2012.12.025

Somers, M.J. and Nel, J.A.J. (2004). Habitat selection by the African clawless otter (*Aonyx capensis*) in rivers in the Western Cape Province, South Africa. *African Journal of Ecology*, 42(4), pp.298–305. https://doi.org/10.1111/j.1365-2028.2004.00526.x

Somers, M.J. and Nel, J.A.J. (2007). Distribution and habitat choice of African clawless otters (*Aonyx capensis*) in South Africa. *South African Journal of Wildlife Research*, 37(1), pp.61–70. Available at: https://journals.co.za/doi/pdf/10.10520/EJC117258 [Accessed 20 October 2025].

Somers, M.J. and Nel, J.A.J. (2013). *Aonyx capensis*. In: J. Kingdon and M. Hoffmann, eds. *Mammals of Africa. Volume V: Carnivores, Pangolins, Equids and Rhinoceroses*. London: Bloomsbury Publishing

South African National Biodiversity Institute (SANBI) (2018). *The Vegetation Map of South Africa, Lesotho and Swaziland, VEGMAP 2018* [dataset]. Version 2018. [Online]. Available at: http://metadata.sanbi.org/srv/api/records/c6aa4ee1-8f13-49d1-980d-2b74cad4b77b [Accessed 26 September 2025].

Specht, H.M., Reich, H.T., Iannarilli, F., Edwards, M.R., Stapleton, S.P., Weegman, M.D., Johnson, M.K., Yohannes, B.J. and Arnold, T.W. (2017). Occupancy surveys with conditional replicates: An alternative sampling design for rare species. *Methods in Ecology and Evolution*, 8(12), pp.1725–1734. https://doi.org/10.1111/2041-210x.12842

Srinivasan, R. and Sorial, G.A. (2011). Treatment of taste and odor causing compounds 2-methyl isoborneol and geosmin in drinking water: A critical review. *Journal of Environmental Sciences*, 23(1), pp.1–13. https://doi.org/10.1016/s1001-0742(10)60367-1.

Stahl, R.G. (1997). Can mammalian and non-mammalian 'sentinel species'; data be used to evaluate the human health implications of environmental contaminants? *Human and Ecological Risk Assessment: An International Journal*, 3(3), pp.329–335. https://doi.org/10.1080/10807039709383689

Steenweg, R., Hebblewhite, M., Whittington, J. and McKelvey, K. (2019). Species-specific differences in detection and occupancy probabilities help drive ability to

detect trends in occupancy. *Ecosphere*, 10(4), e02639. https://doi.org/10.1002/ecs2.2639

Stevens, S., Organ, J.F. and Serfass, T.L. (2011). Otters as flagships: Social and cultural considerations. *IUCN Otter Specialist Group Bulletin*, 28(A), pp.150–161. [Online] Available at: https://www.iucnosgbull.org/Volume28A/Stevens_et_al_2011.html [Accessed 24 April 2024].

Stirnemann, I., Mortelliti, A., Gibbons, P. and Lindenmayer, D.B. (2015). Fine-scale habitat heterogeneity influences occupancy in terrestrial mammals in a temperate region of Australia. *PLOS ONE*, 10(9), e0138681. https://doi.org/10.1371/journal.pone.0138681

Stoffyn-Egli, P. and Willison, J.H.M. (2011). Including wildlife habitat in the definition of riparian areas: The beaver (*Castor canadensis*) as an umbrella species for riparian obligate animals. *Environmental Reviews*, 19, pp.479–494. https://doi.org/10.1139/a11-019

Streicher, J.P., Ramesh, T. and Downs, C.T. (2021). An African urban mesocarnivore: Navigating the urban matrix of Durban, South Africa. *Global Ecology and Conservation*, 26, e01482. https://doi.org/10.1016/j.gecco.2021.e01482.

Stürck, J. and Verburg, P.H. (2016). Multifunctionality at what scale? A landscape multifunctionality assessment for the European Union under conditions of land use change. *Landscape Ecology*, 32(3), pp.481–500. https://doi.org/10.1007/s10980-016-0459-6

Sullivan, T.P. and Sullivan, D.S. (2001). Influence of variable retention harvests on forest ecosystems. II. Diversity and population dynamics of small mammals. *Journal of Applied Ecology*, 38(6), pp.1234–1252. https://doi.org/10.1046/j.0021-8901.2001.00674.x

Suresh, K., Tang, T., Van Vliet, M., Bierkens, M., Strokal, M., Florian Sorger-Domenigg and Wada, Y. (2023). Recent advancement in water quality indicators for eutrophication in global freshwater lakes. 18(6), 063004. https://doi.org/10.1088/1748-9326/acd071

Swan, M., Christie, F., Steel, E., Sitters, H., York, A. and Di Stefano, J. (2020). Ground-dwelling mammal diversity responds positively to productivity and habitat heterogeneity in a fire-prone region. *Ecosphere*, 11(9). https://doi.org/10.1002/ecs2.3248.

Swanepoel, S., Crafford, D. and van Staden, S. 2014. *Aquatic bio-monitoring study of selected river sites upstream and downstream of the existing N3 route*. Scientific Aquatic Services CC, Report SAS 214013-1, prepared for Iliso Consulting. [Online]. Available at: https://www.acerafrica.co.za/debeers/Appendix%20T%20-%20Aquatic%20Biomonitoring%20on%20the%20Existing%20N3/Appendix%20T%20Aquatic%20Final%20report%2009052014.pdf [Accessed 30 December 2025].

Syse, K.V.L. (2013). Otters as symbols in the British environmental discourse. *Landscape Research*, 38(4), pp.540–552. https://doi.org/10.1080/01426397.2013.784244

Talmage, P.J., Perry, J.A. and Goldstein, R.M. (2002). Relation of instream habitat and physical conditions to fish communities of agricultural streams in the northern midwest. *North American Journal of Fisheries Management*, 22(3), pp.825–833. https://doi.org/10.1577/1548-8675(2002)022%3C0825:roihap%3E2.0.co;2

Tanwar, K.S., Sadhu, A. and Jhala, Y.V. (2021). Camera trap placement for evaluating species richness, abundance, and activity. *Scientific Reports*, 11(1), 23050. https://doi.org/10.1038/s41598-021-02459-w

Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M.C., Schwager, M. and Jeltsch, F. (2003). Animal species diversity driven by habitat heterogeneity/diversity: The importance of keystone structures. *Journal of Biogeography*, 31(1), pp.79–92. https://doi.org/10.1046/j.0305-0270.2003.00994.x

Thompson, G.G. and Thompson, S.A. (2007). Using species accumulation curves to estimate trapping effort in fauna surveys and species richness. *Austral Ecology*, 32(5), pp.564–569. https://doi.org/10.1111/j.1442-9993.2007.01728.x

Tickner, D.P., Angold, P.G., Gurnell, A.M. and Mountford, J.O. (2001). Riparian plant invasions: hydrogeomorphological control and ecological impacts. *Progress in Physical Geography: Earth and Environment*, 25(1), pp.22–52. https://doi.org/10.1177/030913330102500102

Tobler, M.W., Carrillo-Percastegui, S.E., Leite Pitman, R., Mares, R. and Powell, G. (2008). An evaluation of camera traps for inventorying large- and medium-sized terrestrial rainforest mammals. *Animal Conservation*, 11(3), pp.169–178. https://doi.org/10.1111/j.1469-1795.2008.00169.x

Tobler, M.W. and Powell, G.V.N. (2013). Estimating jaguar densities with camera traps: Problems with current designs and recommendations for future studies. *Biological Conservation*, 159, pp.109–118. https://doi.org/10.1016/j.biocon.2012.12.009.

Tobler, M.W., Zúñiga Hartley, A., Carrillo-Percastegui, S.E. and Powell, G.V.N. (2015). Spatiotemporal hierarchical modelling of species richness and occupancy using camera trap data. *Journal of Applied Ecology*, 52(2), pp.413–421. https://doi.org/10.1111/1365-2664.12399

Turshak, L.G., Mwansat, G.S. and Manu, S.A. (2011). Effects of habitat structure and altitudinal gradients on avian species diversity at Kurra Falls Forest, Nigeria. *The Zoologist*, 9, pp.31–37. [Online]. Available at: https://www.semanticscholar.org/paper/Effects-of-habitat-structure-and-altitudinal-on-at-Turshak-Mwansat/a22d7f7985911060daccf50b84091742b5befee2 [Accessed 20 June 2025].

Tweddle, D., Bills, R., Swartz, E., Coetzer, W., da Costa, L., Engelbrecht, J. et al. (2009). The status and distribution of freshwater fishes. In: Darwall, W.R.T., Smith, K.G., Tweddle, D. and Skelton, P.H. (eds.) *The Status and Distribution of Freshwater Biodiversity in Southern Africa*. Gland: IUCN and the South African Institute for Aquatic Biodiversity, pp. 21–37. [Online]. Available at: https://portals.iucn.org/library/sites/library/files/documents/RL-68-003.pdf [Accessed 20 April 2024].

United Nations World Water Assessment Programme (2017). *The United Nations World Water Development Report 2017: Wastewater – The Untapped Resource*. UNESCO on behalf of UN-Water. [Online]. Available: https://www.unido.org/sites/default/files/2017-03/UN_World_Water_Development_Report_-_Full_0.pdf [Accessed 25 August 2024].

U.S. Fish and Wildlife Service (2019). *Riparian habitat classification and mapping*. FWS.gov. [Online]. Available at: https://www.fws.gov/program/national-wetlands-inventory/riparian-data [Accessed 21 October 2025].

Ulrich, W. and Ollik, M. (2005). Limits to the estimation of species richness: The use of relative abundance distributions. *Diversity and Distributions*, 11(3), pp.265–273. https://doi.org/10.1111/j.1366-9516.2005.00127.x.

United Nations Educational, Scientific and Cultural Organization (UNESCO) (2025). *Cape Floral Region Protected Areas, South Africa*. [Online] Available at: https://whc.unesco.org/en/list/1007/ [Accessed 25 November 2025].

Venter, J., Ehlers Smith, Y. and Seydack, A., 2016. *A conservation assessment of Potamochoerus larvatus*. The Red List of Mammals of South Africa, Swaziland

and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa. [Online]. Available at: https://speciesstatus.sanbi.org/assessment/last-assessment/2331/. [Accessed 7 October 2025].

Verry, E.S., Dolloff, C.A. and Manning, M.E. (2004). Riparian ecotone: A functional definition and delineation for resource assessment. *Water, Air and Soil Pollution: Focus*, 4(1), pp.67–94. https://doi.org/10.1023/B:WAFO.0000012825.77300.08.

Villar-Rúa, S., Acuña-Alonso, C. and Álvarez, X. (2024). Estimation of the ecological integrity of the Guadiana river using partial least squares path modelling and simulation scenarios. *Science of The Total Environment*, 954, 176305. https://doi.org/10.1016/j.scitotenv.2024.176305

Viljoen, G. and van der Walt, K. (2018). South Africa's water crisis - an interdisciplinary approach. *Tydskrif vir Geesteswetenskappe*, 58(3), pp.483–500. https://doi.org/10.17159/2224-7912/2018/v58n3a3

Watanabe, S. (2010). Asymptotic equivalence of Bayes cross-validation and widely applicable information criterion in singular learning theory. *Journal of Machine Learning Research*, 11, pp.3571–3594. [Online]. Available at: https://www.jmlr.org/papers/volume11/watanabe10a/watanabe10a.pdf [Accessed 5 October 2025].

Watson, M.L., Baehr, L.M., Reichardt, H.M., Tuckermann, J.P., Bodine, S.C. and Furlow, J.D. (2012). A cell-autonomous role for the glucocorticoid receptor in skeletal muscle atrophy induced by systemic glucocorticoid exposure. *American Journal of Physiology-Endocrinology and Metabolism*, 302(10), E1210. https://doi.org/10.1152/ajpendo.00512.2011

Wearn, O.R. and Glover-Kapfer, P. (2017). *Camera-trapping for conservation: a guide to best-practices* (WWF-UK, Woking, UK). WWF Conservation Technology Series., 1(1), 181 pp. [Online]. Available at: https://www.wwf.org.uk/sites/default/files/2019-04/CameraTraps-WWF-guidelines.pdf [Accessed 5 October 2025].

Wearn, O.R., Bell, T.E.M., Bolitho, A., Durrant, J., Haysom, Jessica K., Nijhawan, S., Thorley, J. and Rowcliffe, J.M. (2022). Estimating animal density for a community of species using information obtained only from camera-traps. *Methods in Ecology and Evolution*, 13(10), pp.2248-2261. https://doi.org/10.1111/2041-210x.13930

Weinberger, I.C., Muff, S., Kranz, A. and Bontadina, F. (2019). Riparian vegetation provides crucial shelter for resting otters in a human-dominated landscape.

Mammalian Biology, 98, pp.179–187. https://doi.org/10.1016/j.mambio.2019.09.001

Welsh, A.H., Lindenmayer, D.B. and Donnelly, C.F. (2013). Fitting and Interpreting Occupancy Models. *PLoS ONE*, 8(1), e52015. https://doi.org/10.1371/journal.pone.0052015

Wetzel, R.G. (2001). *Limnology: Lake and River Ecosystems*. 3rd edition. San Diego, CA: Academic Press.

Whipkey, D., Narr, C., Pease, B.S. and Bastille-Rousseau, G. (2025). Terrestrial and aquatic drivers of occupancy in four semiaquatic mammals. Ecosphere, 16(9), e70386. https://doi.org/10.1002/ecs2.70386

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag New York. ISBN 978-3-319-24277-4. Available at: ggplot2 website.

Wickham, H., François, R., Henry, L., Müller, K. and Vaughan, D. (2026). *dplyr: A Grammar of Data Manipulation*. R package version 1.2.1. Available at: CRAN dplyr package page.

Widdows, C.D. and Downs, C.T. (2015). A genet drive-through: are large spotted genets using urban areas for 'fast food'? a dietary analysis. *Urban Ecosystems*, 18(3), pp.907–920. https://doi.org/10.1007/s11252-015-0438-8

Widdows, C.D., Ramesh, T. and Downs, C.T. (2015). Factors affecting the distribution of large spotted genets (*Genetta tigrina*) in an urban environment in South Africa. *Urban Ecosystems*, 18(4), pp.1401–1413. https://doi.org/10.1007/s11252-015-0449-5

Williamson, J.R., Slade, E.M., Luke, S.H., Swinfield, T., Arthur C.K. Chung, Coomes, D.A., Herry Heroin, Tommaso Jucker, Lewis, O.T., Charles Santhanaraju Vairappan, Rossiter, S.J. and Struebig, M.J. (2021). Riparian buffers act as microclimatic refugia in oil palm landscapes. *Journal of Applied Ecology*, 58(2), pp.431–442. https://doi.org/10.1111/1365-2664.13784

Wolff, N.M., Ferreguetti, Á.C., Moraes, W.T. and Bergallo, H.G. (2019). Population density, activity pattern and habitat use of the ocelot *Leopardus pardalis* in an Atlantic Forest protected area, Southeastern Brazil. *Hystrix-italian Journal of Mammalogy*, 30(2), pp.120–125. https://doi.org/10.4404/hystrix-00214-2019.

Wright, W.J., Irvine, K.M. and Rodhouse, T.J. (2016). A goodness-of-fit test for occupancy models with correlated within-season revisits. *Ecology and Evolution*, 6(15), pp.5404–5415. https://doi.org/10.1002/ece3.2292

Xiang, H., Zhang, Y. and Richardson, J.S. (2017). Importance of riparian zone: effects of resource availability at land-water interface. *Riparian Ecology and Conservation*, 3(1), pp.1-17. https://doi.org/10.1515/remc-2016-0001

Yamaura, Y., Royle, A.J., Kuboi, K., Tada, T., Ikeno, S. and Makino, S. (2010). Modelling community dynamics based on species-level abundance models from detection/nondetection data. *Journal of Applied Ecology*, 48(1), pp.67–75. https://doi.org/10.1111/j.1365-2664.2010.01922.x.

Yoccoz, N.G. (2012). Ecological Monitoring. In: *Encyclopedia of Life Sciences (eLS)*. Chichester, UK: John Wiley & Sons. https://doi.org/10.1002/9780470015902.a0023571.

Zacharias, M.A. and Roff, J.C. (2001). Use of focal species in marine conservation and management: a review and critique. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 11(1), pp.59–76. https://doi.org/10.1002/aqc.429

Zamrsky, D., Essink, O. and Bierkens, M. (2024). Global impact of sea level rise on coastal fresh groundwater resources. *Earth's Future*, 12(1), e2023EF003581. https://doi.org/10.1029/2023ef003581.

Zipkin, E.F., DeWan, A. and Royle, A.J. (2009). Impacts of forest fragmentation on species richness: A hierarchical approach to community modelling. *Journal of Applied Ecology*, 46(4), pp.815–822. https://doi.org/10.1111/j.1365-2664.2009.01664.x

Zipkin, E.F., Thorson, J.T., See, K., Lynch, H.J., Campbell, E.H., Kanno, Y., Chandler, R.B., Letcher, B.H. and Royle, J.A. (2013). Modeling structured population dynamics using data from unmarked individuals. *Ecology*, 95(1), pp.22–29. https://doi.org/10.1890/13-1131.1

Zimbres, B., Peres, C.A. and Machado, R.B. (2017). Terrestrial mammal responses to habitat structure and quality of remnant riparian forests in an Amazonian cattle-ranching landscape. *Biological Conservation*, 206, pp.283–292. https://doi.org/10.1016/j.biocon.2016.11.033

Zungu, M.M., Maseko, M.S.T., Kalle, R., Ramesh, T. and Downs, C.T. (2020). Factors affecting the occupancy of forest mammals in an urban-forest mosaic in EThekwini Municipality, Durban, South Africa. *Urban Forestry & Urban Greening*, 48, 126562. https://doi.org/10.1016/j.ufug.2019.126562

Zurell, D., Fandos, G., Malchow, A.-K. and Reeg, J. (2023). *Simple occupancy models*. [Online]. Available at: https://damariszurell.github.io/EEC-QCB/Occ4_SimpleOccModel.html [Accessed 17 June 2025].

