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# Development and Application of a 30-year Vegetation Dataset to Assess the Impacts of Fence Removal within the Kruger to Canyons Biosphere Reserve, South Africa

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Development and Application of a 30-year Vegetation Dataset to Assess the Impacts of Fence  
Removal within the Kruger to Canyons Biosphere Reserve, South Africa

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B.A., Colby College, 2014

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Development and Application of a 30-year Vegetation Dataset to Assess the Impacts of Fence  
Removal within the Kruger to Canyons Biosphere Reserve, South Africa

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## ABSTRACT

Maintaining natural levels of habitat heterogeneity is favorable when conserving protected areas due to its positive relationship with biodiversity. Landscape fragmentation through fencing prevents ecosystems from functioning naturally. Fence removal across protected areas improves connectivity, and allows processes such as animal migration and herbivory to occur over a larger landscape extent. In the mid-1990's, reserves within the Kruger to Canyons Biosphere Reserve in South Africa began removing their fences to become connected to Kruger National Park. The fundamental research objective was to determine how fence removal influenced the vegetation dynamics within the reserves residing in the Kruger to Canyons Biosphere Reserve. In order to do this, we had three goals: 1) determine if multiple satellite sensors could be used within the TIMESAT program to provide robust vegetation metrics that measure historic vegetation changes, 2) determine if vegetation changes occurred within the reserves after fences were removed, and 3) determine which factors contributed to any vegetation changes that occurred after fence removal. We first performed multi-sensor correction to develop a continuous 30-year normalized difference vegetation index (NDVI) dataset at a spatial resolution of 1.1km. The NDVI dataset was then used in the TIMESAT program to extract vegetation metrics across the study area. We used the vegetation metrics to perform two-sample t-tests to compare vegetation within the reserves to Kruger National Park before and after fence removal. We found reserves had similar vegetation to Kruger National Park before fence removal and had different vegetation compared to Kruger National Park after fence removal. We then used linear mixed-effects models to explore potential factors that may have contributed to the vegetation changes. Several factors were associated with changes in vegetation besides fence removal, including geology, seasonality, and rainfall. Additionally, we highlight waterpoint density as it can be manipulated by reserve managers. We found that as waterpoint density increased within the reserves, the vegetation metrics and their variability decreased within open reserves but did not change within closed reserves. We conclude that while fence removal can increase the opportunity for natural animal movements, artificial waterpoints may attract animals and increase local herbivory, and thus

should be considered when managing for vegetation heterogeneity. Reducing the density of artificial waterpoints within reserves that are open to Kruger National Park may help to restore natural heterogeneity within the Kruger to Canyons Biosphere Reserve.

## **CHAPTER 1: Introduction**

### **1.1. Research Context**

The Kruger to Canyons Biosphere Reserve (K2C) includes a vast savanna landscape with complex transitions between tree and grass dominance, influenced by rainfall, soil, and disturbance (Scholes & Archer, 1997). The landscape historically encompassed seasonal mammal dispersal and migration in response to water shortage, droughts, fire, and predation (Mabunda et al., 2003).

Fragmentation through the construction of fences along Kruger National Park (KNP) and smaller reserves within K2C disrupted these natural processes and impacted the vegetation (Peel et al., 1998; Mabunda et al., 2003). Along with the increase in habitat fragmentation, management regimes in the protected areas changed including fire, artificial waterholes, elephant culling, herbivore manipulation, and hunting (Peel et al., 1998; du Toit et al., 2003; Venter et al., 2008).

In the mid-1990's, management objectives shifted in an attempt to restore the natural landscape by removing fences among the reserves adjacent to KNP (Peel et al., 2005). Fence removal has increased animal movement across the reserves and KNP (Kreuter et al., 2010), which in turn influences vegetation dynamics through herbivory (Levick & Rogers, 2008; Scholes & Archer, 1997). Since management goals within these protected areas emphasize maintaining landscape heterogeneity in order to conserve biodiversity (du Toit et al., 2003), it is important to understand how a management action such as fence removal influenced the ecosystem.

Our fundamental research objective was to determine how fence removal influenced vegetation dynamics within the reserves in K2C. In order to do this, we had 3 goals: 1) determine if multiple satellite sensors could be used within the TIMESAT program in order to use robust vegetation metrics to measure historic vegetation changes (addressed in Chapter 2), 2) determine if vegetation changes occurred within the reserves after fences were removed (addressed in Chapter 3), and 3) determine which factors contributed to any vegetation changes that occurred post-fence removal (addressed in Chapter 3). In

answering these questions we hope to provide valuable information to reserve managers for future planning and decision-making.

## **1.2. Literature Review**

### **1.2.1. Disturbance, Heterogeneity, and Biodiversity**

Spatial patterns within a landscape are determined by disturbance, which produces a mosaic of patches differing in size, shape, and arrangement (Turner & Gardner, 2015). These complex patterns are not distributed evenly, with different areas influenced on various levels of severity (Turner & Gardner, 2015). Landscapes large enough to integrate natural levels of disturbance across their patches are considered to support collections of species at an equilibrium, whereas landscapes too small to allow disturbance distributions across patches occur at nonequilibrium (Urban et al., 1987).

Disturbance regimes may be shaped by ecosystem engineers, which are organisms that alter, sustain, and generate habitats by changing physical states in abiotic or biotic resources (Jones et al., 1994). This habitat alteration may directly or indirectly influence resource availability for other species within the ecosystem (Jones et al., 1994). The factors that determine the effect ecosystem engineers have on their environment include the individuals' life time per capita activity, population density and spatial distribution, time a population spends at a particular site, resilience of the ecosystem, the amount and types of resources influenced, and the number of species that are reliant on these resources (Jones et al., 1994). Ecosystem engineers' influence on species abundance and richness has huge variation and may be positive or negative (Jones et al., 1997). However, the natural disturbances across habitats created by ecosystem engineers is thought to ultimately increase species richness within the environment (Jones et al., 1997).

Landscapes altered through disturbance generate heterogeneous formations and vegetation types (Urban et al., 1987). One of the foundations of ecology is the habitat heterogeneity hypothesis, which presumes that complex habitats may offer diverse niches, various resource utilization options, and therefore high species diversity (MacArthur & Levins, 1967; Wilson & MacArthur, 1967). The majority

of studies exploring this idea have found a positive correlation between habitat heterogeneity and species diversity, although the influence of habitat heterogeneity on a single species depends on the spatial scale (Tews et al., 2004).

The amount of spatial heterogeneity within an ecosystem determines the influence of competition and/or predation on species diversity (Menge & Sutherland, 1976). In simple environments with low levels of disturbance, when predation is low competitive exclusion will be intense, reducing species diversity. When predation increases in these simple environments, diversity increases in turn until the intensity of predation reaches a threshold and diversity begins to decrease (Menge & Sutherland, 1976). In contrast, environments with high levels of structural heterogeneity include more microhabitats, therefore allowing a greater number of species to co-exist depending on trophic levels (Menge & Sutherland, 1976). For higher trophic levels within heterogeneous environments, competition retains specialized niches that do not overlap, allowing high levels of diversity to exist (Menge & Sutherland, 1976). For lower trophic levels in complex habitats, predation is regulated, thereby increasing diversity through structural refuges. These refuges are used by prey species to hide from predators and reduce hunting efficiency of the predators (Menge & Sutherland, 1976).

The relationship between species diversity and habitat heterogeneity may be influenced by the presence of keystone structures, which are spatial arrangements that offer survival means for multiple species (Tews et al., 2004). Keystone structures may offer vital habitat characteristics for different groups of species including the ability to escape from predators, foraging effectiveness, and conditions for reproduction (Tews et al., 2004). Studies have found a single vegetation structure to concurrently assist multiple species groups, thereby providing a means to increase species diversity within an ecosystem (Tews et al., 2004). The potential costs across multiple taxonomic groups of losing these keystone structures makes them extremely important to conserve when attempting to maintain biodiversity (Tews et al., 2004).

Heterogeneous landscapes involving numerous types of vegetation provide multiple niches that can be exploited by various organisms, which in turns provides the means to maintain high levels of

biodiversity (Lindenmayer & Fischer, 2006). The number of species that can coexist within an ecosystem is determined by the number of available niches, with coexistence occurring when species' niches do not overlap within a habitat (MacArthur & Levins, 1967; Pulliam, 2000; Dimitrakopoulos & Schmid, 2004). Heterogeneous habitats with numerous niches allow for resource partitioning across species, which increases the total amount of resources used (Tylianakis et al., 2008). A species' niche may determine its distribution across the landscape, depending on niche width, dispersal, and amount of habitat available (Pulliam, 2000).

Spatial distribution of species and niches are influenced by dispersal patterns and the connectivity of populations (Holt, 2009). As landscape mosaics shift and alter species' distributions, niches may evolve in certain environments yet be conserved in others (Holt, 2009). Human-induced landscape homogenization has decreased the number of microhabitats available for various species (Urban et al., 1987), which threatens the amount of biodiversity that may be maintained.

### **1.2.2. Landscape Fragmentation**

#### **1.2.2.1. Habitat Spatial Structure**

The spatial formation of landscape patches has vital consequences on the encompassing ecosystem and its wildlife populations (Wiens, 1996). Landscapes often change with species' habitat divided into smaller pieces, decreasing the size of habitat area as well as isolating the remaining habitat patches (Lindenmayer & Fischer, 2006). Habitat fragmentation decreases patch size, often becoming too small for local populations to persist when the surrounding matrix becomes unsuitable for movement (Fahrig, 2003). The quality of habitat patches is often altered, which impacts a populations' survival and abundance (Fahrig & Merriam, 1994). Fragmentation also increases the amount of habitat edge, which can cause negative effects for inhabiting species (Fahrig, 2003). In order for recolonization to offset local extinction successfully, the amount of habitat patches and their spatial distribution must both be large enough to sustain the disrupted populations (Fahrig, 2003).

Along with the spatial configuration of habitat patches, dispersal routes are important characteristics for species' survival in a fragmented landscape (Fahrig & Merriam, 1994). Dispersal paths include elements within the landscape matrix allowing organism movement (Fahrig & Merriam, 1994). The quality of the habitat that makes up a dispersal route will influence whether an organism uses the path as well as whether the disperser survives (Fahrig & Merriam, 1994). The presence and quality of dispersal routes will therefore determine whether organisms can successfully move to new habitat patches across a landscape matrix.

Landscape spatial structure changes over time, which decreases the likelihood of regional persistence among populations (Fahrig & Merriam, 1994). This occurs when the structure of a landscape changes at a rate that is unnaturally high, which is often the case with human-induced changes (Fahrig & Merriam, 1994). The high rate of change may be too extreme for a population to disperse enough to overcome local extinctions (Fahrig & Merriam, 1994). Anthropogenic changes in landscape structure impose huge challenges for species' survival over fragmented landscapes.

#### **1.2.2.2. Metapopulations**

Landscape fragmentation causes populations to be split into subpopulations through the isolation of habitat sections (Lindenmayer & Fischer, 2006). The theory of island biogeography (Wilson & MacArthur, 1967) has been related to the idea of fragmented populations, with habitat fragments acting as "islands," in order to predict species losses (Wiens, 1996). A species has a greater chance of survival when living in connected habitat compared to divided pieces of land (Burkey, 1989). Considerations for conserving metapopulations within a fragmented landscape include dispersal, genetic variation, and migration.

Natural dispersal across landscapes is impacted by fragmentation as areas unsuitable for organisms block and redirect movement, thereby preventing species from inhabiting suitable habitat (Wiens, 1996; Holt, 2009). If a population is unable to recolonize after it becomes locally extinct within a patch, extinctions over larger areas will occur due to the lack of dispersal ability (Fahrig & Merriam, 1994). The flexibility an organism has in choosing its habitat will determine the distance needed to

disperse before settling in a new area (Wiens, 1996). Changing landscapes cause organisms to travel farther and faster for successful dispersal; their rate of dispersal change must keep up with the rate of landscape change if they are to survive (Fahrig & Merriam, 1994). Since organisms' resource consumption often degrades their habitats (Holt, 2009), animal dispersal allows vegetation to recover (Sinclair & Fryxell, 1985). The local species diversity is maximized at intermediate dispersal rates (Loreau et al., 2003). Dispersal must be high enough to prevent dominant species from competitively excluding all others as well as allowing there to be a balance of extinctions and recolonizations (Wiens, 1996; Loreau et al., 2003). However, too much dispersal may homogenize dynamics within populations and metacommunities (Wiens, 1996; Loreau et al., 2003).

In addition to dispersal ability, habitat fragmentation impacts genetic variation within metapopulations. The main concerns of genetic complexity within fragmented populations include genetic drift, inbreeding, and gene flow (Caughley, 1994; Hedrick, 1996). Lack of genetic drift within isolated populations may lead to small population sizes, bottlenecks, and/or low founding numbers, limiting mate choices and leading to inbreeding (Hedrick, 1996; Boone & Hobbs, 2004). This decreases the genetic variability and increases vulnerability to diseases (Hedrick, 1996; Boone & Hobbs, 2004). Additionally, fragmentation decreases gene flow among populations. In order for positive effects to result from gene flow, migrants must not only be able to move between populations, but there also needs to be high genetic variability within the populations serving as the source for the migrants (Couvett, 2002). Even if migrants are able to move from a mainland population to island populations within a fragmented landscape, low genetic variation within the mainland population may increase the occurrence of deleterious mutations for the subpopulation gathering the new migrants (Couvett, 2002). Successful gene flow must therefore depend on the number of migrants as well as from which populations the migrants originated (Couvett, 2002).

Along with dispersal and genetic variation, migration must also be considered when conserving metapopulations in a fragmented landscape. Changes in habitat suitability often limit the geographic range of species distributions (Pulliam, 2000). Seasonal migration is required by many species to find

patches of suitable habitat that provide food resources or refuges from predators (Whyte & Joubert, 1988). Fragmentation often inhibits natural migration, which may cause a large number of deaths due to the congestion as animals attempt to escape from harmful conditions (Boone & Hobbs, 2004). The negative impact habitat fragmentation has on migration routes is likely to increase as climate change exacerbates stressful environmental conditions (Pulliam, 2000).

### **1.2.2.3. Fenced Protected Areas**

Just as island biogeography theory has been applied to general landscape fragmentation (Wilson & MacArthur, 1967), it has also been used to describe the impacts of fencing (Hayward & Kerley, 2009; Lindsey et al., 2012). Fences are often used to decrease human-wildlife conflict, protect threatened species from human exploitation, and allow small habitat patches to be used by wildlife (Hayward & Kerley, 2009; Lindsey et al., 2012). However, fencing may also limit a species' ability to evolve naturally (Hayward & Kerley, 2009). Problems related with general fragmentation, such as population isolation, dispersal to find available resources, genetic variability, and migration, are also serious problems within fenced ecosystems (Lindsey et al., 2012).

Resource availability decreases as fences prevent animals from naturally dispersing (Hayward & Kerley, 2009; Ferguson et al. 2012; Venter et al., 2015). Fences prevent movement toward high quality biomass patches (Boone & Hobbs, 2004), inhibiting natural population regulation and causing an overuse of resources (Hayward & Kerley, 2009). Habitats fragmented by fencing often overstock wildlife and limit movement, thereby disadvantaging species that would not normally compete for resources in a natural setting (Bond et al., 2004; Lindsey et al., 2008; 2012).

Fencing also impacts genetic variation by preventing gene flow between populations and increasing the likelihood of inbreeding (Lindsey et al., 2008; Hayward & Kerley, 2009). Isolated populations through fencing compromises metapopulation dynamics, and will have a disproportionately negative influence on species that require large areas to sustain their populations (Hayward & Kerley, 2009).

Furthermore, fencing has harsh impacts on wildlife migration (Newmark, 2008). Natural wildebeest migration allows populations to graze on grass plains with high nutrient levels, which is also where birth and growth occurs (Sinclair & Fryxell, 1985). However, fencing has restricted this natural process (Whyte & Joubert, 1988). Douglas-Hamilton et al. (2005) found that unfenced elephants had clear “home” regions connected by “travel corridors” and that their ranges were less homogenous than previously thought. The complexity of elephant movement therefore suggests that it would be difficult to generate similar routes in fenced areas. Furthermore, altitudinal migration is prevented when animals are fenced (Hayward & Kerley, 2009), which alters populations’ natural movement and possible adjustment to climate conditions.

The extent of human modification on natural areas has limited much of the remaining habitat to conservation areas, which often needs to be fenced to effectively protect biodiversity (Hayward & Kerley, 2009). Isolated populations have less prospect of persisting naturally when compared to non-fragmented populations, which contradicts the original objectives of fencing wildlife (Hayward & Kerley, 2009). Thus, the pros and cons of fencing need to be considered on a situational basis to decide when it is in the best interest of conservation management to fence wildlife populations (Hayward & Kerley, 2009).

### **1.2.3. Protected Area Management in South Africa**

Successfully maintaining protected areas is vital in conserving ecosystems (Cumming et al., 2015), with wildlife conservation traditionally depending on nature reserves (Wiens, 1996). Objectives of these protected areas have evolved as conservation biology converted from merely focusing on endangered species to developing a greater mindfulness of entire ecosystems and their potential human services (Cumming et al., 2015). The amount and area encompassed by parks has grown tremendously over the past century in southern Africa, where they are incorporated within matrixes of changing social and economic environments (Cumming, 2004). However, as the overall area and number of parks increased, the average park size has decreased, suggesting that protected areas have become fragmented “ecological islands” (Cumming, 2004). Many of these reserves do not provide enough area to conserve megafauna (Langholz & Lassoie, 2001). Most park boundaries were created without considering natural

environmental processes such as seasonal movements and migration patterns (Cumming, 2004). As large herbivore spatial requirements become restricted, the structure and diversity of plant communities become impacted (Newmark, 2008).

Different types of protected areas within South Africa manage wildlife in various ways depending on their goals. The types of private wildlife systems include game farms, game ranches, and conservancies (Bond et al., 2004). Game farms are small, reaching up to about 5,000 ha, and are encompassed by fences. Wildlife is only one part of game farm management, which also includes livestock and agriculture (Bond et al., 2004). Game ranches are larger, with the types of income including trophy hunting and tourism (Bond et al., 2004). Conservancies involve multiple landowners that cooperate through the removal of internal fences and abide similar management styles (Bond et al., 2004). Conservancies are able to reduce internal fragmentation, therefore enlarging available wildlife habitat (Bond et al., 2004). These practices demonstrate the potential to improve conservation, particularly as almost all South African parks are fenced (Lindsey et al., 2012). In order for natural ecological processes to take place within these fenced protected areas, intensive management needs to occur (Lindsey et al., 2008).

#### **1.2.3.1. Mimicking Nature**

Integrating spatial heterogeneity and ecological variability into savanna conservation planning is a major challenge among wildlife managers (Rogers, 2003). Management needs to mimic ordinary landscape disturbances (Urban et al., 1987) to keep ecosystems processes as natural as possible. This is a difficult task given that ecosystem factors and biodiversity transform and recover at various rates to disturbances (Cumming, 2004). In order to maintain heterogeneity within protected savannas effectively, managers need to incorporate the complexity of spatiotemporal uncertainties due to the systems' constant fluctuations (Rogers, 2003). A strong understanding of which landscapes rely on certain ecosystem engineers and keystone structures is vital in spatially managing biodiversity (Jones et al., 1997; Tews et al., 2004).

In addition to mimicking habitat heterogeneity, managers are often required to manipulate animal populations within protected areas to maintain genetic variability. Managers may need to artificially disperse small, isolated populations for the species to persist (McCullough, 1996; Cumming, 2004). Metapopulation management is a way to counterbalance the genetic threats of small populations by moving individuals among various fenced populations (Hayward & Kerley, 2009). Translocating organisms amongst fragmented populations may allow population dynamics to become comparable to natural metapopulations (Hedrick, 1996; McCullough, 1996). Therefore, if small reserves are isolated within an area, a metapopulation approach comprising reserve networks may pose the best immediate option to mimic processes that would normally occur within larger, open systems (Miller et al., 2013).

#### **1.2.3.2. Balancing Conservation and Ecotourism**

A major reason for the increased number of wildlife parks in South Africa is the growing tourism industry (Cumming, 2004). Over time, park objectives have progressed from merely protecting game, to conserving biodiversity, to emphasizing social and economic benefits (Cumming, 2004). It is important to develop sustainable social-ecological systems that involve the interactions of social, economic, and ecological factors (Cumming et al., 2015). This concept relies on the multiple managers and stakeholders working together as a group to find prospective ways to improve the resilience of protected ecosystems (Cumming et al., 2015).

One challenge in balancing conservation and ecotourism is maintaining sustainable numbers of predators within a system. Lions in particular have gained attention from managers to determine sustainable densities within small reserves due to their large influence on ecotourism (Clements et al., 2016). The positive relationship between high lion densities and ecotourism revenue gives protected areas incentive to overstock predators, although this has consequences for the rest of the ecosystem (Clements et al. 2016). The major challenges in managing lion populations within small reserves include “excess lions,” decreases in genetic variation, higher vulnerability to catastrophic incidents, and cascading effects on other predators or keystone species (Miller et al., 2013). In order to combat the challenges posed by maintaining lion populations within small reserves, managers may choose to relocate lions to

other reserves, use contraception to decrease breeding rates, allow trophy hunting to increase financial revenue, or artificially exchange male coalitions to intermix gene pools (Kettles & Slotow, 2009). Managers must find a way to control reproduction and diversify gene pools while having as little disruption on population behavior and social structure as possible (Miller et al., 2013). Although effective ecological policies concerning viable lion densities within reserves have been established, discrepancies occur between policy guidelines and other predator densities (Clements et al., 2016). It is therefore imperative that sustainable densities are correctly determined for all predators within reserves in order to maximize ecotourism revenue while maintaining natural population dynamics.

Along with maintaining sustainable yet profitable predator numbers, reserves face the challenge of determining appropriate waterhole densities. It is appealing for managers to add artificial waterholes within their reserves because they attract larger numbers of animals that can be viewed by tourists (Owen-Smith, 1996). However, unnaturally high waterhole densities cause many problems within the ecosystem. Faunal and flora structures within savanna ecosystems are shaped by surface water availability and distribution on population, community, and landscape scales (Gaylard et al., 2003). Surface water affects animal populations and communities directly by influencing species composition diversity as well as indirectly by altering processes such as competition and predation (Gaylard et al., 2003). Common ungulate species may benefit from additional water sources while rarer species may be disadvantaged (Owen-Smith, 1996). Predators may also become disproportionally advantaged by higher waterhole densities as their prey becomes more stationary (Tambling & du Toit, 2005). Herbivory pressure surrounding waterpoints is known as the “sacrifice area,” or piosphere, causing heavy trampling that degrades the vegetation and alters the nutrient distributions (Thrash & Derry, 1999; Brits et al., 2002; Gaylard et al., 2003), with waterholes spaced too closely causing starvation during drought periods (Walker et al., 1987). The piosphere effect influences vegetation community composition and diversity. On the landscape-scale, surface water influences animal distributions through water and forage availability (Gaylard et al., 2003). Areas with closely-spaced surface water sources endure heavier animal impacts compared to regions where surface water is more remote (Gaylard et al., 2003). Surface water

spacing across the landscape therefore influences the pattern and intensity of animal impact on vegetation, manipulating landscape heterogeneity. In order to maintain habitat heterogeneity and diversity, waterholes need to be spaced far enough apart to allow vegetation to recover from seasonal grazing (Smit et al., 2007). Habitat heterogeneity may increase if reserves close artificial waterholes, thereby restoring the spatial and temporal heterogeneity qualities of surface water availability (Gaylard et al., 2003; Redfern et al., 2003). Although adding artificial waterholes may initially improve tourism sightings, unnaturally high waterhole densities may have disastrous impacts on the ecosystem. Surface water policies within protected areas should therefore aim to permit natural variability in surface water accessibility (Smit & Grant, 2009).

As balancing ecotourism, socio-economic development and conservation goals becomes increasingly important in South African protected areas, integrating the needs of surrounding communities has become a central part of protected area management. One way this is accomplished is through the establishment of biosphere reserves, which are contiguous systems that combine objectives for conservation management as well as land-use issues of the surrounding communities (Batisse, 1982). Biosphere reserves aim to make conservation more relevant to the social, economic, and cultural needs of surrounding human populations (Batisse, 1982). The principal purpose of a biosphere reserve is long-term conservation by protecting ecosystems within a network that includes the agreement and contribution of the surrounding communities (Batisse, 1982). Indigenous cultures should therefore play a role in managing these landscapes that contain “representative ecosystems” (Batisse, 1982). Through the integration of local communities, biosphere reserves have huge potential to balance sustainable conservation management and ecotourism. Biosphere reserves should strive for economic, social and ecological sustainability.

#### **1.2.3.3. Connectivity**

The isolation of African protected areas and human activities adjacent to reserves have negative effects on the wildlife populations within reserves (Newmark, 2008). Whenever possible, the best way to conserve biodiversity is maintaining large protected areas without fences by connecting isolated protected

areas (Hayward & Kerley, 2009). Wildlife corridors increase landscape connectivity by joining habitat patches (Wiens, 1996), which may determine the survival of metapopulations (Douglas-Hamilton et al., 2005). To prevent population isolation, reserve designs should consider grouping multiple reserves or including corridors to allow animal movement between protected areas (Wiens, 1996). Protected areas often do not account for space required for migration across climatic gradients (Jewitt et al., 2014). Decreasing the isolation of protected areas will be important in allowing species to geographically relocate in response to climate change (Newmark, 2008).

Connectivity can be improved by expanding and joining protected areas through fence removal between adjacent reserves and national parks, which has become an increasingly common practice in southern Africa (Newmark, 2008). This may be done across protected areas spanning multiple countries to create transfrontier conservation areas or between private land to develop conservancies (Lindsey et al., 2012). Fence removal improves conservation for species with large ranges, allows for the maintenance of larger charismatic wildlife species, creates habitat with greater resilience to hazardous events and disturbance, diversifies habitat types, and reduces problems related to genetic variation (Lindsey et al., 2008; 2009; 2012; Durant et al., 2015). These positive outcomes derived from fence removal requires less management intervention and improves ecotourism benefits (Lindsey et al., 2008). However, land owners may be hesitant to develop conservancies with neighboring reserves due to the possibility that wildlife within their land may move into adjacent areas once fences are removed (Lindsey et al., 2012). Tax incentives may be a strategy to combat these hesitations by offering extra financial motivation for managers across neighboring protected areas to merge (Lindsey et al., 2008). Increased fence removal and connectivity across adjacent protected areas would allow for improved ecological processes and is vital in sustaining natural wildlife populations (Hayward & Kerley, 2009; Lindsey et al., 2009; Jewitt et al., 2014; Durant et al., 2015).

#### **1.2.4. Study Area**

##### **1.2.4.1. Tree-Grass Interactions**

The tree-grass interactions within savanna ecosystems are complex, with multiple plant types co-dominating in space and time (Scholes & Archer, 1997). This coexistence occurs through competition and facilitation, niche separation, and leveling disturbances (Scholes & Archer, 1997). These processes occur across the landscape as well as through horizontal interactions within rooting zones (Scholes & Archer, 1997). Trees may have negative impacts on grasses through rainfall interception, the buildup of ground litter, shading, and/or root competition (Scholes & Archer, 1997). Trees may also facilitate grasses by providing nutrients post-decomposition (Scholes & Archer, 1997). On the other hand, grasses may influence tree growth directly by regulating woody recruitment through competition for light, water, and nutrients, or indirectly by controlling fire frequency and intensity with fuel (Scholes & Archer, 1997). A positive feedback loop ultimately develops between tree and grass growth. Increases in woody plant abundance subdue grasses, but also suppress fire. In turn, the influence of grass biomass on fire affects tree density (Scholes & Archer, 1997). Long-term persistence of savannas occur across landscapes since patches are in various states of transition between tree and grass dominance, continually influenced by rainfall, soil, and disturbance (Scholes & Archer, 1997).

#### **1.2.4.2. Abiotic Factors**

Rainfall has a huge influence on savanna vegetation, shaping the ecosystem in different ways. The influence of trees on grasses can be either positive or negative depending on the rainfall amount (Scholes & Archer, 1997). Sankaran et al. (2005) found that 650mm of mean annual rainfall was the upper bound, where woody cover is constrained by rainfall below this level, therefore allowing grasses to coexist. When mean annual rainfall is above 650mm in savannas, fire and herbivory disturbances are required for tree-grass coexistence. The narrow range between 516mm – 784mm mean annual rainfall is considered to be a transition zone where trees and grasses can coexist without fire or herbivory disturbance, but these disturbances modify the ratio between the two vegetation types (Sankaran et al., 2005). The mean annual rainfall within the Kruger to Canyons Biosphere Reserve (K2C) is approximately 400mm – 650mm (Peel et al., 2007). Therefore, the study area falls within the transition zone, amplifying the complexity of how rainfall and disturbance influences vegetation dynamics.

Along with rainfall, soil nutrients in savanna ecosystems have a huge influence on plant community composition and structure. Three main ways soil affects plants are through absorbing and storing water, providing nutrients, and preventing infiltration of plant roots (Scholes & Archer, 1997; Venter et al., 2003). Nutrient-poor soils include granite or sandstone and are dominated by broad-leaved trees and sparse, unpalatable grasses. Nutrient-rich soils include those with high clay content such as basalt and gabbro, and are dominated by fine compound-leaved trees, which are preferred by herbivores (Venter et al., 2003). Soil resources also influence the productivity below tree canopies, creating competition between trees and grasses. Nutrients released from the soil after a disturbance occurs improves grass growth (Scholes & Archer, 1997). Along with soils influencing plant communities, vegetation also affects nutrients levels. Trees have higher levels of nutrients in their leaves compared to herbaceous vegetation, causing soil under sub-canopies to have higher densities of carbon and nitrogen in relation to soils far from trees (Scholes & Archer, 1997). Trees also influence grass nutrients, with grasses containing higher nitrogen and phosphorus levels when located beneath tree canopies (Treydte et al., 2007). Soil properties have a huge influence on creating and retaining heterogeneity through their effect on vegetation composition and structure (Venter et al., 2003).

#### **1.2.4.3. Disturbances**

Fire is a major disturbance that plays a large role in shaping savanna ecosystems. High intensity fires kill aboveground parts of trees, only allowing them to resprout from the base; low intensity fires do not kill aboveground portions, and therefore do not influence their height. Fire is a natural disturbance that is important for the endurance of savanna ecosystems; if fire is fully excluded, there is an unnaturally high growth of woody biomass (Venter et al., 2003). Higgins et al. (2007) found that fire conditions did not influence tree density, but rather the size structure and biomass of tree populations. If humans attempt to suppress fire, fuel buildup occurs, causing higher intensity fires to spread over a greater expanse than would naturally occur. Furthermore, some plant species need periodic fires to distribute their seeds for germination (Urban et al., 1987). Fire disturbance influences how woody vegetation affects grasses. An increase in woody plants subdues grass growth; this decreases the likelihood of fire occurrence, creating a

feedback loop (Scholes & Archer, 1997). Smith et al. (2012) found that fires were more intense in areas with a higher mean annual rainfall (>570mm per year) compared to areas with lower annual rainfall (<510mm per year). Along with rainfall, Smit et al. (2013) found that the geology and distance from the closest river influenced fire heterogeneity. Fire management in Kruger National Park (KNP) has changed over the years (see van Wilgen et al., 2008 for an overview). In 2002, patch burning began within an adaptive management paradigm in the KNP in order to stimulate heterogeneity and increase biodiversity – i.e. assuming “pyrodiversity begets biodiversity” (van Wilgen et al., 2003; 2008; 2011). The current fire management policy has been adopted since 2012, embracing a variable- rather than single- management approach in order to account for the abiotic variables that influence fire regime (Smit et al., 2013; van Wilgen et al., 2014).

