# Determining the long-term effectiveness of an Invasive Alien Plant removal strategy along the Bakwena Toll Route in Gauteng and North-West, South Africa

by

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# DECLARATION

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# ABSTRACT

Alien invasive plants (AIPs) are major drivers of biodiversity loss, and roads often act as effective corridors for their spread. This study analysed long-term changes in AIPs along the Bakwena N4 Toll Route, South Africa, using annual audit data collected between 2011 and 2024. Biodiversity indices and statistical analyses were used to assess rates of change, identify persistent invasion hotspots, and predict future trends. The study further compared invasion dynamics between the open, grass-dominated Marikana Thornveld and the denser woody Savanna vegetation types along the route, examined patterns across urban, peri-urban, and rural land-use zones, and evaluated the influence of vegetation quality and soil type on AIP presence. Results showed clear contrasts between land-use zones and vegetation structures. Urban areas displayed fluctuating but broadly stable AIP counts, with no significant long-term decline, suggesting continued reinvasion linked to disturbance. Peri-urban areas remained persistent hotspots, supporting high diversity and richness despite management interventions. Rural areas showed significant reductions in AIP densities, although diversity trends were more variable, indicating ongoing species turnover. Plots within the more open Marikana Thornveld differed from those in the other, generally more wooded or structurally complex vegetation types, indicating that vegetation structure may influence invasion patterns along the route. Degraded vegetation quality was strongly associated with invasion hotspots, while soil type showed limited explanatory power. Similarity and dissimilarity indices revealed high turnover in urban assemblages, relative stability in peri-urban environments, and gradual compositional shifts in rural areas. The findings indicate that management has reduced AIP densities in rural sections of the route, but has been less effective in urban and peri-urban areas. Persistent reinvasion in disturbed areas highlights the need for targeted, adaptive management. Sustained monitoring, restoration, and context-specific interventions will be important for maintaining ecological integrity along the Bakwena N4 Toll Route.

## KEY WORDS

Alien invasive plants (AIPs), biological invasions, biodiversity loss, roads as dispersal corridors, road ecology, transport infrastructure, Bakwena N4 Toll Route, Savanna vegetation, Marikana Thornveld, vegetation structure, urban invasions, peri-urban invasions, rural invasions, long-term monitoring, species diversity indices, invasion dynamics

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# CHAPTER 1
# INTRODUCTION

Alien invasive plants (AIPs) are recognised as one of the most serious threats to biodiversity worldwide, altering ecosystem structure, reducing native species richness, and increasing the costs of land management and rehabilitation (Richardson *et al.*, 2000; van Wilgen *et al.*, 2004). Alien invasive plants are generally defined as plant species that have been introduced outside their natural range, where they establish self-sustaining populations, spread rapidly, and cause ecological, economic, or social harm (Richardson *et al.*, 2000). In South Africa, where high levels of biodiversity coincide with diverse socio-economic pressures, AIPs have been shown to significantly alter ecological functioning while also imposing economic burdens on conservation agencies, local authorities, and infrastructure managers (van Wilgen *et al.*, 2012). Roads are of particular importance in this regard. Linear infrastructure acts as a pathway for invasion by creating disturbed habitats, providing dispersal corridors, and intersecting ecosystems that would otherwise remain relatively isolated (Forman & Alexander, 1998; Kalwij *et al.*, 2008).

The Bakwena N4 Toll Route provides an appropriate case study through which these processes can be examined in practice. Extending over 300 km, the route traverses a range of Savanna vegetation types and passes through urban, peri-urban, and rural land-use settings. This corridor not only supports national and regional economic connectivity, but also passes through nature reserves, agricultural land, mining areas, and rapidly expanding settlements. These varied environmental and land-use conditions are likely to create multiple opportunities for alien species to establish and spread along the corridor (Forman & Alexander, 1998; Kalwij *et al.*, 2008). Despite the ecological and management importance of the route, published long-term assessments of AIP dynamics specific to this corridor appear to be limited.