Appendix A: Supplementary tables

Appendix A1: Correlation of covariates for winter season

Table A1. Correlation of covariates for the winter season

	Ground veg cover	Canopy cover	SASS	Number of taxa	ASPT	pH	Dissolved O_2	Conductivity	TDS	Temp
Ground veg cover	1.00	-0.28	-0.06	0.06	-0.17	0.08	-0.19	-0.02	-0.02	0.08
Canopy cover	-0.28	1.00	0.18	-0.01	0.39	-0.18	0.40	0.18	0.18	-0.10
SASS	-0.06	0.18	1.00	0.64	-0.01	-0.95	-0.03	-0.34	-0.35	-0.81
Number of taxa	0.06	-0.01	0.64	1.00	-0.47	-0.51	-0.46	-0.05	-0.05	-0.29
ASPT	-0.17	0.39	-0.01	-0.47	1.00	0.03	0.98	0.56	0.55	0.13
pH	0.08	-0.18	-0.95	-0.51	0.03	1.00	0.00	0.53	0.54	0.94
Dissolved O_2	-0.19	0.40	-0.03	-0.46	0.98	0.00	1.00	0.51	0.50	0.08
Conductivity	-0.02	0.18	-0.34	-0.05	0.56	0.53	0.51	1.00	1.00	0.77
Total dissolved solids	-0.02	0.18	-0.35	-0.05	0.55	0.54	0.50	1.00	1.00	0.77
Water temperature	0.08	-0.10	-0.81	-0.29	0.13	0.94	0.08	0.77	0.77	1.00

Appendix A2: Correlation of covariates for the summer season

Table A2. Correlation of covariates for the summer season

	Ground vegetation cover	Canopy cover	SASS	Number of taxa	ASPT	pH	Dissolved O_2	Conductivity	TDS	Temp
Ground veg cover	1.00	-0.02	0.15	0.17	0.12	-0.11	0.04	0.09	0.09	-0.03
Canopy cover	-0.02	1.00	0.01	0.05	-0.08	0.07	0.32	0.36	0.36	0.21
SASS	0.15	0.01	1.00	0.99	0.97	-0.97	-0.50	-0.36	-0.36	-0.79
Number of taxa	0.17	0.05	0.99	1.00	0.93	-0.92	-0.44	-0.24	-0.24	-0.70
ASPT	0.12	-0.08	0.97	0.93	1.00	-1.00	-0.61	-0.56	-0.56	-0.92
pH	-0.11	0.07	-0.97	-0.92	-1.00	1.00	0.60	0.56	0.56	0.92
Dissolved O_2	0.04	0.32	-0.50	-0.44	-0.61	0.60	1.00	0.85	0.85	0.75
Conductivity	0.09	0.36	-0.36	-0.24	-0.56	0.56	0.85	1.00	1.00	0.83
Total dissolved solids	0.09	0.36	-0.36	-0.24	-0.56	0.56	0.85	1.00	1.00	0.83
Watertemperature	-0.03	0.21	-0.79	-0.70	-0.92	0.92	0.75	0.83	0.83	1.00

Appendix A3: Relative abundance index (RAI) of species

Table A3. RAIs for each species per season and location

RAI per seasons total photos	Buffeljags Winter	Grootvadersbosch Winter	Buffeljags Summer	Grootvaderbosch Summer	Total
Bushpig	0,387597	1,500938	0,184162063	0	2,072697
Cape Bushbuck	5,03876	2,251407	6,629834254	7,906977	21,82698
African Clawless Otter	0,968992	0,750469	0,368324125	0	2,087785
Cape Genet	4,069767	0,375235	1,473296501	0	5,918298
Cape Grey Mongoose	0,387597	0	0,368324125	0	0,755921
Cape Porcupine	0,193798	1,313321	0	0	1,507119
Caracal	0,193798	0	0	0	0,193798
Chacma Baboon	16,66667	3,189493	6,445672192	4,186047	30,48788
Honey Badger	0	0,187617	0	0	0,187617

Vervet Monkey	0	0	0,184162063	0	0,184162
Water Mongoose	0,968992	1,125704	0,184162063	5,581395	7,860253
Total	28,87597	10,69418	15,83793738	17,67442	

Appendix A4: Habitat covariates for Buffeljags River sites

Table A4. Habitat covariate results (Buffeljags River sites)

Camera trap name	Bare ground	Rock	Tree	Shrub	Grass	Forb	Reed	Debris	Canopy cover
C1	11,60%	50,30%	0,00%	0,00%	11,90%	0,80%	25,40%	0,00%	4%
C2 survey 1	39,50%	0,00%	0,00%	0,00%	30,00%	21,17%	0,00%	9,33%	50%
C2 survey 2	17,80%	2,00%	0,00%	0,00%	63,60%	10,40%	0,00%	6,20%	35%
C3	5,20%	0,00%	0,00%	0,00%	82,00%	4,00%	0,00%	8,80%	0%
C4	30,00%	0,00%	0,00%	0,00%	0,00%	0,00%	60,00%	10,00%	10%
C5	20,82%	10,46%	0,00%	0,00%	37,50%	7,27%	17,08%	8,58%	6%
C6	1,00%	11,00%	0,00%	0,00%	77,17%	1,67%	9,17%	0,00%	2%
C7	14,60%	9,00%	0,00%	0,00%	56,20%	1,00%	11,60%	7,60%	1%
C8	12,17%	33,36%	0,00%	0,00%	34,22%	1,83%	17,83%	0,58%	0%
C9	2,00%	0,00%	0,00%	0,00%	42,33%	8,00%	47,67%	0,00%	0%
C10 survey 1	32,67%	0,00%	0,00%	0,00%	31,00%	36,17%	0,00%	0,17%	10%
C10 survey 2	32,67%	0,00%	0,00%	0,00%	31,00%	36,17%	0,00%	0,17%	50%
C11	8,00%	40,00%	0,00%	0,00%	20,00%	0,00%	30,00%	2,00%	4%
C12	2,00%	50,00%	0,00%	0,00%	10,00%	0,00%	38,00%	0,00%	0%
C13	3,50%	47,33%	0,00%	0,00%	11,00%	1,33%	36,83%	0,00%	25%
C14 survey 1	12,83%	1,00%	0,00%	0,00%	37,83%	9,50%	0,00%	38,83%	100%
C14 survey 2	14,19%	6,89%	0,00%	0,00%	51,04%	4,31%	2,97%	20,60%	100%
C15 survey 1	64,50%	16,83%	0,00%	0,00%	0,83%	0,00%	17,83%	0,00%	75%
C15 survey 2	7,33%	0,00%	0,00%	4,17%	75,75%	1,17%	10,33%	1,25%	75%
C16	40%	0%	40%	0%	2%	0%	18%	0,00%	97%
C17	31,20%	61,00%	0,00%	0,00%	0,00%	0,00%	7,00%	0,80%	67%
C17 survey 2	39,83%	23,67%	0,00%	0,00%	19,17%	5,17%	11,00%	1,17%	25%
C18	22,60%	1,80%	0,00%	0,00%	72,60%	3,00%	0,00%	0,00%	90%
C19	24,80%	15,80%	0,00%	0,00%	48,20%	0,00%	11,20%	0,00%	10%
C20	17,20%	63,60%	0,00%	0,00%	0,00%	0,00%	18,40%	0,80%	0%

Appendix A5: Habitat covariates for Grootvadersbosch River sites

Table A5. Habitat cover results (Grootvadersbosch River sites)

	Bare ground	Rock	Tree	Shrub	Grass	Forb	Reed	Debris	Canopy cover
C1	7,67%	59,17%	0,00%	0,00%	0,00%	11,00%	0,00%	22,17%	98%
C2	0,00%	0,00%	0,00%	5,75%	94,25%	0,00%	0,00%	0,00%	70%
C3	0,80%	10,60%	0,00%	0,00%	88,40%	0,20%	0,00%	0,00%	0%
C4	7,17%	35,83%	0,00%	0,00%	54,17%	2,83%	0,00%	0,00%	0%
C5	2,40%	3,80%	0,00%	0,00%	83,00%	5,00%	5,80%	0,00%	85%
C6	4,80%	50,00%	0,00%	0,00%	39,60%	0,00%	5,60%	0,00%	60%
C7 survey 1	6,50%	28,00%	0,00%	0,00%	43,83%	0,00%	21,67%	0,00%	25%
C7 survey 2	1,33%	34,17%	1,83%	0,00%	62,67%	0,00%	0,00%	0,00%	80%
C8	2,50%	1,67%	0,00%	0,00%	77,67%	0,00%	17,33%	0,83%	20%
C9 survey 1	13,83%	17,50%	0,00%	0,00%	16,33%	46,33%	5,50%	0,50%	50%
C9 survey 2	0,80%	9,60%	0,00%	0,00%	89,60%	0,00%	0,00%	0,00%	50%
C10 survey 1	2,00%	71,00%	0,00%	0,00%	25,40%	1,60%	0,00%	0,00%	72%
C10 survey 2	2,00%	71,00%	0,00%	0,00%	25,40%	1,60%	0,00%	0,00%	87%
C11 survey 1	25,00%	2,40%	0,00%	0,00%	0,00%	2,00%	69,80%	0,80%	24%
C11 survey 2	15,83%	1,50%	0,00%	0,00%	46,00%	0,50%	36,17%	0,00%	24%
C12	4,50%	12,17%	0,00%	0,00%	61,67%	1,33%	20,33%	0,00%	32%
C13 survey 1	0,00%	0,00%	0,00%	0,00%	100,00%	0,00%	0,00%	0,00%	55%
C13 survey 2	1,50%	6,50%	0,00%	0,00%	82,17%	0,50%	9,33%	0,00%	0%
C14 survey 1	6,80%	84,40%	0,00%	0,00%	8,80%	0,00%	0,00%	0,00%	0%
C14 survey 2	5,83%	0,50%	0,00%	0,00%	93,67%	0,00%	0,00%	0,00%	0%

Table A5. Habitat cover results (Grootvadersbosch River sites) continued									
	Bare ground	**Rock**	**Tree**	**Shrub**	**Grass**	**Forb**	**Reed**	**Debris**	**Canopy cover**
C15	8,00%	8,17%	0,00%	0,00%	56,33%	0,00%	27,00%	0,50%	2%
C16	17,83%	9,50%	0,00%	0,00%	28,17%	0,00%	44,50%	0,00%	0%
C17 survey 1	25,00%	1,83%	0,00%	0,00%	66,33%	6,50%	0,00%	0,33%	0%
C17 survey 2	6,00%	48,40%	0,40%	0,00%	38,00%	1,40%	4,20%	1,60%	0%
C18	22,20%	27,60%	0,00%	0,00%	38,20%	5,60%	5,80%	0,60%	0%
C19	26,50%	17,00%	0,00%	0,00%	40,00%	0,00%	16,50%	0,00%	0%
C20	9,60%	4,00%	0,00%	0,00%	0,00%	0,00%	86,40%	0,00%	20%

Appendix A6: Comparison of single-species and multispecies occupancy estimates for species with the highest detections

Table A6. Comparison of single-species and multispecies occupancy estimates for species with the highest detections

Winter survey					
Species	**Model type**	**Mean ψ**	**95% CI ψ**	**Mean p**	**95% CI p**
Cape Bushbuck	Single-species	0.50	0.23 – 0.84	0.025	0.007 – 0.101
Chacma Baboon	Single-species	0.46	0.24 – 0.71	0.063	0.010 – 0.276
Cape Bushbuck	Multispecies	0.48	0.17–0.87	0.03	0–0.08
Chacma Baboon	Multispecies	0.48	0.16–0.88	0.04	0–0.15
Summer survey					
Species	**Model type**	**Mean ψ**	**95% CI ψ**	**Mean p**	**95% CI p**
Cape Bushbuck	Single-species	0.76	0.40 – 0.96	0.09	0.05 – 0.14
Chacma Baboon	Single-species	0.63	0.30 – 0.92	0.09	0.05 – 0.15
Cape Bushbuck	Multispecies	0.59	0.05–0.99	0.06	0.03–0.10
Chacma Baboon	Multispecies	0.65	0.06–0.99	0.06	0.04–0.09

Appendix A7: Community logit covariate effects

Table A7. Community logit covariate effects

Parameter	Winter Mean (logit)	Winter 95% CrI	Summer Mean (logit)	Summer 95% CrI
Detection probability				
Intercept	-4.8243	-6.1454 – -3.3731	-3.3544	-4.8394 – -1.1105
Effort	1.7491	0.4791 – 3.0397	-0.3785	-1.1939 – 0.4255
Occupancy				
Intercept	-0.1255	-1.2265 – 1.4325	1.4522	-0.0945 – 3.3263
Area Grootvadersbosch	0.1219	-1.2646 – 1.8574	-0.5955	-3.5665 – 2.2123
Canopy cover	-0.0147	-1.0295 – 1.3178	0.3818	-0.8354 – 1.6339
Conductivity	-0.3345	-1.3877 – 0.5947	-0.3746	-2.9150 – 2.5454
Dissolved oxygen	0.4098	-0.9130 – 1.6050	0.4225	-2.2375 – 3.1513
Ground vegetation cover	-0.3854	-1.2788 – 0.5526	0.7219	-0.5183 – 2.4420
SASS	-0.1127	-1.1830 – 0.8928	-0.7956	-2.9319 – 1.3288

Appendix A8: Winter multispecies occupancy – species-level occupancy probabilities

Table A8. Winter species-specific covariate effects

Species	Covariate	Mean coefficient estimate (β, logit scale)	Predicted occupancy probability (ψ)	Lower 95% credible interval (ψ)	Upper 95% credible interval (ψ)
Bushpig	(Intercept)	-0.879	0.335	2.49e-2	0.718
Bushpig	AreaGrootvadersbosch	0.249	0.537	0.127	0.965
Bushpig	Canopy_Cover	0.453	0.589	0.274	0.934
Bushpig	Conductivity	-0.917	0.315	4.09e-2	0.610
Bushpig	Dissolved.oxygen	-0.0421	0.499	7.44e-2	0.843
Bushpig	Ground_vegetation_cover	-0.632	0.368	6.25e-2	0.650
Bushpig	SASS	-0.833	0.339	2.98e-2	0.655
Cape Bushbuck	(Intercept)	0.0229	0.496	0.257	0.872

Table A8. Winter species-specific covariate effects (continued)					
Species	**Covariate**	**Mean coefficient estimate (β, logit scale)**	**Predicted occupancy probability (ψ)**	**Lower 95% credible interval (ψ)**	**Upper 95% credible interval (ψ)**
Cape Bushbuck	AreaGrootvadersbosch	0.752	0.641	0.278	0.971
Cape Bushbuck	Canopy_Cover	0.0736	0.516	0.264	0.786
Cape Bushbuck	Conductivity	-0.851	0.319	5.29e-2	0.550
Cape Bushbuck	Dissolved.oxygen	0.146	0.534	0.233	0.816
Cape Bushbuck	Ground_vegetation_cover	-0.156	0.463	0.253	0.667
Cape Bushbuck	SASS	-0.417	0.405	0.153	0.683
CapeClawless	(Intercept)	0.107	0.472	0.110	0.996
CapeClawless	AreaGrootvadersbosch	0.399	0.551	0.114	0.987
CapeClawless	Canopy_Cover	0.523	0.600	0.244	0.950
CapeClawless	Conductivity	-0.0597	0.484	0.174	0.848
CapeClawless	Dissolved.oxygen	-0.202	0.470	4.94e-2	0.855
CapeClawless	Ground_vegetation_cover	0.321	0.553	0.214	0.960
CapeClawless	SASS	0.359	0.567	0.214	0.955
Cape Genet	(Intercept)	-0.439	0.399	0.173	0.688
Cape Genet	AreaGrootvadersbosch	-1.02	0.303	3.78e-2	0.704
Cape Genet	Canopy_Cover	-0.989	0.292	6.25e-2	0.542
Cape Genet	Conductivity	-0.146	0.465	0.196	0.777
Cape Genet	Dissolved.oxygen	0.334	0.576	0.303	0.828
Cape Genet	Ground_vegetation_cover	-0.937	0.300	7.25e-2	0.513
Cape Genet	SASS	0.204	0.543	0.294	0.844
Cape Porcupine	(Intercept)	0.251	0.509	0.145	0.995
Cape Porcupine	AreaGrootvadersbosch	1.15	0.642	1.06e-1	0.999

Table A8. Winter species-specific covariate effects (continued)					
Species	**Covariate**	**Mean coefficient estimate (β, logit scale)**	**Predicted occupancy probability (ψ)**	**Lower 95% credible interval (ψ)**	**Upper 95% credible interval (ψ)**
Cape Porcupine	Canopy_Cover	0.187	0.533	0.207	0.921
Cape Porcupine	Conductivity	0.00858	0.497	0.156	0.899
Cape Porcupine	Dissolved.oxygen	0.940	0.666	0.236	0.985
Cape Porcupine	Ground_vegetation_cover	-0.258	0.444	0.127	0.777
Cape Porcupine	SASS	0.206	0.539	0.162	0.906
Chacma Baboon	(Intercept)	-0.158	0.462	0.262	0.693
Chacma Baboon	AreaGrootvadersbosch	-0.913	0.311	6.44e-2	0.639
Chacma Baboon	Canopy_Cover	-0.396	0.406	0.221	0.606
Chacma Baboon	Conductivity	-0.390	0.410	0.177	0.631
Chacma Baboon	Dissolved.oxygen	1.46	0.793	0.592	0.954
Chacma Baboon	Ground_vegetation_cover	-0.312	0.425	0.262	0.603
Chacma Baboon	SASS	0.0457	0.509	0.289	0.770
Water Mongoose	(Intercept)	0.551	0.587	0.278	0.986
Water Mongoose	AreaGrootvadersbosch	0.696	0.609	0.186	0.992
Water Mongoose	Canopy_Cover	-0.0916	0.478	0.185	0.814
Water Mongoose	Conductivity	-0.121	0.470	0.181	0.789
Water Mongoose	Dissolved.oxygen	0.517	0.609	0.255	0.911
Water Mongoose	Ground_vegetation_cover	-0.923	0.309	5.08e-2	0.553
Water Mongoose	SASS	-0.366	0.427	7.11e-2	0.714

Appendix A9: Summer multispecies occupancy – species-level occupancy probabilities