In addition to fire, herbivory creates large levels of disturbance within savanna landscapes. Herbivory is known to directly modify the structure and composition of woody vegetation in savannas (Scholes & Archer, 1997; Levick & Rogers, 2008) as well as indirectly by influencing soil nutrient accessibility (van der Waal et al., 2011). Overall, herbivores increase ecosystem heterogeneity, increase herbaceous species richness, and decrease biomass levels (van Coller & Siebert, 2015). The two different vegetation sources for herbivores are grass and browse, which are generally consumed independently (Peel et al., 1998). Woody plant height has shown to be larger when browsing was nonexistent (Levick & Rogers, 2008). Overgrazing may result in bush encroachment, which causes an increase in unpalatable vegetation (Sinclair & Fryxell, 1985). If overgrazing occurs on granite soils, there may be a decrease in grass competition, which in turn increases the woody plant density. However, grass is usually able to recover on nutrient-rich soils, maintaining competition with woody plants (Eckhardt et al., 2000; van der Waal et al., 2011). Herbivory and fire disturbance interact, with browsing maintaining woody plants within flame zones, and fires causing woody plants to remain browsable (Scholes & Archer, 1997).

In large parks, controlling numbers of small-bodied herbivores is rare, since a natural fluctuation is needed to sustain heterogeneity (Grant et al., 2011) and intact predator-prey dynamics will control animal numbers. However, large-bodied herbivore populations are occasionally managed. For example,

KNP managed mega-herbivore numbers for many years through large-scale culling operations. However, since 1994 KNP moved away from a “nature-in-balance” approach towards a “nature-evolving” approach. As such, there has been infrequent population management of large herbivore populations in KNP since the 1990’s when large-scale culling of elephants, buffalo and hippo was ceased (Whyte et al., 2003). In contrast, smaller parks and protected areas often need to manipulate herbivore numbers based on the available forage to keep the system sustainable (Grant et al., 2011).

Along with fire and herbivory, elephants are a large disturbance factor within savannas, playing the role of a keystone species that regulate the ecosystem (Douglas-Hamilton et al., 2005; Asner et al., 2016). Elephant presence influences an ecosystem in both positive and negative ways. Cummings et al. (1997) found that in areas containing high densities of elephants ( $>0.5$  elephants/km<sup>2</sup>) for more than a decade, woodland structure was altered and canopy tree diversity decreased. Shrub layers remained, with the prospective to restore to woodlands (Cummings et al., 1997). Although elephants cause large amounts of vegetation destruction, this has shown to improve the complexity of tree structures, in turn creating microhabitat refuges for smaller species. For example, Pringle (2008) found the population density of a common arboreal lizard increased with the amount of trees disturbed by elephants. These findings suggest that the interactions of mega-herbivores with the surrounding system are essential, and loss of these large animal densities may cause drastic negative cascading effects on the community (Pringle, 2008). Maintaining variable localized elephant densities are vital in maintaining natural levels of disturbance and biodiversity within savanna ecosystems. In order to manage elephants effectively within KNP, elements that influence spatio- and temporal- distribution need to be considered rather than only taking numbers into account (van Aarde et al., 2006). The intricate relationship between elephants and their surrounding environmental factors needs to be carefully contemplated in conservation planning (Douglas-Hamilton et al., 2005).

#### **1.2.4.4. Kruger to Canyons Biosphere Reserve**

Biosphere reserves are globally significant ecosystems neighboring human populations and are recognized in order to encourage sustainable solutions for conserving biodiversity. The United Nations

Educated, Scientific, and Cultural Organization registered the Kruger to Canyons Biosphere Reserve in 2001 (UNESCO, 2015). With an area of 2,474,700ha, the K2C bridges the Limpopo and Mpumalanga provinces. It is South Africa's second-largest biosphere reserve and has been recently named the eleventh most successful biosphere reserve in the world (van Cuong et al., 2017). The objectives of the K2C are to protect the greater KNP bioregion, the eastern savanna, and the eastern escarpment (Pool-Stanvliet, 2013). It aims to conserve biodiversity while promoting sustainable development within the communities, with environmental management playing the major role in developing the economy (UNESCO, 2015).

Human influence altered the naturally-heterogeneous landscapes by condensing ecological communities into small fenced reserves within the K2C (Peel et al., 1998). This compaction reduced the carrying capacity and homogenized vegetation structure and composition (Peel et al., 1998). KNP animal populations historically migrated west during periods of dryness in the direction of the escarpment and higher rainfall (Venter et al., 2003), which was then prevented with the creation of fences (Whyte & Joubert, 1988). Herbivore density within the private reserves decreased due to the lack of migration, overgrazed vegetation and addition of artificial waterholes, which altered the natural herbivore composition (Peel, 2010). Grazing, browsing, and fire interactions were altered, ultimately increasing woody vegetation density and decreasing grass availability (Peel, 2010). This increase in woody vegetation negatively impacted ecotourism because bush encroachment and dense woody vegetation impaired animal visibility (Peel, 2010; Gray & Bond, 2013). The large-scale fence removal across the conservancies adjacent to KNP is the start to positively moderating the negative fencing impacts within the K2C ecosystems (Cumming, 2004; Peel, 2010).

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## **CHAPTER 2: Extending TIMESAT: Use of multi-sensor correction to develop a 30-year vegetation dataset for the Kruger to Canyons Biosphere Reserve**

### **2.1. Abstract**

Remote sensing is a powerful conservation tool as it enables researchers to explore landscape changes over large spatial and temporal scales. Multi-sensor correction is a common practice to extend the time-span of satellite images, and the TIMESAT program increases the amount of phenological vegetation information one may obtain from satellite images. The goal of this study was to combine the two practices in order to extend the time-span of a TIMESAT dataset through the use of multiple sensors over a South African landscape. We first used seven overlapping years between AVHRR and MODIS normalized difference vegetation index (NDVI) datasets to apply multi-sensor correction equations, thereby adjusting AVHRR to MODIS and creating a continuous 30-year dataset. We then applied the TIMESAT program on the extended NDVI data and extracted four vegetation metrics: maximum NDVI (Max), seasonal amplitude (Amp), large integral (LI), and small integral (SI). After standardizing the necessary metrics, we compared the agreement of the vegetation indices across the overlapping years for the two sensors. Although there was a slightly greater difference between the adjusted AVHRR and MODIS NDVI values during the rainy season months compared to the dry season months, there was high overall agreement between the TIMESAT values across the sensors. Robinson's coefficient of agreement for the Max, Amp, LI, and SI metrics were 0.92, 0.88, 0.87, and 0.84, respectively. The high agreement for the TIMESAT metrics between adjusted AVHRR and MODIS validates the approach to use multiple sensors within the TIMESAT program interchangeably. Since the various phenological metrics derived from TIMESAT have been shown to relate to unique ecological phenomena, the ability to extend TIMESAT datasets has powerful management implications in dynamic systems. This methodology provides effective means for temporally extending vegetation analyses, which is important when performing large-scale conservation planning.

### **2.2. Introduction**

Measuring vegetation change through spectral indices is a powerful conservation tool, providing insight for the most effective environmental management practices. The normalized difference vegetation index (NDVI) is one of the most common ways to measure vegetation dynamics using the red and near-infrared (NIR) bands from satellite images (Tucker, 1979). NDVI provides information on vegetation phenology and primary production, which can be valuable in assessing biodiversity (Pettorelli et al., 2011). There are temporal and spatial constraints with ground-based methodologies, rendering NDVI a crucial tool in addressing environmental phenomena across large areas (Pettorelli et al., 2011). The ability to perform landscape-level analyses has led to studies evaluating time-series trends of NDVI within protected areas (Paruelo et al., 2005; Alcaraz-segura et al., 2008; 2009). As environmental challenges continue to increase, analyzing vegetation dynamics with remote sensing across spatial and temporal scales will remain crucial in conservation planning.

The sensors launched over the past few decades possess different temporal and spatial resolutions. The Moderate-resolution Imaging Spectrometer (MODIS) has the advantage of providing a relatively high spatial resolution of 250m. However, this sensor did not start recording until February 2000, limiting its timespan. Although the Advanced Very High Resolution Radiometer (AVHRR) sensor has a coarser spatial resolution of 1.1km, this product provides a longer historic dataset dating back to 1981. Although an AVHRR dataset extends from the 1980's to 2010, it has a much coarser resolution of 8km that is insufficient for many NDVI analyses. Integrating the 1.1km AVHRR and 250m MODIS sensors is therefore often used for long-term NDVI studies to temporally extend the finer spatial resolution datasets (Gallo et al., 2005; Mao et al., 2012; Bao et al., 2014; Guay et al., 2014). Due to slight NDVI value differences from inconsistencies in the sensor band widths (Huete et al., 2002), multi-sensor correction using linear-regression is often performed to improve the agreement between AVHRR and MODIS (van Leeuwen et al., 2006; Mao et al., 2012; Bao et al., 2014).

One of the ways NDVI is used in time-series analyses is through the TIMESAT program. TIMESAT produces seasonality metrics for each growing season within a time-series dataset by using filtering techniques and an upper envelope, which fits a smooth continuous curve to the NDVI data.

TIMESAT decreases the impact of satellite noise by removing extreme spikes in NDVI values as well as by taking into account pixel quality based on cloud coverage (Jönsson & Eklundh, 2002; 2004). In addition to making the NDVI data more robust by smoothing the dataset, TIMESAT identifies several phenological parameters that provide unique information on the vegetation dynamics within an ecosystem along with defining the beginning, middle, and end of each growing season (Wessels et al., 2011; Mbow et al., 2013; Olsen et al., 2015). Many studies have used TIMESAT to analyze vegetation over time through the use of one sensor (Olsson et al., 2005; Heumann et al., 2007; Davis, 2013; van Leeuwen et al., 2013). However, TIMESAT outputs from multiple sensors have not been integrated in a continuous manner, which would increase the temporal application of the program and provide greater perspective on vegetation change in landscapes with dynamic ecological, social, and economic systems.

The aim of this study was to extend the timespan of continuous TIMESAT data through the use of multiple sensors. We performed three steps to complete this process: 1) We applied multi-sensor correction to the NDVI datasets from AVHRR and MODIS, 2) we ran both the adjusted AVHRR and MODIS NDVI datasets in TIMESAT and standardized the necessary vegetation parameters to develop a continuous 30-year span of TIMESAT metrics, and 3) we used seven overlapping years of the AVHRR and MODIS sensors to evaluate the agreement between the TIMESAT metrics.

## **2.3. Methods**

### **2.3.1. Study Area**

The study area consisted of 1.6 million ha of private reserves and the central sections of Kruger National Park (KNP), all located within the Kruger to Canyons Biosphere Reserve (K2C) in the Limpopo and Mpumalanga provinces of South Africa between 30°35' E and 30°40' E and 24°00' S and 25°00' S (Peel et al., 2007) (Figure 2.1). The private reserves and KNP are part of the savanna biome section of the eastern Lowveld (Peel et al., 2007). The mean annual rainfall in the study area ranges from 400mm to 650mm (Peel et al., 2007), which occurs primarily between the months of October and April (Venter et

al., 2003). The geology is predominantly granitic rock, with sections of amphibolitic and gabbro rock woven into some of the reserves (Keyser, 1997; Venter, pers. comm.).

### **2.3.2. Data**

#### **2.3.2.1. AVHRR and MODIS NDVI**

We obtained Terra MODIS NDVI (MOD13Q1) and its corresponding “pixel reliability” rasters for the time period 2000 to 2015 from NASA EOSDIS at 16-day intervals and 250m spatial resolution. The pixel reliability data contained three values corresponding to high cloud coverage, medium cloud coverage, and no clouds.

AVHRR NDVI rasters from the South African Space Agency (Wessels et al., 2012) were processed and calibrated at a 1.1km resolution by the Institute for Soil, Climate, and Water at the Agricultural Research Center (ARC-ISCW) in order to amend for sensor changes and degradation (Rao & Chen, 1995; 1996; Wessels et al., 2006). Due to the lack of available atmospheric water vapor and aerosol optical depth data, the AVHRR rasters could not be corrected for atmospheric effects (Wessels et al., 2006; 2012). Daily data were transformed to 10-day maximum value composites (Wessels et al., 2006), which represented the maximum NDVI value per pixel for the 10-day period. This technique helped to reduce the impact of clouds, atmospheric effects, and fluctuating solar zenith angles (Holben, 1986). A statistical filter was also applied, interpolating pixels impacted by atmospheric aerosols to further reduce cloud and atmospheric effects (Lo Seen Chong et al., 1993; Wessels et al., 2006). Data from 1994 were unavailable due to failure of the NOAA-13 satellite (Wessels et al., 2006). Wessels et al. (2004) provides additional details on the processing of the AVHRR NDVI dataset.

There are several unresolvable differences in the AVHRR and MODIS satellites that are important to consider when integrating the datasets into a continuous time-series. These include differences in red and near infrared (NIR) bandwidths and AVHRR’s lack of atmospheric correction for aerosol impacts (Ji et al., 2008). The narrower MODIS red and NIR bands (620-670; 841-876nm) in comparison to the AVHRR red and NIR bands (570-700; 710-980nm) decreases the impact of water

vapor absorption on the MODIS-NIR band (Huete et al., 2002), typically causing lower AVHRR NDVI values compared to MODIS NDVI values (Cihlar et al., 2001). It is therefore important to keep the bandwidth differences in mind when evaluating the effectiveness of the multi-sensor correction process.

### **2.3.2.2. Adjusting AVHRR NDVI to MODIS NDVI**

An ordinary least-squared linear regression function can successfully adjust AVHRR NDVI values to MODIS NDVI values, making the two sensors comparable (van Leeuwen et al., 2006; Mao et al., 2012). Since the AVHRR and MODIS sensors overlapped during the time period from 2000 to 2006 (Figure 2.2), those years are often used to create an equation that adjusts the NDVI values from one sensor to the other (Mao et al., 2012; Bao et al., 2014). Before extracting the NDVI values from the overlapping years, we resampled the 1.1km AVHRR NDVI to 250m to match the MODIS sensor cell sizes.

The strong relationship between rainfall and seasonal variability in arid ecosystems poses a challenge when using NDVI to evaluate vegetation (Li et al., 2004; Ji & Peters, 2005; Wessels et al., 2007; Vezzoli et al., 2008; Wessels et al., 2012). Since the study area has high rainfall variability (Venter et al., 2003; Vezzoli et al., 2008), we partitioned AVHRR and MODIS NDVI values by month so there would be a month-specific equation for the sensor adjustment.

A common way to prepare AVHRR and MODIS data for sensor adjustment is to generate maximum value composites (MVC) (Piao et al., 2011; Mao et al., 2012; Bao et al., 2014). Computing the maximum NDVI value per pixel for a given time-period decreases the likelihood of cloud and aerosol contamination, since inaccurate pixels usually have lower values (Holben, 1986). We calculated monthly MVC for each year in the overlap period for both the MODIS and AVHRR to use in the least-squared regression.

In addition to the MVC technique, statistical filters are a common way to reduce noise caused by mixed-pixels (Stow & Chen, 2002; Ji et al., 2008). When comparing regressions for rasters with different filter sizes, Ji et al. (2008) found that the 5x5 filter window produced the steadiest intercepts and slopes.

Therefore, once we computed the MVC for each month for the years 2000 to 2006 for the AVHRR and MODIS datasets, we applied a 5x5 median statistical filter to each MVC raster.

Once the MVC and statistical filter techniques performed on the NDVI rasters were complete, we extracted 10,000 random points across the K2C for each month between the years 2000 to 2006 for the AVHRR and MODIS rasters. The MODIS sensor data began in February 2000, so January 2000 was not used in the comparison. Although we had no information on cloud-quality of the AVHRR pixels, we did utilize the MODIS pixel reliability rasters by only using the random points that fell on cloudless MODIS pixels to compare with the AVHRR pixels, leading to a total of 783,047 pixels. An ordinary least-square regression analysis of the AVHRR-MODIS NDVI comparison produced slope and intercept values for each month of the year (Table 2.1). We then used the respective monthly intercept and slope values to adjust the original AVHRR NDVI values to better align with the MODIS NDVI values. For example, the adjustment applied to the original AVHRR NDVI rasters from the month of January was:

$$(AVHRR_{adjusted}) = 1009.65 + (1.04) \times (AVHRR_{original}) \quad (1)$$

After completing the AVHRR sensor adjustment, we extracted a new set of 10,000 random points from the private reserves and KNP central sections to determine how well the adjusted AVHRR corresponded to the MODIS NDVI values. We used the private reserves and central KNP sections for the NDVI agreement analysis because this was the extent used for extracting the TIMESAT metrics.

### **2.3.3. TIMESAT**

TIMESAT has several parameters that need to be established before running the program. We selected sixteen random pixels from two of the private reserves and one section of KNP to test each of the TIMESAT setting parameters and determine which were most appropriate for our study area. We used the Adaptive Savitsky-Golay filter, which uses local polynomial functions in the fitting processing and has been demonstrated to be the most accurate filter method for South African landscapes (Bachoo & Archibald, 2007; Steenkamp et al., 2009; Wessels et al., 2011). We set the season cut-off value to one, signifying that the study area contained one growing season. We used the second spike method (STL

replace) with one envelope iteration and a window size of 3. The AVHRR and MODIS NDVI data were scaled to span the values from 0 to 10,000, so we set this as the data range in TIMESAT to represent positive NDVI values associated with vegetation. Since MODIS cloud coverage data was available, we used the 16-day interval pixel reliability rasters as the quality data within TIMESAT so the program would weight pixels based on their accuracy. We assigned weights of 1.0, 0.5, and 0 to pixels ranked as clear, marginal, and cloudy, respectively.

A user-defined threshold states the percentage of the seasonal amplitude TIMESAT should use to determine the beginning and end of each growing season. Since our study location was within the savanna biome, we used the TIMESAT outputs for the South African savanna biome from Wessels et al. (2011) as a reference for the growing season metrics. After performing trials with the pixel samples using various combinations of beginning/end threshold percentages, we defined the beginning of the growing season as 10% of the seasonal amplitude as measured from the left minima, and the end of the growing season as 45% of the seasonal amplitude as measured from the right minima. Wessels et al. (2011) acknowledged that TIMESAT has a poor ability to accurately detect the end of the growing season. This occurs because although the chlorophyll level will drop off at the end of the growing season, it takes time for the NDVI reflectance to decrease back to a non-growing season level. A higher threshold percentage is therefore required when defining the end of the growing season in TIMESAT (Olsen et al., 2015). This phenomenon occurred within our study area, leading us to choose a larger seasonal amplitude percentage to define the end of the growing season (45%) compared to the beginning of the growing season (10%).

The TIMESAT program requires consistent time-intervals between each satellite image used as the NDVI inputs. Since AVHRR and MODIS have different time-intervals between images (10-day and 16-day, respectively), we processed them separately within the TIMESAT program. We used the adjusted AVHRR NDVI images in TIMESAT for the years 1985 to 2006, and the MODIS NDVI images for the years 2000 to 2015. We analyzed the TIMESAT metrics of maximum NDVI value (Max), amplitude (Amp), large integral (LI), and small integral (SI) (Figure 2.3; Table 2.2). The time-step differences between the sensors made it necessary to standardize the LI and SI outputs in order to have comparable

values derived from the AVHRR and MODIS sensors (Jönsson, pers. comm.). We standardized the AVHRR integral output values by dividing the raw TIMESAT values by 36.5, the total number of days in a year divided by the number of days between each image obtained by the AVHRR sensor:

$$\textbf{Standardized Integral Value} = \frac{(\textbf{Raw TIMESAT Value})}{36.5} \quad (2)$$

We standardized the MODIS integral output values by dividing the raw TIMESAT values by 22.8125, the total number of days in a year divided by the number of days between each image obtained by the MODIS sensor:

$$\textbf{Standardized Integral Value} = \frac{(\textbf{Raw TIMESAT Value})}{22.8125} \quad (3)$$

After standardization, we analyzed the agreement between the AVHRR and MODIS TIMESAT values for the overlapping years of 2000 to 2006 to determine if the sensors could be used interchangeably.

## 2.4. Results

### 2.4.1. NDVI Comparison between Adjusted AVHRR and MODIS

There was strong agreement between the adjusted AVHRR and MODIS NDVI values. The average NDVI values among the overlapping years for the original AVHRR, adjusted AVHRR, and MODIS sensors were compared to determine if the sensor adjustment was effective (Figure 2.4). The linear regression equation applied to the original AVHRR successfully shifted the NDVI values of the adjusted AVHRR to align closer to the MODIS sensor. The dry season months (May to September) appear to have closer agreement between the adjusted AVHRR and MODIS, while the wet season months (October to April) had slightly lower correlation (Figure 2.4). Robinson's coefficient of agreement,  $A$ , and Willmott's index of agreement,  $d$ , measure the agreement between two sets of values (Robinson, 1957; Willmott, 1981;1982), with the higher agreement depicted by values closer to one. Although Robinson's  $A$  did not change, Willmott's  $d$  shows an improved agreement of 0.956 between the adjusted AVHRR and MODIS compared to the agreement of the original AVHRR and MODIS NDVI values of 0.829 (Table 2.3).

The linear relationship between the adjusted AVHRR NDVI and MODIS NDVI for each month had good agreement and correlation (Table 2.3). For example, the month of April had an  $R^2$  of 0.89 (Figure 2.5), while all months combined had an  $R^2$  of 0.86 (Table 2.3). Pearson's correlation coefficient was slightly higher for the adjusted AVHRR compared to the original AVHRR when assessed against the MODIS NDVI (Table 2.3), although the values were similar because the adjustment was largely in the y-intercept, with only minor deviations from 1:1 in the slope term (Table 2.1).

In addition to Robinson's  $A$  and Willmott's  $d$ , we used Ji and Gallo's systematic and unsystematic agreement coefficients to measure the NDVI agreement among the sensors (Ji & Gallo, 2006). This distinction offers additional insight into agreement analyses, as systematic differences among sensors can be adjusted through a regression equation, while unsystematic differences are random errors that may be improved through statistical filters (Ji & Gallo, 2006). The systematic agreement coefficient between the adjusted AVHRR and MODIS NDVI was much higher than between the original AVHRR and MODIS NDVI, with values of 0.982 and 0.782, respectively (Table 2.3). The unsystematic agreement coefficient was higher between the original AVHRR and MODIS compared to the adjusted AVHRR and MODIS, with values of 0.942 and 0.862, respectively (Table 2.3). Taken together, Ji and Gallo (2006) coefficients indicate greater systematic agreement and less unsystematic error for the adjusted AVHRR.

#### **2.4.2. TIMESAT Metric Comparison between Adjusted AVHRR and MODIS**

The TIMESAT metrics derived from the smoothed NDVI data were compared between the adjusted AVHRR and MODIS sensors during the overlapping years (Figure 2.6). TIMESAT smoothed extreme spikes from the raw NDVI data and produced time-series curves for each growing season.

Although the results for the four TIMESAT metrics varied slightly, they all appeared to have a relatively strong agreement between the adjusted AVHRR and MODIS sensors. Both Robinson's  $A$  (0.915) and Willmott's  $d$  (0.866) had the highest agreement between the adjusted AVHRR and MODIS sensors for Max (Table 2.4). Following the Max with the highest agreement were the Amp (Robinson's  $A$  = 0.877, Willmott's  $d$  = 0.821) and LI (Robinson's  $A$  = 0.873, Willmott's  $d$  = 0.821) (Table 2.4). The SI had the least agreement between the adjusted AVHRR and MODIS sensors, with Robinson's  $A$  of 0.840

and Willmott's  $d$  of 0.776 (Table 2.4). For all TIMESAT metrics, MODIS had a slightly higher mean, standard deviation, and standard error compared to the adjusted AVHRR sensor (Table 2.4).

There were linear relationships between the adjusted AVHRR and MODIS for the TIMESAT metrics Max ( $R^2 = 0.702$ ), Amp ( $R^2 = 0.592$ ), LI ( $R^2 = 0.641$ ), and SI ( $R^2 = 0.520$ ) (Figure 2.7). The Max had the highest correlation between the sensors (Pearson's correlation coefficient = 0.838), followed by the LI (0.801), Amp (0.770), and SI (0.721) (Table 2.5).

The systematic agreement coefficient was higher than the unsystematic agreement coefficient for all TIMESAT metrics when comparing between the adjusted AVHRR and MODIS sensors. The systematic and unsystematic agreement coefficients were relatively strong for Max ( $A_s = 0.877$ ,  $A_u = 0.826$ ), Amp ( $A_s = 0.855$ ,  $A_u = 0.748$ ), LI ( $A_s = 0.804$ ,  $A_u = 0.783$ ), and SI ( $A_s = 0.794$ ,  $A_u = 0.709$ ) (Table 2.5).

## 2.5. Discussion

To account for differences between the two sensors, we used linear regression in order to adjust AVHRR NDVI to MODIS NDVI, improving the consistency of the continuous 30-year NDVI dataset. As expected, the MODIS values were slightly higher than the AVHRR values, as found with other studies that compared these sensors after performing linear regression correction (van Leeuwen et al., 2006; Ji et al., 2008; Mao et al., 2012). The difference in mean NDVI values between the MODIS and adjusted AVHRR sensors was slightly greater in the rainy season months compared to the dry season months (Figure 2.4). Correlation between AVHRR and MODIS NDVI is likely to be influenced by the savanna landscape, which has noisy NDVI values due to the mixture of trees and grasses (Archibald & Scholes, 2007). In arid and semi-arid areas, Huete et al. (2002) found NDVI values for the AVHRR and MODIS sensors strongly aligned during the dry season months, but had a greater difference during the wet season months. We found the difference between the wet and dry season agreements across the sensors became more obvious when averaging the values rather than calculating the agreement between all individual pixels. Therefore, studies based on NDVI averages across multiple sensors may be more influenced by seasonal differences compared to per-pixel analyses. These findings are important to consider, as

researchers should be aware that when averaging pixel values from multiple sensors, the rainy season months may have less agreement across sensors compared to the dry season months. The month of February appeared to be impacted the most, as it had the lowest correlation between the sensors. In contrast, the month of April had the highest correlation, which also occurs during the wet season. It has been found that February has most rainfall variation of any month in our study area (Vezzoli et al., 2008). This leads us to believe that although the rainfall may have an occasional impact on NDVI sensor adjustment, it may not have drastic influences apart from the most extreme rainy periods. Overall, our correlation values correspond to those from other studies in arid systems (Gallo et al., 2004; Bao et al., 2014) and the adjusted AVHRR had higher agreement measures to MODIS compared to that of the original AVHRR. This illustrates that the NDVI pixel values became more similar after the linear regression correction was applied to the original AVHRR NDVI dataset.

Once we concluded that the continuous 30-year NDVI dataset was consistent after sensor correction, we explored whether extending the TIMESAT dataset across multiple sensors produced reliable results. We found strong correlation and agreement for the TIMESAT metrics derived from the adjusted AVHRR and MODIS sensors for the overlapping years, validating the approach to use multiple sensors within the TIMESAT program interchangeably. The ability to extend TIMESAT over longer periods of time has powerful implications, since the TIMESAT metrics indicate different ecological phenomena. The LI and SI metrics are robust measures of vegetation biomass (Wessels et al., 2011; Mbow et al., 2013; Olsen et al., 2015). Wessels et al. (2011) found that LI and SI were the most important metrics when classifying vegetation biomes within South Africa, and Olsen et al. (2015) suggested SI should be the metric used to analyze vegetation productivity within herbaceous-dominated ecosystems. Mbow et al. (2013) determined that SI had a positive relationship to annual net primary production (ANPP) without being influenced by species dominance, while Max had a negative relationship to ANPP and was sensitive to species composition. Both Mbow et al. (2013) and Olsson et al. (2005) suggested that Amp trends may be influenced by changes in species dominance. The differences found between the

TIMESAT metrics need to be further explored, but suggest that the range of metrics has strong potential to relay unique information regarding vegetation across landscapes.

Differentiating between the TIMESAT metrics may lead to enhanced interpretation capabilities for decision-makers and managers. Max and Amp appear to be sensitive to species composition (Olsson et al., 2005; Mbow et al., 2013) while LI and SI are robust measures of herbaceous biomass (Wessels et al., 2011; Mbow et al., 2013; Olsen et al., 2015). This separation among the metrics' associations may assist in analyzing types and causes of landscape changes. Scholtz et al. (2017) investigated the relationships between vegetation composition and different forms and degrees of disturbance within the savanna landscape. They found shrub composition was most responsive to fire disturbance, while elephant densities had the largest impact on tree composition. Future studies are needed to continue to gain a better understanding on how both natural- and human- caused disturbances impact vegetation differently. Remote sensing techniques such as the vegetation metrics derived from TIMESAT may prove to have powerful management implications when analyzing changes across large spatial and temporal scales.

Our study highlights several matters to keep in mind when performing multi-sensor correction for arid systems and applying multiple sensors in the TIMESAT program. Due to the seasonal differences in savanna ecosystems, deriving month-specific linear equations when performing multi-sensor corrections may improve the accuracy of the final adjusted datasets. Even so, performing analyses with remote sensing data is never perfect and it is important to understand distinctions between the sensors one is working with. In the case of MODIS and AVHRR, the user should keep in mind that MODIS NDVI values will usually be slightly higher than those from AVHRR, which may be an important distinction when interpreting results. It should also be kept in mind that there may be slightly less sensor agreement for months within the rainy season, especially when averaging pixel values. When utilizing multiple sensors in the TIMESAT program to analyze interchangeably, the sensors must be treated differently when inputting the NDVI datasets into TIMESAT as well as standardizing the outputs. If the sensors have different time-steps between images as in the case with AVHRR and MODIS, the NDVI datasets must be

submitted to TIMESAT separately, as the settings require consistent time-step intervals for each dataset. The output values from TIMESAT often need to be standardized according to the time-step, as is the case with the LI and SI. It should be noted additional TIMESAT metrics that this study did not discuss in detail may also need to be standardized due to the time-step influence on the values, such as the beginning, middle, end, and length of the growing season. It is therefore vital that the user has a strong understanding of the different sensor characteristics when using them interchangeably in TIMESAT. As long as the user properly takes these differences into account, we believe that utilizing multiple sensors in the TIMESAT program will produce accurate results and improve the temporal capability of the program.

## **2.6. Conclusion**

Remote sensing data allows for complex landscape analyses on large spatial and temporal scales. It has become common practice to use multi-sensor correction to extend temporal analyses by combining sensors with different timespans, utilizing periods of overlap for inter-sensor calibration. TIMESAT has become a widely used tool in measuring vegetation parameters in a robust manner. This is the first study to use multiple sensors adjusted to create a temporally lengthened TIMESAT dataset. The results show the TIMESAT metrics have high correlation and agreement between the two sensors, therefore validating the methodology. This approach has powerful applications for future vegetation studies by allowing the TIMESAT metrics, which have been shown to be robust and relay unique phenological and ecological patterns, to be analyzed on an extended temporal scale. These applications are vital as remote sensing measures become increasingly important for large-scale decision-making and conservation planning.