This study is motivated by an apparent lack of empirical, long-term, and vegetation-specific evidence on the spread and impact of AIPs along major transport routes in

South Africa. Although the National Environmental Management: Biodiversity Act (Act 10 of 2004) requires the control and management of invasive species, implementation may be constrained by limited long-term monitoring data and site-specific ecological information. Previous studies have shown that vegetation context, disturbance intensity, and surrounding land use are important drivers of invasion (Henderson & Wilson, 2017). However, there remains a limited integrated understanding of how these factors interact within a single transport corridor spanning multiple vegetation settings and land-use contexts. Without structured, plot-based monitoring across contrasting vegetation groups and land-use zones, evaluation of invasion patterns and management effectiveness is likely to remain constrained.

The aim of this research is to assess the long-term dynamics and spatial patterns of AIPs along the Bakwena N4 Toll Route, with particular emphasis on vegetation structure, land-use context, and site condition. To achieve this aim, the study addresses five objectives. The first is to assess long-term changes in the abundance, diversity, and distribution of AIPs between 2011 and 2024. The second is to compare invasion patterns between plots located within the open, grass-dominated Marikana Thornveld and plots located within the denser woody Savanna vegetation types along the route, and to evaluate differences across urban, peri-urban, and rural land-use zones. The third is to evaluate vegetation condition at the plot level and test whether lower vegetation quality is associated with higher AIP presence. The fourth is to apply a suite of biodiversity indices, including the Shannon–Wiener Index, Pielou’s Evenness Index, Margalef’s Index, Bray–Curtis dissimilarity, and Sørensen’s similarity, to characterise diversity, evenness, and assemblage turnover. The fifth is to identify invasion hotspots and problem areas along the route, and to provide evidence-based recommendations to support the long-term management of AIPs along Bakwena and comparable road networks.

In order to address the research objectives, systematic plot-based surveys were conducted along the Bakwena N4 Toll Route. Plots were established within both road reserve areas and adjacent natural habitats, and were distributed across the open, grass-dominated Marikana Thornveld and the other, generally more woody or structurally complex vegetation types present along the route, as well as across urban, peri-urban, and rural zones. Species were identified and classified in accordance with SANBI's alien plant lists. Data analysis included descriptive statistics, non-parametric tests, and the application of biodiversity indices to detect spatial and ecological patterns. Vegetation condition was further assessed using plant counts, cover-abundance estimates, and visual vegetation quality indicators of both indigenous vegetation and alien invasive plants, which provided an indication of vegetation structure, ecosystem resilience, and disturbance.

The dissertation is organised into seven chapters. Chapter 1 introduces the study by outlining the background, research problem, objectives, and overall approach. Chapter 2 sets out the study area, describing the extent of the Bakwena N4 Toll Route, the vegetation types it traverses, the associated land-use zones. Chapter 3 reviews the literature, focusing on disturbance, land use, and roads as invasion corridors, and how these factors influence the establishment and spread of alien species. Chapter 4 details the methodology, including the annual audit process, the subdivision of the route into 4 km quadrants, the calculation of diversity and similarity indices, hotspot flagging using thresholds, and the application of statistical tests. Chapter 5 assesses long-term changes in alien invasive plants between 2011 and 2024, examining trends in abundance, diversity, and distribution, projecting future invasion dynamics, and identifying persistent problem areas across urban, peri-urban, and rural zones (addressing objectives 1, 4, and 5). Chapter 6 investigates the spatial distribution of alien invasive plants across vegetation and land-use types, incorporating plot-level comparisons between road reserves and adjacent natural areas, and testing links between vegetation quality and invasion hotspots (addressing objectives 2 and 3). Finally, Chapter 7 discusses the findings in relation to the wider literature, draws out ecological and management implications, and concludes the dissertation with key outcomes, limitations, and recommendations for future management and research.