Table A9. Summer species-specific covariate effects

Species	Covariate	Mean coefficient estimate (β, logit scale)	Predicted occupancy probability (ψ)	Lower 95% credible interval (ψ)	Upper 95% credible interval (ψ)
Bushpig	(Intercept)	1.22	0.720	7.96e-2	0.986
Bushpig	AreaGrootvadersbosch	-0.195	0.439	3.34e-5	1.000
Bushpig	Canopy_Cover	0.495	0.596	1.85e-1	0.942
Bushpig	Conductivity	-0.111	0.454	4.19e-2	0.991
Bushpig	Dissolved.oxygen	0.686	0.591	5.30e-2	0.995
Bushpig	Ground_vegetation_cover	0.800	0.655	2.08e-1	0.961
Bushpig	SASS	-1.16	0.336	1.07e-3	0.922
Cape Bushbuck	(Intercept)	1.85	0.820	5.56e-1	0.991
Cape Bushbuck	AreaGrootvadersbosch	1.09	0.513	2.68e-2	1.000
Cape Bushbuck	Canopy_Cover	0.441	0.597	3.27e-1	0.885
Cape Bushbuck	Conductivity	-0.781	0.354	3.33e-2	0.915
Cape Bushbuck	Dissolved.oxygen	0.0485	0.514	4.14e-2	0.953
Cape Bushbuck	Ground_vegetation_cover	0.611	0.627	2.68e-1	0.934
Cape Bushbuck	SASS	-0.643	0.376	3.33e-2	0.854
African Clawless Otter	(Intercept)	1.31	0.737	2.55e-1	0.978
African Clawless Otter	AreaGrootvadersbosch	-3.53	0.252	1.43e-8	0.990
African Clawless Otter	Canopy_Cover	0.507	0.603	2.20e-1	0.935
African Clawless Otter	Conductivity	-0.209	0.461	2.50e-2	0.976
African Clawless Otter	Dissolved.oxygen	0.851	0.621	1.01e-1	0.993

Table A9. Summer species-specific covariate effects (continued)					
Species	**Covariate**	**Mean coefficient estimate (β, logit scale)**	**Predicted occupancy probability (ψ)**	**Lower 95% credible interval (ψ)**	**Upper 95% credible interval (ψ)**
African Clawless Otter	Ground_vegetation_cover	0.775	0.655	2.43e-1	0.944
African Clawless Otter	SASS	-2.20	0.277	8.61e-8	0.798
Cape Genet	(Intercept)	1.85	0.810	4.91e-1	0.995
Cape Genet	AreaGrootvadersbosch	-4.43	0.224	2.98e-9	0.951
Cape Genet	Canopy_Cover	0.506	0.602	2.77e-1	0.917
Cape Genet	Conductivity	-0.572	0.407	1.75e-2	0.950
Cape Genet	Dissolved.oxygen	0.620	0.590	8.73e-2	0.986
Cape Genet	Ground_vegetation_cover	0.918	0.679	2.31e-1	0.966
Cape Genet	SASS	-0.786	0.364	1.16e-2	0.946
Cape Porcupine	(Intercept)	1.53	0.750	1.24e-1	0.994
Cape Porcupine	AreaGrootvadersbosch	-3.97	0.302	1.06e-10	0.989
Cape Porcupine	Canopy_Cover	0.225	0.557	7.82e-2	0.892
Cape Porcupine	Conductivity	-0.623	0.416	2.53e-3	0.960
Cape Porcupine	Dissolved.oxygen	0.441	0.563	4.04e-2	0.989
Cape Porcupine	Ground_vegetation_cover	0.730	0.645	1.68e-1	0.954
Cape Porcupine	SASS	-1.15	0.336	9.63e-4	0.918
Chacma Baboon	(Intercept)	1.89	0.824	5.68e-1	0.993
Chacma Baboon	AreaGrootvadersbosch	1.30	0.535	2.42e-2	1.000
Chacma Baboon	Canopy_Cover	0.372	0.583	2.91e-1	0.840

Table A9. Summer species-specific covariate effects (continued)					
Species	**Covariate**	**Mean coefficient estimate (β, logit scale)**	**Predicted occupancy probability (ψ)**	**Lower 95% credible interval (ψ)**	**Upper 95% credible interval (ψ)**
Chacma Baboon	Conductivity	-0.430	0.420	4.16e-2	0.939
Chacma Baboon	Dissolved.oxygen	0.195	0.532	6.21e-2	0.967
Chacma Baboon	Ground_vegetation_cover	0.861	0.681	4.11e-1	0.940
Chacma Baboon	SASS	0.118	0.500	7.18e-2	0.983
Water mongoose	(Intercept)	1.53	0.768	2.88e-1	0.989
Water mongoose	AreaGrootvadersbosch	1.02	0.487	1.85e-3	1.000
Water mongoose	Canopy_Cover	0.364	0.578	1.93e-1	0.885
Water mongoose	Conductivity	-0.257	0.451	3.11e-2	0.973
Water mongoose	Dissolved.oxygen	0.650	0.603	7.21e-2	0.985
Water mongoose	Ground_vegetation_cover	0.490	0.601	6.03e-2	0.965
Water mongoose	SASS	-1.94	0.244	2.83e-4	0.748

Appendix A10: Winter mean occupancy probability and detection probability per area

Table A10. Winter mean occupancy probability and detection probability per area

Species	Area	Mean ψ	95% CI ψ	Mean p	95% CI p
Bushpig	Buffeljags River	0.36	0.08–0.64	0.81	0.53–0.97
Cape Bushbuck	Buffeljags River	0.48	0.27–0.75	0.88	0.71–0.97
African clawless Otter	Buffeljags River	0.47	0.19–0.98	0.76	0.42–0.95
Cape Genet	Buffeljags River	0.43	0.25–0.63	0.87	0.67–0.98
Cape Porcupine	Buffeljags River	0.52	0.21–0.97	0.73	0.35–0.96
Chacma Baboon	Buffeljags River	0.49	0.31–0.66	0.93	0.78–0.99
Water Mongoose	Buffeljags River	0.56	0.32–0.86	0.82	0.60–0.96
Bushpig	Grootvadersbosch River	0.41	0.11–0.78	0.81	0.53–0.97
Cape Bushbuck	Grootvadersbosch River	0.60	0.34–0.90	0.88	0.71–0.97
African clawless Otter	Grootvadersbosch River	0.52	0.16–0.99	0.76	0.42–0.95
Cape Genet	Grootvadersbosch River	0.28	0.08–0.57	0.87	0.67–0.98
Cape Porcupine	Grootvadersbosch River	0.62	0.23–1.00	0.73	0.35–0.96
Chacma Baboon	Grootvadersbosch River	0.33	0.18–0.52	0.93	0.78–0.99
Water Mongoose	Grootvadersbosch River	0.63	0.30–0.99	0.82	0.60–0.96

Appendix A11: Summer mean occupancy probability and detection probability per area

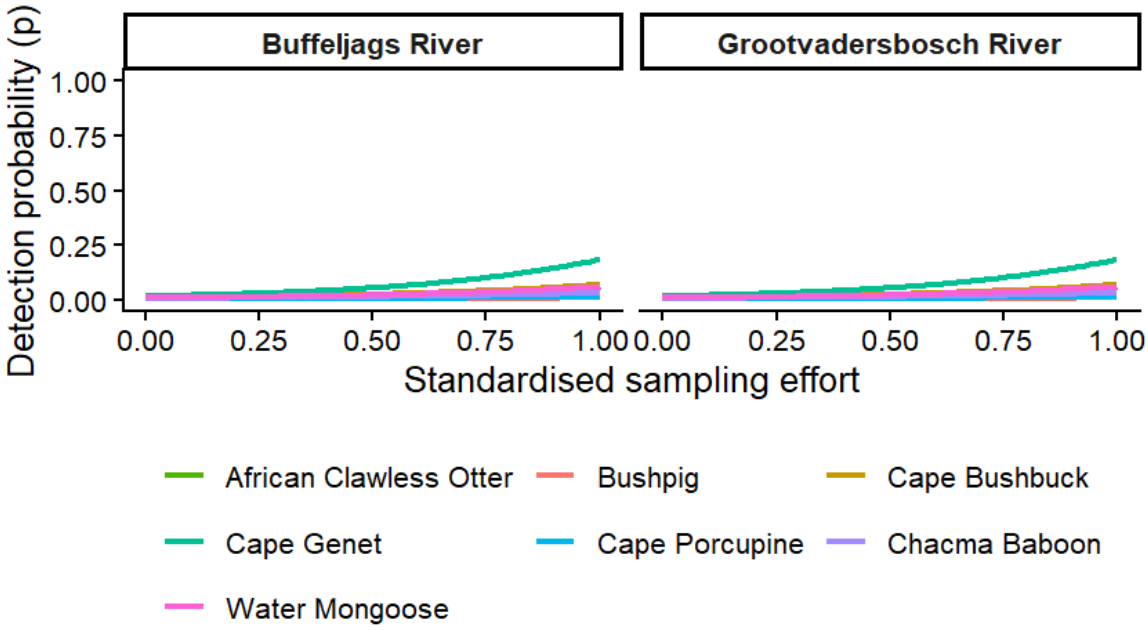
Table A11. Summer mean occupancy probability and detection probability per area

Species	Area	Mean ψ	95% CI ψ	Mean p	95% CI p
Bushpig	Buffeljags River	0.65	0.19–0.90	0.01	0–0.02
Cape Bushbuck	Buffeljags River	0.76	0.54–0.96	0.08	0.05–0.1
African clawless Otter	Buffeljags River	0.68	0.36–0.88	0.01	0–0.02
Cape Genet	Buffeljags River	0.73	0.51–0.96	0.02	0.01–0.04
Cape Porcupine	Buffeljags River	0.67	0.26–0.96	0.00	0–0.01
Chacma Baboon	Buffeljags River	0.75	0.55–0.93	0.06	0.05–0.08
Water Mongoose	Buffeljags River	0.71	0.47–0.93	0.02	0.01–0.05
Bushpig	Grootvadersbosch	0.50	0–1	0.00	0–0
Cape Bushbuck	Grootvadersbosch	0.63	0.26–1	0.07	0.04–0.09
African clawless Otter	Grootvadersbosch	0.33	0–0.99	0.01	0–0.01
Cape Genet	Grootvadersbosch	0.31	0–0.98	0.02	0.01–0.02
Cape Porcupine	Grootvadersbosch	0.38	0–0.99	0.00	0–0
Chacma Baboon	Grootvadersbosch	0.66	0.22–1	0.07	0.04–0.09
Water Mongoose	Grootvadersbosch	0.60	0.14–1	0.02	0.01–0.04

Appendix B: Supplementary figures

Appendix B1: Winter and summer species level detection probability logit effects

Winter



Summer

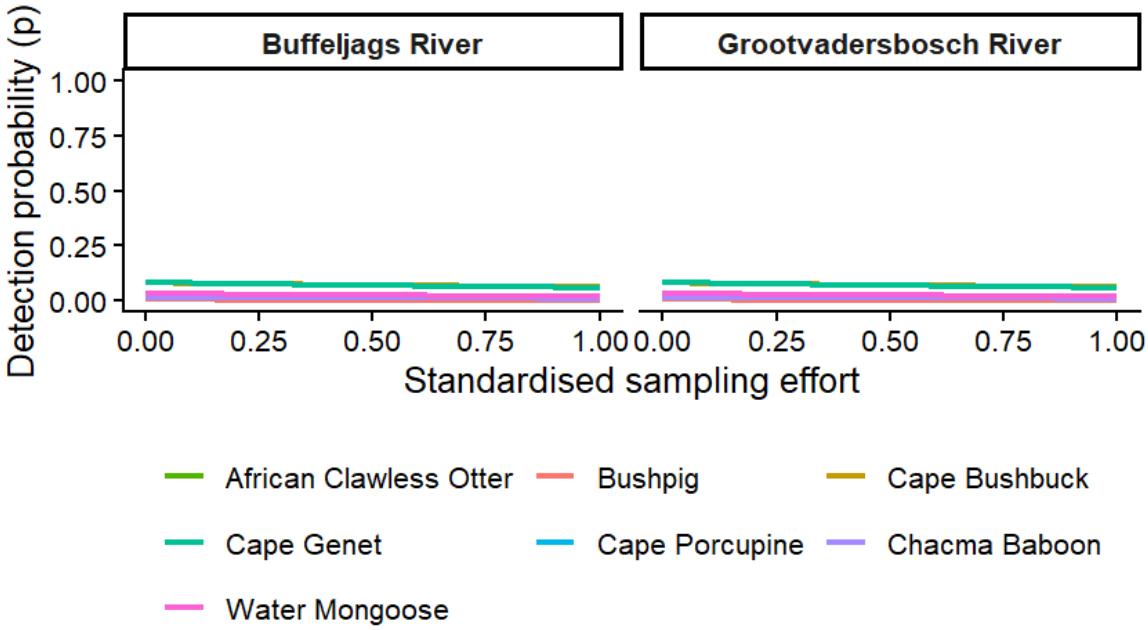


Figure B1: *Predicted detection probability (p) as a function of standardised sampling effort (0–1) for the each species during winter and summer surveys. Predictions were generated from the final multispecies occupancy models with all other detection covariates held at their mean values and random effects excluded. Curves illustrate the direction and relative strength of the estimated effect*

of sampling effort on detection probability; intermediate values along the effort axis represent interpolation of the fitted model rather than independently observed effort levels.

Appendix C: R Scripts

Appendix C1: R script for Z-scaling numeric site covariates for winter and summer

A function was used to *z*-scale all numeric site covariates, standardizing them to have a mean of 0 and a standard deviation of 1. This ensured that continuous covariates were on comparable scales before model fitting. Scaling was applied separately to the winter (site_covs_winter) and summer (site_covs_summer) datasets.

```
# Function to z-scale numeric covariates
z_scale_covariates <- function(df) {
  # Identify numeric columns
  numeric_cols <- sapply(df, is.numeric)

  # Scale only numeric columns
  df_scaled <- df
  df_scaled[, numeric_cols] <- scale(df[, numeric_cols])

  return(df_scaled)
}

# Apply to winter and summer covariates
site_covs_winter_scaled <- z_scale_covariates(site_covs_winter)
site_covs_summer_scaled <- z_scale_covariates(site_covs_summer)

# Optional: check the result
head(site_covs_winter_scaled)
head(site_covs_summer_scaled)
```

Appendix C2: R script for correlation of numeric site covariates

This script calculates Pearson's correlation coefficients among the continuous site covariates to identify any highly correlated covariates before model fitting.

```
# Correlation of Numeric Site Covariates
# ================================

# List of datasets
datasets <- list(
  Winter = site_covs_winter,
  Summer = site_covs_summer
)

# Function to compute correlation of numeric columns
cor_numeric_covs <- function(df) {
  numeric_cols <- sapply(df, is.numeric)
  cor(df[, numeric_cols], use = "pairwise.complete.obs")
}
```

```
# Compute correlations
covariate_correlations <- lapply(datasets, cor_numeric_covs)

# View correlations
covariate_correlations$Winter
covariate_correlations$Summer
```

Appendix C3: R script for the winter full multispecies occupancy model (all covariates)

This R script was used to fit the winter full multispecies occupancy model with all site covariates included, using msPGOcc from the spOccupancy package. Posterior sampling parameters and thread settings are specified within the code.

```
library(spOccupancy)

# -----------------------------
# 1. Load winter detection and covariate data
# -----------------------------
site_covs_winter_scaled <- read.csv(
  "C:/Users/Barba/Dropbox/PC/Downloads/Masters Occupancy data/Occupancy current
models July 2025/Single season_multisp model
data/Detection_unmarked/Winter_sitecovs_trim.csv",
  row.names = 1, sep = ";"
)

Bushpig_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters Occupancy
data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Bushpig_winter_trim.csv", row.names = 1, sep = ";")
CapeBushbuck_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeBushbuck_winter_trim.csv", row.names = 1, sep = ";")
CapeClawless_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeClawless_winter_trim.csv", row.names = 1, sep = ";")
CapeGenet_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeGenet_winter_trim.csv", row.names = 1, sep = ";")
CapePorcu_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapePorcu_winter_trim.csv", row.names = 1, sep = ";")
Chacmabab_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Chacmabab_winter_trim.csv", row.names = 1, sep = ";")
Watermongoose_winter <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Watermongoose_winter_trim.csv", row.names = 1, sep = ";")

Winter_effort <- read.csv(
  "C:/Users/Barba/Dropbox/PC/Downloads/Masters Occupancy data/Occupancy current
models July 2025/Single season_multisp model
data/Detection_unmarked/Winter_effort.csv",
  row.names = 1, sep = ";"
)

# -----------------------------
# 2. Create 3D detection array
```

```
# ------------------------------
winter_species_list <- list(
  Bushpig_winter,
  CapeBushbuck_winter,
  CapeClawless_winter,
  CapeGenet_winter,
  CapePorcu_winter,
  Chacmabab_winter,
  Watermongoose_winter
)

species_names <- c("Bushpig", "CapeBushbuck", "CapeClawless",
                   "CapeGenet", "CapePorcu", "Chacmabab", "Watermongoose")

n_species <- length(winter_species_list)
n_sites <- nrow(winter_species_list[[1]])
n_days  <- ncol(winter_species_list[[1]])

y_array_winter <- array(NA, dim = c(n_species, n_sites, n_days),
                        dimnames = list(
                          species = species_names,
                          site = rownames(winter_species_list[[1]]),
                          day = colnames(winter_species_list[[1]])
                        ))

for(i in 1:n_species){
  y_array_winter[i,,] <- as.matrix(winter_species_list[[i]])
}

# ------------------------------
# 3. Use winter site and observation covariates
# ------------------------------
site_covs_winter <- site_covs_winter_scaled
obs_covs_winter <- list(effort = as.matrix(Winter_effort))

# ------------------------------
# 4. Confirm data consistency
# ------------------------------
cat("\nChecking data consistency before fitting...\n")
cat("Detection array dimensions (species × sites × days):", dim(y_array_winter),
"\n")
cat("Site covariates:", nrow(site_covs_winter), "sites ×",
ncol(site_covs_winter), "covariates\n")
cat("Detection covariate matrix dimensions:", dim(obs_covs_winter$effort), "\n")

stopifnot(
  dim(y_array_winter)[2] == nrow(site_covs_winter),
  dim(y_array_winter)[2] == nrow(obs_covs_winter$effort),
  dim(y_array_winter)[3] == ncol(obs_covs_winter$effort)
)

# ------------------------------
# 5. Define occupancy and detection formulas
# ------------------------------
occ_formula <- ~ Area + Dissolved.oxygen + Canopy_Cover +
                Ground_vegetation_cover + SASS + Conductivity
det_formula <- ~ effort

# ------------------------------
# 6. Fit the multispecies occupancy model
```

```
# ----------------------------
set.seed(123)
ms_model_winter <- msPGOcc(
  occ.formula = occ_formula,
  det.formula = det_formula,
  data = list(
    y = y_array_winter,
    occ.covs = site_covs_winter,
    det.covs = obs_covs_winter
  ),
  n.samples = 3000,
  n.burn = 500,
  n.thin = 5,
  n.chains = 2,
  n.omp.threads = 4,
  verbose = TRUE,
  n.report = 500
)

# ----------------------------
# 7. Save model
# ----------------------------
saveRDS(ms_model_winter, "ms_model_winter_final.rds")
```