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## CHAPTER 2 TABLES

**Table 2.1.** The intercept, slope, and  $R^2$  values calculated from the monthly ordinary least-squared regression of AVHRR and MODIS values.

<b>Month</b>	<b>Intercept</b>	<b>Intercept SE</b>	<b>Slope</b>	<b>Slope SE</b>	<b>R<sup>2</sup></b>	<b># Pixels</b>
January	1009.65	12.33	1.04	<0.01	0.73	53,640
February	1505.88	12.74	0.89	<0.01	0.68	54,687
March	1162.02	10.92	1.04	<0.01	0.77	63,738
April	1059.82	7.93	1.03	<0.01	0.85	69,773
May	1266.98	7.37	0.91	<0.01	0.82	69,930
June	1048.02	6.30	0.99	<0.01	0.83	69,891
July	946.33	5.69	0.98	<0.01	0.83	69,891
August	983.03	5.09	0.98	<0.01	0.80	69,068
September	1006.40	4.65	1.07	<0.01	0.81	68,159
October	1047.35	5.30	0.99	<0.01	0.77	64,127
November	1097.81	9.63	1.00	<0.01	0.68	64,675
December	1154.09	8.95	1.00	<0.01	0.79	65,468
<b>Total</b>						783,047

**Table 2.2.** TIMESAT metric descriptions and interpretations (revised from Jönsson & Eklundh, 2004 & Reed et al., 1994).

<b>TIMESAT Metric</b>	<b>Description</b>	<b>Phenological Interpretation</b>
Maximum NDVI	Largest NDVI value for fitted function during the season.	Maximum measurable level of photosynthetic activity.
Amplitude	Difference between Max and Base.	Range of measurable photosynthetic activity.
Large Integral	Area within the growing season curve between the fitted function and zero level.	Total vegetation production.
Small Integral	Area within the growing season curve between the fitted function and Base.	Seasonally active vegetation.

**Table 2.3.** Agreement and correlation indices comparing the MODIS NDVI values to the adjusted and original AVHRR values.

Month	Robinson's <i>A</i>		Willmott's <i>d</i>		$AC_u$		$AC_s$		$R$		$R^2$		# Pixels
	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	
Jan	0.91	0.90	0.89	0.69	0.72	0.91	0.95	0.65	0.82	0.82	0.66	0.66	60,000
Feb	0.80	0.81	0.74	0.62	0.52	0.82	0.82	0.64	0.63	0.63	0.39	0.39	70,000
Mar	0.93	0.93	0.92	0.68	0.83	0.96	0.92	0.61	0.89	0.89	0.79	0.79	70,000
Apr	0.97	0.97	0.97	0.77	0.90	0.97	0.99	0.71	0.94	0.94	0.89	0.89	70,000
May	0.96	0.95	0.95	0.85	0.82	0.93	1.00	0.82	0.91	0.91	0.83	0.83	70,000
Jun	0.94	0.94	0.94	0.75	0.74	0.94	1.00	0.71	0.88	0.88	0.77	0.77	70,000
Jul	0.93	0.94	0.93	0.72	0.72	0.93	0.99	0.69	0.87	0.87	0.76	0.76	70,000
Aug	0.90	0.91	0.89	0.58	0.66	0.94	0.97	0.57	0.82	0.82	0.67	0.67	70,000
Sep	0.86	0.85	0.84	0.49	0.54	0.94	0.89	0.47	0.75	0.75	0.57	0.57	70,000
Oct	0.89	0.89	0.88	0.63	0.57	0.90	0.99	0.63	0.78	0.78	0.61	0.61	70,000
Nov	0.91	0.91	0.90	0.74	0.71	0.91	0.94	0.70	0.84	0.84	0.71	0.71	70,000
Dec	0.95	0.95	0.94	0.79	0.85	0.95	0.96	0.74	0.91	0.91	0.82	0.82	70,000
Total	0.96	0.95	0.96	0.83	0.86	0.94	0.98	0.78	0.92	0.92	0.86	0.84	830,000

**Table 2.4.** Mean, standard deviation, and standard error for TIMESAT productivity metrics from adjusted AVHRR and MODIS sensors.

TIMESAT Metric	Mean		Standard Deviation		Standard Error	
	Adjusted AVHRR	MODIS	Adjusted AVHRR	MODIS	Adjusted AVHRR	MODIS
Maximum NDVI	6216.24	6626.60	837.07	956.60	0.71	0.81
Amplitude	3727.58	4132.64	773.64	943.54	0.66	0.80
Large Integral	3145.55	3488.09	617.70	907.13	0.52	0.77
Small Integral	1531.51	1785.12	407.28	575.65	0.35	0.49

**Table 2.5.** Agreement and correlation indices comparing TIMESAT metrics from adjusted AVHRR and MODIS sensors.

<b>TIMESAT Metric</b>	<b>Robinson's <i>A</i></b>	<b>Willmott's <i>d</i></b>	<b>AC<sub>u</sub></b>	<b>AC<sub>s</sub></b>	<b>RMSE</b>	<b>R</b>	<b>R<sup>2</sup></b>	<b># Pixels</b>
Maximum NDVI	0.92	0.87	0.83	0.88	664.89	0.84	0.70	1,392,411
Amplitude	0.88	0.82	0.75	0.86	727.56	0.77	0.59	1,392,411
Large Integral	0.87	0.82	0.78	0.80	651.40	0.80	0.64	1,392,411
Small Integral	0.84	0.78	0.71	0.79	472.77	0.72	0.52	1,392,411

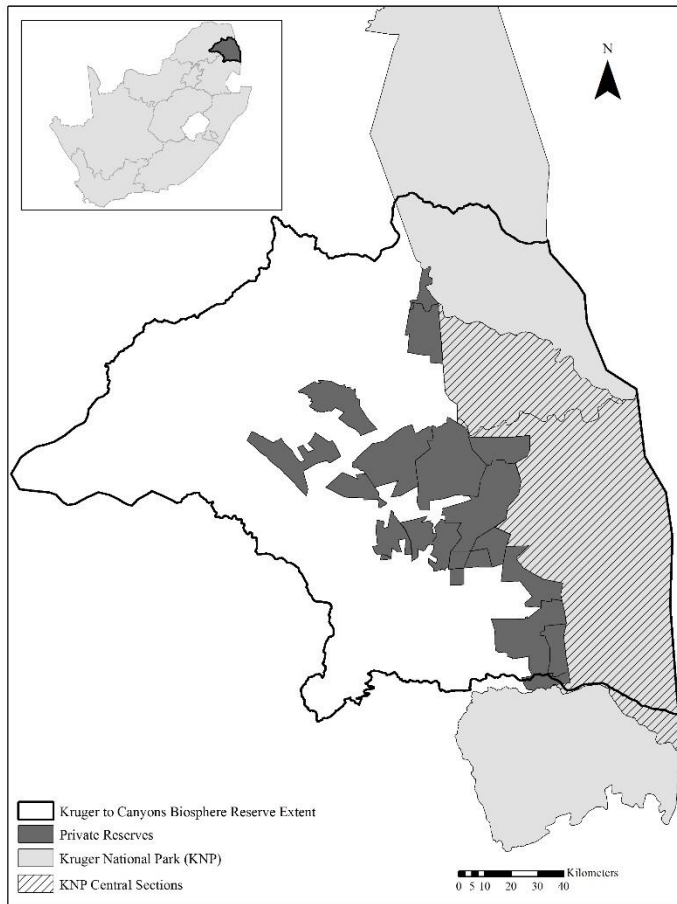
Robinson's *A* = Robinson's coefficient of agreement

Willmott's *d* = Willmott's index of agreement

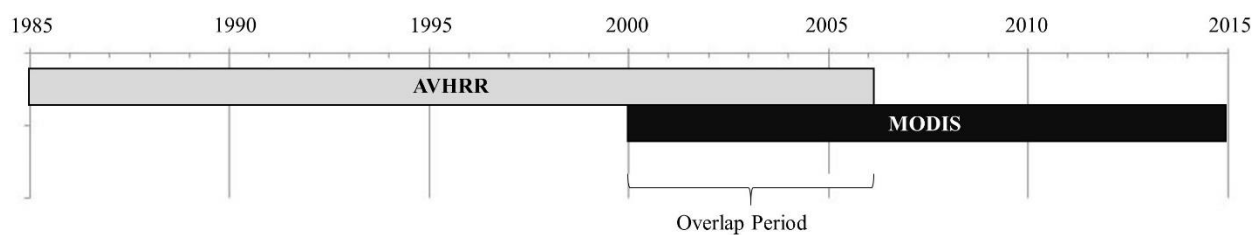
AC<sub>u</sub> = Ji & Gallo's unsystematic agreement coefficient

AC<sub>s</sub> = Ji & Gallo's systematic agreement coefficient

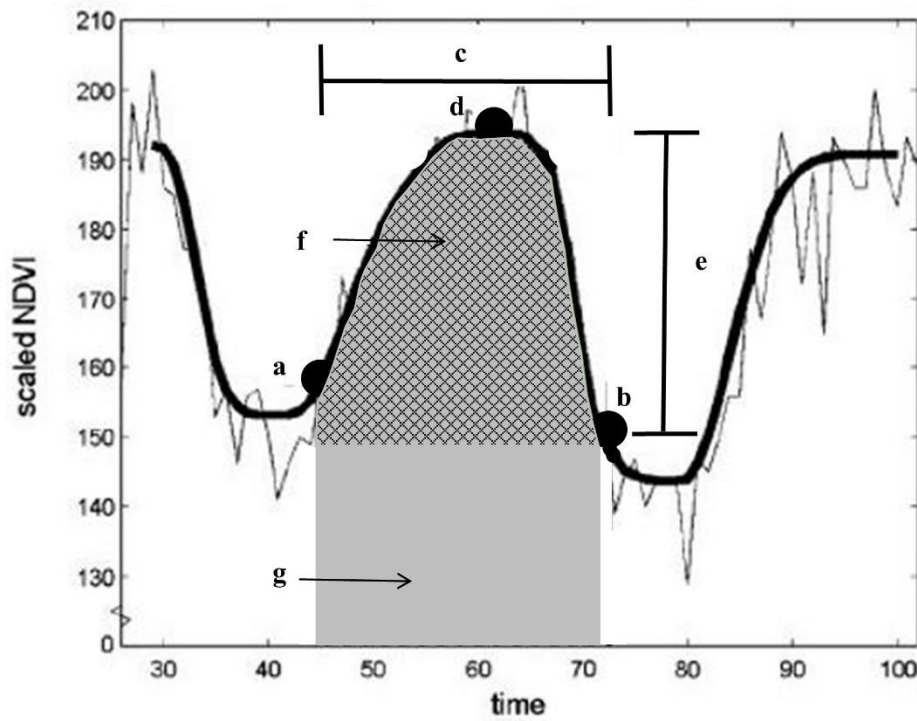
## CHAPTER 2 FIGURES



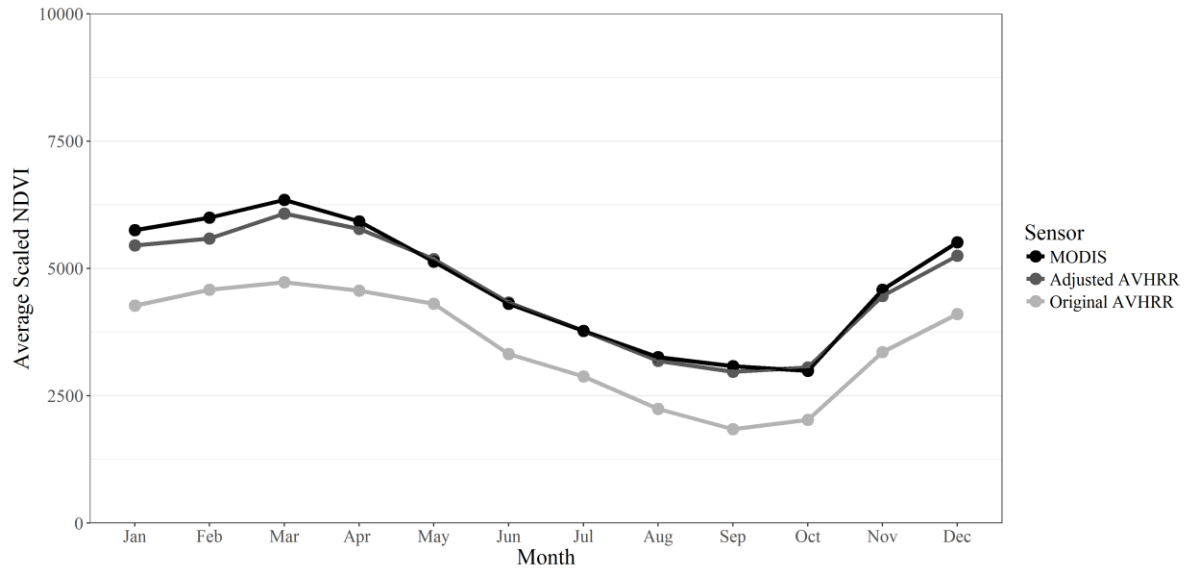
**Figure 2.1.** The study area, depicting the private reserves and central sections of Kruger National Park within the Kruger to Canyons Biosphere Reserve.



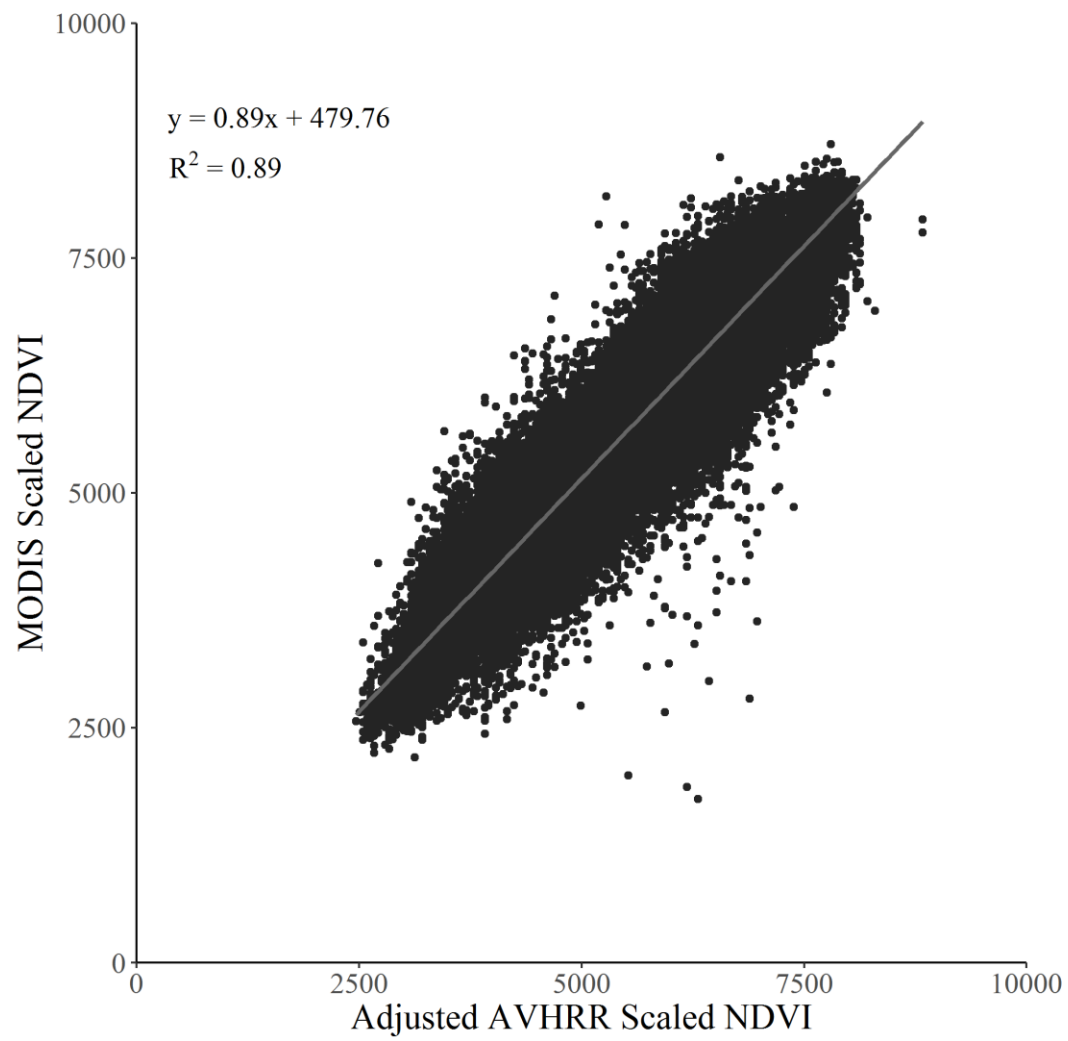
**Figure 2.2.** A timeline illustrating the years covered by the AVHRR and MODIS sensors for the study period 1985 to 2015.



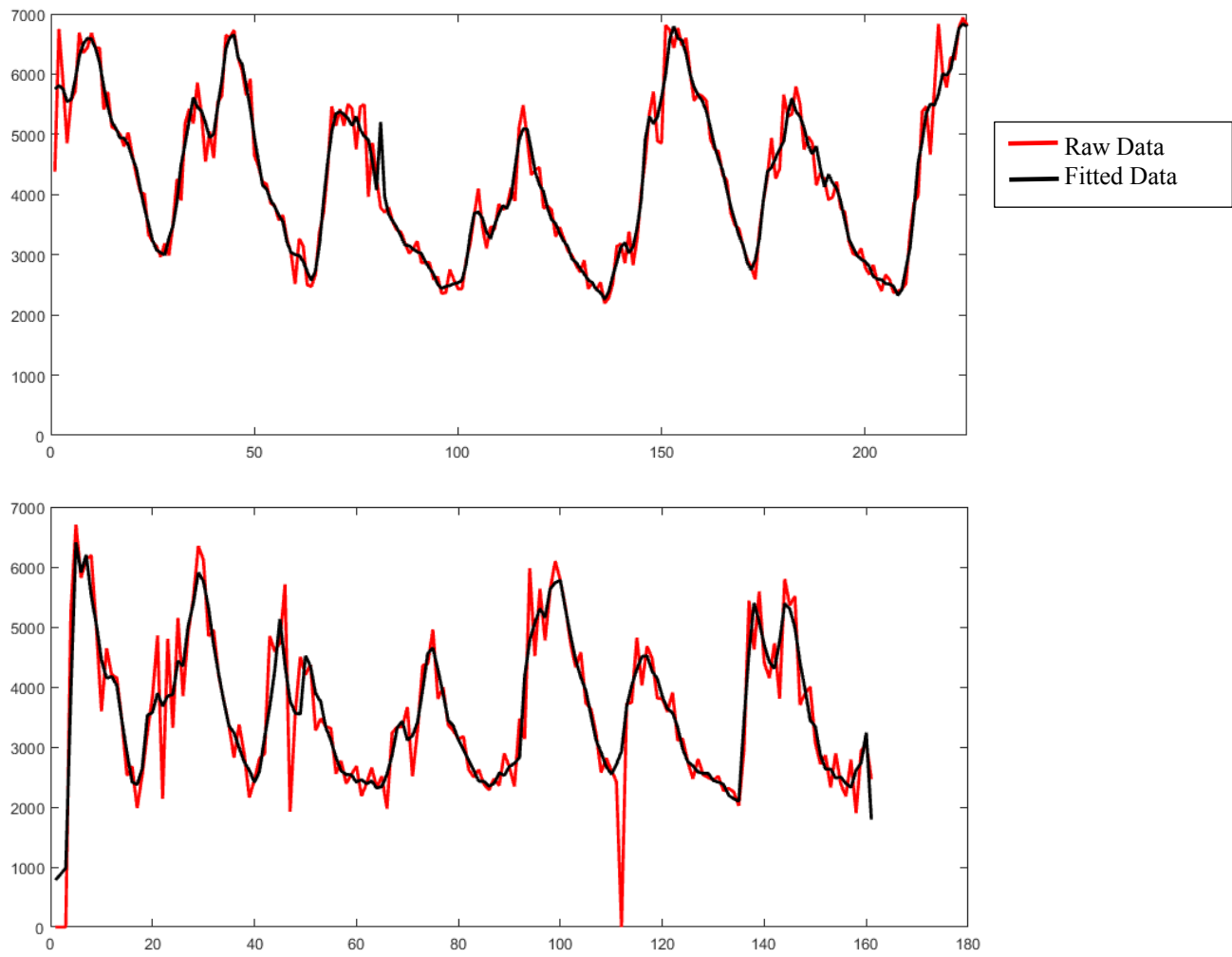
**Figure 2.3.** The seasonality metrics produced from TIMESAT: a) beginning of growing season, b) end of growing season, c) length of growing season, d) maximum NDVI, e) seasonal amplitude, f) small seasonal integral, and f + g) large seasonal integral (revised from Jonsson & Eklundh 2004).



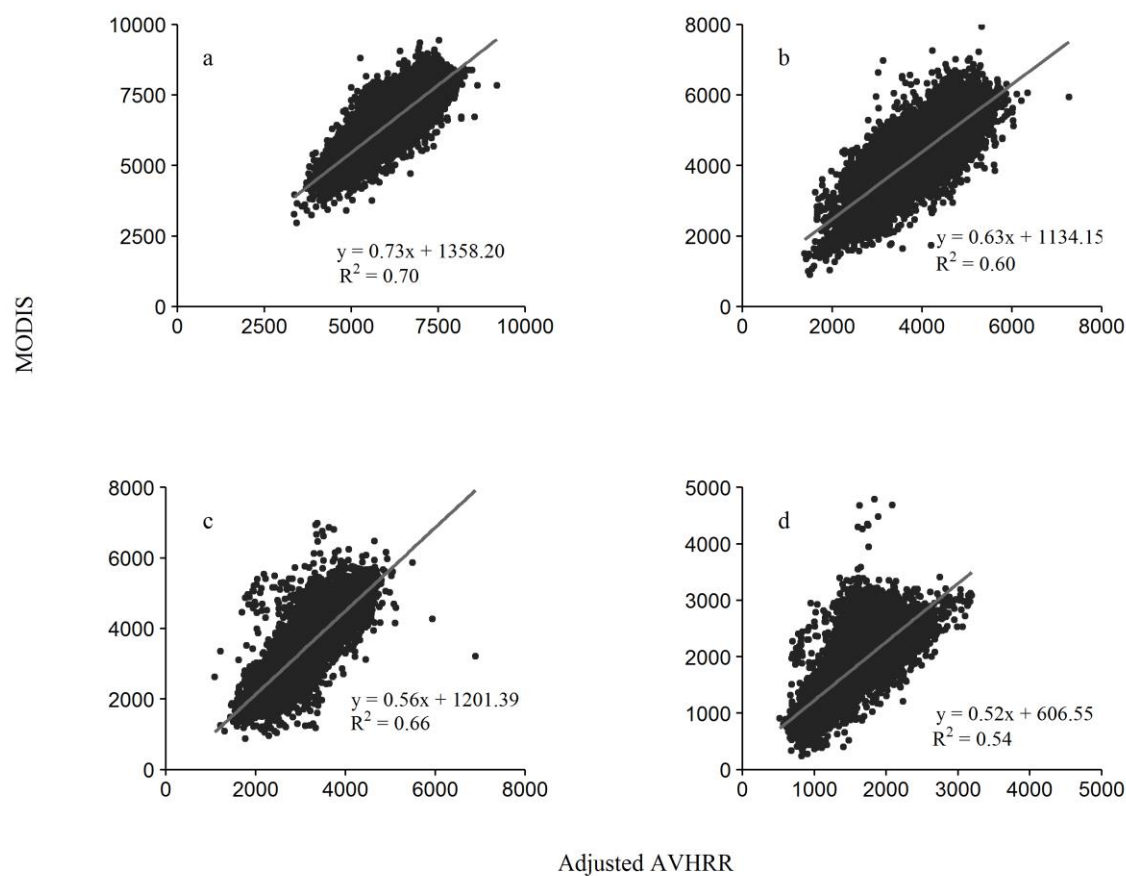
**Figure 2.4.** The average NDVI value per month for the original AVHRR, adjusted AVHRR, and MODIS sensors for the years 2000 to 2006 (the standard error bars are too small to be visible due to the large pixel sample sizes).



**Figure 2.5.** The linear relationship of the adjusted AVHRR NDVI and MODIS NDVI for the month of April for the years 2000 to 2006 (n = 70,000 pixels).



**Figure 2.6.** The time series curves produced from TIMESAT for years 2000 to 2006 for a) adjusted AVHRR (time-step = 10 days) and b) MODIS (time-step = 16 days).



**Figure 2.7.** The linear relationships between the TIMESAT metrics derived from the adjusted AVHRR and MODIS sensors for a) Max, b) Amp, c) LI, and d) SI (a sample of  $n = 10,000$  pixels was used for each scatterplot).

## **CHAPTER 3: The impacts of fence removal on vegetation dynamics within the Kruger to Canyons Biosphere Reserve**

### **3.1. Abstract**

Maintaining natural levels of habitat heterogeneity is favorable in conservation areas due to its relationship with biodiversity. Herbivory is a common disturbance within savanna ecosystems, therefore influencing heterogeneity. Landscape fragmentation by game fencing prevents the natural dispersal of herbivores. The large-scale fence removal that occurred across Kruger National Park (KNP) and adjacent reserves increased opportunities for herbivore movement across a larger ecosystem. The goals of our research were to 1) determine if the vegetation dynamics within reserves differed from KNP before and/or after fences were removed and 2) determine potential factors that may have contributed to any vegetation changes post-fence removal. We used a continuous 30-year NDVI dataset to extract vegetation phenological metrics from the TIMESAT program for each growing season between 1985 and 2015. We compared these vegetation metrics within the reserves and KNP before and after fence removal. We found that vegetation within the reserves was similar to KNP before fence removal, but generally decreased in average vegetation metrics and vegetation metric variability post-fence removal. While many factors were associated with changes in vegetation following fence removal including geology, seasonality, and rainfall, we highlight surface water as it can be manipulated by reserve managers. Supplementing water availability with artificial waterpoints is a common practice in reserves because waterpoints increase animal viewing opportunity for tourists. Yet, as waterpoint and river density increased, average vegetation metrics and vegetation metric variability decreased in open reserves, but remained constant in closed reserves. One potential mechanism for this dynamic is that high surface water density caused an influx of herbivores into the open reserves once the fences were removed, resulting in vegetation homogenization due to overutilization. Thus, while fence removal can increase opportunity for natural, seasonal movements by herbivores, consideration should be given to the role of artificial waterpoints in managing vegetation heterogeneity. Reducing the density of artificial waterpoints within

reserves that are open to KNP may distribute vegetation impacts and subsequently restore natural spatial heterogeneity within the Kruger to Canyons Biosphere Reserve.

### **3.2. Introduction**

Vegetation heterogeneity plays an integral role in savanna ecosystems, where multiple plant types coexist in space and time through competition, facilitation, niche separation, and disturbances (Scholes & Archer, 1997; du Toit et al., 2003). Trees compete with grasses through rainfall interception, ground litter, shading, or root competition, while they facilitate grasses by providing nutrients post-decomposition (Scholes & Archer, 1997). Grasses regulate woody recruitment through competition for light, water, and nutrients or as fire fuel (Scholes & Archer, 1997).

Common disturbance agents in savanna ecosystems include herbivory, fire, and ecosystem engineers. Herbivory creates disturbance within the savanna landscape by modifying the structure and composition of woody vegetation (Scholes & Archer, 1997; Levick & Rogers, 2008) and influencing soil nutrients (van der Waal et al., 2011). Fire influences vegetation communities by permitting the coexistence of trees and grasses, whereas fire suppression may cause woody encroachment (van Wilgen et al., 2003; D’Odorico et al., 2006). Ecosystem engineers alter, sustain, and generate habitats by changing the physical states of resources and altering resource availability for other species within the ecosystem (Jones et al., 1994). Ecosystem engineers’ influence on surrounding species may be positive or negative, although the disturbances they create are thought to ultimately increase species richness within the environment (Jones et al., 1997). In savanna landscapes, elephants are ecosystem engineers because they increase tree structure complexity and create microhabitat refuges for smaller species (Douglas-Hamilton et al., 2005; Pringle, 2008). However, it has also been found that local elephant densities that are too high can homogenize vegetation and result in the loss of specific sensitive plant species (Cummings et al., 1997; Guldemon & van Aarde, 2008).

Landscape fragmentation through fencing alters the natural heterogeneity of savanna ecosystems, preventing the natural dispersal of herbivores to access resources (Hayward & Kerley, 2009; Ferguson et al., 2012; Venter et al., 2015). Since organisms’ resource consumption often degrades their habitats (Holt,

2009), animal dispersal allows vegetation to recover (Sinclair & Fryxell, 1985). Fences prevent movement toward high quality biomass patches (Boone & Hobbs, 2004), inhibiting natural population regulation and causing an overuse of resources (Hayward & Kerley, 2009). Habitats fragmented by fencing often overstock wildlife and limit movement, thereby disadvantaging species that would not normally compete for resources in a natural setting (Bond et al., 2004; Lindsey et al., 2008; 2012). Fencing also impacts wildlife migration by altering populations' natural movement and possible adjustment to climate conditions (Newmark, 2008; Hayward & Kerley, 2009). As large herbivore movements become restricted, the structure and diversity of plant communities become impacted (Newmark, 2008). However, fencing provides some benefits to humans. Fencing can decrease human-wildlife conflict and prevent human exploitation (Hayward & Kerley, 2009; Lindsey et al., 2012). Fenced protected areas also provide economic benefits through tourism (Cumming, 2004; Cumming et al., 2015). The pros and cons of fencing need to be considered on a situational basis to decide when it is in the best interest of conservation management to fence wildlife populations (Hayward & Kerley, 2009).

Protected area objectives evolved as conservation biology converted from merely focusing on endangered species to developing a greater mindfulness of entire ecosystems and their potential human services (Cumming et al., 2015). Although the area and number of parks has grown extensively over the past century in southern Africa, the average park size has decreased, suggesting that protected areas have become fragmented “ecological islands” (Cumming, 2004). Management intensities differ based on the objectives of various protected areas, with intensively managed reserves often focusing on ecotourism revenue and low-intensity management providing a stronger focus on conserving rare species and maintaining sustainable vegetation levels (Child et al., 2013). Even with different management objectives, connecting adjacent protected areas by removing fences along borders has become common in southern Africa (Newmark, 2008). In these instances, wildlife becomes a “common pool resource” where multiple owners attempt to coordinate management decisions impacting the wildlife that inhabit merged properties (Kreuter et al., 2010). However, management priorities do not always align (Kreuter et al., 2010), and owners may be hesitant to join with neighboring reserves due to the possibility that wildlife within their

land may move into adjacent areas once fences are removed (Lindsey et al., 2012). Although many challenges arise when creating wildlife conservancies, connecting isolated protected areas through fence removal is often thought to be one of the best ways to conserve biodiversity (Hayward & Kerley, 2009).