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# CHAPTER 2
# STUDY AREA

## 2.1 Overview of the Bakwena N4 Toll Route

The study area comprises the full length of the Bakwena N4 Toll Route, which stretches approximately 300 kilometres from the Doornpoort Toll Plaza in the City of Tshwane Metropolitan Municipality to the Skilpadshek Border Post adjacent to Botswana (Figure 2.1). This toll road is a strategic part of South Africa's national infrastructure, forming part of the Maputo Development Corridor, a regional transport initiative that facilitates trade and mobility between Gauteng and Mozambique via the N4 and between Gauteng and Botswana via the western link (DBSA, 2014; World Bank, 2010). The route serves as a major economic artery linking inland markets to neighbouring countries and traverses multiple ecological and socio-economic zones. Its linear continuity and exposure to variable disturbance gradients make it an ideal system for studying spatial patterns of alien invasive plant (AIP) colonisation along infrastructure corridors (Foxcroft *et al.*, 2004; Hulme, 2009).

The N4 crosses a wide spectrum of land uses and environmental contexts. Beginning in the densely built-up peri-urban zone of Pretoria North, the route passes through the Onderstepoort Nature Reserve, known for its role in local biodiversity conservation, and Akasia. Westward from Pretoria, the road passes through smallholdings, airstrips, scattered residential areas, and sections of open veld. These transitional zones reflect a mix of formal and informal land use, with varying degrees of land cover and disturbance (Driver *et al.*, 2005). As the route continues, it intersects several high-impact zones near the Elands Platinum Mine and the towns of Brits and Bapong, followed by extensive open-pit mining operations near Tharisa and Kroondal. These areas exhibit intensive landscape modification, soil compaction, and associated edge effects (Milton *et al.*, 2003).