Appendix C4: R script for the summer full multispecies occupancy model (all covariates)

This R script was used to fit the summer full model with all site covariates included, using msPGOcc from the spOccupancy package. Posterior sampling parameters and thread settings are shown within the code.

```
# ============================================================
# Summer multispecies occupancy model – Full model (all covariates)
# ============================================================

library(spOccupancy)

# 1. Load summer detection and covariate data
Summer_sitecovs <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Summer_sitecovs_trim.csv",
                            row.names = 1, sep = ";")

Bushpig_summer       <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Bushpig_summer_trim.csv", row.names=1, sep=";")
CapeBushbuck_summer  <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeBushbuck_summer_trim.csv", row.names=1, sep=";")
CapeClawless_summer  <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeClawless_summer_trim.csv", row.names=1, sep=";")
CapeGenet_summer     <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapeGenet_summer_trim.csv", row.names=1, sep=";")
CapePorcu_summer     <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/CapePorcu_summer_trim.csv", row.names=1, sep=";")
```

```
ChacmaBab_summer     <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/ChacmaBab_summer_trim.csv", row.names=1, sep=";")
Watermongoose_summer <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters
Occupancy data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Watermongoose_summer_trim.csv", row.names=1, sep=";")

Summer_effort <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters Occupancy
data/Occupancy current models July 2025/Single season_multisp model
data/Detection_unmarked/Summer_effort.csv",
                          row.names = 1, sep = ";")

# 2. Convert detection data to matrices
y_list <- list(
  Bushpig       = as.matrix(Bushpig_summer),
  CapeBushbuck  = as.matrix(CapeBushbuck_summer),
  CapeClawless  = as.matrix(CapeClawless_summer),
  CapeGenet     = as.matrix(CapeGenet_summer),
  CapePorcu     = as.matrix(CapePorcu_summer),
  ChacmaBab     = as.matrix(ChacmaBab_summer),
  Watermongoose = as.matrix(Watermongoose_summer)
)

n_species <- length(y_list)
n_sites   <- nrow(y_list[[1]])
n_days    <- ncol(y_list[[1]])
species_names <- names(y_list)

# 3. Build 3D array: species × sites × days
y_clean <- array(NA, dim = c(n_species, n_sites, n_days), dimnames =
list(species_names, NULL, NULL))
for (i in 1:n_species) y_clean[i,,] <- y_list[[i]]

# 4. Prepare effort covariates and sync NAs
effort_matrix <- as.matrix(Summer_effort[, grepl("^Day",
colnames(Summer_effort))])
for (i in 1:n_sites) {
  for (t in 1:n_days) {
    if (is.na(effort_matrix[i, t])) y_clean[, i, t] <- NA
  }
}
obs_covs_list_summer <- list(effort = effort_matrix)

# 5. Prepare full site covariates (all covariates included)
site_covs_full <- as.data.frame(lapply(Summer_sitecovs, function(x)
if(is.numeric(x)) scale(x) else x))

# 6. Specify occupancy and detection formulas
occ_formula_full <- ~ Area + Dissolved.oxygen + Canopy_Cover +
Ground_vegetation_cover + SASS + Conductivity
det_formula_full <- ~ effort

# 7. Fit full summer model
set.seed(123)
ms_model_plus3 <- msPGOcc(
  occ.formula = occ_formula_full,
  det.formula = det_formula_full,
  data = list(
    y = y_clean,
    occ.covs = site_covs_full,
    det.covs = obs_covs_list_summer
```

```
  ),
  n.samples = 2000,
  n.burn = 500,
  n.thin = 5,
  n.chains = 2,
  n.omp.threads = 2,
  verbose = TRUE,
  n.report = 500
)

# 8. Model summary
summary(ms_model_plus3)
```

Appendix C5: R script for single-species occupancy models (winter)

This script fits single-species occupancy models for all seven riparian mammal species during the winter survey using the spOccupancy package. Four occupancy formulas were fitted per species, while detection probability was modelled as a function of effort.

```
# ------------------------------
# Single-species occupancy models (Winter) - spOccupancy
# ------------------------------
library(spOccupancy)

# ------------------------------
# 1. Species detection matrices
# ------------------------------
species_list <- list(
  Bushpig        = Bushpig_winter,
  CapeBushbuck   = CapeBushbuck_winter,
  CapeClawless   = CapeClawless_winter,
  CapeGenet      = CapeGenet_winter,
  CapePorcupine  = CapePorcu_winter,
  ChacmaBaboon   = Chacmabab_winter,
  WaterMongoose  = Watermongoose_winter
)

# ------------------------------
# 2. Site covariates
# ------------------------------
site_covs <- Winter_sitecovs_unmarked[, c(
  "Area", "SASS", "Ground_vegetation_cover",
  "Canopy_Cover", "Conductivity", "Dissolved.oxygen"
)]

# Scale numeric covariates (exclude Area if factor)
site_covs_scaled <- site_covs
site_covs_scaled[, -1] <- scale(site_covs[, -1])
site_covs_scaled$Area <- as.factor(site_covs_scaled$Area)

# ------------------------------
# 3. Detection covariates
# ------------------------------
# Keep all rows, raw effort matrix
det_covs <- list(effort = Winter_effort_unmarked)
```

```
# ----------------------------
# 4. Occupancy formulas
# ----------------------------
occ_formulas <- list(
  null    = ~ Area,
  habitat = ~ Area + SASS + Ground_vegetation_cover + Canopy_Cover,
  water   = ~ Area + Conductivity + Dissolved.oxygen,
  full    = ~ Area + SASS + Ground_vegetation_cover + Canopy_Cover + Conductivity
+ Dissolved.oxygen
)

det_formula <- ~ effort

# ----------------------------
# 5. Fit models in a loop per species
# ----------------------------
ss_models <- list()

for(sp in names(species_list)) {
  cat("\nRunning single-species models for:", sp, "\n")

  y <- species_list[[sp]]  # detection matrix for this species

  # List to store 4 models per species
  ss_models[[sp]] <- list()

  for(mod_name in names(occ_formulas)) {
    cat("  Fitting model:", mod_name, "\n")

    ss_models[[sp]][[mod_name]] <- PGOcc(
      occ.formula = occ_formulas[[mod_name]],
      det.formula = det_formula,
      data = list(
        y = y,
        occ.covs = site_covs_scaled,
        det.covs = det_covs
      ),
      n.samples = 1000,
      n.burn    = 200,
      n.thin    = 2,
      n.chains  = 2,
      verbose   = TRUE
    )

    cat("  Model", mod_name, "completed.\n")
  }
}

# ----------------------------
# 6. Example: summary for Bushpig full model
# ----------------------------
summary(ss_models$Bushpig$full)
```

Appendix C6: R script for single-species occupancy models (summer)

This script fits single-species occupancy models for all seven riparian mammal species during the summer survey using the spOccupancy package. Four occupancy formulas were fitted per species, while detection probability was modelled as a function of effort.

```
Summer single species
# ----------------------------
# Single-species occupancy models (Summer) - spOccupancy
# ----------------------------
library(spOccupancy)

# ----------------------------
# 1. Species detection matrices
# ----------------------------
species_list <- list(
  Bushpig        = Bushpig_summer_trim,
  CapeBushbuck   = CapeBushbuck_summer_trim,
  CapeClawless   = CapeClawless_summer_trim,
  CapeGenet      = CapeGenet_summer_trim,
  CapePorcupine  = CapePorcu_summer_trim,
  ChacmaBaboon   = ChacmaBab_summer_trim,
  WaterMongoose  = Watermongoose_summer_trim
)

# ----------------------------
# 2. Site covariates (trimmed summer)
# ----------------------------
site_covs <- Summer_sitecovs_trim[, c(
  "Area", "Ground_vegetation_cover", "Canopy_Cover",
  "Conductivity", "Dissolved.oxygen", "SASS"
)]

# Scale numeric covariates (exclude Area if factor)
site_covs_scaled <- site_covs
site_covs_scaled[, -1] <- scale(site_covs[, -1])
site_covs_scaled$Area <- as.factor(site_covs_scaled$Area)

# ----------------------------
# 3. Detection covariates
# ----------------------------
det_covs <- list(effort = Summer_effort)  # your cleaned summer effort matrix

# ----------------------------
# 4. Occupancy formulas (SASS in water, not habitat)
# ----------------------------
occ_formulas <- list(
  null    = ~ Area,
  habitat = ~ Area + Ground_vegetation_cover + Canopy_Cover,
  water   = ~ Area + Conductivity + Dissolved.oxygen + SASS,
  full    = ~ Area + Ground_vegetation_cover + Canopy_Cover + Conductivity +
Dissolved.oxygen + SASS
```

```
)

det_formula <- ~ effort

# ----------------------------
# 5. Fit models in a loop per species
# ----------------------------
ss_models_summer <- list()

for(sp in names(species_list)) {
  cat("\nRunning single-species models for:", sp, "\n")

  y <- species_list[[sp]]  # detection matrix for this species

  # List to store 4 models per species
  ss_models_summer[[sp]] <- list()

  for(mod_name in names(occ_formulas)) {
    cat("  Fitting model:", mod_name, "\n")

    ss_models_summer[[sp]][[mod_name]] <- PGOcc(
      occ.formula = occ_formulas[[mod_name]],
      det.formula = det_formula,
      data = list(
        y = y,
        occ.covs = site_covs_scaled,
        det.covs = det_covs
      ),
      n.samples = 1000,
      n.burn    = 200,
      n.thin    = 2,
      n.chains  = 2,
      verbose   = TRUE
    )

    cat("  Model", mod_name, "completed.\n")
  }
}

# ----------------------------
# 6. Example: summary for Bushpig full model
# ----------------------------
summary(ss_models_summer$Bushpig$full)
```

Appendix C7: R script for calculating winter species-level occupancy summary

This R script summarizes the mean occupancy probabilities (ψ) and corresponding 95% credible intervals for each riparian mammal species from the winter multispecies occupancy model. Posterior samples were extracted from the fitted model (ms_model_winter) and converted from the logit to the probability scale using the logistic transformation (plogis()).

```
Winter occupancy summary per species
# ------------------------------------------

library(dplyr)

# --------------------------------
# Inputs
# --------------------------------
# beta_species: ms_model_winter$beta.samples (logit scale)
# species_names: vector of species names, cleaned
# Example:
# species_names <- gsub(".*-", "", colnames(beta_species)) %>% unique()

# --------------------------------
# Initialize summary table
# --------------------------------
winter_occ_summary <- data.frame(
  Species = species_names,
  Mean_Logit = NA,
  Mean_Probability = NA,
  CI2.5_Probability = NA,
  CI97.5_Probability = NA
)

# --------------------------------
# Loop over species to calculate:
# - mean logit
# - mean probability
# - 95% credible interval on probability scale
# --------------------------------
for (i in seq_along(species_names)) {
  # Extract MCMC samples for this species
  species_beta <- beta_species[, grep(species_names[i], colnames(beta_species))]

  # Mean logit
  mean_logit <- mean(species_beta)

  # Mean probability
  mean_prob <- plogis(mean_logit)

  # 95% credible interval (probability scale)
  ci_prob <- plogis(quantile(species_beta, c(0.025, 0.975)))

  # Fill table
  winter_occ_summary[i, "Mean_Logit"] <- mean_logit
```

```
  winter_occ_summary[i, "Mean_Probability"] <- mean_prob
  winter_occ_summary[i, "CI2.5_Probability"] <- ci_prob[1]
  winter_occ_summary[i, "CI97.5_Probability"] <- ci_prob[2]
}

# ------------------------------
# Format CI as a range and keep relevant columns
# ------------------------------
winter_occ_summary_final <- winter_occ_summary %>%
  mutate(Probability_CI_Range = paste0(
    round(CI2.5_Probability, 2), "-", round(CI97.5_Probability, 2)
  )) %>%
  select(Species, Mean_Probability, Probability_CI_Range)

# ------------------------------
# Output final table
# ------------------------------
winter_occ_summary_final
```

Appendix C8: R script for calculating summer species-level occupancy and detection summary

This R script summarizes the mean occupancy (ψ) and detection (p) probabilities, along with their 95% credible intervals, for each riparian mammal species from the summer multispecies occupancy model. Posterior samples were extracted from the fitted model (ms_model_plus3) and converted from the logit to the probability scale using the logistic transformation (plogis()).

```
library(dplyr)

species_names_summer <- c("Bushpig", "CapeBushbuck", "CapeClawless", "CapeGenet",
                          "CapePorcu", "ChacmaBab", "Watermongoose")

# ------------------------------
# Occupancy summary
# ------------------------------
beta_samples <- ms_model_plus3$beta.samples
occ_summary <- data.frame(Species = species_names_summer,
                          Mean_Occupancy = NA,
                          Occupancy_CI = NA)

for (i in seq_along(species_names_summer)) {
  sp <- species_names_summer[i]
  sp_beta <- beta_samples[, grep(sp, colnames(beta_samples))]

  mean_prob <- plogis(mean(sp_beta))
  ci_prob <- plogis(quantile(sp_beta, c(0.025, 0.975)))

  occ_summary[i, "Mean_Occupancy"] <- mean_prob
  occ_summary[i, "Occupancy_CI"] <- paste0(round(ci_prob[1], 2), "-",
round(ci_prob[2], 2))
}
```

```
# --------------------------------
# Detection summary
# --------------------------------
alpha_samples <- ms_model_plus3$alpha.samples
det_summary <- data.frame(Species = species_names_summer,
                          Mean_Detection = NA,
                          Detection_CI = NA)

for (i in seq_along(species_names_summer)) {
  sp <- species_names_summer[i]

  intercept_col <- grep(paste0("\\(Intercept\\)-", sp), colnames(alpha_samples),
value = TRUE)
  effort_col <- grep(paste0("effort-", sp), colnames(alpha_samples), value =
TRUE)

  if (length(intercept_col) == 0) stop(paste("Intercept column missing for", sp))

  intercept <- alpha_samples[, intercept_col]

  if (length(effort_col) > 0) {
    effort <- alpha_samples[, effort_col]
    det_prob <- plogis(intercept + effort)
  } else {
    det_prob <- plogis(intercept)
  }

  mean_det <- mean(det_prob)
  ci_det <- quantile(det_prob, c(0.025, 0.975))

  det_summary[i, "Mean_Detection"] <- mean_det
  det_summary[i, "Detection_CI"] <- paste0(round(ci_det[1], 2), "-",
round(ci_det[2], 2))
}

# --------------------------------
# Combine occupancy + detection
# --------------------------------
summer_summary <- occ_summary %>%
  left_join(det_summary, by = "Species")

print(summer_summary)
```

Appendix C9: R script to generate community-level occupancy and detection summaries for the winter multispecies model

This script was used to extract posterior samples of community-level occupancy (ψ) and detection (p) parameters from the winter multispecies model (ms_model_winter). It calculates mean values and 95% credible intervals for each community-level parameter, including the intercept and the effect of effort on detection probability. The resulting tables summarize overall community patterns rather than species-specific estimates.

```
library(dplyr)

# ----------------------------
# Community-level occupancy (probability scale)
# ----------------------------
community_beta_summary <- apply(ms_model_winter$beta.comm.samples, 2, function(x)
{
  c(
    Mean = mean(x),
    CI2.5 = quantile(x, 0.025),
    CI97.5 = quantile(x, 0.975)
  )
})
community_beta_summary <- t(community_beta_summary) %>% as.data.frame()

comm_param_names <- colnames(ms_model_winter$X)

community_psi_table_prob <- data.frame(
  Parameter = comm_param_names,
  Mean.ψ = round(plogis(community_beta_summary$Mean), 2),
  `95% CI ψ` = paste0(
    round(plogis(community_beta_summary$CI2.5), 2),
    "-",
    round(plogis(community_beta_summary$CI97.5), 2)
  )
)

cat("\nCommunity-level occupancy parameters (probability scale) — Winter:\n")
print(community_psi_table_prob)

# ----------------------------
# Community-level detection (probability scale)
# ----------------------------
community_alpha_summary <- apply(ms_model_winter$alpha.comm.samples, 2,
function(x) {
  c(
    Mean = mean(x),
    CI2.5 = quantile(x, 0.025),
    CI97.5 = quantile(x, 0.975)
  )
})
community_alpha_summary <- t(community_alpha_summary) %>% as.data.frame()

comm_det_names <- c("(Intercept)", "effort")  # same as model

community_p_table_prob <- data.frame(
  Parameter = comm_det_names,
  Mean.p = round(plogis(community_alpha_summary$Mean), 2),
  `95% CI p` = paste0(
    round(plogis(community_alpha_summary$CI2.5), 2),
    "-",
    round(plogis(community_alpha_summary$CI97.5), 2)
  )
)

cat("\nCommunity-level detection parameters (probability scale) — Winter:\n")
print(community_p_table_prob)
```

Appendix C10: R script to generate community-level occupancy and detection summaries for the summer multispecies model