Large-scale research is needed to better understand the impacts of fencing on savanna landscapes. Boone and Hobbs (2004) modelled scenarios with different levels of fragmentation and suggested that as a landscape becomes more fragmented through fencing, herbivore carrying capacity and vegetation heterogeneity decreases. They called for researchers to use the opportunity of large-scale fence removal in Africa to study the response of herbivory. Graz et al. (2012) also used scenarios to predict the impact of herbivores on vegetation after removing fences and/or waterholes across a landscape. They predicted that in comparison to the default scenario of retaining fences and high numbers of waterpoints, removing fences while maintaining high waterpoint numbers would increase the overall grazing impact by 44.3%, reducing waterpoints while retaining fences would decrease overall grazing by 78.4%, and removing fences along with reducing waterpoint numbers would counterbalance one-another and increase overall grazing by 11.1%. Similarly to Boone and Hobbs (2004), Graz et al. (2012) stated that there was no landscape-level field research investigating the concurrent impact of waterpoints and fencing. Until now, no study has provided empirical evidence to examine these hypotheses by measuring the landscape-scale impact of fence removal. The aims of our research were to 1) determine if the vegetation dynamics within reserves differed from Kruger National Park before and/or after fences were removed and 2) determine potential factors that may have contributed to any vegetation changes post-fence removal, with a specific focus on artificial water provision.

### **3.3. Methods**

#### **3.3.1. Study Area**

Our study area included the reserves and the central sections of Kruger National Park (KNP) located within the Kruger to Canyons Biosphere Reserve (K2C) in the eastern Lowveld of South Africa between 30°35' E and 30°40' E and 24°00' S and 25°00' S (Peel et al., 2007) (Figure 3.1). These managed

protected areas are part of the Limpopo and Mpumalanga provinces and fall within the savanna biome (Peel et al., 2007). The protected areas have a total area of about 1.6 million ha. The average annual rainfall varies over a gradient in the study area, ranging from about 400mm to 650mm with the rainfall levels increasing from north to south (Peel et al., 2007). Rainfall occurs primarily between the months of October and April (Venter et al., 2003). The geology is predominantly granitic rock, with sections of amphibolitic and gabbro rock woven into some of the reserves (Keyser, 1997; Venter, pers. comm.).

Cattle ranching was the dominant land use within the K2C until the 1970's when the focus switched to wildlife tourism (Peel et al., 2005). In the early 1990's reserves adjacent to KNP began to remove their fences (Peel et al., 2005) creating a continuous landscape known as the "Greater Kruger Ecosystem." The changing land uses from ranching to fenced ecotourism to an unfenced open system has influenced the structure and function of the vegetation within the landscape (Peel et al., 2005).

We used KNP as the "control" representing a natural, open system to compare with the reserves. Although in reality KNP is not truly natural, its large size and management objective to "maintain biodiversity in all its natural facets and fluxes" (Mabunda et al., 2008) makes it as close to a natural ecosystem as possible. KNP management acknowledges that landscape-scale heterogeneity allows long-term ecosystem sustainability by maintaining biodiversity (Venter et al., 2008). It should be noted that within our study time period 1985 to 2015, several changes in KNP management regimes occurred. From 1985 to 1995, management "*Laissez-faire*" was adopted to allow natural ecosystem processes to flow (Venter et al., 2008). Strategic adaptive management was implemented in 1995 and is the current management regime within KNP. Through this management regime, the goal is still to allow the ecosystem to function naturally, but with the addition of progressive structured learning through experimentation with unexpected incidents that occur within the complex system (Levin, 1999; Venter et al., 2008). As scientific knowledge improved the understanding of the ecosystem, specific management regimes such as artificial waterholes, fire, and elephant culling were altered in order to achieve a natural functioning system (du Toit et al., 2003; SANParks, 2012; Smit, 2013; van Wilgen et al. 2014).

We used eighteen reserves within the K2C to determine the impact of fence removal on vegetation dynamics. Ten of the reserves removed their fences within the period 1985 to 2015, therefore becoming open and connecting to the Greater Kruger Ecosystem (Figure 3.1). Eight of the reserves retained their fences during this time period and remained closed to other reserves and the Greater Kruger Ecosystem (Figure 3.1). Although the reserves adjacent to KNP aim to incorporate the management goals of the KNP, they operate at smaller spatial extents. Similarly, the smaller fenced reserves encounter various management challenges due to their size and restricted animal movement (Peel et al., 1998; 2005). Child et al. (2013) found considerable variation among management intensities across the reserves in the Lowveld savanna biome, meaning that the vegetation changes following fence removal likely varies depending on different management regimes within the reserves.

### **3.3.2. Data**

#### **3.3.2.1. Reserve Boundaries**

We obtained reserve boundaries from the reserve managers when available, and supplemented with shapefiles<sup>1</sup> provided from the K2C Biosphere Reserve when necessary. Since ownership and reserve boundaries have changed over time, we used a parcel shapefile obtained from the Association for Water and Rural Development to update necessary boundaries to represent the current reserves (Holness et al., 2014).

#### **3.3.2.2. Geology**

Data from the Council of Geoscience were used to account for different geology types within the reserves. The original shapefile contained 21 geology types located within the reserves. This list was reclassified and simplified to six classes: amphibolitic rock, basaltic rock, gabbro, granitic rock, rhyolitic rock, sandstone, and shale (Venter, pers. comm.; Appendix A). We calculated the percentage of each rock

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<sup>1</sup> Developed by ESRI for its GIS software, a shapefile is “A vector data storage format for storing the location, shape, and attributes of geographic features. A shapefile is stored in a set of related files and contains one feature class.” (<http://support.esri.com/sitecore/content/support/Home/other-resources/gis-dictionary/term/shapefile>)

class within each reserve. All reserves had a majority of granitic rock, with sections of amphibolitic and gabbro included in some of the reserves. The percentages of amphibolitic rock and gabbro rock were used as explanatory variables in the analysis.

### **3.3.2.3. Vegetation Metrics**

The normalized difference vegetation index (NDVI) is commonly used to measure vegetation from satellite images. We obtained the 16-day interval 250m MODIS/Terra NDVI (MOD12Q1) dataset for the time period 2000 to 2015 from NASA EOSDIS. We used 10-day interval AVHRR NDVI that covered the time period 1985 to 2006 from the South African Space Agency (Wessels et al., 2012), which was processed and calibrated at a resolution of 1.1km by the Institute for Soil, Climate, and Water at the Agricultural Research Center (ARC-ISCW) to improve sensor changes and degradation (Rao & Chen, 1995; 1996; Wessels et al., 2006). AVHRR NDVI data were unavailable for 1994 due to the failure of the NOAA-13 satellite (Wessels et al., 2006). Wessels et al. (2004) provides additional details on the processing of the AVHRR NDVI dataset.

We used the seven overlapping years (2000 to 2006) where both AVHRR and MODIS NDVI data were available to adjust the AVHRR rasters to MODIS (Chapter 2). We first resampled the 1.1km AVHRR NDVI to 250m to align with the MODIS NDVI pixel sizes. We created maximum value composites (MVC) for each month of the overlapping years, which reduces the chance of cloud and aerosol contamination because pixels that are influenced by clouds or noise usually have low cell values (Holben, 1986). We then applied a 5x5 median statistical filter to each MVC raster to further reduce noise from mixed-pixels (Stow & Chen, 2002; Ji et al., 2008). Once the NDVI datasets were pre-processed, we extracted 10,000 random points across the K2C for each month of the overlapping years for both the AVHRR and MODIS satellite images. We compared pixel values between the AVHRR and MODIS NDVI datasets using an ordinary least-square regression analysis to produce slope and intercept values for each month. We then used the respective monthly slope and intercept values to adjust the original AVHRR NDVI to the MODIS NDVI. We performed an agreement analysis to ensure the adjusted AVHRR and MODIS NDVI values were compatible and found strong agreement between the datasets.

See Chapter 2 for more details on the sensor correction process between the AVHRR and MODIS datasets.

We used the TIMESAT program to fit a smooth continuous curve for the adjusted AVHRR and MODIS NDVI time-series datasets (Chapter 2). TIMESAT decreases the impact of satellite noise by using filtering techniques and an upper envelope to remove extreme spikes in NDVI values and weighting pixels based on cloud coverage (Jönsson & Eklundh, 2002; 2004). We applied the adaptive Savitsky-Golay filter with the season cut-off value set to one. We used the second spike method (STL replace) with one envelope iteration and a spatial window size of three. We utilized MODIS cloud coverage data to weight the pixels based on their quality, with cloudy, mixed, and clear pixels having weights of 0, 0.5, and 1, respectively. We defined the beginning of the growing season as 10% of the seasonal amplitude as measured from the left minima, and the end of the growing season as 45% of the seasonal amplitude as measured from the right minima (Chapter 2). The adjusted AVHRR and MODIS NDVI rasters were processed separately in the TIMESAT program because they had different time-intervals between images (10-day for AVHRR, 16-day for MODIS). We extracted TIMESAT vegetation metrics from AVHRR for the time period 1985 to 2006, and from MODIS for the time period 2000 to 2015. Similarly to the comparison of adjusted AVHRR and MODIS NDVI values, we used the overlapping years to compare the similarity between the TIMESAT output values from the AVHRR and MODIS sensors, which were found to have high agreement. For the analysis, the TIMESAT outputs from the AVHRR sensor were used for the years 1985 to 2000, and the outputs from the MODIS sensor were used for the years 2000 to 2015. See Chapter 2 for additional details on the process of creating the continuous 30-year TIMESAT datasets using multiple sensors.

The TIMESAT vegetation metrics used in the analysis included the beginning of the growing season, length of growing season, maximum NDVI value, amplitude, large integral, and small integral. The beginning of the growing season and the length of the growing season were used as explanatory variables. Since these TIMESAT metrics were influenced by the two different time-steps for AVHRR and MODIS, the outputs were standardized so as to be used interchangeably (Appendix B). The Maximum

NDVI value (Max), amplitude (Amp), large integral (LI), and small integral (SI) were also extracted from TIMESAT and used as the four response variables. Studies have suggested Max and Amp are influenced by vegetation composition (Olsson et al., 2005; Mbow et al., 2013), while LI and SI have been shown to be related to herbaceous biomass (Wessels et al., 2011; Mbow et al., 2013; Olsen et al., 2015). SI only measures seasonally active vegetation, while LI measures the total vegetation production (Jönsson & Eklundh, 2004; Wessels et al., 2011). LI and SI normally behave similarly in savanna landscapes (Wessels et al., 2011). Since SI has been shown to be a more robust measure compared to LI (Mbow et al., 2013), we often focused on the SI when interpreting the results. Similar to the beginning and length of the growing season, LI and SI were standardized since the time-steps differed between AVHRR and MODIS (Chapter 2).

For all TIMESAT metrics, we calculated the average and standard deviation per growing season for each reserve as well as for the central sections of KNP. Since we used KNP as our control, for each response variable we calculated the difference between KNP and each reserve per growing season. When calculating the difference, we compared most reserves to the central section of KNP between the Olifants and Sabi rivers. However, we compared three reserves (O1, O2, and C1) to the central section of KNP between the Letaba and Olifants rivers due to their latitudinal locations. By comparing the vegetation within the reserves and KNP before and after fence removal, we hoped to detect any changes that occurred and to determine if those changes transpired on local- and/or landscape-scales.

Of the seven simplified geological rock classes, the reserves in the analysis only contained three classes of rocks: granitic, amphibolitic, and gabbro. In order to keep the comparison between KNP and the reserves consistent, we only used the vegetation values that fell on sections of granitic, amphibolitic, and gabbro rock within the central sections of KNP.

#### **3.3.2.4. Rainfall**

Rainfall interpolated surfaces were kindly provided by Johan Malherbe and Philip Beukes from the Agricultural Research Council: Institute for Soil, Climate and Water (ARC-ISCW) in South Africa. See Malherbe et al. (2016) for details on the development of the historic interpolated 1km rainfall rasters.

The interpolated rasters depicted average total rainfall for a 12-month period beginning in July and ending in June of each year. We calculated the average total annual rainfall for each reserve per year. In addition to the average total annual rainfall, we included the rainfall in each of the previous three years (rainfall lags 1, 2 and 3, respectively).

#### **3.3.2.5. Topography**

We obtained USGS global multi-resolution terrain elevation data at 250m resolution to depict elevation as well as aspect, which was calculated in ArcGIS 10.3 (ESRI, 2014). We calculated the average elevation and average aspect for each reserve to acquire a constant value for each explanatory variable.

#### **3.3.2.6. Surface Water**

The two types of surface water we considered in the analysis were waterpoints and rivers. We used historic imagery from Google Earth to digitize waterpoints within the reserves. Of the historic images available in Google Earth, we used only those images that were in the dry season months, regardless of the year, to determine the locations of waterpoints that held water year-round. Therefore, ephemeral waterpoints present only during the wet season and waterpoints that were added or closed over time were not accounted for using historic images. However, this method provided the most consistent results given the difficulty in finding dry season images due to the irregular historic aerial images.

For KNP, waterpoint data were provided by the South African National Parks (SANParks). In order to stay consistent with the digitizing methodology of the reserve waterpoints, we only used waterpoints within KNP that contained water in the dry season.

We obtained river data for KNP and all of the reserves from the South African National Biodiversity Institute (SANBI). This data contained rivers with year-round flowing water.

We calculated the density of waterpoints per km<sup>2</sup> and the density of total river length (km) per km<sup>2</sup>. For closed reserves this was straightforward, using the total number of waterpoints, total river length, and total area within each reserve. For the open reserves, we accounted for changes in surface water density once the fences were removed and herbivorous animals had access to surrounding reserves.

Since 15 km is the suggested minimal spacing of waterpoints due to herbivore movement (Owen-Smith, 1996), we applied a 15 km buffer to open reserves and calculated waterpoint and river density for all open reserves within the buffer.

### **3.3.2.7. Ownership**

Reserves were split into two ownership categories: Private and State. State-owned reserves included those managed by Mpumalanga Tourism and Parks Authority (MTPA), the Limpopo Economic Development, Environment, and Tourism Department (LEDET), or the South African National Parks (SANParks) (Marisa Coetzee, pers. comm.).

### **3.3.3. Statistical Analysis**

#### **3.3.3.1. Comparison of Vegetation Before/After Fence Removal**

We compared the vegetation within each reserve to the KNP vegetation before and after each reserve removed its fence. We used three time periods relative to fence removal: closed, transition, and open. The transition period included the first five years after a fence was removed from a reserve, connecting it to the Greater Kruger Ecosystem. Previous studies found that it took time for animals to adjust to newly connected reserves, and that five years was roughly the amount of time it took for the ecosystem to be fully impacted by the removal of a fence (Hiscocks, 1999; Druce et al., 2008; de Boer et al., 2015). The years incorporated in the transition period for each reserve were therefore not included in the closed/open comparison. Although some reserves removed fences bordering each other before connecting to the KNP landscape, for this analysis a fence was not considered removed until the reserve became part of the Greater Kruger Ecosystem.

We performed Welch's two-sample t-test to determine if the vegetation in each reserve was significantly different than KNP for the years before fence removal as well as after fence removal for the four TIMESAT vegetation metrics (Max, Amp, LI, and SI). When comparing the average vegetation metrics (hereafter average), the continuous 30-year NDVI dataset was used that included both AVHRR and MODIS sensors. When comparing the standard deviation of the vegetation metrics (hereafter standard

deviation), we used only one sensor to compare the before/after changes per reserve to ensure that the resolution difference (1.1km for AVHRR, 250m for MODIS), did not influence the results. The t-test analysis used AVHRR for the years 1985 to 2006 for the reserves that removed their fences during the 1990's. The t-test analysis used MODIS for the years 2000 to 2015 for the reserves that removed their fences after 2000.

### **3.3.3.2. Evaluation of Factors Associated with Changes in Vegetation**

Since we wanted to compare vegetation within the reserves to KNP, we calculated the average of the difference between the reserves and KNP (hereafter, average difference) and the standard deviation of the difference between reserves and KNP (hereafter, standard deviation of the difference) for each vegetation index. We used the reserve-KNP difference for each TIMESAT metric as response variables in linear mixed-effects models. We compared several random effects and their structures, and determined via model selection and likelihood ratio tests that the model with the most support included a global intercept, random effect intercepts for Reserve, and the correlation between intercept deviations and Fence Status deviations across levels of Reserve. We used a Pearson's correlation coefficient matrix to determine correlations between all explanatory variables (Appendix C). For all analyses we only included explanatory variables that were not correlated with one another.

We had three steps in our linear mixed-effect model process. In step 1, we created *a priori* models, each representing hypotheses of factors associated with changes in vegetation over time (Appendix D). Due to the complexity of the South African savanna landscape, none of our *a priori* models had greater support than the global models for each response variable. See Appendix E for the fits of the *a priori* models assessed using Akaike's weights constructed from Akaike's information criterion corrected for small sample sizes (AICc) (Burnham & Anderson, 2002) for each response variable. We then used the R package "MuMIn" (Bartón, 2010) to create and rank models of all possible combinations of explanatory variables for the second and third steps of the model selection process. In step 2 we fit models that contained each explanatory variable without any interaction terms for all four response variables. In step 3, we added an interaction term with Fence Status to all explanatory variables that were

retained in the most-supported models from step 2, and re-fit the models using the “MuMIn” package. We used the most-supported model from step 3 as the overall most-supported model for each response variable. This process was completed for the average of each of the four response variables as well as the standard deviation of each of the four response variables. All statistical analyses were performed in R Studio 3.3.1 (R Core Team, 2016; RStudio Team, 2016). We used the “lmer” function from the lme4 package in R (Bates et al., 2007) to fit all linear mixed-effects models. In order to estimate the significance of the fixed effects we used the R package “lmerTest,” which calculates p-values based on Satterthwaite’s approximations (Kuznetsova et al., 2016). It should be noted that there is disagreement on the best way to take into account random effects when determining the significance of fixed effects in mixed-effects models (Pinheiro & Bates, 2000).

### **3.4. Results**

#### **3.4.1. Changes in Vegetation Before/After Fence Removal**

All ten reserves that removed their fences had statistically similar averages compared to KNP before fence removal for all four vegetation response variables. Eight of the ten reserves that opened to the KNP had significantly different averages compared to KNP after fence removal for at least one of the four response variables (Table 3.1). Of those eight reserves, five reserves had lower average Max after fence removal, and three reserves had higher average Max after fence removal. Five reserves had lower average Amp after fence removal. Five reserves had different average LI compared to KNP after fence removal, with the metric decreasing in three reserves and increasing in two reserves. Two reserves had lower average SI compared to KNP after fence removal (Table 3.1).

Six of the eight reserves that remained closed during the study period significantly differed in averages compared to KNP for at least one of the four response variables (Table 3.1). Five reserves had higher average Max and one reserve had lower average Max. One reserve had higher average Amp and another reserve had lower average Amp compared to KNP. Three reserves had higher average LI while one reserve had lower average LI compared to KNP. Two closed reserves had higher average SI

compared to KNP (Table 3.1). See Appendix F for full summary statistics of the t-tests comparing the averages between the reserves and KNP.

All ten reserves that removed their fences had significantly different standard deviation compared to KNP for at least one of the four response variables (Table 3.2). In all cases of significant differences, reserves had lower standard deviation compared to KNP. Nine reserves had similar Max standard deviation compared to KNP before fence removal. Six reserves had lower Max standard deviation post-fence removal and one reserve had lower Max standard deviation both before and after fence removal. Nine reserves had similar Amp standard deviation compared to KNP before fence removal. Six reserves had lower Amp standard deviation post-fence removal and one reserve had lower Amp standard deviation both before and after fence removal. Nine reserves had similar LI standard deviation compared to KNP before fence removal. Eight reserves had lower LI standard deviation after fence removal, while one reserve had lower LI standard deviation before fence removal but similar LI standard deviation to KNP after fence removal. Eight reserves had similar SI standard deviation compared to KNP before fence removal. Seven reserves had lower SI standard deviation post-fence removal, one reserve had lower SI standard deviation before and after fence removal, and one reserve had lower SI standard deviation before fence removal with similar SI standard deviation to KNP after fence removal (Table 3.2).

Seven of the eight reserves that remained closed during the study period had significantly different standard deviation compared to KNP for at least one of the four response variables (Table 3.2). The same five closed reserves had lower Max, LI, and SI standard deviation, while seven closed reserves had lower Amp standard deviation (Table 3.2). See Appendix F for full summary statistics of the t-tests comparing standard deviation between the reserves and KNP.

### **3.4.2. Factors Associated with Vegetation Changes**

Across all four response variables, the most-supported model for average differences contained nearly all variables included in the analysis (i.e., the global model; Tables 3.3 – 3.6). The AICc tables displaying the top ten models for the average differences can be found in Appendix G.

We found a similar pattern for vegetation variability in that the most supported model for standard deviation of the differences contained nearly all variables included in the analysis (i.e., the global model; Tables 3.7 – 3.10). The AICc tables displaying the top ten models for the standard deviation of the differences can be found in Appendix G.

Since waterpoint density and fence status are the only explanatory variables that management can manipulate, we created interaction plots for those variables for the average and standard deviation of all four vegetation metrics. We also plotted the interactions for all explanatory variables that had statistically significant fence status interactions. The y-axis of the interaction plots illustrates the difference in vegetation metrics between the reserves and KNP. As a reference, KNP's vegetation is represented by the black dotted line at zero.

#### *Surface Water*

In general, the averages in both closed and open reserves decreased as waterpoint density increased (Figure 3.2). Open reserves had mostly lower averages compared to closed reserves at higher waterpoint densities and higher averages at lower waterpoint densities. The difference in the averages between closed and open reserves increased as waterpoint density increased. Reserves in the transition period had higher averages for higher waterpoint densities (Figure 3.2). The standard deviation within the reserves responded with the same pattern to waterpoint density (Figure 3.3).

River density exemplified the same pattern for averages for closed and open reserves as waterpoint density (Figure 3.4). However, reserves in the transition period had decreasing averages as river density increased, and Amp standard deviation increased in open reserves as river density increased (Figure 3.4). For both waterpoint and river densities, the temporal changes in vegetation metrics within open reserves had much steeper slopes compared to closed reserves (Figures 3.2-3.4).

#### *Geology*

Open reserves had lower averages than closed reserves on gabbro geology (Figure 3.5). The difference between closed and open reserves was greater for reserves with higher percentages of gabbro

rock. In both cases, the transition period had lower averages compared to the closed period, but averages increased with higher percentages of gabbro rock (Figure 3.5).

### *Seasonality*

For Max and Amp, open reserves had lower averages than closed reserves when the beginning of the growing season was earlier in the year and higher averages when the beginning of the growing season was later in the year (Figure 3.6). The closed reserves had steeper slopes compared to open reserves which had mostly flat slopes. The temporal vegetation metric changes during the transition period were in the opposite direction compared to the closed reserves for average Max and Amp. Open reserves had lower Amp standard deviation compared to closed reserves. The transition pattern was similar to that from the averages (Figure 3.6). For both average Amp and LI standard deviation the open reserves had lower vegetation metrics than closed reserves with shorter growing seasons and higher vegetation metrics with longer growing seasons. However, the reserves changed differently during the transition period, showing a decrease in averages as the length of the growing season increased. While closed reserves continued to illustrate a negative slope as the length of the growing season increased for standard deviation, open reserves had positive slopes (Figure 3.7).

### *Rainfall Lag 3*

Open reserves had lower Amp standard deviation and had higher SI standard deviation compared to closed reserves (Figure 3.8). The slopes for both closed and open reserves were positive, with the standard deviation increasing as rainfall increased. Reserves in the transition period changed in the opposite direction, with the standard deviation decreasing as rainfall increased (Figure 3.8).

## **3.5. Discussion**

The goals of this study were to determine if fence removal across reserves impacted vegetation dynamics and what factors contributed to these changes. In general, we found that fence removal was associated with a decrease in the average and standard deviation of TIMESAT derived vegetation metrics within the reserves. We used vegetation metric standard deviation as a proxy for vegetation variability, although it should be noted that it may not necessarily relate 1:1 with ecological heterogeneity. Short-term

vegetation changes within the five-year transition period varied greatly, but there were clear patterns in the long-term vegetation changes during the open period. Long-term decreases in the vegetation metrics were associated with geology containing soils with high clay-content, waterpoint density, and river density. Long-term increases in the vegetation metrics were associated with rainfall in the growing season three years previous.

### **3.5.1. Changes in Vegetation Before/After Fence Removal**

When comparing the average vegetation metrics within the open reserves to KNP, all significant changes that occurred showed that the reserves were similar to KNP before fence removal and became dissimilar post-fence removal. There were more changes with the Max (8 out of 10 comparisons) compared to the Amp (5 out of 10 comparisons), LI (5 out of 10 comparisons), and SI (2 out of 10 comparisons). Since studies have suggested that Max is influenced by vegetation composition while LI and SI are related to herbaceous biomass (Olsson et al., 2005; Wessels et al., 2011; Mbow et al., 2013; Olsen et al., 2015), these findings suggest that fence removal influenced vegetation composition more than herbaceous biomass. Future work is needed to better distinguish the ecological difference between the TIMESAT metrics in order to interpret the ecological significance and meaning of these vegetation metrics in greater detail. In most cases, reserves that became dissimilar to KNP after fence removal decreased in average vegetation metrics. These changes are likely due to an influx of herbivores that occurred once the reserves became open to the KNP ecosystem, which may have been a result of the higher waterpoint densities within the reserves compared to KNP (Child et al., 2013). Many studies have found herbivores influence woody plant diversity and density (Wigley et al., 2014), species richness (Burkepile et al. 2016), and overall plant species composition (Levick & Rogers, 2008; Wigley et al., 2014; Burkepile et al., 2016). Browsers can shift vegetation species composition (Augustine & Mcnaughton, 2004; Shannon et al., 2011; Levick & Asner, 2013; Wigley et al., 2014). Changes in the browser: grazer ratios within the open reserves may possibly contribute towards changes in vegetation composition.

Three open reserves, O8, O9, and O10, increased in average vegetation metrics. Different factors may have played a role in vegetation changes within these reserves compared to the others, since O8, O9, and O10 are all adjacent to one another and further south than the rest of the reserves. One possible factor is the higher rainfall in the southern sections of the study area where O8, O9, and O10 are located (Venter et al., 2003). Stevens et al. (2016) found that conservation areas encompassing elephants at higher rainfall gradients experienced greater levels of increasing woody cover (i.e. woody encroachment/densification) compared to lower rainfall areas which showed less evidence of increases in woody cover over time. Similarly, Asner and Levick (2012) found elephants caused higher treefall rates in drier landscapes. An increase in elephant numbers within the reserves with lower rainfall levels may therefore have resulted in a decrease in woody cover, while the higher rainfall levels in O8, O9, and O10 may have caused herbaceous biomass to remain at higher levels or increase with reduced woody cover (Smit & Prins, 2015). This reveals the complexity of the savanna ecosystem and suggests that management regimes regarding factors such as elephants need to consider the influence of abiotic factors, since open reserves appear to respond differently to fence removal depending on where they fall on the rainfall gradient.

In contrast to the majority of the open reserves experiencing a decrease in average vegetation metrics, most closed reserves that had significantly different average vegetation metrics to KNP had higher averages. A potential reason for this is that managers within closed reserves are likely to be very attentive to herbivore carrying capacity numbers, and would likely respond to any “unwanted” decreases in vegetation by adjusting grazer and browser numbers appropriately (i.e. “conservative” stocking). This differs from an open reserve, which has little control over which animals enter the property.

In most cases, reserves had statistically similar vegetation metric variation compared to KNP before fence removal, and had significantly less vegetation metric variation after fence removal. The decrease in vegetation metric variation is likely due to increased herbivore numbers within the open reserves, which appears to have homogenized the vegetation. Browsers impact woody vegetation structure at both community and species scales (Levick & Rogers, 2008). Elephants in particular influence woody canopy heights, altering the vertical structure and therefore the structural heterogeneity

of the ecosystem (Asner et al., 2009; Fisher et al., 2014; Hilbers et al., 2015). An increase in elephant numbers along with other herbivores may have contributed to the changes in vegetation composition and biomass.

One reserve had less vegetation metric variation before fence removal but similar variation to KNP after the fences were removed. O3 was historically fragmented, composed of numerous fenced land parcels. These properties slowly began to merge in the 1990's, and it was not connected to the KNP ecosystem until 2005 (Marisa Coetzee & Craig Spencer, pers. comm.). These findings likely show the increase in vegetation heterogeneity as the vegetation recovered from fragmentation in the recent-past. This vegetation change aligns with the findings from Boone and Hobbs (2004) that increased fragmentation among smaller land parcels exhibits less heterogeneity.

Two open reserves experienced both local- and landscape-scale changes. O7 had statistically different Max variation both before and after fence removal, while the LI and SI variations became less similar to KNP after fence removal. This suggests that local-scale management differences may have been occurring in O7 that made the vegetation composition different than KNP both before and after fence removal, while herbaceous biomass responded to landscape-scale differences (e.g. fence removal). O5 had different Amp and SI variation compared to KNP both before and after fence removal, but experienced a decrease in Max variation. This suggests local-scale management differences that influenced Amp and SI variation, while landscape-scale differences influenced Max variation. Although vegetation composition has been found to influence both Max and Amp, these findings show Max responded differently than Amp to fence removal within this reserve. Further investigation is needed to determine possible factors that may be influencing vegetation within the reserves on local- versus landscape-scales.

All closed reserves that had significantly different vegetation metric variation compared to KNP had lower vegetation metrics. These results are intriguing, since in most cases when the closed reserves differed from KNP in average vegetation metrics, they had higher metrics compared to KNP. Although managers of closed reserves may be aware of and respond to average amounts of vegetation as it directly

influences forage, they may not be as concerned with the heterogeneity of the reserves. This is an important differentiation, since low levels of vegetation heterogeneity make ecosystems more susceptible to extreme events such as drought (Walker et al., 1987).

### **3.5.2. Short-Term vs. Long-Term Vegetation Changes**

We considered two different time periods for reserve vegetation changes after fence removal: the initial vegetation change during the first five years post-removal (the transition period), and the long-term vegetation change that included all years after the initial transition period (the open period). Vegetation within the first five years post-fence removal may respond in three ways: 1) respond in the opposite direction that the open period ultimately shows, 2) foreshadow the vegetation change to the open period by responding in the same direction but not as extreme, or 3) respond in the same direction as the open period but more extreme. Therefore, the reserve managers may see a range of vegetation changes in the first five years post-fence removal, but this signal was not a reliable indication of the long-term vegetation impact. For one, the transition period is shorter compared to the open period, making the results from the transition period less statistically robust. In addition, there was a severe drought in 1991/1992 that coincided with the time that several reserves removed their fences. The transition period may therefore reflect a post-drought response rather than a response from the fence removal. Also, initial changes in herbivore abundance may be a major cause of vegetation changes during the transition period. de Boer et al. (2015) found that the vegetation within one of the reserves adjacent to the KNP was severely impacted by the increase in elephant numbers for about five years. After five years the vegetation seemed to still be impacted, although not as severely. In addition to the vegetation changes, de Boer et al (2015) found that the influx of elephants shifted the composition of the herbivore community. Since there are factors constantly at flux, such as rainfall and temperature, that influence herbivore populations (Ogutu & Owen-Smith, 2003; Gandiwa et al., 2016), it makes sense that the initial impact on vegetation would be highly variable.