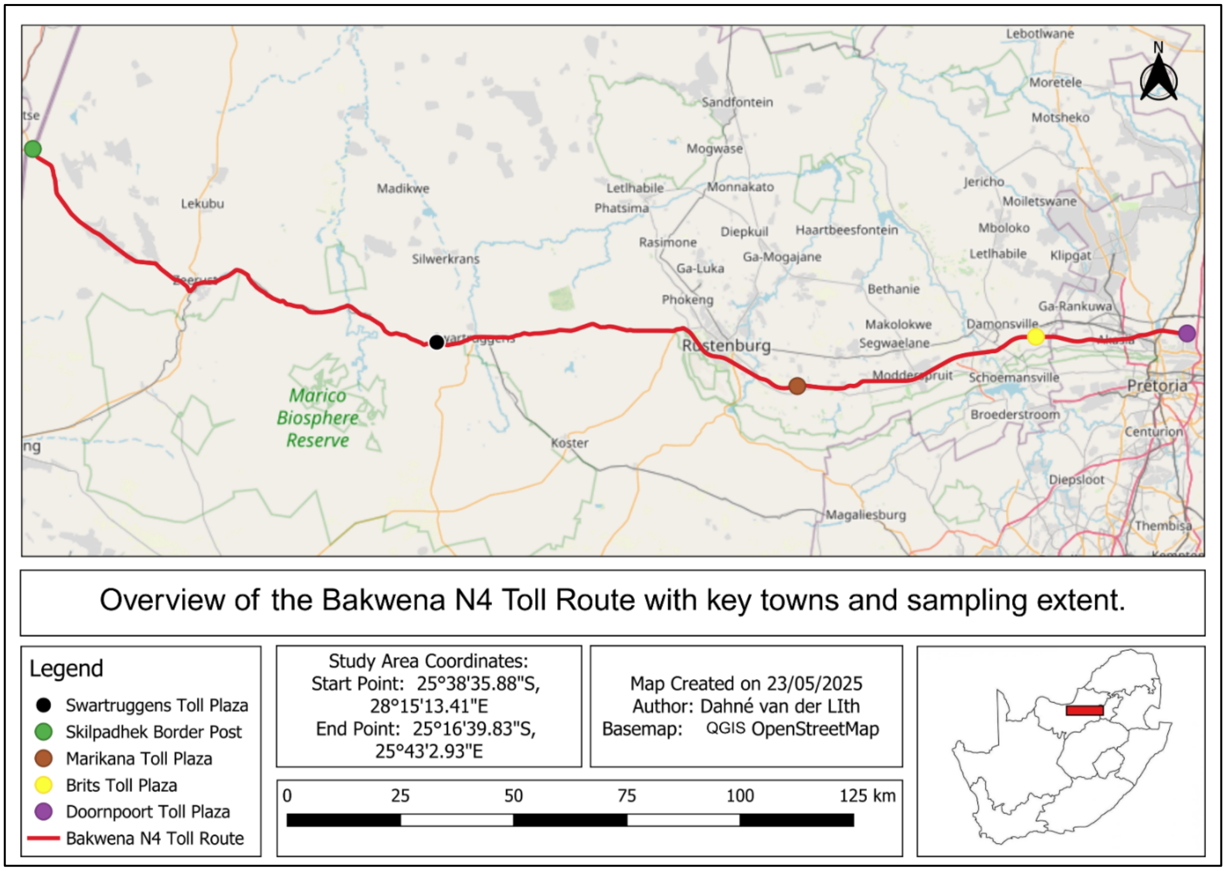


Figure 2.1 Overview of the Bakwena N4 Toll Route with key towns.

Beyond Rustenburg, the corridor enters a more rural landscape characterised by broader conservation matrices and large-scale agricultural holdings. The route skirts the Kgaswane Mountain Nature Reserve, transitions through the towns of Swartruggens and Groot Marico, and then passes through the Marico Biosphere Reserve, a nationally recognised biosphere region under the Man and the Biosphere (MAB) programme (UNESCO, 2018). Finally, the corridor reaches the town of Zeerust before terminating at the Skilpadshek Border Post. These later sections are comparatively less developed and offer an opportunity to examine AIP spread in low-disturbance zones.

The route spans a broad environmental gradient across the Savanna Biome, with variation in elevation, slope, geology, rainfall, and land use influencing the distribution of soil types, vegetation structure, and local ecological conditions (Mucina &

Rutherford, 2006). The corridor traverses a range of Savanna vegetation types, including Marikana Thornveld, Moot Plains Bushveld, Gold Reef Mountain Bushveld, Dwarsberg-Swartruggens Mountain Bushveld, and Zeerust Thornveld. These vegetation units differ in structure, with some sections being more open and grass-dominated and others more wooded, shrubby, or topographically complex, allowing for comparison of AIP dynamics across contrasting Savanna vegetation settings (Le Maitre *et al.*, 2011). Frequent transitions between natural, semi-natural, and highly transformed areas create disturbed ecological edges that may act as invasion hotspots (Donaldson *et al.*, 2014; van Wilgen *et al.*, 2020).

As a heavily trafficked freight and passenger corridor, the N4 experiences ongoing anthropogenic pressure, including vegetation clearance, verge mowing, stormwater discharge, dumping, and mechanical edge degradation (Baard & Kraaij, 2019). These activities, especially when combined with high propagule pressure from adjacent urban gardens, informal settlements, or construction sites, create ideal conditions for AIP establishment and spread (Meunier & Lavoie, 2012). Long-distance dispersal of seeds and vegetative fragments is also facilitated by traffic, especially trucks and construction vehicles, which can act as unintentional vectors (von der Lippe & Kowarik, 2007).

### 2.2 Savanna Vegetation Types

The Bakwena N4 Toll Route traverses a range of vegetation types within the Savanna Biome, commonly referred to as Bushveld (Figure 2.2). These include Marikana Thornveld, Moot Plains Bushveld, Gold Reef Mountain Bushveld, Dwarsberg-Swartruggens Mountain Bushveld, and Zeerust Thornveld (SANBI, 2017). While these vegetation types all fall within the Savanna Biome, they differ in floristic composition, vegetation structure, topographic setting, and disturbance context. These differences provide a useful basis for analysing variation in AIP patterns along the route (Mucina & Rutherford, 2006; Rouget *et al.*, 2015).