This script was used to extract posterior samples of community-level occupancy (ψ) and detection (p) parameters from the winter multispecies model (ms_model_plus3). It calculates mean values and 95% credible intervals for each community-level parameter, including the intercept and the effect of effort on detection probability. The resulting tables summarize overall community patterns rather than species-specific estimates.

```
# Community-level occupancy parameters
community_beta_summary <- apply(ms_model_plus3$beta.comm.samples, 2, function(x)
{
  c(
    Mean = mean(x),
    CI2.5 = quantile(x, 0.025),
    CI97.5 = quantile(x, 0.975)
  )
})
community_beta_summary <- t(community_beta_summary) %>% as.data.frame()

# Name the occupancy parameters
comm_param_names <- colnames(ms_model_plus3$X)
community_psi_table_named <- data.frame(
  Parameter = comm_param_names,
  Mean.ψ = round(community_beta_summary$Mean, 2),
  X95..CI.ψ = paste0(round(community_beta_summary$CI2.5, 2), "-",
round(community_beta_summary$CI97.5, 2))
)

# Print community-level occupancy table
cat("\nCommunity-level occupancy parameters:\n")
print(community_psi_table_named)

# Community-level detection parameters
community_alpha_summary <- apply(ms_model_plus3$alpha.comm.samples, 2,
function(x) {
  c(
    Mean = mean(x),
    CI2.5 = quantile(x, 0.025),
    CI97.5 = quantile(x, 0.975)
  )
})
community_alpha_summary <- t(community_alpha_summary) %>% as.data.frame()

# Name the detection parameters
comm_det_names <- c("(Intercept)", "effort")  # exactly as used
community_p_table_named <- data.frame(
  Parameter = comm_det_names,
  Mean.p = round(community_alpha_summary$Mean, 3),
  X95..CI.p = paste0(round(community_alpha_summary$CI2.5, 3), "-",
round(community_alpha_summary$CI97.5, 3))
)
```

```
# Print community-level detection table
cat("\nCommunity-level detection parameters:\n")
print(community_p_table_named)
```

Appendix C11: R script for winter mean occupancy and detection probability per area

This R script calculates the mean probability of occupancy (ψ) and detection (p) for each species per study area (Buffeljags River and Grootvadersbosch River) from the fitted winter multispecies occupancy model (ms_model_winter). Estimates were derived from posterior samples and summarized with 95% credible intervals using dplyr.

```
library(dplyr)

# ----------------------------
# Common names and number of species
# ----------------------------
common_names <- c("Bushpig", "Cape Bushbuck", "African clawless Otter",
                  "Cape Genet", "Cape Porcupine", "Chacma Baboon",
                  "Water Mongoose")
n_species <- length(common_names)

# ----------------------------
# Area columns (site indices)
# ----------------------------
buf_cols <- 1:20
gvb_cols <- 21:40  # adjust if your GVB survey has fewer sites

# ----------------------------
# Occupancy per area with 95% CI
# ----------------------------
psi_area_summary <- lapply(1:n_species, function(s) {
  psi_s <- ms_model_winter$psi.samples[, s, ]  # draws × sites

  # Posterior mean per area
  buf_draws <- rowMeans(psi_s[, buf_cols])
  gvb_draws <- rowMeans(psi_s[, gvb_cols])

  # Convert all to numeric explicitly
  list(
    Buffeljagsriver_psi = as.numeric(mean(buf_draws)),
    Buffeljagsriver_CI_lower = as.numeric(quantile(buf_draws, 0.025)),
    Buffeljagsriver_CI_upper = as.numeric(quantile(buf_draws, 0.975)),
    Grootvadersbosch_psi = as.numeric(mean(gvb_draws)),
    Grootvadersbosch_CI_lower = as.numeric(quantile(gvb_draws, 0.025)),
    Grootvadersbosch_CI_upper = as.numeric(quantile(gvb_draws, 0.975))
  )
})

psi_area_summary <- do.call(rbind, lapply(psi_area_summary, function(x)
as.data.frame(x)))
```

```
# ----------------------------
# Detection probability per area with 95% CI
# ----------------------------
effort_area_summary <- lapply(1:n_species, function(s) {
  # Convert alpha samples to probability
  p_s <- plogis(ms_model_winter$alpha.samples[, 8 + s - 1])

  list(
    Buffeljagsriver_p = as.numeric(mean(p_s)),
    Buffeljagsriver_p_CI_lower = as.numeric(quantile(p_s, 0.025)),
    Buffeljagsriver_p_CI_upper = as.numeric(quantile(p_s, 0.975)),
    Grootvadersbosch_p = as.numeric(mean(p_s)),
    Grootvadersbosch_p_CI_lower = as.numeric(quantile(p_s, 0.025)),
    Grootvadersbosch_p_CI_upper = as.numeric(quantile(p_s, 0.975))
  )
})

effort_area_summary <- do.call(rbind, lapply(effort_area_summary, function(x)
as.data.frame(x)))

# ----------------------------
# Combine into final table
# ----------------------------
species_table_winter_area <- data.frame(
  Common_Name = common_names,
  Species = ms_model_winter$sp.names,
  Buffeljagsriver_psi = round(psi_area_summary$Buffeljagsriver_psi, 2),
  Buffeljagsriver_psi_CI =
paste0(round(psi_area_summary$Buffeljagsriver_CI_lower, 2), "-",

round(psi_area_summary$Buffeljagsriver_CI_upper, 2)),
  Grootvadersbosch_psi = round(psi_area_summary$Grootvadersbosch_psi, 2),
  Grootvadersbosch_psi_CI =
paste0(round(psi_area_summary$Grootvadersbosch_CI_lower, 2), "-",

round(psi_area_summary$Grootvadersbosch_CI_upper, 2)),
  Buffeljagsriver_p = round(effort_area_summary$Buffeljagsriver_p, 2),
  Buffeljagsriver_p_CI =
paste0(round(effort_area_summary$Buffeljagsriver_p_CI_lower, 2), "-",

round(effort_area_summary$Buffeljagsriver_p_CI_upper, 2)),
  Grootvadersbosch_p = round(effort_area_summary$Grootvadersbosch_p, 2),
  Grootvadersbosch_p_CI =
paste0(round(effort_area_summary$Grootvadersbosch_p_CI_lower, 2), "-",

round(effort_area_summary$Grootvadersbosch_p_CI_upper, 2))
)

# ----------------------------
# Print final table
# ----------------------------
print(species_table_winter_area)
```

Appendix C12: R script for summer mean occupancy and detection probability per area

This R script calculates the mean probability of occupancy (ψ) and detection (p) for each species per study area (Buffeljags River and Grootvadersbosch River) from the fitted summer multispecies occupancy model (ms_model_plus3). Estimates were derived from posterior samples and summarized with 95% credible intervals using dplyr.

```
library(dplyr)

# ----------------------------
# Common names and number of species
# ----------------------------
common_names <- c("Bushpig", "Cape Bushbuck", "African clawless Otter",
                  "Cape Genet", "Cape Porcupine", "Chacma Baboon",
                  "Water Mongoose")
n_species <- length(common_names)

# ----------------------------
# Area columns (site indices) for summer
# ----------------------------
buf_cols <- 1:20         # Buffeljagsriver sites
gvb_cols <- 21:32        # Grootvadersbosch summer sites (12 deployed sites)

# ----------------------------
# Occupancy per area with 95% CI
# ----------------------------
psi_area_summary <- lapply(1:n_species, function(s) {
  # Slice draws × sites for species s
  psi_s <- ms_model_plus3$psi.samples[, s, ]  # 600 × 32

  buf_draws <- rowMeans(psi_s[, buf_cols], na.rm = TRUE)
  gvb_draws <- rowMeans(psi_s[, gvb_cols], na.rm = TRUE)

  list(
    Buffeljagsriver_psi = mean(buf_draws),
    Buffeljagsriver_CI_lower = quantile(buf_draws, 0.025),
    Buffeljagsriver_CI_upper = quantile(buf_draws, 0.975),
    Grootvadersbosch_psi = mean(gvb_draws),
    Grootvadersbosch_CI_lower = quantile(gvb_draws, 0.025),
    Grootvadersbosch_CI_upper = quantile(gvb_draws, 0.975)
  )
})

psi_area_summary <- do.call(rbind, lapply(psi_area_summary, as.data.frame))

# ----------------------------
# Detection probability per species (same for both areas)
# ----------------------------
effort_area_summary <- lapply(1:n_species, function(s) {
  p_s <- plogis(ms_model_plus3$alpha.samples[, s])
```

```
  list(
    Buffeljagsriver_p = mean(p_s),
    Buffeljagsriver_p_CI_lower = quantile(p_s, 0.025),
    Buffeljagsriver_p_CI_upper = quantile(p_s, 0.975),
    Grootvadersbosch_p = mean(p_s),
    Grootvadersbosch_p_CI_lower = quantile(p_s, 0.025),
    Grootvadersbosch_p_CI_upper = quantile(p_s, 0.975)
  )
})

effort_area_summary <- do.call(rbind, lapply(effort_area_summary, as.data.frame))

# ----------------------------
# Combine into final summer table
# ----------------------------
species_table_summer_area <- data.frame(
  Common_Name = common_names,
  Species = ms_model_plus3$sp.names,
  Buffeljagsriver_psi = round(psi_area_summary$Buffeljagsriver_psi, 2),
  Buffeljagsriver_psi_CI =
paste0(round(psi_area_summary$Buffeljagsriver_CI_lower, 2), "-",

round(psi_area_summary$Buffeljagsriver_CI_upper, 2)),
  Grootvadersbosch_psi = round(psi_area_summary$Grootvadersbosch_psi, 2),
  Grootvadersbosch_psi_CI =
paste0(round(psi_area_summary$Grootvadersbosch_CI_lower, 2), "-",

round(psi_area_summary$Grootvadersbosch_CI_upper, 2)),
  Buffeljagsriver_p = round(effort_area_summary$Buffeljagsriver_p, 2),
  Buffeljagsriver_p_CI =
paste0(round(effort_area_summary$Buffeljagsriver_p_CI_lower, 2), "-",

round(effort_area_summary$Buffeljagsriver_p_CI_upper, 2)),
  Grootvadersbosch_p = round(effort_area_summary$Grootvadersbosch_p, 2),
  Grootvadersbosch_p_CI =
paste0(round(effort_area_summary$Grootvadersbosch_p_CI_lower, 2), "-",

round(effort_area_summary$Grootvadersbosch_p_CI_upper, 2))
)

# ----------------------------
# Print final summer table
# ----------------------------
print(species_table_summer_area)
```

Appendix C13: R script for species-level covariate effects for winter

This R script extracts species-specific posterior estimates for occupancy covariates from the winter multispecies occupancy model. For each species and covariate, the script calculates the mean logit value, converts it to probability scale, and computes the 95% credible interval.

```
# ========================================================
```

```
# Species-level covariate extraction: Winter (ms_model_winter)
# ============================================================

library(dplyr)
library(tidyr)
library(stringr)

# Extract posterior samples from the winter model
beta_samples_winter <- as.data.frame(ms_model_winter$beta.samples)

# Convert to long format and separate covariate and species
species_cov_table_winter <- beta_samples_winter %>%
  pivot_longer(
    cols = everything(),
    names_to = "Parameter",
    values_to = "Logit"
  ) %>%
  # Split at the LAST dash (covariate-species separator)
  mutate(
    Covariate = sub("-[^-]+$", "", Parameter),
    Species = sub("^.*-", "", Parameter),
    Prob = plogis(Logit)
  ) %>%
  group_by(Species, Covariate) %>%
  summarise(
    Mean_logit = mean(Logit, na.rm = TRUE),
    Mean_prob = mean(Prob, na.rm = TRUE),
    LCL_prob = quantile(Prob, 0.025, na.rm = TRUE),
    UCL_prob = quantile(Prob, 0.975, na.rm = TRUE)
  ) %>%
  ungroup()

# Preview the table
print(species_cov_table_winter)
```

Appendix C14: R script for species-level covariate effects for summer

This R script extracts species-specific posterior estimates for occupancy covariates from the summer multispecies occupancy model. For each species and covariate, the script calculates the mean logit value, converts it to probability scale, and computes the 95% credible interval.

```
# ============================================================
# Species-level covariate extraction: Summer (msPGOcc)
# ============================================================

library(dplyr)
library(tidyr)
library(stringr)

# Extract posterior samples
beta_samples <- as.data.frame(ms_model_plus3$beta.samples)
```

```
# Convert to long format
species_cov_table_summer <- beta_samples %>%
  pivot_longer(
    cols = everything(),
    names_to = "Parameter",
    values_to = "Logit"
  ) %>%
  # Split at the LAST dash (covariate-species separator)
  mutate(
    Covariate = sub("-[^-]+$", "", Parameter),
    Species = sub("^.*-", "", Parameter),
    Prob = plogis(Logit)
  ) %>%
  group_by(Species, Covariate) %>%
  summarise(
    Mean_logit = mean(Logit, na.rm = TRUE),
    Mean_prob = mean(Prob, na.rm = TRUE),
    LCL_prob = quantile(Prob, 0.025, na.rm = TRUE),
    UCL_prob = quantile(Prob, 0.975, na.rm = TRUE)
  ) %>%
  ungroup()

# Preview
print(species_cov_table_summer)
```

Appendix C15: R script for species response curves for winter

This R script extracts species-specific posterior estimates for occupancy covariates from the winter multispecies occupancy model. For each species and covariate, the script calculates posterior predictions of occupancy probability across the observed range of covariate values, converts estimates from the logit scale to the probability scale, and computes 95% credible intervals. The predicted relationships are visualised as species response curves and arranged in a grid layout to allow comparison across species and study areas.

```
############################################################
## WINTER COVARIATE PANELS — GRID VERSION
## 2 plots per row, bold species names
############################################################

library(ggplot2)
library(patchwork)

inv_logit <- function(x) 1 / (1 + exp(-x))

cat(
  "SCRIPT STARTED AT:",
  format(Sys.time(), "%Y-%m-%d %H:%M:%S"),
  "\n"
)

# ------------------------------------------------
# MODEL & DATA  (WINTER)
# ------------------------------------------------

model    <- ms_model_winter
cov_data <- Masters_covariate.data_2026.unscaled.winter

# ------------------------------------------------
# SPECIES
# ------------------------------------------------

species <- c(
  "CapeClawless",
  "Bushpig",
  "CapeBushbuck",
  "CapeGenet",
  "CapePorcu",
```

```
  "Chacmabab",
  "Watermongoose"
)

species_labels <- c(
  CapeClawless  = "African Clawless Otter",
  Bushpig       = "Bushpig",
  CapeBushbuck  = "Cape Bushbuck",
  CapeGenet     = "Cape Genet",
  CapePorcu     = "Cape Porcupine",
  Chacmabab     = "Chacma Baboon",
  Watermongoose = "Water Mongoose"
)

# ------------------------------------------------
# COVARIATES
# ------------------------------------------------

covariates <- c(
  "Dissolved.oxygen",
  "Canopy_Cover",
  "Ground_vegetation_cover",
  "Conductivity",
  "SASS"
)

cov_labels <- c(
  Dissolved.oxygen        = "Dissolved oxygen (%)",
  Canopy_Cover            = "Canopy cover (%)",
  Ground_vegetation_cover = "Ground vegetation cover (%)",
  Conductivity            = "Conductivity (µS/cm)",
  SASS                    = "SASS score"
)

percent_covs <- c(
  "Dissolved.oxygen",
  "Canopy_Cover",
  "Ground_vegetation_cover"
)

# ------------------------------------------------
# AREAS
# ------------------------------------------------

areas <- c(
  "Buffeljagsriver",
```

```
  "Grootvadersbosch River"
)

# ------------------------------------------------
# OUTPUT DIRECTORY
# ------------------------------------------------

out_dir <- "Winter_covariate_panels_GRID_BOLD"
dir.create(out_dir, showWarnings = FALSE)

cat("SAVING TO:\n", out_dir, "\n")

# ------------------------------------------------
# SINGLE SPECIES PLOT
# ------------------------------------------------

plot_winter_effect <- function(sp, cov, area) {

  x_all <- as.numeric(gsub(",", ".", cov_data[[cov]]))
  x_center <- mean(x_all, na.rm = TRUE)
  x_scale  <- sd(x_all, na.rm = TRUE)

  x_area <- as.numeric(
    gsub(",", ".", cov_data[cov_data$Area == area, cov])
  )

  if (cov %in% percent_covs) {
    x_raw <- seq(0, 100, length.out = 200)
  } else {
    x_raw <- seq(
      min(x_area, na.rm = TRUE),
      max(x_area, na.rm = TRUE),
      length.out = 200
    )
  }

  x_scaled <- (x_raw - x_center) / x_scale

  beta <- model$beta.samples
  b0   <- beta[, paste0("(Intercept)-", sp)]
  b1   <- beta[, paste0(cov, "-", sp)]

  b_area <- rep(0, length(b0))
  if (area == "Grootvadersbosch River") {
    b_area <- beta[, paste0("AreaGrootvadersbosch River-", sp)]
  }
```

```
  psi <- inv_logit(
    sapply(x_scaled, function(x) b0 + b_area + b1 * x)
  )

  df <- data.frame(
    x  = x_raw,
    mu = colMeans(psi),
    lo = apply(psi, 2, quantile, 0.025),
    hi = apply(psi, 2, quantile, 0.975)
  )

  ggplot(df, aes(x = x, y = mu)) +
    geom_line(linewidth = 0.9) +
    geom_line(aes(y = lo), linetype = "dotted", linewidth = 0.5) +
    geom_line(aes(y = hi), linetype = "dotted", linewidth = 0.5) +
    coord_cartesian(ylim = c(0, 1)) +
    labs(
      x = cov_labels[cov],
      y = expression(paste("Occupancy probability (", psi, ")"))
    ) +
    ggtitle(species_labels[sp]) +
    theme_classic(base_size = 11) +
    theme(
      plot.title = element_text(
        size = 11,
        face = "bold",
        hjust = 0.5
      ),
      plot.margin = margin(t = 16, r = 10, b = 6, l = 10)
    )
}

# ------------------------------------------------
# BUILD PANELS (GRID — 2 PER ROW)
# ------------------------------------------------

for (cov in covariates) {
  for (area in areas) {

    cat("BUILDING:", area, "-", cov, "\n")

    plots <- lapply(species, function(sp) {
      plot_winter_effect(sp, cov, area)
    })
```

```
    panel <- wrap_plots(plots, ncol = 2) +
      plot_annotation(
        title    = cov_labels[cov],
        subtitle = paste0(area, " (Winter)"),
        theme = theme(
          plot.title    = element_text(size = 16, face = "bold", hjust = 0.5),
          plot.subtitle = element_text(size = 12, hjust = 0.5)
        )
      )

    ggsave(
      file.path(
        out_dir,
        paste0(
          "WINTER_",
          gsub(" ", "", area),
          "_",
          cov,
          "_GRID.png"
        )
      ),
      panel,
      width  = 11.69,
      height = 8.27,
      dpi    = 300
    )
  }
}

cat("DONE ✓\n")
```