### **3.5.3. Factors Associated with Vegetation Changes**

## *Surface Water*

As waterpoint density increased, the average and standard deviation of all vegetation metrics decreased in open reserves. The difference between closed and open reserves also became more extreme as waterpoint density increased. These findings suggest that once reserves removed their fences and became open to the Greater Kruger Ecosystem, there was an influx of animals into the open reserves to take advantage of this increased water supply. The open reserves have higher waterpoint densities than KNP (Child et al., 2013), and waterpoints are known to have a strong influence on herbivore distribution (Owen-Smith, 1996; Chamaillé-Jammes et al., 2007; Smit et al., 2007a; Smit, 2011). Artificial waterpoints can result in unnaturally high pressure on the surrounding vegetation, with abundant water subjecting larger amounts of the landscape to high herbivore impact (Owen-Smith, 1996; Thrash, 1998; Chamaillé-Jammes et al., 2007). The area immediately surrounding a waterpoint is known as the “sacrifice zone,” which contains the highest amount of herbivore impact through trampling, preventing woody plant seed survival (Walker et al., 1987; Parker & Witkowski, 1999; Thrash, 2000; Brits et al., 2002). Unnaturally close spacing between waterpoints causes adjacent impacted zones to overlap and increases starvation during drought periods (Walker et al., 1987; Owen-Smith, 1996). Waterpoints therefore need be spaced appropriate distances in order to increase grazing variation and ecosystem resilience (Smit et al., 2007a). Owen-Smith (1996) suggested minimal waterpoint spacing of 15km in order to reduce herbivore pressure. Closure of artificial water sources has shown to be related to higher vegetation biomass, palatable species, and structural heterogeneity (Grant et al., 2002; Hilbers et al., 2015). Removing artificial waterpoints is therefore encouraged in order to increase the spatial and temporal heterogeneity of the landscape (Redfern et al., 2005).

Our results relating fence removal to waterpoint density support the findings modeled from Graz et al. (2012). When predicting the outcome of retaining high levels of waterpoints with fence removal, Graz et al. (2012) found that the grazing activity spread across the landscape. This increased accessibility to several waterpoints, expanding grazing impact. In contrast, the scenario that modeled the closure of some waterpoints with fence removal resulted in the two management changes counterbalancing one-

another. In this scenario, there was a spatial expansion of grazing, with some areas experiencing an increase in herbivory while other areas experienced a decrease (Graz et al., 2012). We agree with their conclusion that due to the impacts that both fencing and waterpoint have on animal movement, these management regimes need to be jointly considered.

For the most part, higher river densities had similar impacts on vegetation within open and closed reserves compared to waterpoint densities, with closed reserves being unaffected by the river density gradient and vegetation metrics mostly decreasing in open reserves. Again, this may be due to an influx of animals from the KNP ecosystem utilizing the rivers within the open reserves. In particular, mixed elephant herds prefer rivers compared to waterpoints, possibly because of the extra resources provided such as shade and forage (Smit et al., 2007b; Smit & Ferreira, 2010). The increase in elephant numbers into the reserves adjacent to KNP may therefore be the main reason high river densities were impacted in open compared to closed reserves. The exception to this pattern is the increase in Amp standard deviation with higher river densities in open reserves. This suggests that even with the influx of herbivores into open reserves, vegetation composition may have been influenced differently across various river densities. Further study is needed to better understand this exception.

### *Geology*

Gabbro rock has higher clay content and nutrients compared to granite, which dominates the reserves (Venter et al., 2003). Average vegetation metrics decreased in open reserves and increased in closed reserves as gabbro rock percentage increased. Geology may have had a greater impact on open compared to closed reserves because animals have more foraging options and may therefore have sought out areas with high clay-content. Since herbivores are known to prefer vegetation associated with clay-like soils (Whyte & Joubert, 1988; Thrash, 2000), they likely selected areas with higher proportions of these underlying rock types. Elephants have been found to prefer gabbro soil types and have a proportionally higher impact on trees in areas with high-nutrient soils (Shannon et al., 2008; Asner & Levick, 2012). The impact on average vegetation metrics by herbivory and tree destruction by elephant likely increases in areas with high proportions of clay-like soils.

### *Seasonality*

Findings related to the beginning and length of the growing season provide insight into how seasonal grazing pressure differed within closed and open reserves. The vegetation metrics within the closed and open reserves generally responded differently to the beginning and the length of the growing season. The average vegetation metrics within open reserves did not seem to be influenced by the beginning of the growing season, as indicated by level slopes, while the average vegetation metrics decreased in the closed reserves as the beginning of the growing season was later in the year. This may be due to herbivores within closed reserves consuming forage faster than it could regrow, whereas herbivores within open reserves had the ability to move in response to different forage levels across the landscape. Further investigation is required to determine possible landscape-scale factors that be contributing to these differences.

### *Rainfall Lag 3*

Similar to other studies that used rainfall in the previous season (Thrash, 2000) or rainfall from the actual year, 2-year, and year-3 running averages (Peel et al., 2005), our analysis maintained the interaction between rainfall lag 3 and fence status in several of most-supported models. Rainfall lag 3 without an interaction with fence status was significant in all four vegetation metric standard deviation models, although not significant with the fence status interaction terms. This suggests that vegetation metric variation may respond differently to rainfall in KNP compared to the reserves, but not differently when comparing closed and open reserves. Both closed and open reserves had lower vegetation variation compared to KNP with low levels of rainfall lag 3. It appears that the vegetation within both closed and open reserves may be more susceptible to lower rainfall levels compared to KNP, possibly making the ecosystem less resilient. Further investigation is required as to why the vegetation metric variation within KNP may be responding differently to rainfall compared to smaller reserves.

#### **3.5.4. Additional Potential Factors**

Although elephants are known to influence vegetation, we were not able to include changes in elephant densities in our model because we lacked consistent data across reserves. It is known that

elephant numbers increased in reserves adjacent to the KNP once fences were removed, with de Boer et al. (2015) documenting that elephant numbers increased 17-fold in one reserve after fence removal between 1992 and 2011. It must be noted that elephant culling stopped during 1994 in KNP and hence their numbers also increased in KNP (Smit, 2011). However, because of the high water provision in the areas adjacent to KNP, it is postulated that the elephant densities in the adjacent reserves went up disproportionately due to higher high waterpoint densities. Elephants are known to be the key determinant of treefall in savannas, with treefall rates increasing 6-fold in areas containing elephants versus areas where elephants were excluded (Asner & Levick, 2012). Asner and Levick (2012) found that elephants had the largest impact on trees that were 5-9m in height, although the model from Hilbers et al. (2015) suggested that elephants altered the woody biomass across all heights classes. Along with elephants' impact on vegetation, they also have cascading effects on the distribution of other herbivores (Hilbers et al., 2015), shifting herbivore communities to become grazer-dominated through competition with other browsers (de Boer et al., 2015). Elephants also impact other herbivores through habitat modification, with browsers and mixed feeders selecting sites where vegetation was uprooted by elephants (Valeix et al., 2011). Shrader et al. (2010) found that the combination of fences and artificial waterpoints caused elephants to have a negative impact on vegetation and additional species during dry years. It is therefore important to consider the influence of elephants on vegetation and other species when interpreting the interaction between fence removal and waterpoint densities.

In addition to elephants, we were unable to include fire within our model, which is known to modify savanna vegetation because we lacked consistent data across reserves. The impacts of fire on vegetation have been related to herbivory (Bond & Keeley, 2005), with fire disturbance influencing tree-grass coexistence (Venter et al., 2003). Full exclusion of fire causes an unnaturally high growth of woody biomass (Venter et al., 2003). In addition to the direct impact fire has on vegetation, it has complex interactions with elephants. The probability of tree mortality by elephants increases after being disturbed by fire (Shannon et al., 2011). Fire management regimes differ across the reserves and KNP (Child et al., 2013), which therefore may have influenced vegetation differently in the various protected areas.

However, rainfall has been found to have a larger influence on fire extent than management policy (van Wilgen et al., 2004; Smit et al., 2013). This suggests that differing fire management regimes across reserves may not cause majorly different fire extents. Future studies are needed to determine the potential impacts of elephants and fire on the vegetation after fence removal across reserves.

### **3.5.5. Importance of Heterogeneity**

In order to effectively maintain spatial and temporal heterogeneity, reserve managers must allow savanna ecosystems to undergo their natural complex fluctuations (Rogers, 2003). Landscape heterogeneity creates microhabitats and subsequent niches which maintains higher levels of biodiversity (Menge & Sutherland, 1976; Lindenmayer & Fischer, 2006). Our findings that fence removal decreased vegetation metric variation is concerning, as landscape homogenization decreases potential microhabitats, therefore limiting the amount of biodiversity the system can retain (Urban et al., 1987).

In addition to increasing potential biodiversity, heterogeneity improves the resilience of the ecosystem from severe natural disturbances. For example, when water becomes unnaturally abundant, species that are dependent on water are favored and become uniformly distributed during all seasons (Walker et al., 1987; Owen-Smith, 1996). This homogenization of vegetation and herbivore species increases the susceptibility to drought-related mortality, as shown in the 1982 drought within a formerly-fenced reserve (Walker et al., 1987). This reserve endured higher mortality rates of ungulate species and woody plants compared to the KNP and required five years to recover compared to the KNP, which was able to recapture previous conditions within one to two years (Walker et al., 1987). It is thought that along with immobility from the fences, the extreme mortalities within the reserve were due to the high artificial waterpoint densities (Walker et al., 1987).

### **3.5.6. Management Implications**

Our primary finding that fence removal reduced vegetation metric variability within the reserves compared to KNP has potential implications for reserve managers. While many factors played a role including river density, geology, seasonality, and rainfall, only waterpoint density can be managed on

local-scales. Artificial waterpoints likely contributed to increased densities of water-dependent herbivores, especially elephant, within the open reserves. Both fences and artificial water have been shown to decrease the seasonal movement of elephants, increasing the impact on the vegetation (Loarie et al., 2009). Since herbivores influence the spatial heterogeneity within savanna ecosystems (de Knecht et al., 2008), it is vital to consider management regimes that will influence their movement across the landscape. We propose that managers within reserves connected to the Greater Kruger Ecosystem strongly consider reducing the density of artificial water sources in order to produce conditions conducive for restoring and maintaining habitat heterogeneity.

### **3.6. Conclusion**

Landscape heterogeneity increases the potential for biodiversity within savanna ecosystems. Fragmentation through fencing is known to decrease habitat heterogeneity and causes systems to become less resilient to change. We used historic satellite imagery to detect the impacts of fence removal on vegetation dynamics within reserves adjacent to KNP. We found evidence that suggests vegetation metric variation within reserves decreased compared to KNP post-fence removal. We postulate that a possible reason for this homogenization was the artificially high waterpoint densities within the reserves, which may have caused an influx of animals, especially water dependent species like elephants, thus leading to increased herbivory. Within the K2C, different reserves have varying management objectives, ranging from conservation to ecotourism. While tempting to maintain wide-scale artificial surface water to attract big game and to act as tourism viewpoints, this practice appears to have decreased vegetation metric variability within the reserves of the Greater Kruger Ecosystem, which may render these areas less resilient during extreme events such as drought. Management regimes need to consider the larger picture in order to maintain habitat heterogeneity and ecosystem resilience.

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## CHAPTER 3 TABLES

**Table 3.1.** The t-test results comparing the average vegetation metrics within the reserves to KNP.

Reserve	Fence Status	Max		Amp		LI		SI	
		p-value	Direction	p-value	Direction	p-value	Direction	p-value	Direction
O1	Closed	0.712	-	0.904	+	0.913	+	0.882	+
	Open	<b>0.007</b>	-	<b>0.009</b>	-	0.205	-	0.111	-
O2	Closed	0.772	+	0.538	+	0.957	-	0.888	+
	Open	<b>0.014</b>	-	<b>0.041</b>	-	<b>0.032</b>	-	<i>0.052</i>	-
O3	Closed	0.159	-	0.464	-	0.327	-	0.452	-
	Open	0.217	-	0.377	-	0.725	-	0.687	-
O4	Closed	0.338	-	0.527	-	0.672	-	0.719	-
	Open	<b>0.002</b>	-	<b>0.036</b>	-	<b>0.008</b>	-	<b>0.008</b>	-
O5	Closed	0.300	-	0.335	-	0.799	-	0.708	-
	Open	<b>&lt;0.001</b>	-	<b>0.001</b>	-	<b>0.008</b>	-	<b>0.002</b>	-
O6	Closed	0.583	-	0.662	-	0.811	-	0.842	-
	Open	<b>0.004</b>	-	<b>0.044</b>	-	<i>0.080</i>	-	<i>0.059</i>	-
O7	Closed	0.654	+	0.702	+	0.826	+	0.838	+
	Open	0.986	+	0.650	-	0.667	+	0.926	-
O8	Closed	0.153	+	0.277	+	0.574	+	0.594	+
	Open	<b>0.034</b>	+	0.178	+	0.143	+	0.183	+
O9	Closed	0.123	+	0.256	+	0.382	+	0.523	+
	Open	<b>0.028</b>	+	0.453	+	<b>0.036</b>	+	0.148	+
O10	Closed	0.105	+	0.209	+	0.502	+	0.567	+
	Open	<b>0.005</b>	+	0.230	+	<b>0.027</b>	+	<i>0.070</i>	+
C1	Closed	<b>&lt;0.001</b>	+	<b>0.010</b>	+	<b>0.017</b>	+	<b>0.024</b>	+
C2	Closed	<b>0.014</b>	+	0.375	+	<i>0.074</i>	+	0.233	+
C3	Closed	<b>&lt;0.001</b>	-	<b>0.025</b>	-	<b>0.040</b>	-	0.107	-
C4	Closed	0.225	+	0.855	+	0.111	+	0.377	+
C5	Closed	0.263	+	0.822	+	0.209	+	0.513	+
C6	Closed	<b>0.011</b>	+	0.336	+	0.226	-	0.256	+
C7	Closed	<b>0.002</b>	+	0.796	+	<b>0.009</b>	+	0.574	+
C8	Closed	<b>&lt;0.001</b>	+	0.331	+	<b>&lt;0.001</b>	+	<b>0.039</b>	+

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.2.** The t-test results comparing the vegetation metric standard deviation within the reserves to KNP.

Reserve	Fence Status	Max		Amp		LI		SI	
		p-value	Direction	p-value	Direction	p-value	Direction	p-value	Direction
O1	Closed	0.102	-	0.214	-	0.265	-	0.202	-
	Open	<b>0.014</b>	-	<b>0.005</b>	-	<b>&lt;0.001</b>	-	<b>0.002</b>	-
O2	Closed	0.235	-	0.408	-	0.524	-	0.342	-
	Open	0.554	-	0.583	-	<b>0.001</b>	-	<b>&lt;0.001</b>	-
O3	Closed	0.775	+	0.552	-	<b>0.020</b>	-	<b>0.019</b>	-
	Open	0.920	-	0.176	-	0.836	-	0.197	-
O4	Closed	0.396	-	0.277	-	0.624	-	0.379	-
	Open	<b>0.007</b>	-	<b>0.005</b>	-	<b>0.002</b>	-	<b>0.005</b>	-
O5	Closed	<i>0.076</i>	-	<b>0.015</b>	-	0.136	-	<b>0.023</b>	-
	Open	<b>0.002</b>	-	<b>0.002</b>	-	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-
O6	Closed	0.835	-	0.645	-	0.369	-	0.165	-
	Open	<i>0.081</i>	-	<b>0.031</b>	-	<i>0.066</i>	-	<i>0.073</i>	-
O7	Closed	<b>0.040</b>	-	<i>0.094</i>	-	0.147	-	0.143	-
	Open	<b>0.044</b>	-	0.153	-	<b>&lt;0.001</b>	-	<b>0.007</b>	-
O8	Closed	0.179	-	0.142	-	<i>0.090</i>	-	<i>0.061</i>	-
	Open	<b>0.050</b>	-	<b>0.034</b>	-	<b>0.030</b>	-	<b>0.023</b>	-
O9	Closed	0.231	-	0.423	-	0.852	+	0.853	-
	Open	<b>0.007</b>	-	<b>0.031</b>	-	<b>0.021</b>	-	<b>0.010</b>	-
O10	Closed	0.284	-	0.127	-	0.766	-	0.415	-
	Open	<b>0.001</b>	-	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-
C1	Closed	0.425	-	0.538	-	0.359	-	0.101	-
C2	Closed	0.133	-	<b>0.042</b>	-	0.691	-	0.211	-
C3	Closed	<b>0.005</b>	-	<b>&lt;0.001</b>	-	<b>0.012</b>	-	<b>&lt;0.001</b>	-
C4	Closed	0.433	-	<b>0.043</b>	-	0.543	-	0.158	-
C5	Closed	<b>0.003</b>	-	<b>0.006</b>	-	<b>0.035</b>	-	<b>&lt;0.001</b>	-
C6	Closed	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-	<b>0.004</b>	-	<b>&lt;0.001</b>	-
C7	Closed	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-	<b>0.009</b>	-	<b>&lt;0.001</b>	-
C8	Closed	<b>&lt;0.001</b>	-	<b>&lt;0.001</b>	-	<b>0.003</b>	-	<b>&lt;0.001</b>	-

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.3.** The most-supported model for average Max differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	2,589.00	1,982.00	8.90	1.31	0.224
Amphibolitic	498.60	1,518.00	7.70	0.33	0.751
Aspect	-3.74	7.82	8.20	-0.48	0.645
BegMean	-5.03	0.96	451.30	-5.24	<b>&lt;0.001</b>
Elevation	2.85	1.91	7.70	1.50	0.175
FenceStatusOpen	-93.05	1,584.00	7.20	-0.06	0.955
FenceStatusTransition	-1,765.00	2,223.00	32.10	-0.79	0.433
Gabbro	4,499.00	3,958.00	7.80	1.14	0.289
LengthMean	-2.30	0.54	452.20	-4.29	<b>&lt;0.001</b>
OwnershipState	10.50	301.00	4.10	0.04	0.974
RainfallLag1	-0.31	0.08	451.10	-3.75	<b>&lt;0.001</b>
RiverDensity	-4,036.00	2,819.00	13.30	-1.43	0.175
WaterpointDensity	-1,171.00	987.80	9.70	-1.19	0.264
Amphibolitic:FenceStatusOpen	-180.40	2,294.00	6.40	-0.08	0.940
Amphibolitic:FenceStatusTransition	-936.30	2,780.00	36.80	-0.34	0.738
Aspect:FenceStatusOpen	-6.29	5.31	3.10	-1.18	0.320
Aspect:FenceStatusTransition	0.63	7.28	15.50	0.09	0.932
BegMean:FenceStatusOpen	4.23	1.76	451.20	2.40	<b>0.017</b>
BegMean:FenceStatusTransition	9.47	3.43	414.00	2.76	<b>0.006</b>
Elevation:FenceStatusOpen	4.45	4.18	4.50	1.07	0.340
Elevation:FenceStatusTransition	-0.22	3.91	128.70	-0.06	0.954
FenceStatusOpen:Gabbro	-7,836.00	2,748.00	3.10	-2.85	<i>0.062</i>
FenceStatusTransition:Gabbro	-2,417.00	3,159.00	6.20	-0.77	0.472
FenceStatusOpen:LengthMean	1.64	1.06	452.20	1.55	0.121
FenceStatusTransition:LengthMean	-4.13	2.24	410.40	-1.85	<i>0.066</i>
FenceStatusOpen:OwnershipState	-279.10	331.70	6.60	-0.84	0.429
FenceStatusTransition:OwnershipState	88.74	338.90	10.60	0.26	0.798
FenceStatusOpen:RiverDensity	-12,920.00	4,969.00	17.60	-2.60	<b>0.018</b>
FenceStatusTransition:RiverDensity	-6,087.00	5,294.00	197.90	-1.15	0.252
FenceStatusOpen:WaterpointDensity	-1,477.00	3,652.00	2.60	-0.40	0.717
FenceStatusTransition:WaterpointDensity	3,388.00	3,699.00	22.60	0.92	0.369

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.4.** The most-supported model for average Amp differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	1,844.00	1,259.00	12.00	1.47	0.169
Amphibolitic	541.20	887.40	7.90	0.61	0.559
Aspect	-2.16	4.79	10.10	-0.45	0.662
BegMean	-3.69	0.89	457.30	-4.16	<b>&lt;0.001</b>
Elevation	1.26	1.12	8.20	1.12	0.293
FenceStatusOpen	-170.70	1,240.00	66.80	-0.14	0.891
FenceStatusTransition	-1,649.00	2,052.00	13.60	-0.80	0.436
Gabbro	3,011.00	2,346.00	8.80	1.28	0.232
LengthMean	-1.72	0.49	457.40	-3.47	<b>0.001</b>
OwnershipState	-12.33	207.10	10.50	-0.06	0.954
RainfallLag1	-0.22	0.08	458.20	-2.82	<b>0.005</b>
RiverDensity	-1,945.00	1,793.00	10.70	-1.09	0.302
WaterpointDensity	-486.20	625.50	10.90	-0.78	0.454
Amphibolitic:FenceStatusOpen	-820.00	1,990.00	5.20	-0.41	0.697
Amphibolitic:FenceStatusTransition	-1,963.00	2,836.00	6.90	-0.69	0.512
Aspect:FenceStatusOpen	-6.35	3.92	67.60	-1.62	0.110
Aspect:FenceStatusTransition	-1.45	6.70	6.80	-0.22	0.835
BegMean:FenceStatusOpen	3.07	1.63	457.10	1.89	<i>0.060</i>
BegMean:FenceStatusTransition	7.58	3.20	390.70	2.37	<b>0.018</b>
Elevation:FenceStatusOpen	5.66	3.77	5.40	1.50	0.189
Elevation:FenceStatusTransition	0.35	3.90	13.90	0.09	0.929
FenceStatusOpen:Gabbro	-6,565.00	2,047.00	86.50	-3.21	<b>0.002</b>
FenceStatusTransition:Gabbro	-82.93	2,761.00	5.70	-0.03	0.977
FenceStatusOpen:LengthMean	1.90	0.97	457.50	1.95	<i>0.051</i>
FenceStatusTransition:LengthMean	-4.79	2.09	383.00	-2.29	<b>0.022</b>
FenceStatusOpen:OwnershipState	-306.90	254.20	46.10	-1.21	0.233
FenceStatusTransition:OwnershipState	91.37	279.90	239.50	0.33	0.744
FenceStatusOpen:RiverDensity	-11,380.00	4,284.00	9.80	-2.66	<b>0.024</b>
FenceStatusTransition:RiverDensity	1,511.00	5,153.00	15.80	0.29	0.773
FenceStatusOpen:WaterpointDensity	-3,023.00	3,257.00	3.60	-0.93	0.411
FenceStatusTransition:WaterpointDensity	1,057.00	3,782.00	10.10	0.28	0.785

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.5.** The most-supported model for average LI differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	1,058.00	1,917.00	12.40	0.55	0.591
Amphibolitic	178.80	1,125.00	8.70	0.16	0.877
Aspect	2.26	6.93	10.30	0.33	0.751
BegMean	-3.36	1.79	467.70	-1.88	<i>0.061</i>
Elevation	3.39	1.45	10.00	2.34	<b>0.042</b>
FenceStatusOpen	2,434.00	2,340.00	12.60	1.04	0.318
FenceStatusTransition	605.30	3,682.00	93.50	0.16	0.870
Gabbro	4,751.00	3,098.00	11.40	1.53	0.152
LengthMean	-4.84	0.99	469.00	-4.87	<b>&lt;0.001</b>
OwnershipState	271.40	360.50	9.60	0.75	0.470
RainfallLag1	-0.61	0.15	469.20	-3.96	<b>&lt;0.001</b>
RainfallLag2	0.49	0.15	468.00	3.25	<b>0.001</b>
RainfallLag3	-0.91	0.15	470.20	-5.88	<b>&lt;0.001</b>
RiverDensity	-717.90	2,453.00	9.80	-0.29	0.776
WaterpointDensity	-673.90	949.10	9.70	-0.71	0.494
Amphibolitic:FenceStatusOpen	-1,058.00	3,132.00	317.60	-0.34	0.736
Amphibolitic:FenceStatusTransition	-449.80	4,485.00	340.70	-0.10	0.920
Aspect:FenceStatusOpen	-1.60	7.77	5.40	-0.21	0.845
Aspect:FenceStatusTransition	3.05	11.89	32.00	0.26	0.799
BegMean:FenceStatusOpen	4.42	3.26	466.70	1.36	0.175
BegMean:FenceStatusTransition	1.28	6.31	462.70	0.20	0.839
Elevation:FenceStatusOpen	-0.14	6.04	236.40	-0.02	0.982
Elevation:FenceStatusTransition	-5.98	6.78	452.70	-0.88	0.378
FenceStatusOpen:Gabbro	-5,441.00	4,043.00	9.00	-1.35	0.211
FenceStatusTransition:Gabbro	-734.70	5,086.00	18.30	-0.14	0.887
FenceStatusOpen:LengthMean	-2.05	1.98	467.40	-1.03	0.302
FenceStatusTransition:LengthMean	0.93	4.11	458.60	0.23	0.822
FenceStatusOpen:OwnershipState	-256.60	466.80	27.00	-0.55	0.587
FenceStatusTransition:OwnershipState	192.70	556.40	28.80	0.35	0.732
FenceStatusOpen:RiverDensity	-18,550.00	6,757.00	77.60	-2.75	<b>0.008</b>
FenceStatusTransition:RiverDensity	-1,336.00	8,881.00	371.80	-0.15	0.881
FenceStatusOpen:WaterpointDensity	-835.60	4,847.00	93.30	-0.17	0.863
FenceStatusTransition:WaterpointDensity	3,697.00	6,202.00	267.90	0.60	0.552

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.6.** The most-supported model for average SI differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	-971.60	646.70	11.50	-1.50	0.160
Amphibolitic	358.90	391.90	8.90	0.92	0.384
Aspect	1.44	2.41	10.50	0.60	0.562
BegMean	0.69	0.57	467.10	1.20	0.233
Elevation	1.17	0.50	10.10	2.33	<b>0.042</b>
FenceStatusOpen	1,534.00	800.90	10.00	1.92	<i>0.084</i>
FenceStatusTransition	861.50	1,238.00	194.60	0.70	0.488
Gabbro	1,819.00	1,078.00	11.60	1.69	0.118
OwnershipState	152.50	124.90	9.60	1.22	0.251
Rainfall	0.21	0.05	466.80	4.06	<b>&lt;0.001</b>
RainfallLag3	-0.25	0.05	467.90	-4.73	<b>&lt;0.001</b>
RiverDensity	-245.10	853.90	10.10	-0.29	0.780
WaterpointDensity	152.10	329.10	9.90	0.46	0.654
Amphibolitic:FenceStatusOpen	-1,143.00	1,089.00	119.70	-1.05	0.296
Amphibolitic:FenceStatusTransition	-863.20	1,543.00	66.30	-0.56	0.578
Aspect:FenceStatusOpen	-3.23	2.79	5.40	-1.16	0.296
Aspect:FenceStatusTransition	-1.52	4.02	57.20	-0.38	0.706
BegMean:FenceStatusOpen	1.22	1.10	465.40	1.11	0.268
BegMean:FenceStatusTransition	0.24	2.16	450.10	0.11	0.910
Elevation:FenceStatusOpen	1.90	2.09	254.60	0.91	0.366
Elevation:FenceStatusTransition	-1.06	2.38	165.30	-0.44	0.658
FenceStatusOpen:Gabbro	-3,837.00	1,448.00	7.90	-2.65	<b>0.030</b>
FenceStatusTransition:Gabbro	-908.00	1,706.00	27.00	-0.53	0.599
FenceStatusOpen:OwnershipState	-264.90	162.10	25.80	-1.64	0.114
FenceStatusTransition:OwnershipState	-27.91	187.00	28.50	-0.15	0.882
FenceStatusOpen:RiverDensity	-11,410.00	2,327.00	277.20	-4.90	<b>&lt;0.001</b>
FenceStatusTransition:RiverDensity	-2,288.00	3,095.00	125.40	-0.74	0.461
FenceStatusOpen:WaterpointDensity	-1,360.00	1,667.00	456.10	-0.82	0.415
FenceStatusTransition:WaterpointDensity	399.70	2,189.00	77.40	0.18	0.856

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.7.** The most-supported model for Max standard deviation differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	-575.82	338.78	18.00	-1.70	0.106
Amphibolitic	312.78	256.26	11.00	1.22	0.248
Aspect	0.81	1.46	15.80	0.56	0.587
FenceStatusOpen	-85.98	378.29	68.50	-0.23	0.821
FenceStatusTransition	3.06	554.27	100.10	0.01	0.996
Gabbro	400.33	670.23	13.30	0.60	0.560
LengthMean	1.08	0.19	469.20	5.72	<b>&lt;0.001</b>
OwnershipState	-72.59	69.18	22.40	-1.05	0.305
RainfallLag3	0.17	0.03	473.30	5.53	<b>&lt;0.001</b>
RiverDensity	-347.80	545.31	14.10	-0.64	0.534
WaterpointDensity	82.57	196.19	17.00	0.42	0.679
Amphibolitic:FenceStatusOpen	313.67	534.92	24.10	0.59	0.563
Amphibolitic:FenceStatusTransition	670.59	776.15	37.10	0.86	0.393
Aspect:FenceStatusOpen	-0.30	1.47	53.30	-0.21	0.836
Aspect:FenceStatusTransition	-0.46	2.32	74.50	-0.20	0.843
FenceStatusOpen:Gabbro	31.45	611.34	45.30	0.05	0.959
FenceStatusTransition:Gabbro	137.69	978.40	47.90	0.14	0.889
FenceStatusOpen:LengthMean	-0.70	0.40	468.50	-1.75	<i>0.081</i>
FenceStatusTransition:LengthMean	-0.29	0.85	454.50	-0.34	0.735
FenceStatusOpen:OwnershipState	45.36	76.75	88.40	0.59	0.556
FenceStatusTransition:OwnershipState	-49.71	107.69	277.00	-0.46	0.645
FenceStatusOpen:RiverDensity	1786.16	894.05	16.00	2.00	<i>0.063</i>
FenceStatusTransition:RiverDensity	729.51	1484.10	36.70	0.49	0.626
FenceStatusOpen:WaterpointDensity	-141.24	539.86	12.70	-0.26	0.798
FenceStatusTransition:WaterpointDensity	-17.65	777.32	28.30	-0.02	0.982

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.8.** The most-supported model for Amp standard deviation differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	249.30	336.70	18.90	0.74	0.468
Amphibolitic	222.30	234.90	11.20	0.95	0.364
Aspect	-0.40	1.37	14.40	-0.29	0.776
BegMean	-1.53	0.32	466.80	-4.79	<b>&lt;0.001</b>
FenceStatusOpen	-690.20	368.70	68.40	-1.87	<i>0.065</i>
FenceStatusTransition	-1296.00	619.70	154.20	-2.09	<b>0.038</b>
Gabbro	560.80	611.60	13.50	0.92	0.375
LengthMean	0.40	0.16	466.70	2.58	<b>0.010</b>
OwnershipState	-66.91	68.04	14.90	-0.98	0.341
RainfallLag3	0.22	0.03	472.50	6.63	<b>&lt;0.001</b>
RiverDensity	-299.10	505.30	13.60	-0.59	0.564
WaterpointDensity	-57.84	189.50	14.00	-0.31	0.765
Amphibolitic:FenceStatusOpen	346.30	434.30	62.30	0.80	0.428
Amphibolitic:FenceStatusTransition	381.30	642.30	387.10	0.59	0.553
Aspect:FenceStatusOpen	0.92	1.32	46.40	0.70	0.490
Aspect:FenceStatusTransition	0.03	2.10	34.50	0.01	0.989
BegMean:FenceStatusOpen	0.25	0.62	466.10	0.40	0.692
BegMean:FenceStatusTransition	4.31	1.21	455.50	3.57	<b>&lt;0.001</b>
FenceStatusOpen:Gabbro	238.20	547.30	40.90	0.44	0.666
FenceStatusTransition:Gabbro	80.66	888.90	22.60	0.09	0.928
FenceStatusOpen:OwnershipState	103.00	72.33	31.30	1.42	0.164
FenceStatusTransition:OwnershipState	36.05	100.40	50.30	0.36	0.721
FenceStatusOpen:RainfallLag3	0.10	0.06	468.30	1.48	0.140
FenceStatusTransition:RainfallLag3	-0.31	0.11	465.30	-2.74	<b>0.006</b>
FenceStatusOpen:RiverDensity	2440.00	732.00	37.80	3.33	<b>0.002</b>
FenceStatusTransition:RiverDensity	156.30	1220.00	314.80	0.13	0.898
FenceStatusOpen:WaterpointDensity	-282.10	415.60	18.30	-0.68	0.506
FenceStatusTransition:WaterpointDensity	624.50	613.90	106.50	1.02	0.311

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.9.** The most-supported model for LI standard deviation differences between the reserves and KNP.

Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	14270.00	7637.00	16.00	1.87	<i>0.080</i>
Amphibolitic	8902.00	3984.00	6.80	2.24	<i>0.062</i>
Aspect	-4.98	25.94	10.40	-0.19	0.851
BegMean	-37.99	10.73	1097.00	-3.54	<b>&lt;0.001</b>
Elevation	-3.14	5.50	10.30	-0.57	0.580
FenceStatusOpen	-19740.00	15970.00	7.00	-1.24	0.256
FenceStatusTransition	-24620.00	27350.00	7.20	-0.90	0.397
Gabbro	9238.00	12190.00	13.80	0.76	0.461
LengthMean	-10.01	5.98	1098.00	-1.68	<i>0.094</i>
OwnershipState	-1606.00	1297.00	9.30	-1.24	0.246
Rainfall	-1.67	0.90	1101.00	-1.84	<i>0.066</i>
RainfallLag2	-3.18	0.91	1100.00	-3.50	<b>&lt;0.001</b>
RainfallLag3	5.21	1.10	1098.00	4.72	<b>&lt;0.001</b>
RiverDensity	-4589.00	8873.00	7.40	-0.52	0.620
WaterpointDensity	-1248.00	3419.00	8.10	-0.37	0.724
Amphibolitic:FenceStatusOpen	-729.00	26940.00	4.70	-0.03	0.980
Amphibolitic:FenceStatusTransition	-5075.00	41100.00	4.20	-0.12	0.907
Aspect:FenceStatusOpen	-29.00	56.09	4.50	-0.52	0.630
Aspect:FenceStatusTransition	24.87	93.56	5.30	0.27	0.800
BegMean:FenceStatusOpen	24.64	20.77	1093.00	1.19	0.236
BegMean:FenceStatusTransition	47.61	40.40	812.40	1.18	0.239
Elevation:FenceStatusOpen	63.31	51.65	5.70	1.23	0.269
Elevation:FenceStatusTransition	15.14	53.97	9.00	0.28	0.785
FenceStatusOpen:Gabbro	-28150.00	30360.00	5.90	-0.93	0.390
FenceStatusTransition:Gabbro	1192.00	39600.00	5.10	0.03	0.977
FenceStatusOpen:LengthMean	23.41	12.23	1096.00	1.91	0.056
FenceStatusTransition:LengthMean	4.52	25.53	642.60	0.18	0.859
FenceStatusOpen:OwnershipState	-3413.00	2980.00	7.50	-1.15	0.287
FenceStatusTransition:OwnershipState	235.80	3677.00	14.10	0.06	0.950
FenceStatusOpen:RainfallLag3	-1.55	2.14	1098.00	-0.73	0.468
FenceStatusTransition:RainfallLag3	-9.34	3.77	1059.00	-2.48	<b>0.013</b>
FenceStatusOpen:RiverDensity	-42810.00	55730.00	5.60	-0.77	0.473
FenceStatusTransition:RiverDensity	6734.00	70770.00	6.90	0.10	0.927
FenceStatusOpen:WaterpointDensity	-27420.00	43120.00	4.80	-0.64	0.554
FenceStatusTransition:WaterpointDensity	11220.00	49890.00	9.30	0.23	0.827

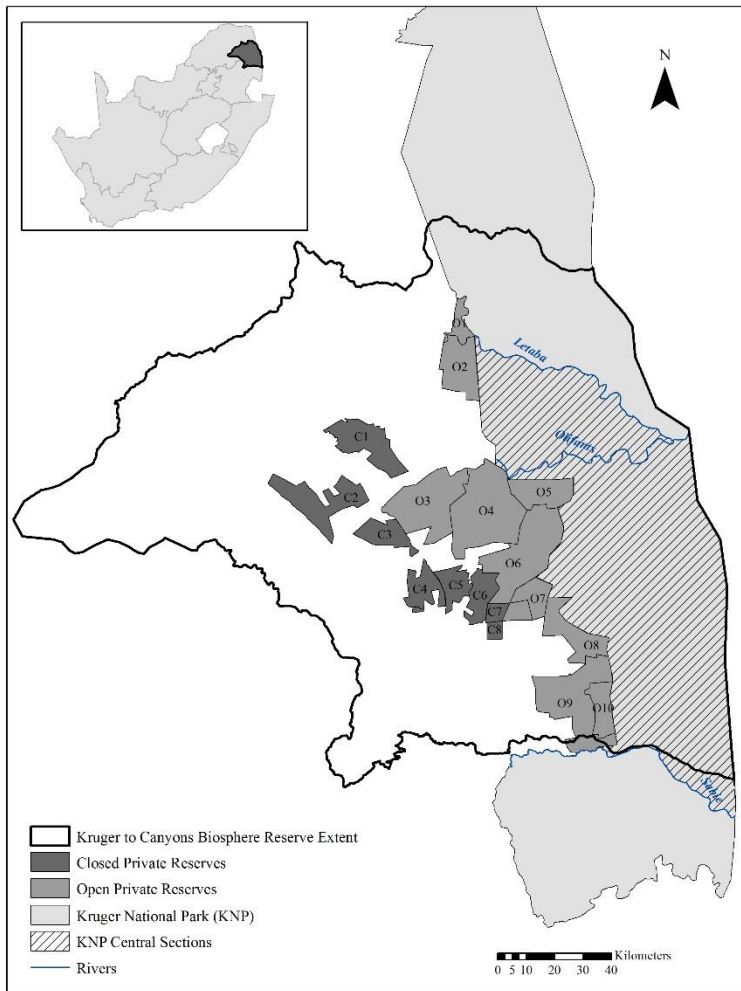
Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

**Table 3.10.** The most-supported model for SI standard deviation differences between the reserves and KNP.

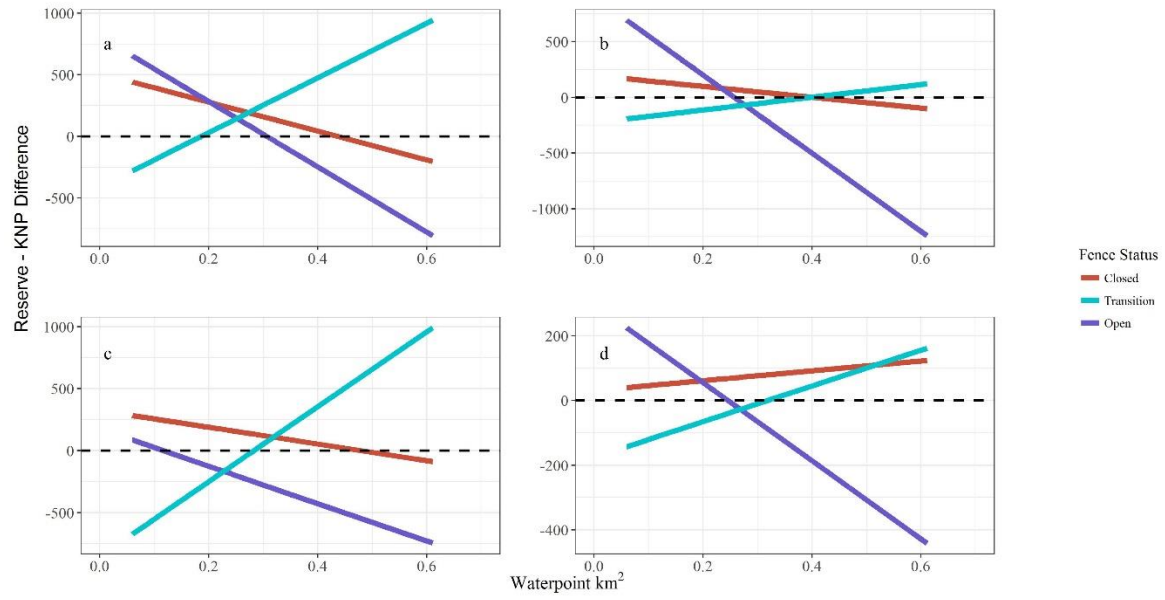
Explanatory Variables	Estimate	SE	df	t value	Pr(> t )
(Intercept)	8752.00	4013.00	16.44	2.18	<b>0.044</b>
Amphibolitic	5005.00	2227.00	8.55	2.25	<i>0.053</i>
Aspect	-2.63	14.02	12.01	-0.19	0.854
BegMean	-24.41	5.07	240.00	-4.82	<b>&lt;0.001</b>
Elevation	-1.89	2.99	11.58	-0.63	0.540
FenceStatusOpen	-12020.00	7523.00	7.02	-1.60	0.154
FenceStatusTransition	-9259.00	11560.00	11.64	-0.80	0.439
Gabbro	6205.00	6530.00	14.67	0.95	0.357
LengthMean	-8.16	2.82	241.30	-2.89	<b>0.004</b>
OwnershipState	-581.20	697.40	9.25	-0.83	0.426
RainfallLag1	-0.76	0.44	242.50	-1.73	<i>0.085</i>
RainfallLag2	-0.82	0.43	240.40	-1.92	<i>0.056</i>
RainfallLag3	1.97	0.52	244.30	3.76	<b>&lt;0.001</b>
RiverDensity	-2509.00	4912.00	9.52	-0.51	0.621
WaterpointDensity	260.70	1867.00	9.65	0.14	0.892
Amphibolitic:FenceStatusOpen	-1531.00	11310.00	5.60	-0.14	0.897
Amphibolitic:FenceStatusTransition	11550.00	14920.00	3.97	0.77	0.482
Aspect:FenceStatusOpen	-5.58	26.78	4.07	-0.21	0.845
Aspect:FenceStatusTransition	-7.24	39.68	7.52	-0.18	0.860
BegMean:FenceStatusOpen	0.17	9.93	239.50	0.02	0.987
BegMean:FenceStatusTransition	18.34	19.11	231.90	0.96	0.338
Elevation:FenceStatusOpen	39.21	21.84	6.53	1.80	0.119
Elevation:FenceStatusTransition	13.34	21.46	8.99	0.62	0.550
FenceStatusOpen:Gabbro	-14270.00	14360.00	5.11	-0.99	0.365
FenceStatusTransition:Gabbro	-3911.00	17610.00	7.03	-0.22	0.831
FenceStatusOpen:LengthMean	14.49	5.73	239.50	2.53	<b>0.012</b>
FenceStatusTransition:LengthMean	5.97	11.93	197.40	0.50	0.617
FenceStatusOpen:OwnershipState	-1479.00	1339.00	9.95	-1.11	0.295
FenceStatusTransition:OwnershipState	-906.10	1643.00	18.05	-0.55	0.588
FenceStatusOpen:RainfallLag3	1.22	1.02	240.80	1.20	0.231
FenceStatusTransition:RainfallLag3	-3.02	1.80	239.60	-1.68	<i>0.094</i>
FenceStatusOpen:RiverDensity	-17870.00	23440.00	6.63	-0.76	0.472
FenceStatusTransition:RiverDensity	-8729.00	27750.00	8.44	-0.32	0.761
FenceStatusOpen:WaterpointDensity	-21080.00	18020.00	5.28	-1.17	0.292
FenceStatusTransition:WaterpointDensity	8114.00	19650.00	9.29	0.41	0.689

Significant values in bold ( $p \leq 0.05$ ) and nearly significant values in italics ( $p \leq 0.10$ )

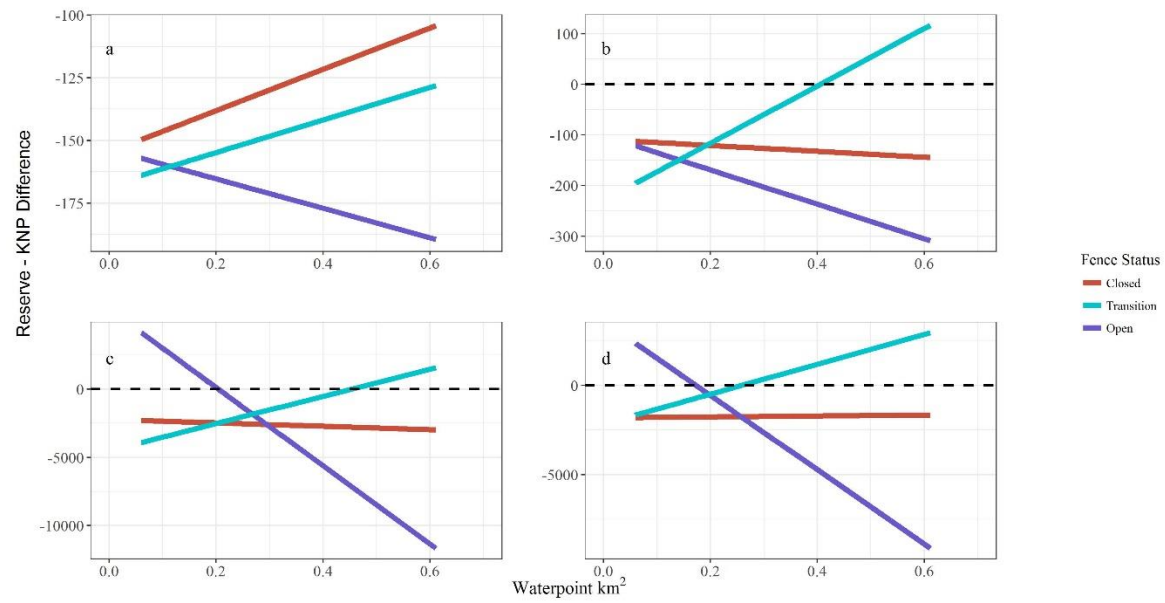
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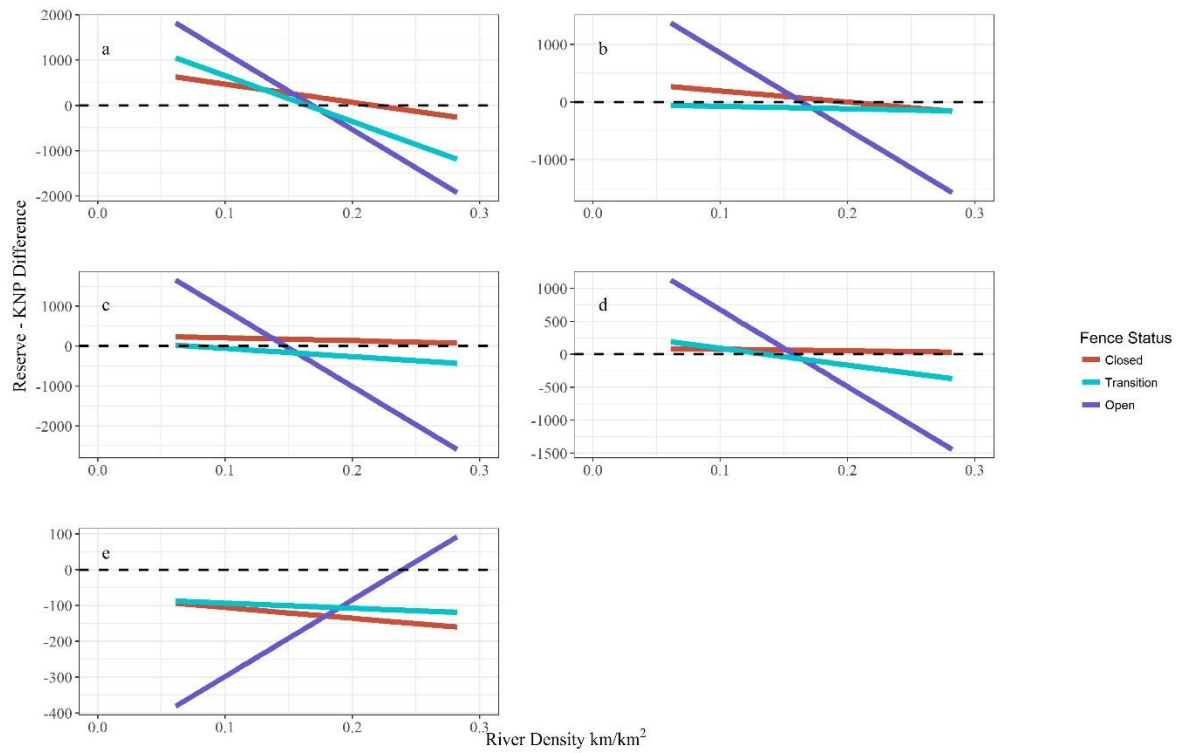
**Figure 3.1.** The Kruger to Canyons Biosphere Reserve includes open and closed private reserves and central sections of Kruger National Park.



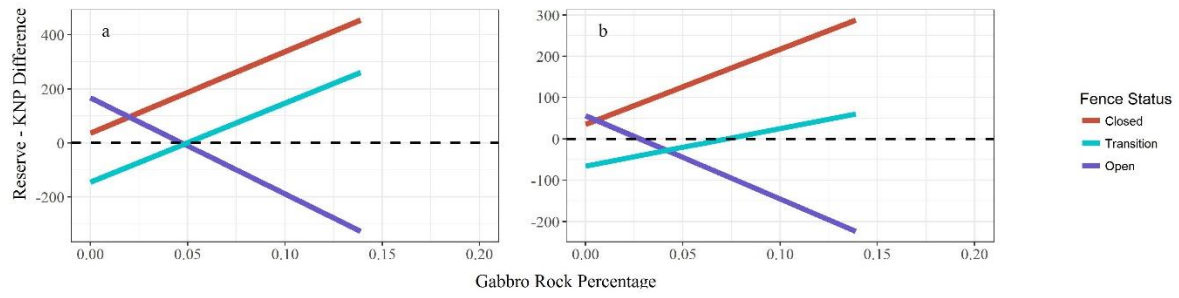
**Figure 3.2.** The interaction between waterpoint density and fence status for average Max (a), Amp (b), LI (c), and SI (d).



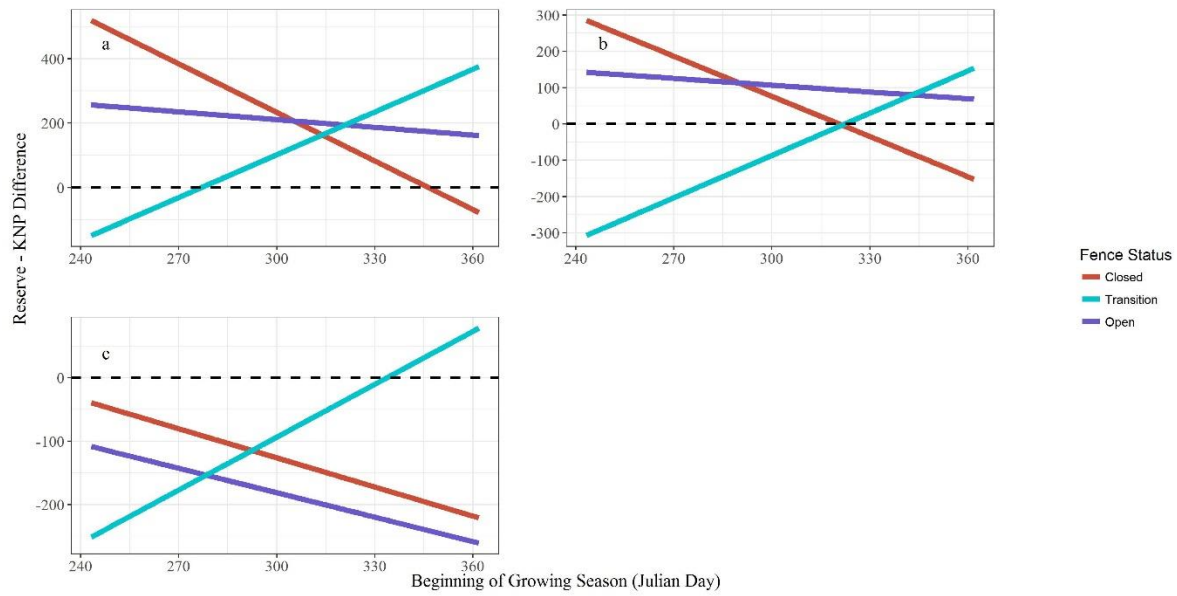
**Figure 3.3.** The interaction between waterpoint density and fence status for the standard deviation of Max (a), Amp (b), LI (c), and SI (d).



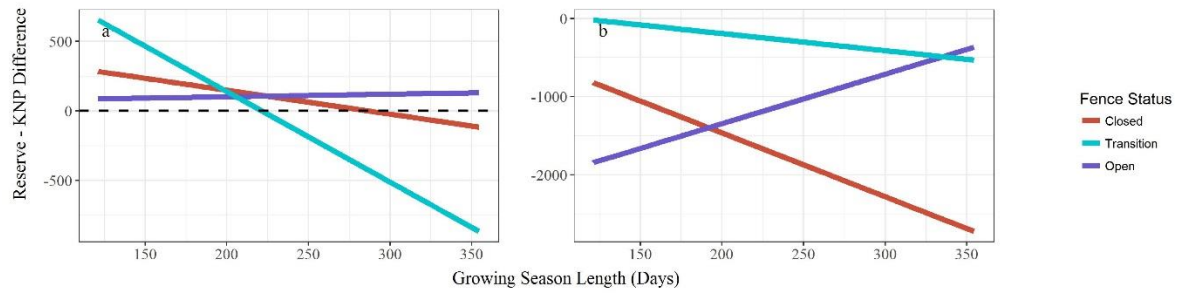
**Figure 3.4.** The interaction between river density and fence status for average Max (a), Amp (b), LI (c), SI (d); and Amp standard deviation (e).



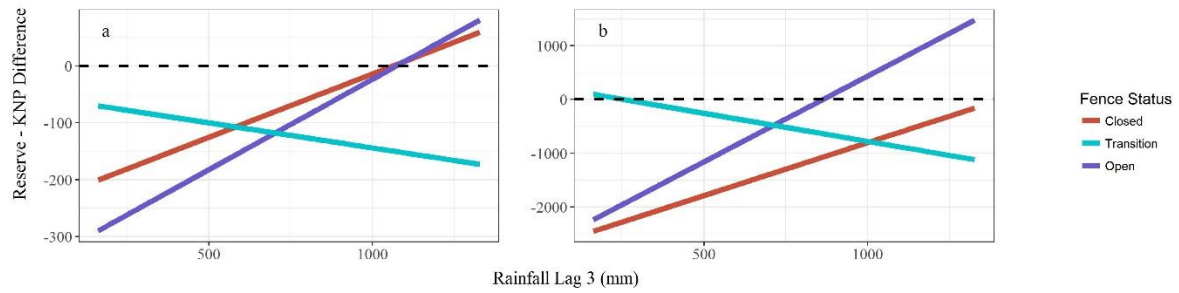
**Figure 3.5.** The interaction between gabbro rock percentage and fence status on average Amp (a) and SI (b).



**Figure 3.6.** The interaction between the beginning of the growing season and fence status for average Max (a) and Amp (b); and for Amp standard deviation (c).



**Figure 3.7.** The interaction between growing season length and fence status for average Amp (a) and for SI standard deviation (b).



**Figure 3.8.** The interaction between rainfall lag 3 and fence status for Amp standard deviation (a) and SI standard deviation (b).

## APPENDIX A: Simplified Geology Types

**Table A1.** Geology types of the Kruger to Canyons Biosphere Reserve, reclassified into 6 simplified geology categories (Venter, pers. comm.).

Simplified Geology	Geology Type Descriptions
Amphibolitic Rock	<p>Mafic and ultramafic lavas, felsic tuffs</p> <p>Mafic lavas, various chlorite schists, quartzites, conglomerates, grits, porphyritic tuffs, banded iron-formation, ultramafic lavas</p> <p>Ultramafic chlorite-amphibole-talc-serpentine-rich rocks and subordinate amphibolites, acid igneous rocks and sedimentary rocks</p>
Basaltic Rock	<p>Dark grey gabbro-norite forming irregular vein-like intrusions as well as plutons</p> <p>Basic volcanic rocks (tholeiites, picrite basalts and nephelinites)</p>
Gabbro Rock	Carbonatite, basalt, trachyte, andesite, rhyolite, volcanic breccia, agglomerate, ignimbrite, tuff
Granitic Rock	<p>Basalt/basaltic andesite, tuff, quartzite, minor conglomerate</p> <p>Pink, coarse-grained, porphyritic, potassic granite</p> <p>Dark-grey mudrock</p> <p>Medium- to coarse-grained sandstone (pebbly in places), conglomerate, trachytic lava, quartz porphyry</p> <p>Pink to red, fine- to medium-grained biotite granite, minor grey granite and granophyre</p> <p>Granophyric quartz gabbro</p> <p>Red sandstone/quartzite, interbedded red siltstone and shale</p> <p>Potassic granite, gneiss</p> <p>Grey, medium-grained (porphyritic in places), granodioritic biotite-muscovite granite</p> <p>Grey, medium-grained, equigranular tonalite</p> <p>Medium- to coarse-grained, homogeneous hornblende and hornblende -biotite tonalite</p>
Rhyolitic Rock	<p>Acid lavas (rhyolites with some dacites), minor tuffs</p> <p>Sill-like bodies and dykes of granophyre</p>
Sandstone	Pink-weathering granular or augen quartz-feldspar gneiss
Shale	Shale, sandstone, mudstone, coal

## **APPENDIX B:** Standardizing TIMESAT Metric Equations

**Equation B1.** The AVHRR sensor adjustment equation for the beginning of the growing season values.

$$\text{Adjusted Value} = \left( \text{Raw Value} - \left( 36 \times (\text{TIMESAT Season Number} - 1) \right) \right) \times 10$$

**Equation B2.** The MODIS sensor adjustment equation for the beginning of the growing season values.

$$\text{Adjusted Value} = \left( \text{Raw Value} - \left( 23 \times (\text{TIMESAT Season Number} - 1) \right) \right) \times 16$$

**Equation B3.** The AVHRR sensor adjustment equation for the length of the growing season values.

$$\text{Adjusted Value} = \text{Raw Value} \times 10$$

**Equation B4.** The MODIS sensor adjustment equation for the length of the growing season values.

$$\text{Adjusted Value} = \text{Raw Value} \times 16$$

**APPENDIX C:** The coefficient of determination ( $R^2$ ) between all explanatory variables.

**Table C1.** The coefficient of determination ( $R^2$ ) between all explanatory variables considered for the linear mixed-effects analysis.

	Beg	Mid	End	Length	Base	Waterpoint Density	River Density	Rainfall	Rainfall Lag1	Rainfall Lag2	Rainfall Lag3	Aspect	Elevation	Slope	Amphibolitic	Gabbro	Granitic	Area	Perimeter: Area Ratio
Beg	1.00	0.00	0.11	0.07	0.31	0.01	0.01	0.04	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01
Mid	0.00	1.00	0.01	0.01	0.01	0.00	0.00	0.17	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
End	0.11	0.01	1.00	0.68	0.06	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.02
Length	0.07	0.01	0.68	1.00	0.01	0.00	0.01	0.00	0.01	0.01	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.03	0.04
Base	0.31	0.01	0.06	0.01	1.00	0.06	0.07	0.21	0.07	0.05	0.01	0.01	0.18	0.11	0.00	0.00	0.00	0.09	0.13
Waterpoint Density	0.01	0.00	0.00	0.00	0.06	1.00	0.07	0.02	0.03	0.03	0.03	0.11	0.14	0.00	0.00	0.02	0.01	0.03	0.04
River Density	0.01	0.00	0.00	0.01	0.07	0.07	1.00	0.10	0.10	0.09	0.09	0.00	0.28	0.46	0.00	0.00	0.00	0.07	0.16
Rainfall	0.04	0.17	0.01	0.00	0.21	0.02	0.10	1.00	0.11	0.02	0.04	0.05	0.13	0.09	0.01	0.00	0.01	0.05	0.11
Rainfall Lag1	0.01	0.00	0.00	0.01	0.07	0.03	0.10	0.11	1.00	0.14	0.03	0.06	0.14	0.09	0.02	0.00	0.02	0.06	0.12
Rainfall Lag2	0.00	0.00	0.00	0.01	0.05	0.03	0.09	0.02	0.14	1.00	0.13	0.04	0.14	0.09	0.01	0.00	0.01	0.06	0.11
Rainfall Lag3	0.02	0.01	0.00	0.00	0.01	0.03	0.09	0.04	0.03	0.13	1.00	0.04	0.13	0.09	0.01	0.00	0.01	0.05	0.11
Aspect	0.00	0.00	0.00	0.00	0.01	0.11	0.00	0.05	0.06	0.04	0.04	1.00	0.00	0.03	0.00	0.01	0.00	0.01	0.07
Elevation	0.01	0.00	0.01	0.02	0.18	0.14	0.28	0.13	0.14	0.14	0.13	0.00	1.00	0.67	0.00	0.07	0.01	0.23	0.45
Slope	0.00	0.00	0.00	0.01	0.11	0.00	0.46	0.09	0.09	0.09	0.09	0.03	0.67	1.00	0.01	0.03	0.03	0.12	0.35
Amphibolitic	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.01	0.01	0.00	0.00	0.01	1.00	0.03	0.85	0.03	0.05
Gabbro	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.07	0.03	0.03	1.00	0.05	0.18	0.10
Granitic	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.02	0.01	0.01	0.00	0.01	0.03	0.85	0.05	1.00	0.11	0.11
Area	0.00	0.00	0.02	0.03	0.09	0.03	0.07	0.05	0.06	0.06	0.05	0.01	0.23	0.12	0.03	0.18	0.11	1.00	0.68
Perimeter: Area Ratio	0.01	0.00	0.02	0.04	0.13	0.04	0.16	0.11	0.12	0.11	0.11	0.07	0.45	0.35	0.05	0.10	0.11	0.68	1.00

## APPENDIX D: Explanatory variables included in all *a priori* models

**Table D1.** The explanatory variables included in each *a priori* model without interaction terms between Fence Status.