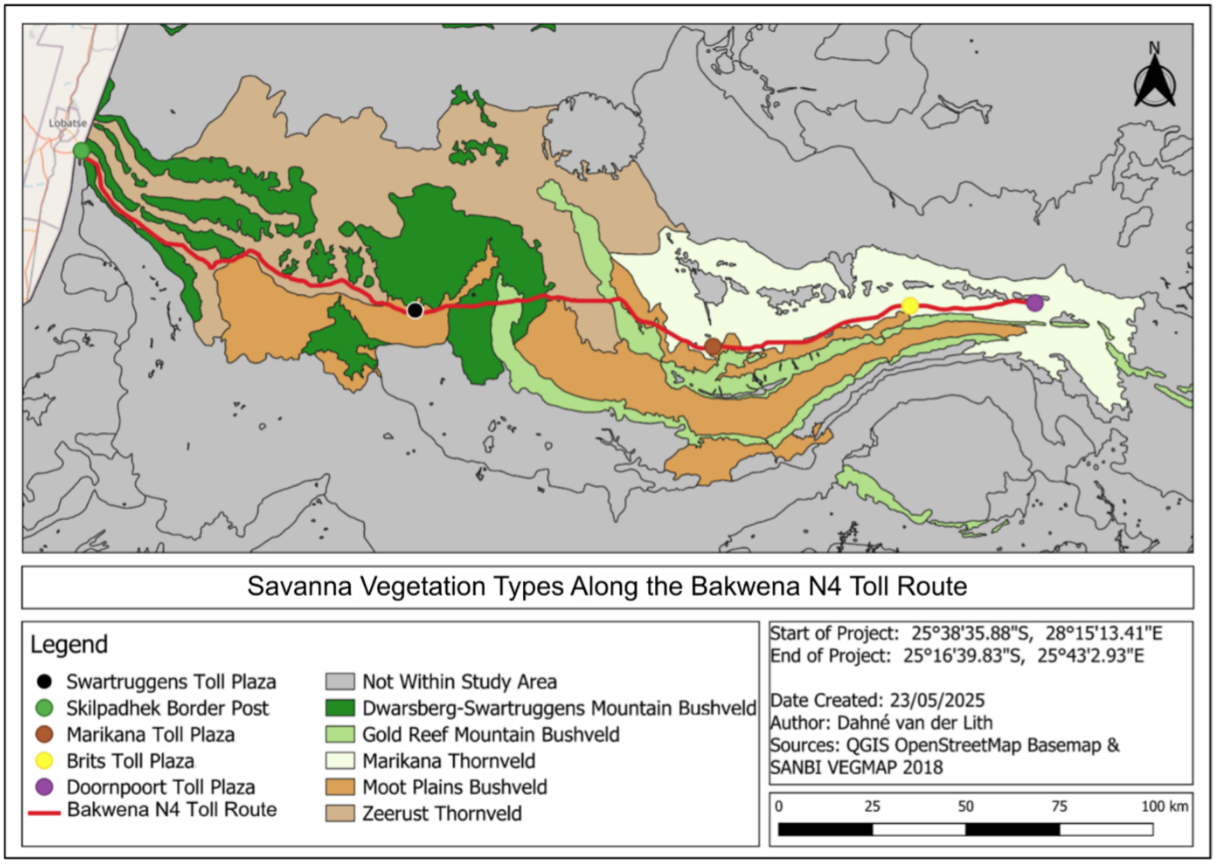


Figure 2.2 Savanna Vegetation Types Along the Bakwena N4 Toll Route

Although all five vegetation types along the Bakwena N4 Toll Route fall within the Savanna Biome, they are not structurally uniform and should not be treated as ecologically identical units (Mucina & Rutherford, 2006; Scholes & Walker, 1993). Marikana Thornveld differs from the remaining vegetation types in that it is associated with more open, grass-dominated vegetation on moderately undulating clay-rich plains, with a relatively continuous herbaceous layer and less prominent woody cover than the other Savanna units occurring further west along the corridor (Mangwane *et al.*, 2025; Mucina & Rutherford, 2006). This results in a simpler vertical structure and a more open ground layer, in contrast to vegetation types where shrubs, woodland elements, or terrain-related variation contribute more strongly to overall vegetation complexity (Fynn *et al.*, 2005; Mucina & Rutherford, 2006; Scholes & Walker, 1993). In addition, Marikana Thornveld has been substantially transformed by surrounding agriculture, mining, and associated disturbance, which may further increase its exposure to alien invasive plant establishment along road verges and adjacent disturbed land (Le Maitre *et al.*, 2011; Milton *et al.*, 2003). Within the context of this study, Marikana Thornveld therefore represents the more open end of the structural vegetation gradient present along the Bakwena N4 Toll Route, rather than a separate biome or unrelated vegetation system (Mucina & Rutherford, 2006; Scholes & Walker, 1993).