Appendix C16: R script for species response curves for summer

This R script extracts species-specific posterior estimates for occupancy covariates from the summer multispecies occupancy model. For each species and covariate, the script calculates posterior predictions of occupancy probability across the observed range of covariate values, converts estimates from the logit scale to the probability scale, and computes 95% credible intervals. The predicted relationships are visualised as species response curves and arranged in a grid layout to allow comparison across species and study areas.

```
############################################################
## SUMMER COVARIATE PANELS — FINAL, FULL SCRIPT
## Big grid layout (INTENTIONAL)
## Y-axis title retained, spacing fixed
############################################################

library(ggplot2)
library(patchwork)

inv_logit <- function(x) 1 / (1 + exp(-x))

cat(
  "SCRIPT STARTED AT:",
  format(Sys.time(), "%Y-%m-%d %H:%M:%S"),
  "\n"
)

# ------------------------------------------------
# MODEL & DATA
# ------------------------------------------------

model    <- ms_model_plus3
cov_data <- Masters.cov.data.summer.2026

# ------------------------------------------------
# SPECIES
# ------------------------------------------------

species <- c(
  "CapeClawless",
  "Bushpig",
```

```
  "CapeBushbuck",
  "CapeGenet",
  "CapePorcu",
  "ChacmaBab",
  "Watermongoose"
)

species_labels <- c(
  CapeClawless  = "African Clawless Otter",
  Bushpig       = "Bushpig",
  CapeBushbuck  = "Cape Bushbuck",
  CapeGenet     = "Cape Genet",
  CapePorcu     = "Cape Porcupine",
  ChacmaBab     = "Chacma Baboon",
  Watermongoose = "Water Mongoose"
)

# ------------------------------------------------
# COVARIATES
# ------------------------------------------------

covariates <- c(
  "Dissolved.oxygen",
  "Canopy_Cover",
  "Ground_vegetation_cover",
  "Conductivity",
  "SASS"
)

cov_labels <- c(
  Dissolved.oxygen        = "Dissolved oxygen (%)",
  Canopy_Cover            = "Canopy cover (%)",
  Ground_vegetation_cover = "Ground vegetation cover (%)",
  Conductivity            = "Conductivity (µS/cm)",
  SASS                    = "SASS score"
)

# ------------------------------------------------
# AREAS
# ------------------------------------------------
```

```
areas <- c(
  "Buffeljags River",
  "Grootvadersbosch River"
)

area_labels <- c(
  "Buffeljags River"       = "Buffeljagsriver",
  "Grootvadersbosch River" = "Grootvadersbosch River"
)

# ------------------------------------------------
# OUTPUT DIRECTORY
# ------------------------------------------------

out_dir <-
"C:/Users/Barba/Dropbox/PC/Documents/Summer_covariate_panels_FINAL_FIXED"
cat("SAVING TO:\n", out_dir, "\n")

# ------------------------------------------------
# SINGLE SPECIES PLOT
# ------------------------------------------------

plot_summer_effect <- function(sp, cov, area) {

  x_all    <- as.numeric(cov_data[[cov]])
  x_center <- mean(x_all, na.rm = TRUE)
  x_scale  <- sd(x_all, na.rm = TRUE)

  x_area <- as.numeric(cov_data[cov_data$Area == area, cov])

  x_raw <- seq(
    min(x_area, na.rm = TRUE),
    max(x_area, na.rm = TRUE),
    length.out = 200
  )

  x_scaled <- (x_raw - x_center) / x_scale

  beta <- model$beta.samples
  b0   <- beta[, paste0("(Intercept)-", sp)]
  b1   <- beta[, paste0(cov, "-", sp)]
```

```
  psi <- inv_logit(sapply(x_scaled, function(x) b0 + b1 * x))

  df <- data.frame(
    x  = x_raw,
    mu = colMeans(psi),
    lo = apply(psi, 2, quantile, 0.025),
    hi = apply(psi, 2, quantile, 0.975)
  )

  ggplot(df, aes(x = x, y = mu)) +
    geom_line(linewidth = 0.9) +
    geom_line(aes(y = lo), linetype = "dotted", linewidth = 0.5) +
    geom_line(aes(y = hi), linetype = "dotted", linewidth = 0.5) +
    coord_cartesian(ylim = c(0, 1)) +
    labs(
      x = cov_labels[cov],
      y = expression(paste("Occupancy probability (", psi, ")"))
    ) +
    ggtitle(species_labels[sp]) +
    theme_classic(base_size = 11) +
    theme(
      plot.title = element_text(
        size = 11,
        face = "bold",
        hjust = 0.5
      ),
      # 🔑 THIS is what fixes the overlap
      plot.margin = margin(t = 16, r = 10, b = 6, l = 10),
      axis.title.y = element_text(margin = margin(r = 8))
    )
}

# ------------------------------------------------
# BUILD PANELS (BIG GRID)
# ------------------------------------------------

for (cov in covariates) {
  for (area in areas) {

    cat("BUILDING:", area_labels[area], "-", cov, "\n")
```

```
    plots <- lapply(species, function(sp) {
      plot_summer_effect(sp, cov, area)
    })

    panel <- wrap_plots(plots, ncol = 2) +
      plot_annotation(
        title    = cov_labels[cov],
        subtitle = paste0(area_labels[area], " (Summer)"),
        theme = theme(
          plot.title    = element_text(size = 16, face = "bold", hjust = 0.5),
          plot.subtitle = element_text(size = 12, hjust = 0.5),
          plot.margin   = margin(t = 14)
        )
      )

    ggsave(
      file.path(
        out_dir,
        paste0(
          "SUMMER_",
          gsub(" ", "", area_labels[area]),
          "_",
          cov,
          ".png"
        )
      ),
      plot   = panel,
      width  = 11.69,
      height = 8.27,
      dpi    = 300
    )
  }
}

cat("DONE ✓\n")
```

Appendix C17: R script for plotting winter species-level occupancy by area

This R script visualizes and compares the mean occupancy probabilities (ψ) with 95% credible intervals for each riparian mammal species between the Buffeljags River and Grootvadersbosch River sites during winter. The plot was produced using ggplot2, showing grouped bars per species for the two areas.

```
library(ggplot2)
library(dplyr)

# ----------------------------
# Data setup
# ----------------------------
area_data <- data.frame(
  Species = c("Bushpig", "Cape Bushbuck", "African clawless Otter",
              "Cape Genet", "Cape Porcupine", "Chacma Baboon", "Water Mongoose",
              "Bushpig", "Cape Bushbuck", "African clawless Otter",
              "Cape Genet", "Cape Porcupine", "Chacma Baboon", "Water Mongoose"),
  Area = rep(c("Buffeljags", "Grootvadersbosch"), each = 7),
  Mean = c(0.36, 0.48, 0.47, 0.43, 0.52, 0.49, 0.56,
           0.41, 0.60, 0.52, 0.28, 0.62, 0.33, 0.63),
  CI_lower = c(0.08, 0.27, 0.19, 0.25, 0.21, 0.31, 0.32,
               0.11, 0.34, 0.16, 0.08, 0.23, 0.18, 0.30),
  CI_upper = c(0.64, 0.75, 0.98, 0.63, 0.97, 0.66, 0.86,
               0.78, 0.90, 0.99, 0.57, 1.00, 0.52, 0.99)
)

# ----------------------------
# Factor levels for proper ordering
# ----------------------------
# Species: Bushpig top
area_data$Species <- factor(area_data$Species, levels = rev(c("Bushpig", "Cape
Bushbuck", "African clawless Otter",
                                                              "Cape Genet", "Cape
Porcupine", "Chacma Baboon", "Water Mongoose")))

# Area: Buffeljags bar on top, Grootvadersbosch on bottom, also legend order
area_data$Area <- factor(area_data$Area, levels = c("Grootvadersbosch",
"Buffeljags"))

# ----------------------------
# Position dodge for grouped bars
# ----------------------------
dodge <- position_dodge(width = 0.7)

# ----------------------------
# Plot
# ----------------------------
ggplot(area_data, aes(x = Species, y = Mean, fill = Area)) +
  geom_col(position = dodge, width = 0.6, color = "grey20") +
  geom_errorbar(aes(ymin = CI_lower, ymax = CI_upper),
                position = dodge,
                width = 0,               # remove "T" ends
                color = "black",
```

```
                linewidth = 0.6) +
  scale_fill_manual(values = c("Buffeljags" = "#90caf9", "Grootvadersbosch" =
"#1565c0")) +
  coord_flip() +
  scale_y_continuous(limits = c(0,1)) +
  labs(
    title = "Species-level Occupancy Probability",
    subtitle = "Comparison between Buffeljags and Grootvadersbosch",
    y = "Occupancy probability (ψ)",
    x = NULL,
    fill = "Area"
  ) +
  theme_bw(base_size = 13) +
  theme(
    panel.grid = element_blank(),
    plot.title = element_text(hjust = 0.48, face = "bold"),  # nudge slightly
left
    plot.subtitle = element_text(hjust = 0.5),
    legend.position = "bottom"
  )
```

Appendix C18: R script for plotting summer species-level occupancy by area

This R script visualizes and compares the mean occupancy probabilities (ψ) with 95% credible intervals for each riparian mammal species between the Buffeljags River and Grootvadersbosch River sites during summer. The plot was produced using ggplot2, showing grouped bars per species for the two areas.

```
library(ggplot2)
library(dplyr)

# ------------------------------
# Summer occupancy data
# ------------------------------
area_data <- data.frame(
  Species = c("Bushpig", "Cape Bushbuck", "African clawless Otter",
              "Cape Genet", "Cape Porcupine", "Chacma Baboon", "Water Mongoose",
              "Bushpig", "Cape Bushbuck", "African clawless Otter",
              "Cape Genet", "Cape Porcupine", "Chacma Baboon", "Water Mongoose"),
  Area = rep(c("Buffeljags", "Grootvadersbosch"), each = 7),
  Mean = c(0.65, 0.76, 0.68, 0.73, 0.67, 0.75, 0.71,
           0.50, 0.63, 0.33, 0.31, 0.38, 0.66, 0.60),
  CI_lower = c(0.19, 0.54, 0.36, 0.51, 0.26, 0.55, 0.47,
               0, 0.26, 0, 0, 0, 0.22, 0.14),
  CI_upper = c(0.90, 0.96, 0.88, 0.96, 0.96, 0.93, 0.93,
               1, 1, 0.99, 0.98, 0.99, 1, 1)
)

# ------------------------------
# Factor levels for proper ordering
# ------------------------------
# Species: Bushpig top
area_data$Species <- factor(area_data$Species,
```

```
                            levels = rev(c("Bushpig", "Cape Bushbuck", "African
clawless Otter",
                                           "Cape Genet", "Cape Porcupine",
"Chacma Baboon", "Water Mongoose")))

# Area: Buffeljags first in legend, Grootvadersbosch second
area_data$Area <- factor(area_data$Area, levels = c("Buffeljags",
"Grootvadersbosch"))

# Arrange data by Species then Area to ensure Buffeljags bar always first
area_data <- area_data %>% arrange(Species, Area)

# ----------------------------
# Position dodge for grouped bars
# ----------------------------
dodge <- position_dodge(width = 0.7)

# ----------------------------
# Plot
# ----------------------------
ggplot(area_data, aes(x = Species, y = Mean, fill = Area)) +
  geom_col(position = dodge, width = 0.6, color = "grey20") +
  geom_errorbar(aes(ymin = CI_lower, ymax = CI_upper),
                position = dodge,
                width = 0,
                color = "black",
                linewidth = 0.6) +
  scale_fill_manual(
    values = c("Buffeljags" = "#a5d6a7",      # light green
               "Grootvadersbosch" = "#2e7d32"), # dark green
    breaks = c("Buffeljags", "Grootvadersbosch")
  ) +
  coord_flip() +
  scale_y_continuous(limits = c(0,1)) +
  labs(
    title = "Species-level Occupancy Probability",
    subtitle = "Comparison between Buffeljags and Grootvadersbosch",
    y = "Occupancy probability (ψ)",
    x = NULL,
    fill = "Area"
  ) +
  theme_bw(base_size = 13) +
  theme(
    panel.grid = element_blank(),
    plot.title = element_text(hjust = 0.48, face = "bold"),
    plot.subtitle = element_text(hjust = 0.5),
    legend.position = "bottom"
  )
```

Appendix C19: R script for species-level detection probability vs. standardized sampling effort

This R script generates species-specific detection probability curves across a standardized sampling effort gradient (0–1) for both winter and summer seasons, using posterior estimates from the fitted multispecies occupancy model (spOccupancy msPGOcc). For each species and study area, the script constructs

a detection design matrix holding all covariates at their mean values except for sampling effort, predicts detection probabilities, and calculates posterior mean estimates. The resulting curves illustrate how detection probability changes with increasing effort, are colour-coded by species, and are facetted by study area to allow clear visual comparison across taxa and sites

```
############################################################
## DETECTION PROBABILITY vs STANDARDISED SAMPLING EFFORT
## spOccupancy msPGOcc
## CONSISTENT FOR WINTER & SUMMER
############################################################

library(spOccupancy)
library(ggplot2)
library(dplyr)

# ----------------------------------------------------------
# 0. CHOOSE MODEL (CHANGE ONLY THIS LINE)
# ----------------------------------------------------------

# model_obj <- ms_model_plus3   # summer (model plus 3)
model_obj <- ms_model_winter  # winter

# ----------------------------------------------------------
# 1. Species order
# ----------------------------------------------------------

species <- c(
  "African Clawless Otter",
  "Cape Bushbuck",
  "Cape Porcupine",
  "Water Mongoose",
  "Bushpig",
  "Cape Genet",
  "Chacma Baboon"
)

# ----------------------------------------------------------
# 2. Species colour palette (FINAL)
# ----------------------------------------------------------

species_cols <- c(
  "Bushpig"                = "#F8766D",
  "Cape Bushbuck"          = "#C49A00",
  "African Clawless Otter" = "#53B400",
  "Cape Genet"             = "#00C094",
  "Cape Porcupine"         = "#00B6EB",
  "Chacma Baboon"          = "#A58AFF",
  "Water Mongoose"         = "#FB61D7"
)

# ----------------------------------------------------------
# 3. Effort gradient (MODEL SCALE: 0–1)
# ----------------------------------------------------------

effort_seq <- seq(0, 1, length.out = 100)

# ----------------------------------------------------------
# 4. Build detection design matrix
# ----------------------------------------------------------

X_p <- model_obj$X.p
```

```
X_p_means <- colMeans(X_p)

X_p_new <- matrix(
  rep(X_p_means, each = length(effort_seq)),
  nrow = length(effort_seq),
  byrow = FALSE
)

colnames(X_p_new) <- colnames(X_p)

# Replace effort column with gradient
X_p_new[, "effort"] <- effort_seq

# ------------------------------------------------------------
# 5. Predict detection probability
# ------------------------------------------------------------

pred <- predict(
  object    = model_obj,
  type      = "detection",
  X.0       = X_p_new,
  ignore.RE = TRUE
)

# Posterior mean p
p_mean <- apply(pred$p.0.samples, c(2, 3), mean)

# ------------------------------------------------------------
# 6. Tidy dataframe
# ------------------------------------------------------------

det_df_base <- do.call(
  rbind,
  lapply(seq_along(species), function(i) {
    data.frame(
      Species = species[i],
      Effort  = effort_seq,
      p       = p_mean[i, ]
    )
  })
)

det_df <- bind_rows(
  transform(det_df_base, Area = "Buffeljags River"),
  transform(det_df_base, Area = "Grootvadersbosch River")
)

# ------------------------------------------------------------
# 7. Plot (FINAL)
# ------------------------------------------------------------

ggplot(det_df, aes(x = Effort, y = p, colour = Species)) +
  geom_line(linewidth = 1.0) +
  facet_wrap(~ Area, ncol = 2) +
  scale_colour_manual(values = species_cols) +
  scale_x_continuous(limits = c(0, 1)) +
  scale_y_continuous(limits = c(0, 1)) +
  guides(colour = guide_legend(nrow = 3, byrow = TRUE)) +
  labs(
    x = "Standardised sampling effort",
    y = "Detection probability (p)"
  ) +
  theme_classic(base_size = 13) +
  theme(
    legend.position = "bottom",
    legend.title    = element_blank(),
    strip.text      = element_text(face = "bold")
  )
```

Appendix C20: R script for plotting winter species-level logit covariate effects

This R script was used to create a dot plot of the estimated effects of site-level covariates on species-level occupancy in logit scale for the summer survey.

```
library(ggplot2)
library(dplyr)

# Winter logit data manually input
winter_logit <- data.frame(
  Species = c(
    rep("Bushpig",7), rep("Cape Bushbuck",7), rep("African clawless",7),
    rep("Cape Genet",7), rep("Cape Porcupine",7), rep("Chacma Baboon",7),
    rep("Water Mongoose",7)
  ),
  Covariate = rep(c("(Intercept)", "AreaGrootvadersbosch", "Canopy_Cover",
                    "Conductivity", "Dissolved.oxygen",
"Ground_vegetation_cover", "SASS"),7),
  Mean_logit = c(
    -0.879,0.249,0.453,-0.917,-0.0421,-0.632,-0.833,
    0.0229,0.752,0.0736,-0.851,0.146,-0.156,-0.417,
    0.107,0.399,0.523,-0.0597,-0.202,0.321,0.359,
    -0.439,-1.02,-0.989,-0.146,0.334,-0.937,0.204,
    0.251,1.15,0.187,0.00858,0.940,-0.258,0.206,
    -0.158,-0.913,-0.396,-0.390,1.46,-0.312,0.0457,
    0.551,0.696,-0.0916,-0.121,0.517,-0.923,-0.366
  )
)

# Clean covariate names
winter_logit <- winter_logit %>%
  mutate(Covariate = recode(Covariate,
                            "(Intercept)" = "Intercept",
                            "Ground_vegetation_cover" = "Ground veg",
                            "Dissolved.oxygen" = "DO",
                            "AreaGrootvadersbosch" = "AreaGVB",
                            "Canopy_Cover" = "Canopy"
  ))

# Set factor levels: Intercept first, then alphabetically
cov_levels <- c("Intercept",
sort(unique(winter_logit$Covariate[winter_logit$Covariate != "Intercept"])))
winter_logit$Covariate <- factor(winter_logit$Covariate, levels = cov_levels)

# Vertical plot
ggplot(winter_logit, aes(x = Covariate, y = Mean_logit, color = Species)) +
  geom_point(size = 3, position = position_dodge(width = 0.6)) +
  geom_hline(yintercept = 0, linetype = "dashed", color = "black") +
  labs(title = "Winter Species-level Logit Covariate Effects",
       x = "Covariate",
       y = "Logit effect") +
  theme_minimal(base_size = 14) +
  theme(axis.text.x = element_text(angle = 45, hjust = 1)) +
```

```
scale_y_continuous(limits = c(-4.5, 3), breaks = seq(-4,3,1))
```