<i>A Priori</i> Model	Explanatory Variables
Max.Global	BegMean + LengthMean + WaterpointDensity + RiverDensity + RainfallLag1 + Aspect + Elevation + Amphibolitic + Gabbro + Area + Ownership
Amp.Global	BegMean + LengthMean + WaterpointDensity + RiverDensity + RainfallLag1 + Aspect + Elevation + Amphibolitic + Gabbro + Area + Ownership
LI.Global	BegMean + LengthMean + WaterpointDensity + RiverDensity + RainfallLag1 + Aspect + Elevation + Amphibolitic + Gabbro + Area + Ownership
SI.Global	BegMean + LengthMean + WaterpointDensity + RiverDensity + RainfallLag3 + Aspect + Elevation + Amphibolitic + Gabbro + Area + Ownership
Surface Water	WaterpointDensity + RiverDensity
Waterpoint Density	WaterpointDensity
River Density	RiverDensity
Geology	Amphibolitic + Gabbro
Amphibolitic	Amphibolitic
Gabbro	Gabbro
Topography	Aspect + Elevation
Aspect	Aspect
Elevation	Elevation
Ownership	Ownership
Rainfall	Rainfall
RainfallLag1	RainfallLag1
RainfallLag2	RainfallLag2
RainfallLag3	RainfallLag3
Seasonality	BegMean + LengthMean
Beg	BegMean
Length	LengthMean

**Table D2.** The explanatory variables included in each *a priori* model with Fence Status interaction terms.

<i>A Priori</i> Model	Explanatory Variables
Max.Global.Fence	BegMean + BegMean*FenceStatus + LengthMean + LengthMean*FenceStatus + WaterpointDensity + WaterpointDensity*FenceStatus + RiverDensity + RiverDensity*FenceStatus + RainfallLag1 + RainfallLag1*FenceStatus + Aspect + Aspect*FenceStatus + Elevation + Elevation*FenceStatus + Amphibolitic + Amphibolitic*FenceStatus + Gabbro + Gabbro*FenceStatus + Area + Area*FenceStatus + Ownership + Ownership*FenceStatus
Amp.Global.Fence	BegMean + BegMean*FenceStatus + LengthMean + LengthMean*FenceStatus + WaterpointDensity + WaterpointDensity*FenceStatus + RiverDensity + RiverDensity*FenceStatus + RainfallLag1 + RainfallLag1*FenceStatus + Aspect + Aspect*FenceStatus + Elevation + Elevation*FenceStatus + Amphibolitic + Amphibolitic*FenceStatus + Gabbro + Gabbro*FenceStatus + Area + Area*FenceStatus + Ownership + Ownership*FenceStatus
LI.Global.Fence	BegMean + BegMean*FenceStatus + LengthMean + LengthMean*FenceStatus + WaterpointDensity + WaterpointDensity*FenceStatus + RiverDensity + RiverDensity*FenceStatus + RainfallLag1 + RainfallLag1*FenceStatus + Aspect + Aspect*FenceStatus + Elevation + Elevation*FenceStatus + Amphibolitic + Amphibolitic*FenceStatus + Gabbro + Gabbro*FenceStatus + Area + Area*FenceStatus + Ownership + Ownership*FenceStatus
SI.Global.Fence	BegMean + BegMean*FenceStatus + LengthMean + LengthMean*FenceStatus + WaterpointDensity + WaterpointDensity*FenceStatus + RiverDensity + RiverDensity*FenceStatus + RainfallLag3 + RainfallLag3*FenceStatus + Aspect + Aspect*FenceStatus + Elevation + Elevation*FenceStatus + Amphibolitic + Amphibolitic*FenceStatus + Gabbro + Gabbro*FenceStatus + Area + Area*FenceStatus + Ownership + Ownership*FenceStatus
SurfaceWater.Fence	WaterpointDensity + RiverDensity + WaterpointDensity*FenceStatus + RiverDensity*FenceStatus
WaterpointDensity.Fence	WaterpointDensity + WaterpointDensity*FenceStatus
RiverDensity.Fence	RiverDensity + RiverDensity*FenceStatus
Geology.Fence	Amphibolitic + Gabbro + Amphibolitic*FenceStatus + Gabbro*FenceStatus
Amphibolitic.Fence	Amphibolitic + Amphibolitic*FenceStatus
Gabbro.Fence	Gabbro + Gabbro*FenceStatus
Topography.Fence	Aspect + Elevation + Aspect*FenceStatus + Elevation*FenceStatus
Aspect.Fence	Aspect + Aspect*FenceStatus
Elevation.Fence	Elevation + Elevation*FenceStatus
Ownership.Fence	Ownership + Ownership*FenceStatus
Rainfall.Fence	Rainfall + Rainfall*FenceStatus
RainfallLag1.Fence	RainfallLag1 + RainfallLag1*FenceStatus
RainfallLag2.Fence	RainfallLag2 + RainfallLag2*FenceStatus
RainfallLag3.Fence	RainfallLag3 + RainfallLag3*FenceStatus
Seasonality.Fence	BegMean + LengthMean + BegMean*FenceStatus + LengthMean*FenceStatus
Beg.Fence	BegMean + BegMean*FenceStatus
Length.Fence	LengthMean + LengthMean*FenceStatus

## APPENDIX E: AICc comparisons between *a priori* and global models

**Table E1.** The AICc comparison for average Max between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
Max.Global.Fence	46	7019.01	0	1	1	-3458.77
Max.Global	20	7233.70	214.68	0	1	-3595.98
Max.Geology.Fence	16	7240.73	221.72	0	1	-3603.81
Max.Surface.Water.Fence	16	7241.28	222.26	0	1	-3604.08
Max.Ownership.Fence	16	7264.02	245.01	0	1	-3615.45
Max.Gabbro.Fence	13	7284.20	265.19	0	1	-3628.73
Max.RiverDensity.Fence	13	7285.19	266.18	0	1	-3629.22
Max.WaterpointDensity.Fence	13	7287.47	268.46	0	1	-3630.37
Max.Amphibolitic.Fence	13	7287.97	268.96	0	1	-3630.62
Max.Seasonality.Fence	16	7298.47	279.45	0	1	-3632.67
Max.Topography.Fence	16	7318.47	299.46	0	1	-3642.68
Max.Beg.Fence	13	7318.55	299.53	0	1	-3645.90
Max.Length.Fence	13	7321.68	302.67	0	1	-3647.47
Max.Aspect.Fence	13	7322.11	303.10	0	1	-3647.68
Max.Surface.Water	10	7324.49	305.48	0	1	-3652.02
Max.Elevation.Fence	13	7328.28	309.27	0	1	-3650.77
Max.Geology	10	7329.03	310.02	0	1	-3654.29
Max.RainfallLag1.Fence	13	7334.68	315.67	0	1	-3653.97
Max.Ownership	10	7338.28	319.27	0	1	-3658.92
Max.Rainfall.Fence	13	7339.08	320.06	0	1	-3656.17
Max.RainfallLag3.Fence	13	7339.70	320.69	0	1	-3656.48
Max.WaterpointDensity	9	7339.90	320.89	0	1	-3660.77
Max.RainfallLag2.Fence	13	7341.78	322.77	0	1	-3657.52
Max.Gabbro	9	7344.17	325.16	0	1	-3662.90
Max.Seasonality	10	7344.98	325.96	0	1	-3662.26
Max.Amphibolitic	9	7346.98	327.97	0	1	-3664.31
Max.RiverDensity	9	7347.30	328.28	0	1	-3664.47
Max.Beg	9	7354.89	335.88	0	1	-3668.26
Max.Topography	10	7356.85	337.83	0	1	-3668.20
Max.Length	9	7357.73	338.72	0	1	-3669.68
Max.Aspect	9	7357.75	338.74	0	1	-3669.69
Max.RainfallLag1	9	7358.59	339.57	0	1	-3670.11
Max.Elevation	9	7360.50	341.48	0	1	-3671.07
Max.RainfallLag3	9	7364.80	345.78	0	1	-3673.22
Max.Rainfall	9	7366.32	347.31	0	1	-3673.98
Max.RainfallLag2	9	7366.89	347.88	0	1	-3674.26

**Table E2.** The AICc comparison for average Amp between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
Amp.Global.Fence	46	6939.89	0	1	1	-3419.20
Amp.Geology.Fence	16	7132.76	192.87	0	1	-3549.82
Amp.Surface.Water.Fence	16	7135.85	195.96	0	1	-3551.37
Amp.Global	20	7146.11	206.22	0	1	-3552.18
Amp.Ownership.Fence	16	7157.06	217.17	0	1	-3561.97
Amp.Gabbro.Fence	13	7175.52	235.62	0	1	-3574.39
Amp.Rainfall.Fence	13	7178.31	238.42	0	1	-3575.78
Amp.Amphibolitic.Fence	13	7179.5	239.61	0	1	-3576.38
Amp.WaterpointDensity.Fence	13	7179.97	240.08	0	1	-3576.61
Amp.Seasonality.Fence	16	7198.52	258.63	0	1	-3582.70
Amp.Topography.Fence	16	7213.10	273.21	0	1	-3589.99
Amp.Beg.Fence	13	7213.90	274.01	0	1	-3593.58
Amp.Length.Fence	13	7213.93	274.03	0	1	-3593.59
Amp.Aspect.Fence	13	7214.16	274.26	0	1	-3593.71
Amp.Geology	10	7214.72	274.83	0	1	-3597.14
Amp.Surface.Water	10	7215.08	275.18	0	1	-3597.31
Amp.Elevation.Fence	13	7221.77	281.88	0	1	-3597.51
Amp.Ownership	10	7226.87	286.98	0	1	-3603.21
Amp.Gabbro	9	7229.15	289.26	0	1	-3605.39
Amp.RainfallLag1.Fence	13	7229.21	289.32	0	1	-3601.23
Amp.WaterpointDensity	9	7229.33	289.44	0	1	-3605.48
Amp.RainfallLag3.Fence	13	7233.05	293.16	0	1	-3603.15
Amp.Amphibolitic	9	7233.09	293.20	0	1	-3607.36
Amp.RiverDensity	9	7233.34	293.44	0	1	-3607.49
Amp.RainfallLag2.Fence	13	7234.14	294.24	0	1	-3603.70
Amp.RiverDensity.Fence	13	7234.35	294.46	0	1	-3603.80
Amp.Seasonality	10	7237.91	298.02	0	1	-3608.73
Amp.Beg	9	7242.51	302.62	0	1	-3612.07
Amp.Aspect	9	7244.52	304.63	0	1	-3613.08
Amp.Topography	10	7244.53	304.63	0	1	-3612.04
Amp.Length	9	7246.10	306.20	0	1	-3613.87
Amp.Elevation	9	7247.10	307.20	0	1	-3614.37
Amp.RainfallLag1	9	7247.74	307.85	0	1	-3614.69
Amp.RainfallLag3	9	7251.44	311.55	0	1	-3616.54
Amp.Rainfall	9	7251.46	311.56	0	1	-3616.55
Amp.RainfallLag2	9	7252.25	312.35	0	1	-3616.94

**Table E3.** The AICc comparison for average LI between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
LI.Global.Fence	46	7606.19	0	1	1	-3752.35
LI.Surface.Water.Fence	16	7835.28	229.09	0	1	-3901.08
LI.Global	20	7835.35	229.17	0	1	-3896.81
LI.Geology.Fence	16	7843.77	237.59	0	1	-3905.33
LI.Ownership.Fence	16	7866.08	259.89	0	1	-3916.48
LI.Rainfall.Fence	13	7878.74	272.56	0	1	-3926.00
LI.Gabbro.Fence	13	7888.83	282.65	0	1	-3931.05
LI.Amphibolitic.Fence	13	7890.87	284.68	0	1	-3932.06
LI.WaterpointDensity.Fence	13	7892.16	285.97	0	1	-3932.71
LI.Seasonality.Fence	16	7893.41	287.23	0	1	-3930.15
LI.Length.Fence	13	7905.70	299.52	0	1	-3939.48
LI.RainfallLag3.Fence	13	7916.77	310.58	0	1	-3945.01
LI.Topography.Fence	16	7922.00	315.81	0	1	-3944.44
LI.Aspect.Fence	13	7926.91	320.72	0	1	-3950.08
LI.Beg.Fence	13	7927.90	321.71	0	1	-3950.58
LI.Geology	10	7927.96	321.77	0	1	-3953.75
LI.Elevation.Fence	13	7930.82	324.63	0	1	-3952.04
LI.Surface.Water	10	7930.95	324.77	0	1	-3955.25
LI.RiverDensity.Fence	13	7930.98	324.79	0	1	-3952.12
LI.Seasonality	10	7930.99	324.80	0	1	-3955.27
LI.RainfallLag3	9	7935.10	328.91	0	1	-3958.37
LI.Ownership	10	7936.10	329.91	0	1	-3957.83
LI.RainfallLag2.Fence	13	7937.36	331.18	0	1	-3955.31
LI.RainfallLag1.Fence	13	7939.01	332.83	0	1	-3956.13
LI.Length	9	7939.93	333.74	0	1	-3960.78
LI.Gabbro	9	7942.07	335.88	0	1	-3961.85
LI.RiverDensity	9	7943.53	337.35	0	1	-3962.58
LI.Amphibolitic	9	7944.04	337.85	0	1	-3962.84
LI.WaterpointDensity	9	7945.48	339.29	0	1	-3963.55
LI.Topography	10	7952.73	346.54	0	1	-3966.14
LI.Aspect	9	7955.15	348.97	0	1	-3968.39
LI.Elevation	9	7955.32	349.13	0	1	-3968.48
LI.Beg	9	7955.55	349.37	0	1	-3968.59
LI.RainfallLag1	9	7957.58	351.39	0	1	-3969.61
LI.RainfallLag2	9	7959.26	353.08	0	1	-3970.45
LI.Rainfall	9	7961.18	354.99	0	1	-3971.41

**Table E4.** The AICc comparison for average SI between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
SI.Global.Fence	46	6586.57	0	1	1	-3242.54
SI.Surface.Water.Fence	16	6754.76	168.19	0	1	-3360.82
SI.Geology.Fence	16	6758.63	172.06	0	1	-3362.75
SI.Global	20	6778.35	191.79	0	1	-3368.30
SI.Ownership.Fence	16	6786.46	199.89	0	1	-3376.67
SI.RiverDensity.Fence	13	6792.48	205.91	0	1	-3382.87
SI.Gabbro.Fence	13	6799.24	212.67	0	1	-3386.25
SI.Amphibolitic.Fence	13	6801.05	214.48	0	1	-3387.15
SI.WaterpointDensity.Fence	13	6803.25	216.68	0	1	-3388.25
SI.Geology	10	6830.98	244.42	0	1	-3405.27
SI.Seasonality.Fence	16	6832.56	246.00	0	1	-3399.72
SI.Length.Fence	13	6833.59	247.03	0	1	-3403.43
SI.Surface.Water	10	6833.62	247.05	0	1	-3406.59
SI.Topography.Fence	16	6838.04	251.48	0	1	-3402.46
SI.Aspect.Fence	13	6839.14	252.57	0	1	-3406.20
SI.Elevation.Fence	13	6840.64	254.08	0	1	-3406.95
SI.Ownership	10	6841.24	254.67	0	1	-3410.39
SI.RainfallLag3.Fence	13	6841.89	255.32	0	1	-3407.57
SI.Rainfall.Fence	13	6842.00	255.43	0	1	-3407.63
SI.Beg.Fence	13	6842.00	255.43	0	1	-3407.63
SI.Gabbro	9	6844.00	257.43		1	-3412.82
SI.RiverDensity	9	6845.56	258.99	0	1	-3413.60
SI.Amphibolitic	9	6845.82	259.25	0	1	-3413.73
SI.WaterpointDensity	9	6846.25	259.69	0	1	-3413.94
SI.RainfallLag3	9	6849.78	263.22	0	1	-3415.71
SI.RainfallLag2.Fence	13	6850.57	264.00	0	1	-3411.91
SI.Rainfall	9	6852.37	265.81	0	1	-3417.00
SI.RainfallLag1.Fence	13	6853.38	266.81	0	1	-3413.32
SI.Topography	10	6856.04	269.47	0	1	-3417.79
SI.Elevation	9	6856.98	270.41	0	1	-3419.31
SI.Aspect	9	6857.16	270.59	0	1	-3419.40
SI.RainfallLag2	9	6860.05	273.48	0	1	-3420.84
SI.Beg	9	6860.14	273.57	0	1	-3420.89
SI.Length	9	6860.70	274.14	0	1	-3421.17
SI.RainfallLag1	9	6862.01	275.44	0	1	-3421.82
SI.Seasonality	10	6862.17	275.60	0	1	-3420.86

**Table E5.** The AICc comparison for Max standard deviation between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
Max.Global.Fence	46	6127.10	0	1	1	-3012.81
Max.Surface.Water.Fence	16	6262.16	135.06	0	1	-3114.52
Max.Geology.Fence	16	6263.17	136.07	0	1	-3115.03
Max.Global	20	6275.00	147.91	0	1	-3116.63
Max.Ownership.Fence	16	6285.01	157.91	0	1	-3125.94
Max.RiverDensity.Fence	13	6297.50	170.40	0	1	-3135.38
Max.Amphibolitic.Fence	13	6300.84	173.74	0	1	-3137.05
Max.Gabbro.Fence	13	6301.28	174.18	0	1	-3137.27
Max.WaterpointDensity.Fence	13	6302.52	175.42	0	1	-3137.89
Max.Seasonality.Fence	16	6316.73	189.63	0	1	-3141.80
Max.Length.Fence	13	6321.44	194.34	0	1	-3147.35
Max.Beg.Fence	13	6331.38	204.29	0	1	-3152.32
Max.Surface.Water	10	6331.65	204.55	0	1	-3155.60
Max.Geology	10	6331.78	204.68	0	1	-3155.67
Max.RainfallLag3.Fence	13	6332.32	205.23	0	1	-3152.79
Max.Ownership	10	6332.76	205.66	0	1	-3156.16
Max.Length	9	6335.26	208.17	0	1	-3158.45
Max.Seasonality	10	6337.96	210.86	0	1	-3158.75
Max.Aspect.Fence	13	6338.45	211.35	0	1	-3155.85
Max.WaterpointDensity	9	6343.56	216.46	0	1	-3162.60
Max.Amphibolitic	9	6343.84	216.75	0	1	-3162.74
Max.Topography.Fence	16	6344.47	217.38	0	1	-3155.68
Max.Elevation.Fence	13	6344.99	217.89	0	1	-3159.12
Max.Gabbro	9	6345.10	218.01	0	1	-3163.37
Max.RiverDensity	9	6345.68	218.59	0	1	-3163.66
Max.RainfallLag3	9	6346.03	218.94	0	1	-3163.83
Max.RainfallLag1.Fence	13	6348.78	221.69	0	1	-3161.02
Max.RainfallLag2.Fence	13	6353.63	226.53	0	1	-3163.44
Max.Rainfall.Fence	13	6354.23	227.14	0	1	-3163.74
Max.Aspect	9	6356.72	229.62	0	1	-3169.18
Max.Beg	9	6358.34	231.25	0	1	-3169.99
Max.Topography	10	6359.52	232.42	0	1	-3169.53
Max.Elevation	9	6360.18	233.08	0	1	-3170.91
Max.RainfallLag1	9	6362.41	235.31	0	1	-3172.02
Max.RainfallLag2	9	6363.43	236.34	0	1	-3172.53
Max.Rainfall	9	6364.30	237.21	0	1	-3172.97

**Table E6.** The AICc comparison for Amp standard deviation between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
Amp.Global.Fence	46	6065.55	0	1	1	-2982.03
Amp.Geology.Fence	16	6189.54	123.99	0	1	-3078.21
Amp.Surface.Water. Fence	16	6193.74	128.19	0	1	-3080.31
Amp.Global	20	6211.46	145.91	0	1	-3084.86
Amp.Ownership.Fence	16	6219.42	153.87	0	1	-3093.15
Amp.Rainfall.Fence	13	6227.18	161.63	0	1	-3100.22
Amp.Amphibolitic.Fence	13	6228.12	162.57	0	1	-3100.69
Amp.Gabbro.Fence	13	6228.42	162.87	0	1	-3100.84
Amp.WaterpointDensity.Fence	13	6232.23	166.68	0	1	-3102.74
Amp.RainfallLag3.Fence	13	6238.34	172.79	0	1	-3105.80
Amp.Seasonality.Fence	16	6248.15	182.61	0	1	-3107.52
Amp.Beg.Fence	13	6248.75	183.20	0	1	-3111.00
Amp.RainfallLag3	9	6251.79	186.24	0	1	-3116.71
Amp.Geology	10	6257.37	191.83	0	1	-3118.46
Amp.Length.Fence	13	6260.24	194.69	0	1	-3116.75
Amp.Surface.Water	10	6261.44	195.89	0	1	-3120.50
Amp.Ownership	10	6266.05	200.51	0	1	-3122.80
Amp.Aspect.Fence	13	6267.78	202.23	0	1	-3120.52
Amp.Amphibolitic	9	6269.95	204.40	0	1	-3125.79
Amp.Elevation.Fence	13	6270.93	205.38	0	1	-3122.09
Amp.Gabbro	9	6272.08	206.53	0	1	-3126.86
Amp.RiverDensity	9	6272.56	207.01	0	1	-3127.10
Amp.Seasonality	10	6272.69	207.14	0	1	-3126.12
Amp.Topography.Fence	16	6273.00	207.45	0	1	-3119.94
Amp.WaterpointDensity	9	6273.23	207.68	0	1	-3127.43
Amp.RainfallLag1.Fence	13	6273.85	208.30	0	1	-3123.55
Amp.Length	9	6274.21	208.66	0	1	-3127.92
Amp.Beg	9	6277.29	211.74	0	1	-3129.46
Amp.RainfallLag2.Fence	13	6277.63	212.08	0	1	-3125.44
Amp.RiverDensity.Fence	13	6280.45	214.90	0	1	-3126.85
Amp.RainfallLag1	9	6284.89	219.35	0	1	-3133.26
Amp.Aspect	9	6285.16	219.61	0	1	-3133.40
Amp.RainfallLag2	9	6286.93	221.39	0	1	-3134.28
Amp.Elevation	9	6287.31	221.76	0	1	-3134.47
Amp.Rainfall	9	6288.09	222.54	0	1	-3134.86
Amp.Topography	10	6288.60	223.06	0	1	-3134.08

**Table E7.** The AIC comparison for LI standard deviation between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
LI.Global.Fence	46	9248.68	0	1	1	-4573.60
LI.Geology.Fence	16	9553.77	305.09	0	1	-4760.33
LI.Surface.Water.Fence	16	9558.01	309.33	0	1	-4762.44
LI.Global	20	9567.44	318.76	0	1	-4762.85
LI.Ownership.Fence	16	9580.13	331.45	0	1	-4773.51
LI.Amphibolitic.Fence	13	9612.00	363.32	0	1	-4792.63
LI.Rainfall.Fence	13	9612.97	364.29	0	1	-4793.11
LI.Gabbro.Fence	13	9614.84	366.16	0	1	-4794.05
LI.WaterpointDensity.Fence	13	9617.62	368.94	0	1	-4795.44
LI.Seasonality.Fence	16	9631.59	382.92	0	1	-4799.24
LI.Topography.Fence	16	9635.38	386.70	0	1	-4801.13
LI.RainfallLag3.Fence	13	9649.60	400.92	0	1	-4811.43
LI.Aspect.Fence	13	9650.93	402.25	0	1	-4812.09
LI.Beg.Fence	13	9652.48	403.80	0	1	-4812.87
LI.Length.Fence	13	9654.12	405.44	0	1	-4813.69
LI.Elevation.Fence	13	9656.19	407.51	0	1	-4814.72
LI.Geology	10	9658.05	409.37	0	1	-4818.80
LI.RainfallLag2.Fence	13	9659.09	410.41	0	1	-4816.17
LI.Surface.Water	10	9666.17	417.49	0	1	-4822.86
LI.RainfallLag1.Fence	13	9666.37	417.69	0	1	-4819.81
LI.Ownership	10	9666.82	418.15	0	1	-4823.19
LI.RiverDensity.Fence	13	9669.24	420.56	0	1	-4821.25
LI.Amphibolitic	9	9677.30	428.62	0	1	-4829.47
LI.Gabbro	9	9681.18	432.50	0	1	-4831.41
LI.RiverDensity	9	9681.86	433.18	0	1	-4831.75
LI.WaterpointDensity	9	9683.76	435.08	0	1	-4832.70
LI.RainfallLag3	9	9688.28	439.60	0	1	-4834.96
LI.Seasonality	10	9689.55	440.87	0	1	-4834.55
LI.Topography	10	9690.93	442.25	0	1	-4835.24
LI.RainfallLag2	9	9692.42	443.74	0	1	-4837.03
LI.Beg	9	9693.64	444.96	0	1	-4837.64
LI.Aspect	9	9693.81	445.13	0	1	-4837.72
LI.Length	9	9696.04	447.36	0	1	-4838.84
LI.Elevation	9	9696.53	447.85	0	1	-4839.08
LI.RainfallLag1	9	9697.98	449.30	0	1	-4839.81
LI.Rainfall	9	9699.84	451.16	0	1	-4840.74

**Table E8.** The AIC comparison for SI standard deviation between all *a priori* and global models.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Res.LL
SI.Global.Fence	46	8536.91	0	1	1	-4217.71
SI.Global	20	8827.27	290.37	0	1	-4392.77
SI.Geology.Fence	16	8837.71	300.80	0	1	-4402.30
SI.Surface.Water.Fence	16	8842.44	305.53	0	1	-4404.66
SI.Ownership.Fence	16	8867.63	330.72	0	1	-4417.26
SI.Gabbro.Fence	13	8893.51	356.60	0	1	-4433.38
SI.Amphibolitic.Fence	13	8893.68	356.77	0	1	-4433.47
SI.RiverDensity.Fence	13	8893.69	356.78	0	1	-4433.47
SI.WaterpointDensity.Fence	13	8896.93	360.02	0	1	-4435.09
SI.Seasonality.Fence	16	8900.63	363.72	0	1	-4433.76
SI.Topography.Fence	16	8919.34	382.43	0	1	-4443.11
SI.Beg.Fence	13	8927.08	390.17	0	1	-4450.17
SI.Length.Fence	13	8928.87	391.96	0	1	-4451.06
SI.RainfallLag3.Fence	13	8930.54	393.63	0	1	-4451.90
SI.Aspect.Fence	13	8933.16	396.25	0	1	-4453.21
SI.Elevation.Fence	13	8934.66	397.75	0	1	-4453.96
SI.Geology	10	8935.44	398.53	0	1	-4457.49
SI.Surface.Water	10	8942.16	405.25	0	1	-4460.86
SI.RainfallLag2.Fence	13	8945.49	408.59	0	1	-4459.37
SI.Ownership	10	8946.35	409.44	0	1	-4462.95
SI.RainfallLag1.Fence	13	8946.65	409.74	0	1	-4459.95
SI.Rainfall.Fence	13	8947.90	410.99	0	1	-4460.58
SI.Amphibolitic	9	8954.56	417.66	0	1	-4468.10
SI.Seasonality	10	8957.59	420.68	0	1	-4468.57
SI.Gabbro	9	8957.62	420.71	0	1	-4469.63
SI.WaterpointDensity	9	8958.66	421.75	0	1	-4470.15
SI.RiverDensity	9	8958.81	421.90	0	1	-4470.22
SI.Beg	9	8961.61	424.70	0	1	-4471.62
SI.RainfallLag3	9	8963.59	426.68	0	1	-4472.61
SI.Topography	10	8968.16	431.25	0	1	-4473.86
SI.Aspect	9	8971.13	434.22	0	1	-4476.38
SI.Elevation	9	8972.20	435.29	0	1	-4476.92
SI.Length	9	8973.41	436.50	0	1	-4477.52
SI.Rainfall	9	8974.92	438.01	0	1	-4478.28
SI.RainfallLag1	9	8975.02	438.11	0	1	-4478.33
SI.RainfallLag2	9	8975.24	438.33	0	1	-4478.44

## APPENDIX F: Summary statistics for welch two sample t-tests

**Table F1.** Summary statistics for welch two sample t-test comparing average Max between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	6020.23	6150.21	17.08	-0.38	0.712
	Open	5906.53	6694.21	12.84	-3.22	0.007
O2	Closed	6167.20	6059.10	15.17	0.30	0.772
	Open	5944.16	6741.33	10.21	-2.97	0.014
O3	Closed	6031.75	6380.94	33.56	-1.44	0.159
	Open	6296.84	6651.36	6.78	-1.36	0.217
O4	Closed	5643.97	6059.10	13.97	-0.99	0.338
	Open	6207.00	6741.33	31.92	-3.40	0.002
O5	Closed	5623.58	6059.10	13.99	-1.08	0.300
	Open	5929.77	6741.33	31.99	-5.00	<0.001
O6	Closed	5842.04	6059.10	13.79	-0.56	0.583
	Open	6255.23	6741.33	31.89	-3.10	0.004
O7	Closed	6256.77	6059.10	13.86	0.46	0.654
	Open	6744.17	6741.33	31.99	0.02	0.986
O8	Closed	6738.16	6150.21	15.97	1.50	0.153
	Open	7164.49	6694.21	24.98	2.24	0.034
O9	Closed	6755.39	6059.10	13.93	1.64	0.123
	Open	7121.18	6741.33	31.93	2.30	0.028
O10	Closed	6772.17	6059.10	14.00	1.73	0.105
	Open	7207.75	6741.33	31.82	3.00	0.005
C1	Closed	6633.44	5780.94	53.01	4.83	<0.001
C2	Closed	6925.64	6493.45	53.80	2.54	0.014
C3	Closed	5871.64	6493.45	52.96	-3.80	<0.001
C4	Closed	6702.78	6493.45	53.85	1.23	0.225
C5	Closed	6689.25	6493.45	53.98	1.13	0.263
C6	Closed	6944.07	6493.45	53.94	2.62	0.011
C7	Closed	7019.30	6493.45	52.07	3.28	0.002
C8	Closed	7099.95	6493.45	52.29	3.77	<0.001

**Table F2.** Summary statistics for welch two sample t-test comparing average Amp between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	2953.33	2911.92	19.59	0.12	0.904
	Open	3524.42	4184.19	18.03	-2.93	0.009
O2	Closed	3024.44	2810.48	17.73	0.63	0.538
	Open	3535.55	4136.94	11.95	-2.29	0.041
O3	Closed	3149.62	3356.54	32.98	-0.74	0.464
	Open	3782.53	4009.43	8.00	-0.94	0.377
O4	Closed	2569.43	2810.48	13.86	-0.65	0.527
	Open	3783.77	4136.94	28.32	-2.20	0.036
O5	Closed	2454.47	2810.48	13.99	-1.00	0.335
	Open	3478.90	4136.94	31.73	-3.68	0.001
O6	Closed	2657.77	2810.48	13.94	-0.45	0.662
	Open	3785.86	4136.94	30.13	-2.10	0.044
O7	Closed	2960.26	2810.48	13.67	0.39	0.702
	Open	4049.09	4136.94	31.92	-0.46	0.650
O8	Closed	3282.05	2911.92	15.88	1.13	0.277
	Open	4535.10	4184.19	24.78	1.39	0.178
O9	Closed	3205.33	2810.48	13.80	1.19	0.256
	Open	4292.75	4136.94	31.11	0.76	0.453
O10	Closed	3232.39	2810.48	13.37	1.32	0.209
	Open	4381.58	4136.94	31.50	1.22	0.230
C1	Closed	3589.82	3069.65	52.93	2.65	0.010
C2	Closed	3832.57	3640.89	53.02	0.90	0.375
C3	Closed	3168.69	3640.89	50.81	-2.31	0.025
C4	Closed	3680.45	3640.89	53.21	0.18	0.855
C5	Closed	3691.37	3640.89	53.92	0.23	0.822
C6	Closed	3856.27	3640.89	53.82	0.97	0.336
C7	Closed	3696.77	3640.89	53.18	0.26	0.796
C8	Closed	3859.63	3640.89	53.88	0.98	0.331