Appendix C21: R script for plotting summer species-level logit covariate effects

This R script was used to create a dot plot of the estimated effects of site-level covariates on species-level occupancy in logit scale for the summer survey.

```
library(ggplot2)
library(dplyr)

# Update species names for readability
logit_df <- logit_df %>%
  mutate(Species = recode(Species,
                          "CapeBushbuck" = "Cape Bushbuck",
                          "CapeClawless" = "African clawless",
                          "CapeGenet" = "Cape Genet",
                          "CapePorcu" = "Cape Porcupine",
                          "ChacmaBab" = "Chacma Baboon",
                          "WaterMongoose" = "Water Mongoose"
  ))

# Update covariate names
logit_df <- logit_df %>%
  mutate(Covariate = recode(Covariate,
                            "Ground_vegetation_cover" = "Ground veg",
                            "Dissolved.oxygen" = "DO",
                            "AreaGrootvadersbosch" = "AreaGVB",
                            "Canopy_Cover" = "Canopy",
                            "(Intercept)" = "Intercept"
  ))

# Set factor levels to have Intercept first, then alphabetically
cov_levels <- c("Intercept", sort(unique(logit_df$Covariate[logit_df$Covariate !=
"Intercept"])))
logit_df$Covariate <- factor(logit_df$Covariate, levels = cov_levels)

# Plot
ggplot(logit_df, aes(x = Covariate, y = Mean_logit, color = Species)) +
  geom_point(size = 3, position = position_dodge(width = 0.6)) +
  geom_hline(yintercept = 0, linetype = "dashed", color = "black") +
  labs(title = "Summer Species-level Logit Covariate Effects",
       x = "Covariate",
       y = "Logit effect") +
  theme_minimal(base_size = 14) +
  theme(axis.text.x = element_text(angle = 45, hjust = 1)) +
  scale_y_continuous(limits = c(-4.5, 3), breaks = seq(-4, 3, 1))
```

Appendix C22: R script for plotting species accumulation curves

This R script generates temporal and spatial species accumulation curves for riparian mammal communities at the Buffeljags River and Grootvadersbosch River study areas. Using camera trap detection data, the script constructs presence–absence matrices for each species by aggregating detections across cameras and survey days. Temporal species accumulation curves are calculated by treating survey days as sampling units, while spatial species accumulation curves are calculated by treating camera traps as sampling units. Species accumulation curves were produced separately for winter and summer surveys using functions from the *vegan* package, with spatial curves smoothed using randomised permutations and associated confidence intervals.

```
# ================================
# SPECIES ACCUMULATION CURVES
# Buffeljags & Grootvadersbosch
# ================================

library(vegan)

# --- Load your CSV ---
df <- read.csv("C:/Users/Barba/Dropbox/PC/Downloads/Masters occupancy
data_2025_all sites_species accumulation curves.csv", sep=";")

# --- Replace NAs with 0 ---
day_cols <- paste0("Day", 1:60)
df[, day_cols][is.na(df[, day_cols])] <- 0

# --- Helper function to create presence/absence matrix for specaccum ---
create_pa_matrix <- function(df, cam_start, cam_end, day_start, day_end){
  day_cols <- paste0("Day", day_start:day_end)
  cams <- paste0("C", cam_start:cam_end)

  mat <- matrix(0, nrow=length(day_cols), ncol=11)  # rows = days, cols = species
  colnames(mat) <- unique(df$Species)
  rownames(mat) <- day_cols

  for(i in seq_along(day_cols)){
    day <- day_cols[i]
    # For this day, is each species detected in any camera?
    for(sp in unique(df$Species)){
      rows <- df$Camera.trap %in% cams & df$Species == sp
      mat[i, sp] <- as.numeric(any(df[rows, day] > 0))
    }
  }
  return(mat)
}

# --- Buffeljags ---
buf_winter <- create_pa_matrix(df, 1, 20, 1, 30)
buf_summer <- create_pa_matrix(df, 1, 20, 31, 60)
```

```
# --- Grootvadersbosch ---
gvb_winter <- create_pa_matrix(df, 21, 40, 1, 30)
gvb_summer <- create_pa_matrix(df, 21, 40, 31, 60)

# --- Build specaccum objects (temporal) ---
accum_buf_winter <- specaccum(buf_winter, method="collector")
accum_buf_summer <- specaccum(buf_summer, method="collector")

accum_gvb_winter <- specaccum(gvb_winter, method="collector")
accum_gvb_summer <- specaccum(gvb_summer, method="collector")

# --- Plot Buffeljags (Temporal) ---
par(mar=c(5,5,4,7))
plot(accum_buf_winter, col="blue", lwd=2,
     xlab="Days", ylab="Cumulative Species Richness",
     xlim=c(1,30), ylim=c(0,12),
     xaxt="n", yaxt="n")
lines(accum_buf_summer, col="green4", lwd=2)
title(main="Buffeljags: Temporal Species Accumulation", line=1)
axis(1, at=seq(0,30,5))
axis(2, at=seq(0,12,2))
usr <- par("usr")
xrange <- usr[2] - usr[1]
legend(x=usr[2]+0.02*xrange, y=usr[4],
       legend=c("Winter","Summer"), col=c("blue","green4"),
       lwd=2, xpd=TRUE, bty="n")

# --- Plot Grootvadersbosch (Temporal) ---
par(mar=c(5,5,4,7))
plot(accum_gvb_winter, col="blue", lwd=2,
     xlab="Days", ylab="Cumulative Species Richness",
     xlim=c(1,30), ylim=c(0,12),
     xaxt="n", yaxt="n")
lines(accum_gvb_summer, col="green4", lwd=2)
title(main="Grootvadersbosch: Temporal Species Accumulation", line=2)
axis(1, at=seq(0,30,5))
axis(2, at=seq(0,12,2))
usr <- par("usr")
xrange <- usr[2] - usr[1]
legend(x=usr[2]+0.02*xrange, y=usr[4],
       legend=c("Winter","Summer"), col=c("blue","green4"),
       lwd=2, xpd=TRUE, bty="n")


# ============================================================
# SPATIAL SPECIES ACCUMULATION CURVES (Cameras as Samples)
# ============================================================

# --- Helper function for spatial accumulation ---
create_spatial_matrix <- function(df, cam_start, cam_end, day_start, day_end){
  day_cols <- paste0("Day", day_start:day_end)
  cams <- paste0("C", cam_start:cam_end)

  mat <- matrix(0, nrow=length(cams), ncol=11)  # rows = cameras, cols = species
  colnames(mat) <- unique(df$Species)
  rownames(mat) <- cams

  for(cam in cams){
```

```
    for(sp in unique(df$Species)){
      rows <- df$Camera.trap == cam & df$Species == sp
      mat[cam, sp] <- as.numeric(any(df[rows, day_cols] > 0))
    }
  }
  return(mat)
}

# --- Buffeljags (Spatial) ---
buf_spatial_winter <- create_spatial_matrix(df, 1, 20, 1, 30)
buf_spatial_summer <- create_spatial_matrix(df, 1, 20, 31, 60)

# --- Grootvadersbosch (Spatial) ---
gvb_spatial_winter <- create_spatial_matrix(df, 21, 40, 1, 30)
gvb_spatial_summer <- create_spatial_matrix(df, 21, 40, 31, 60)

# --- Build specaccum objects (spatial, smoothed random method) ---
accum_buf_spatial_winter <- specaccum(buf_spatial_winter, method="random",
permutations=100)
accum_buf_spatial_summer <- specaccum(buf_spatial_summer, method="random",
permutations=100)

accum_gvb_spatial_winter <- specaccum(gvb_spatial_winter, method="random",
permutations=100)
accum_gvb_spatial_summer <- specaccum(gvb_spatial_summer, method="random",
permutations=100)

# --- Plot Buffeljags (Spatial, smoothed) ---
par(mar=c(5,5,4,7))
plot(accum_buf_spatial_winter, ci.type="poly", col="blue", lwd=2,
     ci.lty=0, ci.col=adjustcolor("blue", alpha.f=0.2),
     xlab="Cameras", ylab="Cumulative Species Richness",
     xlim=c(1,20), ylim=c(0,12),
     xaxt="n", yaxt="n")
lines(accum_buf_spatial_summer, ci.type="poly", col="green4", lwd=2,
      ci.lty=0, ci.col=adjustcolor("green4", alpha.f=0.2))
title(main="Buffeljags: Spatial Species Accumulation", line=1)
axis(1, at=seq(0,20,5))
axis(2, at=seq(0,12,2))
usr <- par("usr")
xrange <- usr[2] - usr[1]
legend(x=usr[2]+0.02*xrange, y=usr[4],
       legend=c("Winter","Summer"),
       col=c("blue","green4"), lwd=2,
       xpd=TRUE, bty="n")

# --- Plot Grootvadersbosch (Spatial, smoothed) ---
par(mar=c(5,5,4,7))
plot(accum_gvb_spatial_winter, ci.type="poly", col="blue", lwd=2,
     ci.lty=0, ci.col=adjustcolor("blue", alpha.f=0.2),
     xlab="Cameras", ylab="Cumulative Species Richness",
     xlim=c(1,20), ylim=c(0,12),
     xaxt="n", yaxt="n")
lines(accum_gvb_spatial_summer, ci.type="poly", col="green4", lwd=2,
      ci.lty=0, ci.col=adjustcolor("green4", alpha.f=0.2))
title(main="Grootvadersbosch: Spatial Species Accumulation", line=2)
axis(1, at=seq(0,20,5))
axis(2, at=seq(0,12,2))
usr <- par("usr")
xrange <- usr[2] - usr[1]
```

```
legend(x=usr[2]+0.02*xrange, y=usr[4],
       legend=c("Winter","Summer"),
       col=c("blue","green4"), lwd=2,
       xpd=TRUE, bty="n")
```

Appendix D: Console results

Full R console output for posterior summaries of winter and summer multispecies occupancy models

Appendix D1: Winter multispecies model console results

```
Species codes:
sp1 = Bushpig
sp2 = Cape Bushbuck
sp3 = African clawless Otter
sp4 = Cape Genet
sp5 = Cape Porcupine
sp6 = Chacma Baboon
sp7 = Water Mongoose

Call:
msPGOcc(occ.formula = occ_formula, det.formula = det_formula,
    data = list(y = y_clean, occ.covs = site_covs_scaled, det.covs = obs_covs_list),
    n.samples = 3000, n.omp.threads = 4, verbose = TRUE, n.report = 500,
    n.burn = 500, n.thin = 5, n.chains = 2)

Samples per Chain: 3000
Burn-in: 500
Thinning Rate: 5
Number of Chains: 2
Total Posterior Samples: 1000
Run Time (min): 0.4103

----------------------------------------
        Community Level
----------------------------------------
Occurrence Means (logit scale):
                            Mean     SD    2.5%     50%  97.5%   Rhat ESS
(Intercept)               -0.0451 0.7207 -1.1854 -0.1752 1.7650 1.0782  84
AreaGrootvadersbosch River  0.1615 0.8077 -1.3926  0.1240 1.9073 1.0646 307
Dissolved.oxygen            0.3976 0.6330 -0.9090  0.3888 1.7088 1.0170 173
Canopy_Cover              -0.1034 0.5586 -1.1178 -0.1531 1.1916 1.0442 395
Ground_vegetation_cover   -0.4111 0.5218 -1.4708 -0.4028 0.5935 1.1211 360
SASS                      -0.2080 0.5214 -1.3237 -0.1997 0.8022 1.0149 327
Conductivity              -0.4196 0.5265 -1.3688 -0.4052 0.6523 1.0023 265

Occurrence Variances (logit scale):
                             Mean     SD   2.5%    50%   97.5%   Rhat ESS
(Intercept)                2.1505 5.8014 0.0582 0.5193 16.0394 1.2940  37
AreaGrootvadersbosch River 3.1261 6.4006 0.0616 1.2369 17.6736 1.0993 145
Dissolved.oxygen           2.0060 3.9431 0.0963 0.8583 12.6446 1.1408  49
Canopy_Cover               1.2169 2.0797 0.0520 0.5625  6.9926 1.0408  99
Ground_vegetation_cover    1.5410 3.2885 0.0634 0.6009  9.2736 1.0282 114
SASS                       0.8945 1.3587 0.0544 0.4416  4.9541 1.0710 324
Conductivity               0.9526 1.7340 0.0479 0.3822  6.3107 1.0222 125
```

```
Detection Means (logit scale):
               Mean     SD    2.5%     50%   97.5%   Rhat ESS
(Intercept) -4.9789 0.8762 -6.3692 -5.0566 -2.9491 1.0302 324
effort       2.0342 0.6873  0.6477  2.0523  3.3995 1.1630 154

Detection Variances (logit scale):
              Mean     SD   2.5%    50%   97.5%   Rhat ESS
(Intercept) 2.0442 4.4629 0.0695 0.8345 11.8635 1.2404 314
effort      1.0349 1.2968 0.0519 0.6483  4.5802 1.0444 388

----------------------------------------
        Species Level
----------------------------------------
Occurrence (logit scale):
                                  Mean     SD    2.5%     50%  97.5%   Rhat
(Intercept)-sp1                -0.7416 1.2280 -3.0136 -0.7386 1.5238 1.0517
(Intercept)-sp2                 0.0582 0.7372 -1.0174 -0.0660 2.0355 1.0000
(Intercept)-sp3                 0.2419 1.6512 -1.8213 -0.1012 5.4688 1.0567
(Intercept)-sp4                -0.4601 0.6357 -1.6951 -0.4735 0.8789 1.0026
(Intercept)-sp5                 0.2943 1.5111 -1.8990 -0.0120 4.3635 1.0453
(Intercept)-sp6                -0.2175 0.4687 -1.1139 -0.2246 0.7236 0.9992
(Intercept)-sp7                 0.8423 2.0297 -0.8915  0.3797 6.3617 1.2897
AreaGrootvadersbosch River-sp1  0.3649 1.1675 -1.9247  0.2882 2.9199 1.1272
AreaGrootvadersbosch River-sp2  0.7780 1.2564 -1.1540  0.6825 3.0589 1.1067
AreaGrootvadersbosch River-sp3  0.5142 1.6981 -2.1058  0.2334 4.2674 1.0893
AreaGrootvadersbosch River-sp4 -0.8609 1.1880 -3.4336 -0.7928 1.4932 1.0341
AreaGrootvadersbosch River-sp5  1.1359 1.9982 -1.4562  0.6260 5.7879 1.0361
AreaGrootvadersbosch River-sp6 -0.7730 0.8553 -2.5844 -0.7118 0.7519 1.0011
AreaGrootvadersbosch River-sp7  0.8791 1.5612 -1.3009  0.5749 4.7887 1.0077
Dissolved.oxygen-sp1           -0.2295 1.1696 -3.1785 -0.0436 1.5444 1.0426
Dissolved.oxygen-sp2            0.1203 0.6798 -1.2985  0.1398 1.3583 1.0060
Dissolved.oxygen-sp3           -0.1590 1.3746 -3.2590 -0.0580 2.0395 1.0849
Dissolved.oxygen-sp4            0.3454 0.7158 -1.1113  0.3224 1.7657 1.0086
Dissolved.oxygen-sp5            1.0245 1.7639 -1.2119  0.7457 6.4599 1.5366
Dissolved.oxygen-sp6            1.5273 0.6307  0.4839  1.4897 2.8935 1.0006
Dissolved.oxygen-sp7            0.4150 1.0241 -1.9404  0.4196 2.3161 1.0379
Canopy_Cover-sp1                0.3182 0.9426 -1.1129  0.1590 2.6194 1.1675
Canopy_Cover-sp2                0.0422 0.6029 -1.0881  0.0079 1.4138 1.0424
Canopy_Cover-sp3                0.4107 1.1536 -1.3292  0.2068 3.3862 1.0085
Canopy_Cover-sp4               -1.0750 0.7813 -2.8937 -0.9528 0.1527 1.0149
Canopy_Cover-sp5                0.0281 0.9788 -1.6013 -0.0626 2.3882 1.0276
Canopy_Cover-sp6               -0.3981 0.4175 -1.2505 -0.3893 0.4370 1.0436
Canopy_Cover-sp7               -0.1172 0.7095 -1.4911 -0.1365 1.3979 1.0354
Ground_vegetation_cover-sp1    -0.7923 0.8784 -2.8821 -0.6703 0.5834 1.0130
Ground_vegetation_cover-sp2    -0.1658 0.5254 -1.1563 -0.1736 0.9397 1.0832
Ground_vegetation_cover-sp3     0.5488 1.6154 -1.4289  0.2733 4.2217 1.1897
Ground_vegetation_cover-sp4    -1.1088 0.9248 -3.1518 -0.9312 0.0441 1.0388
```

```
Ground_vegetation_cover-sp5    -0.2475 0.9114 -2.1273 -0.2400 1.5060 1.0436
Ground_vegetation_cover-sp6    -0.2861 0.4041 -1.0372 -0.2938 0.5069 1.0070
Ground_vegetation_cover-sp7    -1.0265 0.8521 -2.9626 -0.8981 0.3661 1.0299
SASS-sp1                       -0.7779 0.8542 -2.7140 -0.6955 0.6661 1.0133
SASS-sp2                       -0.4968 0.5994 -1.8083 -0.4512 0.6948 1.0108
SASS-sp3                        0.2212 0.8892 -1.2699  0.1037 2.2390 1.1085
SASS-sp4                        0.1897 0.6788 -0.9245  0.0980 1.6953 1.0796
SASS-sp5                       -0.0774 1.0110 -2.1776 -0.0935 2.0888 1.0839
SASS-sp6                       -0.0057 0.5167 -0.9425 -0.0225 1.0750 1.0280
SASS-sp7                       -0.4251 0.7837 -2.0579 -0.3764 1.1163 1.0008
Conductivity-sp1               -0.9629 0.9854 -3.5184 -0.7808 0.4528 1.0432
Conductivity-sp2               -0.9086 0.6216 -2.3946 -0.8352 0.0841 1.0390
Conductivity-sp3               -0.2500 0.8923 -2.0036 -0.2996 1.6828 1.0863
Conductivity-sp4               -0.2294 0.6463 -1.4092 -0.2627 1.1985 1.0081
Conductivity-sp5                0.0239 1.0777 -1.5871 -0.1483 2.5689 1.0101
Conductivity-sp6               -0.4873 0.4950 -1.6092 -0.4673 0.4254 1.0001
Conductivity-sp7               -0.2633 0.7410 -1.7174 -0.2676 1.3113 1.0226
                               ESS
(Intercept)-sp1                107
(Intercept)-sp2                147
(Intercept)-sp3                 52
(Intercept)-sp4                451
(Intercept)-sp5                 50
(Intercept)-sp6                682
(Intercept)-sp7                 39
AreaGrootvadersbosch River-sp1 364
AreaGrootvadersbosch River-sp2 201
AreaGrootvadersbosch River-sp3 118
AreaGrootvadersbosch River-sp4 254
AreaGrootvadersbosch River-sp5 102
AreaGrootvadersbosch River-sp6 575
AreaGrootvadersbosch River-sp7 146
Dissolved.oxygen-sp1           178
Dissolved.oxygen-sp2           292
Dissolved.oxygen-sp3            90
Dissolved.oxygen-sp4           510
Dissolved.oxygen-sp5            41
Dissolved.oxygen-sp6           325
Dissolved.oxygen-sp7           206
Canopy_Cover-sp1               131
Canopy_Cover-sp2               359
Canopy_Cover-sp3               143
Canopy_Cover-sp4               313
Canopy_Cover-sp5               223
Canopy_Cover-sp6               791
Canopy_Cover-sp7               415
Ground_vegetation_cover-sp1    274
Ground_vegetation_cover-sp2    614
```

```
Ground_vegetation_cover-sp3     96
Ground_vegetation_cover-sp4    129
Ground_vegetation_cover-sp5    227
Ground_vegetation_cover-sp6    812
Ground_vegetation_cover-sp7    289
SASS-sp1                       254
SASS-sp2                       428
SASS-sp3                       246
SASS-sp4                       237
SASS-sp5                       245
SASS-sp6                       432
SASS-sp7                       326
Conductivity-sp1               159
Conductivity-sp2               245
Conductivity-sp3               172
Conductivity-sp4               416
Conductivity-sp5               137
Conductivity-sp6               333
Conductivity-sp7               370

Detection (logit scale):
                   Mean     SD    2.5%     50%   97.5%   Rhat ESS
(Intercept)-sp1 -5.7099 0.9251 -7.7527 -5.6649 -4.0486 1.2705 117
(Intercept)-sp2 -4.9525 0.6745 -6.2825 -4.9383 -3.6517 1.0469 154
(Intercept)-sp3 -6.0518 1.0689 -8.4838 -5.9400 -4.3224 1.4401  78
(Intercept)-sp4 -5.1788 0.8437 -6.7882 -5.1618 -3.5783 1.1168 138
(Intercept)-sp5 -6.2520 1.1095 -8.6176 -6.1704 -4.2840 1.2480 119
(Intercept)-sp6 -4.4884 0.8539 -6.1252 -4.4933 -2.9446 1.0107 205
(Intercept)-sp7 -5.6113 0.8186 -7.3187 -5.5625 -4.1360 1.1749 153
effort-sp1       1.9863 0.8777  0.3150  1.9598  3.6668 1.1243 165
effort-sp2       2.3543 0.6655  1.0684  2.3406  3.6895 1.0388 219
effort-sp3       1.6653 0.9278 -0.0871  1.6831  3.5921 1.2962 167
effort-sp4       2.3904 0.8147  0.9112  2.3623  4.0094 1.1055 153
effort-sp5       1.4889 1.0548 -0.4760  1.4914  3.5865 1.3450 148
effort-sp6       3.0065 0.8563  1.4713  2.9799  4.6554 1.0063 232
effort-sp7       2.1130 0.7973  0.6015  2.0925  3.6987 1.1108 170
```

Appendix D2: Summer multispecies model console results

```
Call:
msPGOcc(occ.formula = occ_formula_plus3, det.formula = ~effort,
    data = list(y = y_clean, occ.covs = site_covs_plus3, det.covs = obs_covs_list_summer),
    n.samples = 2000, n.omp.threads = 2, verbose = TRUE, n.report = 500,
    n.burn = 500, n.thin = 5, n.chains = 2)

Samples per Chain: 2000
Burn-in: 500
Thinning Rate: 5
```

```
Number of Chains: 2
Total Posterior Samples: 600
Run Time (min): 0.313

----------------------------------------
        Community Level
----------------------------------------
Occurrence Means (logit scale):
                           Mean     SD    2.5%     50%  97.5%   Rhat ESS
(Intercept)              1.4522 0.9009 -0.0945  1.3816 3.3263 1.0119  28
AreaGrootvadersbosch    -0.5955 1.4961 -3.5665 -0.5741 2.2123 1.0286  89
Dissolved.oxygen         0.4225 1.3423 -2.2375  0.3550 3.1513 1.0435  59
Canopy_Cover             0.3818 0.6286 -0.8354  0.3988 1.6339 1.0341 145
Ground_vegetation_cover  0.7219 0.7174 -0.5183  0.6658 2.4420 1.5175  78
SASS                    -0.7956 1.0512 -2.9319 -0.7286 1.3288 1.1878  75
Conductivity            -0.3746 1.4084 -2.9150 -0.3541 2.5454 1.3975  25

Occurrence Variances (logit scale):
                           Mean      SD   2.5%    50%    97.5%   Rhat ESS
(Intercept)              2.4524 11.1517 0.0483 0.5268  17.0749 1.4014  60
AreaGrootvadersbosch    40.7508 84.2005 0.0846 4.4647 295.6643 3.9189   9
Dissolved.oxygen         2.3357  7.7946 0.0531 0.6132  13.6965 1.1520 117
Canopy_Cover             0.8908  2.4522 0.0421 0.3107   4.9643 1.5517  46
Ground_vegetation_cover  1.0769  2.2394 0.0440 0.3882   7.2373 1.5304 166
SASS                     7.6519 26.3887 0.0642 1.1130  64.9835 2.0277  54
Conductivity             2.0456  5.0643 0.0582 0.5882  12.9774 1.3045 112

Detection Means (logit scale):
               Mean     SD    2.5%     50%   97.5%   Rhat ESS
(Intercept) -3.3544 0.9261 -4.8394 -3.4683 -1.1105 1.1149 155
effort      -0.3785 0.4003 -1.1939 -0.3673  0.4255 1.0543 145

Detection Variances (logit scale):
              Mean     SD   2.5%    50%   97.5%   Rhat ESS
(Intercept) 5.8806 8.0825 0.5074 3.2874 27.3742 1.2703  24
effort      0.4213 0.8131 0.0349 0.2006  2.0468 1.0363 377

----------------------------------------
        Species Level
----------------------------------------
Occurrence (logit scale):
                                       Mean     SD     2.5%     50%   97.5%
(Intercept)-Bushpig                  1.2187 1.5299  -2.4477  1.2537  4.2233
(Intercept)-CapeBushbuck             1.8496 1.2092   0.2258  1.5796  4.6673
(Intercept)-CapeClawless             1.3109 1.3018  -1.0701  1.2594  3.8027
(Intercept)-CapeGenet                1.8529 1.3086  -0.0361  1.6039  5.2192
(Intercept)-CapePorcu                1.5295 2.0250  -1.9579  1.3211  5.0452
(Intercept)-ChacmaBab                1.8878 1.2708   0.2757  1.6297  4.9381
```

```
(Intercept)-Watermongoose              1.5316 1.3636  -0.9073  1.4313  4.5114
AreaGrootvadersbosch-Bushpig          -0.1949 5.4619 -10.3097 -0.5358 12.4366
AreaGrootvadersbosch-CapeBushbuck      1.0919 4.4849  -3.5917 -0.0059 14.6302
AreaGrootvadersbosch-CapeClawless     -3.5288 5.6222 -18.0668 -2.1337  4.6249
AreaGrootvadersbosch-CapeGenet        -4.4270 5.8834 -19.6309 -2.5614  2.9673
AreaGrootvadersbosch-CapePorcu        -3.9728 6.9497 -22.9732 -1.6151  4.4668
AreaGrootvadersbosch-ChacmaBab         1.2968 4.6697  -3.6968  0.0492 14.0919
AreaGrootvadersbosch-Watermongoose     1.0150 5.2626  -6.2951 -0.2366 14.7439
Dissolved.oxygen-Bushpig               0.6861 2.1399  -2.8838  0.4160  5.2741
Dissolved.oxygen-CapeBushbuck          0.0485 1.6420  -3.1411  0.0639  3.0055
Dissolved.oxygen-CapeClawless          0.8507 1.8932  -2.1893  0.6180  5.0028
Dissolved.oxygen-CapeGenet             0.6195 1.7920  -2.3475  0.4127  4.2361
Dissolved.oxygen-CapePorcu             0.4406 2.0037  -3.1665  0.3095  4.5383
Dissolved.oxygen-ChacmaBab             0.1950 1.6676  -2.7157  0.1440  3.3718
Dissolved.oxygen-Watermongoose         0.6500 1.6875  -2.5552  0.4874  4.2004
Canopy_Cover-Bushpig                   0.4948 1.0870  -1.4814  0.4498  2.7891
Canopy_Cover-CapeBushbuck              0.4415 0.7008  -0.7206  0.4103  2.0358
Canopy_Cover-CapeClawless              0.5066 0.9457  -1.2628  0.4920  2.6621
Canopy_Cover-CapeGenet                 0.5063 0.9166  -0.9590  0.4437  2.4024
Canopy_Cover-CapePorcu                 0.2254 1.1168  -2.4676  0.3014  2.1126
Canopy_Cover-ChacmaBab                 0.3721 0.6682  -0.8884  0.3367  1.6550
Canopy_Cover-Watermongoose             0.3641 0.8685  -1.4324  0.3534  2.0392
Ground_vegetation_cover-Bushpig        0.8002 1.2157  -1.3361  0.7507  3.2078
Ground_vegetation_cover-CapeBushbuck   0.6115 0.9028  -1.0048  0.5732  2.6492
Ground_vegetation_cover-CapeClawless   0.7751 1.0061  -1.1365  0.7262  2.8267
Ground_vegetation_cover-CapeGenet      0.9175 1.0910  -1.2050  0.8139  3.3398
Ground_vegetation_cover-CapePorcu      0.7296 1.1253  -1.6029  0.7028  3.0384
Ground_vegetation_cover-ChacmaBab      0.8607 0.7600  -0.3586  0.7414  2.7463
Ground_vegetation_cover-Watermongoose  0.4898 1.3192  -2.7460  0.5180  3.3186
SASS-Bushpig                          -1.1618 2.2448  -6.8363 -0.8720  2.4686
SASS-CapeBushbuck                     -0.6427 1.2245  -3.3688 -0.5747  1.7669
SASS-CapeClawless                     -2.2019 4.1051 -16.2822 -1.2046  1.3741
SASS-CapeGenet                        -0.7862 1.7039  -4.4430 -0.7079  2.8584
SASS-CapePorcu                        -1.1502 2.2588  -6.9445 -0.9093  2.4199
SASS-ChacmaBab                         0.1181 1.6820  -2.5598 -0.0122  4.0837
SASS-Watermongoose                    -1.9376 2.4962  -8.1705 -1.5789  1.0891
Conductivity-Bushpig                  -0.1112 2.1008  -3.1292 -0.2849  4.7263
Conductivity-CapeBushbuck             -0.7812 1.5373  -3.3678 -0.8703  2.3782
Conductivity-CapeClawless             -0.2089 1.8493  -3.6625 -0.2352  3.6894
Conductivity-CapeGenet                -0.5722 1.8314  -4.0283 -0.5305  2.9426
Conductivity-CapePorcu                -0.6234 2.1304  -5.9772 -0.4373  3.1665
Conductivity-ChacmaBab                -0.4302 1.5230  -3.1361 -0.4123  2.7292
Conductivity-Watermongoose            -0.2575 1.8180  -3.4406 -0.2290  3.5741
                                        Rhat ESS
(Intercept)-Bushpig                   1.1701  55
(Intercept)-CapeBushbuck              1.0235  45
(Intercept)-CapeClawless              1.0104  49
(Intercept)-CapeGenet                 1.0059  39
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(Intercept)-CapePorcu                 1.2499  71
(Intercept)-ChacmaBab                 1.0333  33
(Intercept)-Watermongoose             1.1231  50
AreaGrootvadersbosch-Bushpig          1.4116  21
AreaGrootvadersbosch-CapeBushbuck     2.6439  12
AreaGrootvadersbosch-CapeClawless     1.7316  18
AreaGrootvadersbosch-CapeGenet        3.1126  14
AreaGrootvadersbosch-CapePorcu        3.6858   8
AreaGrootvadersbosch-ChacmaBab        1.6164  24
AreaGrootvadersbosch-Watermongoose    3.4504  22
Dissolved.oxygen-Bushpig              1.1611  55
Dissolved.oxygen-CapeBushbuck         1.0080  66
Dissolved.oxygen-CapeClawless         1.0610  73
Dissolved.oxygen-CapeGenet            1.0518  86
Dissolved.oxygen-CapePorcu            1.1576  56
Dissolved.oxygen-ChacmaBab            1.0274  71
Dissolved.oxygen-Watermongoose        1.0310  77
Canopy_Cover-Bushpig                  1.1390 112
Canopy_Cover-CapeBushbuck             1.1186 189
Canopy_Cover-CapeClawless             1.0950 174
Canopy_Cover-CapeGenet                1.0838 180
Canopy_Cover-CapePorcu                1.0387  96
Canopy_Cover-ChacmaBab                1.0608 198
Canopy_Cover-Watermongoose            1.0012 163
Ground_vegetation_cover-Bushpig       1.1342 117
Ground_vegetation_cover-CapeBushbuck  1.1439  84
Ground_vegetation_cover-CapeClawless  1.4728 131
Ground_vegetation_cover-CapeGenet     1.1850 121
Ground_vegetation_cover-CapePorcu     1.0364 114
Ground_vegetation_cover-ChacmaBab     1.3490  70
Ground_vegetation_cover-Watermongoose 1.6170  63
SASS-Bushpig                          1.5177  87
SASS-CapeBushbuck                     1.1199  87
SASS-CapeClawless                     1.9680  28
SASS-CapeGenet                        1.0022  72
SASS-CapePorcu                        1.6120  65
SASS-ChacmaBab                        1.0865  38
SASS-Watermongoose                    1.3745  39
Conductivity-Bushpig                  1.5416  46
Conductivity-CapeBushbuck             1.4798  32
Conductivity-CapeClawless             1.1582  55
Conductivity-CapeGenet                1.0958  49
Conductivity-CapePorcu                1.1061  49
Conductivity-ChacmaBab                1.2500  46
Conductivity-Watermongoose            1.5962  29

Detection (logit scale):
                           Mean     SD     2.5%     50%   97.5%   Rhat ESS
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(Intercept)-Bushpig	-5.7895	1.1305	-8.3113	-5.7734	-3.5014	1.5475	45
(Intercept)-CapeBushbuck	-2.4759	0.3230	-3.1220	-2.4598	-1.8869	1.0054	140
(Intercept)-CapeClawless	-4.5898	0.8017	-6.1762	-4.5531	-3.0237	1.1377	136
(Intercept)-CapeGenet	-3.5155	0.5009	-4.5455	-3.5041	-2.5642	1.2611	72
(Intercept)-CapePorcu	-6.4981	1.8426	-12.4267	-6.1740	-3.7996	1.2809	17
(Intercept)-ChacmaBab	-2.4541	0.3224	-3.1480	-2.4358	-1.8752	1.0117	201
(Intercept)-Watermongoose	-4.3024	0.6228	-5.5528	-4.3100	-3.1651	1.0046	141
effort-Bushpig	-0.4125	0.6657	-1.8537	-0.3860	0.7892	1.0011	217
effort-CapeBushbuck	-0.2749	0.3747	-0.9621	-0.2934	0.4267	1.0187	273
effort-CapeClawless	-0.1098	0.6462	-1.2121	-0.1657	1.2861	1.0030	201
effort-CapeGenet	-0.5326	0.5070	-1.6001	-0.4913	0.4174	0.9984	207
effort-CapePorcu	-0.5517	0.6989	-2.0282	-0.5285	0.6907	1.0021	224
effort-ChacmaBab	-0.3540	0.3439	-0.9750	-0.3762	0.3797	1.0182	290
effort-Watermongoose	-0.4921	0.5976	-1.8072	-0.4628	0.5381	1.0338	213

Appendix E: Grootvadersbosch Conservancy Letter of permission

Letter of permission for the use of the Grootvadersbosch Conservancy data (via FBIS).

GROOTVADERSBOSCH
CONSERVANCY

Groot Vaders Bosch Conservancy Trust
IT2623/97
Strawberry Hill Farm
Grootvadersbosch Estate
Po Box 17 Heidelberg 6665
10 April 2023

The Grootvadersbosch Conservancy is pleased to give Barbara Anne-Marie le Roux access to data (SASS5 data and fish presence and absence) that is available on the Freshwater Biodiversity Information System. This data is owned by the Grootvadersbosch Conservancy but available to share on the FBIS platform. The data will be used to complete a Masters of Science in Life Sciences (Zoology) for thesis on "The relationship between the population density of African Clawless otters (Aonyx capensis), prey presence and water quality in rivers of the Breede Water Management Area of the Western Cape"

The following conditions apply:

- The data is downloaded off FBIS
- Credit is given to the Grootvadersbosch Conservancy and FBIS
- Ms Le Roux shares images collected on site for use by the Grootvadersbosch Conservancy which may include use on social media (credit will be given and posts can be viewed/checked before submitting).
- The results/findings are shared with interested landowners (which may include a presentation to interested landowners).
- The conservancy and landowner are informed when accessing the sites. In some cases, the conservancy may be able to inform the landowner but this depends on the final sites chosen.
- The conservancy staff be invited along for fieldwork to better understand and learn from the research process.

We will offer support in whatever way possible within our capacity
All the best
Aileen Anderson