**Table F3.** Summary statistics for welch two sample t-test comparing average LI between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	3722.04	3674.48	13.50	0.11	0.913
	Open	3252.51	3637.53	11.01	-1.35	0.205
O2	Closed	3616.01	3641.84	10.56	-0.06	0.957
	Open	3227.67	3770.77	14.44	-2.37	0.032
O3	Closed	3419.19	3695.09	33.20	-1.00	0.327
	Open	3657.47	3779.21	7.89	-0.36	0.725
O4	Closed	3409.08	3641.84	13.53	-0.43	0.672
	Open	3287.62	3770.77	29.91	-2.87	0.008
O5	Closed	3493.47	3641.84	13.97	-0.26	0.799
	Open	3243.45	3770.77	31.93	-2.84	0.008
O6	Closed	3505.18	3641.84	13.87	-0.24	0.811
	Open	3460.29	3770.77	30.45	-1.81	0.080
O7	Closed	3769.78	3641.84	13.96	0.22	0.826
	Open	3846.43	3770.77	30.89	0.44	0.667
O8	Closed	3961.27	3674.48	15.91	0.57	0.574
	Open	3937.74	3637.53	25.77	1.51	0.143
O9	Closed	4149.14	3641.84	13.89	0.90	0.382
	Open	4161.06	3770.77	31.47	2.19	0.036
O10	Closed	4034.75	3641.84	13.96	0.69	0.502
	Open	4225.80	3770.77	31.85	2.31	0.027
C1	Closed	3896.42	3398.55	53.85	2.46	0.017
C2	Closed	4049.19	3700.73	53.55	1.82	0.074
C3	Closed	3328.82	3700.73	49.89	-2.11	0.040
C4	Closed	4006.07	3700.73	53.17	1.62	0.111
C5	Closed	3926.37	3700.73	50.17	1.27	0.209
C6	Closed	3408.50	3700.73	49.63	-1.22	0.226
C7	Closed	4218.00	3700.73	53.49	2.71	0.009
C8	Closed	4461.86	3700.73	51.91	4.17	<0.001

**Table F4.** Summary statistics for welch two sample t-test comparing average SI between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	1417.15	1379.89	15.16	0.15	0.882
	Open	1561.93	1895.27	11.59	-1.72	0.111
O2	Closed	1382.49	1345.06	11.33	0.14	0.888
	Open	1532.45	1922.09	10.74	-2.18	0.052
O3	Closed	1429.07	1560.49	31.92	-0.76	0.452
	Open	1821.86	1886.86	7.67	-0.42	0.687
O4	Closed	1232.59	1345.06	13.84	-0.37	0.719
	Open	1646.46	1922.09	27.96	-2.84	0.008
O5	Closed	1221.67	1345.06	14.00	-0.38	0.708
	Open	1537.97	1922.09	31.98	-3.41	0.002
O6	Closed	1281.63	1345.06	13.93	-0.20	0.842
	Open	1727.32	1922.09	28.94	-1.97	0.059
O7	Closed	1414.09	1345.06	13.96	0.21	0.838
	Open	1912.01	922.09	31.48	-0.09	0.926
O8	Closed	1528.96	1379.89	15.82	0.54	0.594
	Open	2102.00	1895.27	24.98	1.37	0.183
O9	Closed	1549.80	1345.06	13.94	0.66	0.523
	Open	2091.37	1922.09	32.00	1.48	0.148
O10	Closed	1526.09	1345.06	13.89	0.59	0.567
	Open	2154.11	1922.09	31.29	1.88	0.070
C1	Closed	1721.15	1430.25	2.32	53.99	0.024
C2	Closed	1854.23	1691.33	53.91	1.21	0.233
C3	Closed	1494.88	1691.33	48.92	-1.64	0.107
C4	Closed	1812.63	1691.33	53.98	0.89	0.377
C5	Closed	1779.72	1691.33	53.85	0.66	0.513
C6	Closed	1842.98	1691.33	53.57	1.15	0.256
C7	Closed	1765.47	1691.33	53.42	0.57	0.574
C8	Closed	1993.39	1691.33	53.71	2.11	0.039

**Table F5.** Summary statistics for welch two sample t-test comparing Max standard deviation between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	325.02	462.75	6.35	-1.91	0.102
	Open	270.21	473.21	6.48	-3.34	0.014
O2	Closed	392.97	478.05	7.00	-1.30	0.235
	Open	386.91	422.94	13.99	-0.61	0.554
O3	Closed	592.31	565.84	7.98	0.30	0.775
	Open	512.15	519.99	7.40	-0.10	0.920
O4	Closed	414.29	478.05	9.65	-0.89	0.396
	Open	265.41	422.94	10.49	-3.38	0.007
O5	Closed	319.58	478.05	13.05	-1.93	0.076
	Open	224.50	422.94	10.08	-4.31	0.002
O6	Closed	460.82	478.05	12.75	-0.21	0.835
	Open	335.72	422.94	8.96	-1.97	0.081
O7	Closed	306.35	478.05	10.17	-2.35	0.040
	Open	248.37	422.94	11.84	-2.25	0.044
O8	Closed	357.78	462.75	14.72	-1.41	0.179
	Open	290.44	473.21	7.93	-2.31	0.050
O9	Closed	375.83	478.05	12.83	-1.26	0.231
	Open	262.98	422.94	11.47	-3.31	0.007
O10	Closed	371.81	478.05	13.96	-1.12	0.284
	Open	212.18	422.94	11.97	-4.27	0.001
C1	Closed	354.90	385.55	27.24	-0.81	0.425
C2	Closed	392.63	454.61	31.42	-1.54	0.133
C3	Closed	341.03	454.61	26.59	-3.03	0.005
C4	Closed	421.02	454.61	33.91	-0.79	0.433
C5	Closed	319.89	454.61	33.17	-3.24	0.003
C6	Closed	297.33	454.61	30.17	-3.99	<0.001
C7	Closed	202.44	454.61	27.49	-6.66	0.000
C8	Closed	153.75	454.61	21.53	-8.59	<0.001

**Table F6.** Summary statistics for welch two sample t-test comparing the Amp standard deviation between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	279.44	418.15	1.97	-1.81	0.214
	Open	239.10	446.64	8.09	-3.84	0.005
O2	Closed	384.95	430.80	6.92	-0.88	0.408
	Open	363.03	396.83	13.21	-0.56	0.583
O3	Closed	489.92	540.89	7.81	-0.62	0.552
	Open	398.64	514.42	6.82	-1.51	0.176
O4	Closed	367.86	430.80	12.76	-1.14	0.277
	Open	248.52	396.83	9.61	-3.69	0.005
O5	Closed	279.39	430.80	12.10	-2.82	0.015
	Open	223.35	396.83	10.23	-4.22	0.002
O6	Closed	402.22	430.80	13.88	-0.47	0.645
	Open	294.44	396.83	10.24	-2.49	0.031
O7	Closed	328.05	430.80	13.22	-1.80	0.094
	Open	321.00	396.83	13.91	-1.51	0.153
O8	Closed	328.48	418.15	15.99	-1.54	0.142
	Open	278.69	446.64	7.90	-2.56	0.034
O9	Closed	379.03	430.80	13.99	-0.82	0.423
	Open	294.61	396.83	10.08	-2.50	0.031
O10	Closed	325.49	430.80	13.97	-1.62	0.127
	Open	188.88	396.83	12.20	-4.70	<0.001
C1	Closed	4.58	324.44	25.12	-0.62	0.538
C2	Closed	332.12	414.65	35.04	-2.11	0.042
C3	Closed	275.34	414.65	30.52	-4.66	<0.001
C4	Closed	344.58	414.65	35.25	-2.10	0.043
C5	Closed	318.04	414.65	35.12	-2.92	0.006
C6	Closed	276.42	414.65	35.23	-4.15	0.000
C7	Closed	194.31	414.65	34.15	-6.86	<0.001
C8	Closed	154.37	414.65	27.39	-9.12	<0.001

**Table F7.** Summary statistics for welch two sample t-test comparing the LI standard deviation between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	8481.24	15551.43	1.54	-1.71	0.265
	Open	5129.23	14003.26	8.65	-7.68	<0.001
O2	Closed	11278.09	16049.98	1.30	-0.85	0.524
	Open	7568.04	14127.31	12.89	-4.49	0.001
O3	Closed	8035.56	11249.45	7.54	-2.95	0.020
	Open	8889.90	9217.02	7.35	-0.21	0.836
O4	Closed	14808.82	16049.98	12.81	-0.50	0.624
	Open	8653.60	14127.31	11.25	-4.02	0.002
O5	Closed	11992.96	16049.98	13.38	-1.59	0.136
	Open	6353.43	14127.31	10.31	-5.91	<0.001
O6	Closed	13651.29	16049.98	13.49	-0.93	0.369
	Open	10743.30	14127.31	13.98	-2.00	0.066
O7	Closed	11930.06	16049.98	13.83	-1.54	0.147
	Open	6069.55	14127.31	11.56	-5.85	<0.001
O8	Closed	11197.77	15551.43	15.65	-1.81	0.090
	Open	8622.35	14003.26	6.14	-2.82	0.030
O9	Closed	16873.58	16049.98	10.56	0.19	0.852
	Open	9326.75	14127.31	13.55	-2.61	0.021
O10	Closed	14668.28	16049.98	10.19	-0.31	0.766
	Open	6326.16	14127.31	13.93	-4.86	<0.001
C1	Closed	11771.87	12853.24	29.34	-0.93	0.359
C2	Closed	13749.81	14565.16	28.58	-0.40	0.691
C3	Closed	10500.54	14565.16	35.35	-2.66	0.012
C4	Closed	12924.19	14565.16	23.84	-0.62	0.543
C5	Closed	9965.79	14565.16	28.04	-2.21	0.035
C6	Closed	9623.88	14565.16	34.52	-3.10	0.004
C7	Closed	8824.79	14565.16	28.48	-2.82	0.009
C8	Closed	7929.34	14565.16	28.19	-3.22	0.003

**Table F8.** Summary statistics for welch two sample t-test comparing the SI standard deviation between the reserves and KNP.

Reserve	Fence Status	Reserve Mean	KNP Mean	df	t value	Pr(> t )
O1	Closed	3906.31	8134.94	1.43	-2.24	0.202
	Open	2473.82	8543.77	4.78	-6.41	0.002
O2	Closed	5591.89	8332.33	1.47	-1.37	0.342
	Open	3900.90	8284.63	9.78	-5.12	<0.001
O3	Closed	5176.01	6899.95	4.87	-3.45	0.019
	Open	5142.39	6718.06	7.05	-1.42	0.197
O4	Closed	7255.29	8332.33	14.00	-0.91	0.379
	Open	5087.78	8284.63	12.10	-3.43	0.005
O5	Closed	5506.74	8332.33	13.62	-2.56	0.023
	Open	3818.79	8284.63	10.45	-5.10	<0.001
O6	Closed	6741.86	8332.33	13.41	-1.47	0.165
	Open	6261.87	8284.63	13.82	-1.94	0.073
O7	Closed	6262.94	8332.33	13.49	-1.56	0.143
	Open	4939.50	8284.63	13.83	-3.20	0.007
O8	Closed	6215.62	8134.94	14.43	-2.03	0.061
	Open	4951.97	8543.77	8.00	-2.82	0.023
O9	Closed	8033.73	8332.33	11.82	-0.19	0.853
	Open	5295.07	8284.63	13.38	-2.99	0.010
O10	Closed	7166.50	8332.33	13.13	-0.84	0.415
	Open	3325.42	8284.63	8.04	-6.14	<0.001
C1	Closed	5996.14	6843.30	31.12	-1.69	0.101
C2	Closed	7078.12	7996.21	35.89	-1.27	0.211
C3	Closed	5234.19	7996.21	35.89	-4.05	<0.001
C4	Closed	6536.93	7996.21	28.44	-1.45	0.158
C5	Closed	5563.64	7996.21	35.55	-3.67	0.001
C6	Closed	5128.12	7996.21	35.13	-3.76	0.001
C7	Closed	4276.54	7996.21	34.78	-4.79	<0.001
C8	Closed	3752.66	7996.21	35.94	-5.93	<0.001

# APPENDIX G: Top ten most-supported models for each response variable

**Table G1.** Top ten most supported models for average Max.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	7051.37	7055.99	7056.50	7058.52	7058.61	7059.74	7060.15	7061.10	7062.44	7063.16
delta	0.00	4.62	5.13	7.15	7.24	8.37	8.78	9.73	11.07	11.78
weight	0.78	0.08	0.06	0.02	0.02	0.01	0.01	0.01	0.00	0.00
df	38.00	40.00	36.00	36.00	36.00	37.00	35.00	38.00	38.00	38.00
logLik	-3484.49	-3484.45	-3489.39	-3490.40	-3490.45	-3489.85	-3492.38	-3489.36	-3490.03	-3490.38
(Intercept)	2589.06	2676.98	2733.90	2247.54	3874.63	2137.27	4766.42	2833.03	2328.80	4009.92
Amphibolitic	498.64	475.89	352.95	570.75	407.64	635.13	717.16	327.48	547.49	375.96
Aspect	-3.74	-3.76	-4.45	-2.88	-9.33	-3.33	-7.85	-4.50	-2.94	-9.54
BegMean	-5.03	-5.15	-5.04	-4.87	-5.03	-4.49	-4.99	-5.15	-4.96	-5.15
Elevation	2.85	2.90	3.21	2.84	2.58	2.59	NA	3.26	2.89	2.62
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	4499.25	4519.34	4564.44	4576.32	3775.47	4384.69	2478.85	4578.10	4589.19	3772.90
LengthMean	-2.30	-2.32	-2.30	-2.06	-2.31	-2.17	-2.29	-2.33	-2.05	-2.34
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag1	-0.31	-0.38	-0.32	-0.31	-0.31	NA	-0.31	-0.38	-0.38	-0.38
RiverDensity	-4036.24	-4188.28	-4817.83	-3893.23	-4729.04	-3499.73	-5112.44	-5001.03	-4036.61	-4912.92
WaterpointDensity	-1171.05	-1173.72	-1325.80	-959.78	-1466.38	-1002.60	-1078.49	-1331.02	-967.28	-1495.16
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	NA	+	+	+	+	NA
BegMean:FenceStatus	+	+	+	+	+	+	+	+	+	+
Elevation:FenceStatus	+	+	NA	+	+	+	NA	NA	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	+	+	NA	+	+	+	+	NA	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag1	NA	+	NA	NA	NA	NA	NA	+	+	+
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G2.** Top ten most supported models for average Amp.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	6968.08	6970.32	6972.88	6974.34	6975.47	6975.73	6976.66	6977.65	6977.85	6978.23
delta	0.00	2.24	4.81	6.27	7.40	7.65	8.59	9.57	9.78	10.16
weight	0.64	0.21	0.06	0.03	0.02	0.01	0.01	0.01	0.00	0.00
df	38.00	37.00	40.00	36.00	36.00	35.00	35.00	34.00	35.00	36.00
logLik	-3442.84	-3445.14	-3442.89	-3448.31	-3448.88	-3450.17	-3450.63	-3452.28	-3451.23	-3450.26
(Intercept)	1844.40	1622.48	1923.66	1795.01	3295.99	2656.38	1601.92	2309.21	3072.43	1508.98
Amphibolitic	541.21	594.61	522.51	490.71	377.59	682.06	551.89	727.58	431.85	518.81
Aspect	-2.16	-2.09	-2.22	-2.14	-8.16	-3.54	-2.13	-3.21	-8.09	-2.17
BegMean	-3.69	-3.31	-3.78	-3.70	-3.70	-3.65	-3.32	-3.28	-3.32	-2.48
Elevation	1.26	1.10	1.29	1.49	1.01	NA	1.28	NA	0.85	1.20
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	3011.37	2920.68	3017.21	3117.02	2259.04	2150.80	2991.61	2172.85	2164.51	3016.84
LengthMean	-1.72	-1.63	-1.74	-1.72	-1.74	-1.70	-1.63	-1.61	-1.66	-1.46
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag1	-0.22	NA	-0.27	-0.22	-0.22	-0.21	NA	NA	NA	-0.24
RiverDensity	-1945.06	-1832.78	-2081.79	-2284.84	-2886.58	-2256.57	-2174.46	-2094.25	-2739.93	-2247.59
WaterpointDensity	-486.22	-433.16	-489.26	-488.34	-962.98	-329.03	-425.45	-262.81	-918.37	-438.75
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	NA	+	+	+	NA	+
BegMean:FenceStatus	+	+	+	+	+	+	+	+	+	NA
Elevation:FenceStatus	+	+	+	NA	+	NA	NA	NA	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	+	+	+	+	+	+	+	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag1	NA	NA	+	NA	NA	NA	NA	NA	NA	NA
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G3.** Top ten most supported models for average LI.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	7613.43	7616.01	7617.77	7619.56	7619.90	7620.26	7620.37	7621.38	7622.53	7622.60
delta	0.00	2.57	4.34	6.13	6.46	6.83	6.94	7.95	9.10	9.17
weight	0.61	0.17	0.07	0.03	0.02	0.02	0.02	0.01	0.01	0.01
df	40.00	42.00	38.00	39.00	40.00	38.00	38.00	38.00	41.00	37.00
logLik	-3763.17	-3762.08	-3767.69	-3767.41	-3766.40	-3768.94	-3768.99	-3769.50	-3766.53	-3771.28
(Intercept)	1057.99	1043.14	1296.77	1094.65	1247.99	694.37	1104.91	1217.35	1076.40	-23.51
Amphibolitic	178.84	186.21	154.28	107.57	169.11	167.43	191.70	178.20	110.25	160.45
Aspect	2.26	2.32	2.10	2.72	2.18	2.17	2.21	1.47	2.82	2.32
BegMean	-3.36	-3.38	-3.62	-3.73	-3.63	-2.15	-3.35	-3.36	-3.75	NA
Elevation	3.39	3.38	3.42	3.71	3.40	3.31	3.28	3.36	3.72	3.21
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	4751.31	4756.27	4760.21	4941.31	4754.21	4750.90	4688.29	4661.60	4960.70	4808.45
LengthMean	-4.84	-4.84	-5.31	-4.60	-5.23	-4.56	-4.84	-4.84	-4.61	-4.16
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag1	-0.61	-0.62	-0.62	-0.57	-0.62	-0.63	-0.61	-0.61	-0.57	-0.58
RainfallLag2	0.49	0.50	0.49	NA	0.50	0.49	0.49	0.49	NA	0.51
RainfallLag3	-0.91	-0.89	-0.94	-0.85	-0.90	-0.88	-0.90	-0.91	-0.85	-0.93
RiverDensity	-717.87	-705.69	-777.20	-776.72	-753.05	-766.37	-686.86	-756.55	-766.67	-776.46
WaterpointDensity	-673.88	-662.45	-712.77	-661.47	-694.74	-682.84	-668.36	-704.12	-649.70	-655.80
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	+	+	+	NA	+	+
BegMean:FenceStatus	+	+	+	+	+	NA	+	+	+	NA
Elevation:FenceStatus	+	+	+	+	+	+	NA	+	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	+	NA	+	NA	+	+	+	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	NA	+	NA	NA	+	NA	NA	NA	+	NA
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G4.** Top ten most supported models for average SI.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	6611.53	6613.51	6614.50	6615.56	6616.49	6616.57	6616.73	6618.42	6618.55	6618.59
delta	0.00	1.98	2.97	4.04	4.96	5.04	5.20	6.89	7.02	7.07
weight	0.44	0.16	0.10	0.06	0.04	0.04	0.03	0.01	0.01	0.01
df	36.00	34.00	34.00	33.00	32.00	34.00	33.00	33.00	31.00	32.00
logLik	-3266.91	-3270.21	-3270.71	-3272.39	-3274.00	-3271.74	-3272.97	-3273.82	-3276.17	-3275.05
(Intercept)	-971.56	-1050.22	-1021.16	-796.41	-1099.92	-406.44	-639.90	-286.12	-846.26	-480.99
Amphibolitic	358.89	356.15	352.40	374.63	350.55	301.90	332.01	482.82	369.82	298.35
Aspect	1.44	1.44	1.54	1.44	1.54	-0.94	NA	0.33	1.55	-0.95
BegMean	0.69	0.98	0.69	NA	0.98	0.69	0.68	0.75	NA	0.98
Elevation	1.17	1.16	1.24	1.20	1.22	1.07	1.12	NA	1.25	1.05
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	1818.63	1827.02	1866.54	1787.81	1874.87	1512.98	1643.59	1056.59	1834.30	1518.96
Ownership	+	+	+	+	+	+	+	+	+	+
Rainfall	0.21	0.20	0.21	0.18	0.20	0.21	0.21	0.21	0.19	0.20
RainfallLag3	-0.25	-0.24	-0.25	-0.21	-0.24	-0.25	-0.25	-0.25	-0.21	-0.24
RiverDensity	-245.09	-255.57	-213.26	-215.87	-219.62	-593.28	-481.36	-167.70	-180.81	-609.00
WaterpointDensity	152.14	152.31	142.41	159.39	143.77	-18.79	78.38	203.55	152.73	-20.17
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	+	NA	NA	+	+	NA
BegMean:FenceStatus	+	NA	+	NA	NA	+	+	+	NA	NA
Elevation:FenceStatus	+	+	NA	+	NA	+	+	NA	NA	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G5.** Top ten most supported models for Max standard deviation.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	6118.50	6118.51	6119.94	6119.99	6120.23	6120.24	6123.77	6124.91	6125.24	6125.55
delta	0.00	0.01	1.44	1.49	1.73	1.74	5.27	6.41	6.74	7.05
weight	0.25	0.25	0.12	0.12	0.11	0.10	0.02	0.01	0.01	0.01
df	32.00	30.00	30.00	28.00	29.00	27.00	34.00	32.00	32.00	31.00
logLik	-3025.00	-3027.29	-3028.00	-3030.28	-3029.27	-3031.53	-3025.34	-3028.21	-3028.37	-3029.67
(Intercept)	-575.82	-514.38	-547.75	-482.88	-398.58	-349.87	-567.56	-496.36	-541.59	-380.03
Amphibolitic	312.78	305.58	310.50	303.21	290.44	285.39	312.90	303.61	311.03	289.73
Aspect	0.81	0.76	0.67	0.60	NA	NA	0.86	0.78	0.73	NA
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	400.33	404.45	386.01	388.55	318.65	328.44	410.32	410.48	396.73	323.68
LengthMean	1.08	0.91	1.08	0.91	1.08	0.91	1.05	0.87	1.05	1.05
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag3	0.17	0.15	0.17	0.15	0.17	0.15	0.14	0.13	0.14	0.14
RiverDensity	-347.80	-359.74	-343.25	-354.07	-460.98	-458.20	-341.35	-359.22	-331.15	-456.56
WaterpointDensity	82.57	70.12	69.49	55.87	20.62	12.01	91.86	73.26	78.66	26.10
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	NA	NA	NA	NA	+	+	NA	NA
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	NA	+	NA	+	NA	+	NA	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	NA	NA	NA	NA	NA	NA	+	+	+	+
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G6.** Top ten most supported models for Amp standard deviation.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	6018.09	6018.48	6018.91	6019.28	6019.56	6019.98	6020.48	6020.76	6021.41	6021.63
delta	0.00	0.40	0.82	1.19	1.48	1.89	2.40	2.67	3.32	3.54
weight	0.17	0.14	0.11	0.09	0.08	0.07	0.05	0.05	0.03	0.03
df	35.00	33.00	32.00	30.00	33.00	31.00	34.00	37.00	31.00	34.00
logLik	-2971.34	-2973.85	-2975.21	-2977.67	-2974.39	-2976.88	-2973.70	-2970.35	-2977.60	-2974.27
(Intercept)	249.33	217.99	166.60	151.51	135.37	106.02	442.18	219.76	344.36	146.75
Amphibolitic	222.30	228.78	231.24	235.64	234.56	240.80	215.76	225.30	227.15	232.61
Aspect	-0.40	-0.32	NA	NA	0.15	0.21	-0.47	-0.36	NA	NA
BegMean	-1.53	-1.53	-1.53	-1.52	-1.53	-1.52	-1.78	-1.50	-1.78	-1.50
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	560.82	570.30	603.64	605.48	617.24	625.42	563.11	564.73	613.26	603.43
LengthMean	0.40	0.43	0.40	0.43	0.40	0.43	NA	0.45	NA	0.45
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag3	0.22	0.23	0.22	0.23	0.22	0.23	0.21	0.22	0.21	0.22
RiverDensity	-299.15	-281.93	-282.31	-275.69	-262.88	-244.30	-314.55	-292.96	-288.48	-283.69
WaterpointDensity	-57.84	-43.79	-24.05	-15.78	-14.49	-1.98	-72.06	-50.18	-31.11	-20.25
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	NA	NA	NA	NA	+	+	NA	NA
BegMean:FenceStatus	+	+	+	+	+	+	+	+	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	NA	NA	NA	NA	NA	NA	NA	+	NA	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	+	NA	+	NA	+	NA	+	+	+	+
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G7.** Top ten most supported models for LI standard deviation.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	9278.83	9279.32	9281.45	9281.65	9288.09	9288.28	9290.07	9290.15	9290.35	9290.38
delta	0.00	0.49	2.63	2.82	9.27	9.45	11.24	11.32	11.52	11.55
weight	0.43	0.33	0.12	0.10	0.00	0.00	0.00	0.00	0.00	0.00
Df	42.00	43.00	41.00	42.00	40.00	41.00	42.00	41.00	40.00	41.00
logLik	-4593.49	-4592.54	-4595.99	-4594.90	-4600.50	-4599.40	-4599.11	-4600.34	-4601.62	-4600.45
(Intercept)	14272.11	14404.63	13063.73	13336.31	13326.96	13590.92	13936.32	13674.86	12612.36	13334.70
Amphibolitic	8901.73	8858.27	8972.84	8900.84	8818.07	8751.30	9245.09	9342.26	8806.68	9243.71
Aspect	-4.98	-4.46	-7.00	-6.04	-1.78	-1.16	-8.23	-9.43	-2.83	-8.99
BegMean	-37.99	-38.46	-34.34	-35.30	-36.30	-37.11	-34.89	-33.85	-34.21	-33.21
Elevation	-3.14	-2.86	-4.76	-4.22	-2.36	-1.95	-5.47	-6.12	-3.33	-6.12
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	9237.93	9491.56	8494.52	8943.28	10234.98	10584.33	8012.52	7453.83	9979.19	7775.06
LengthMean	-10.01	-10.16	-8.26	-8.59	-10.76	-10.96	-10.97	-10.70	-9.40	-9.96
Ownership	+	+	+	+	+	+	+	+	+	+
Rainfall	-1.67	-1.61	NA	NA	-1.62	-1.53	-0.93	-1.03	NA	NA
RainfallLag1	NA	-0.42	NA	-0.66	NA	-0.65	-0.81	NA	-0.88	-0.93
RainfallLag2	-3.18	-3.13	-2.84	-2.78	-3.15	-3.06	NA	NA	-2.73	NA
RainfallLag3	5.21	5.14	4.90	4.81	4.23	4.16	4.70	4.83	3.94	4.53
RiverDensity	-4588.95	-4511.44	-4835.10	-4691.89	-4159.03	-4070.60	-4783.28	-4958.23	-4267.12	-4887.13
WaterpointDensity	-1247.97	-1213.07	-1377.22	-1310.56	-917.70	-884.84	-1398.82	-1483.95	-987.87	-1455.34
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	+	+	+	+	+	+
BegMean:FenceStatus	+	+	+	+	+	+	+	+	+	+
Elevation:FenceStatus	+	+	+	+	+	+	+	+	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	+	+	+	+	+	+	+	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	+	+	+	+	NA	NA	+	+	NA	+
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+

**Table G8.** Top ten most supported models for SI standard deviation.

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10
AICc	8584.81	8585.60	8586.24	8587.69	8589.84	8590.79	8591.02	8592.63	8595.11	8596.31
delta	0.00	0.78	1.42	2.88	5.03	5.97	6.21	7.82	10.30	11.50
weight	0.38	0.26	0.19	0.09	0.03	0.02	0.02	0.01	0.00	0.00
df	42.00	41.00	41.00	40.00	40.00	39.00	39.00	38.00	40.00	39.00
logLik	-4246.48	-4248.06	-4248.38	-4250.30	-4251.37	-4253.02	-4253.14	-4255.12	-4254.01	-4255.79
(Intercept)	8752.15	8409.52	8767.48	8393.95	8734.90	8688.88	8259.06	8154.94	8400.70	7757.44
Amphibolitic	5005.08	5097.01	5104.72	5215.99	5045.31	5141.48	5147.93	5266.09	4994.66	5097.78
Aspect	-2.63	-3.69	-3.57	-4.87	-2.51	-3.16	-3.21	-3.98	-2.27	-3.03
BegMean	-24.41	-23.32	-23.81	-22.54	-24.60	-24.01	-23.37	-22.59	-23.55	-21.80
Elevation	-1.88	-2.50	-2.45	-3.19	-1.95	-2.44	-2.54	-3.15	-1.83	-2.47
FenceStatus	+	+	+	+	+	+	+	+	+	+
Gabbro	6205.34	5690.39	5844.94	5232.08	6184.51	5902.23	5742.40	5386.91	6330.88	5872.65
LengthMean	-8.16	-7.76	-8.55	-8.15	-8.12	-8.50	-7.74	-8.13	-7.99	-7.45
Ownership	+	+	+	+	+	+	+	+	+	+
RainfallLag1	-0.76	NA	-0.84	NA	-0.80	-0.88	NA	NA	-0.80	NA
RainfallLag2	-0.82	-0.89	NA	NA	-0.76	NA	-0.84	NA	-0.85	-0.93
RainfallLag3	1.97	2.08	1.89	2.01	2.07	1.98	2.14	2.05	1.95	2.06
RiverDensity	-2508.68	-2602.30	-2568.38	-2680.80	-2528.78	-2564.30	-2584.55	-2625.86	-2424.27	-2481.05
WaterpointDensity	260.74	193.90	208.69	125.89	310.48	287.01	286.76	259.97	283.59	253.68
Amphibolitic:FenceStatus	+	+	+	+	+	+	+	+	+	+
Aspect:FenceStatus	+	+	+	+	+	+	+	+	+	+
BegMean:FenceStatus	+	+	+	+	+	+	+	+	NA	NA
Elevation:FenceStatus	+	+	+	+	+	+	+	+	+	+
FenceStatus:Gabbro	+	+	+	+	+	+	+	+	+	+
FenceStatus:LengthMean	+	+	+	+	+	+	+	+	+	+
FenceStatus:Ownership	+	+	+	+	+	+	+	+	+	+
FenceStatus:RainfallLag3	+	+	+	+	NA	NA	NA	NA	+	+
FenceStatus:RiverDensity	+	+	+	+	+	+	+	+	+	+
FenceStatus:WaterpointDensity	+	+	+	+	+	+	+	+	+	+