The remaining vegetation types, namely Zeerust Thornveld, Moot Plains Bushveld, Gold Reef Mountain Bushveld, and Dwarsberg-Swartruggens Mountain Bushveld, can be grouped together for comparative purposes because they are generally more similar to one another in woody expression and structural complexity than any of them is to Marikana Thornveld (Mucina & Rutherford, 2006; Scholes & Walker, 1993). Although they differ in soils, slope, aridity, and floristic composition, these vegetation types all tend to include a stronger shrub or woodland component, greater woody biomass, or a stronger topographic influence on vegetation pattern than is typical of Marikana Thornveld (Fynn *et al.*, 2005; Mucina & Rutherford, 2006).

Moot Plains Bushveld is characterised by short woodland and a mixed grass layer on clay-rich plains, while Gold Reef Mountain Bushveld and Dwarsberg-Swartruggens Mountain Bushveld are associated with rocky or elevated terrain that supports more structurally heterogeneous woody vegetation. Zeerust Thornveld, although relatively open, is still more comparable to these vegetation types in its woody and shrub-dominated Savanna character than to the grass-dominated structure of Marikana Thornveld (Mucina & Rutherford, 2006). For this reason, the four vegetation types may reasonably be treated as a broader group of relatively more wooded or structurally complex Savanna units against which Marikana Thornveld can be compared in terms of vegetation openness, disturbance response, and susceptibility to invasion (Fynn *et al.*, 2005; Le Maitre *et al.*, 2011; Scholes & Walker, 1993). This grouping does not imply that the four vegetation types are identical, but rather that they share sufficient structural similarity to function as a defensible comparison group within the aims of this study (Baard & Kraaij, 2019; Meunier & Lavoie, 2012; Mucina & Rutherford, 2006).

Marikana Thornveld occurs on moderately undulating clay-rich plains and supports open to moderately dense stands of *Vachellia karroo*, *V. tortilis*, and *Senegalia mellifera*, with a continuous grass layer (Mucina & Rutherford, 2006). It is widespread near Brits and Rustenburg and has been significantly transformed by agriculture and mining, making it particularly vulnerable to colonisation by herbaceous invaders such as *Parthenium hysterophorus* and *Tithonia rotundifolia* (Le Maitre *et al.*, 2011; Mucina & Rutherford, 2006). Zeerust Thornveld, found near Zeerust and the Skilpadshek border, is more arid and open, with *Vachellia erioloba*, *Boscia albitrunca*, and sparse grasses like *Stipagrostis uniplumis*. It occurs on calcareous soils and is prone to patchy but persistent invasion under grazing pressure (Mucina & Rutherford, 2006).

Moot Plains Bushveld occurs on clay-rich plains between Hartbeespoort and Rustenburg. It contains short woodland species such as *Ziziphus mucronata*, *Vachellia nilotica*, and *Dichrostachys cinerea*, with a mixed grass layer including *Themeda triandra* and *Eragrostis lehmanniana* (Mucina & Rutherford, 2006; SANBI, 2021). This unit is fire-prone and often subject to bush encroachment where fire suppression or grazing intensity is high (Mucina & Rutherford, 2006). Gold Reef Mountain Bushveld occurs in elevated, rocky areas along the corridor and is less transformed due to its shallow, leached soils. Vegetation includes *Englerophytum magalismontanum*, *Olea europaea subsp. africana*, and scattered tall shrubs (Mucina & Rutherford, 2006).

Dwarsberg–Swartruggens Mountain Bushveld is located in mountainous terrain west of Rustenburg, with diverse topography and a mix of *Strychnos pungens*, *Combretum apiculatum*, and *Terminalia sericea*, underlain by a variable grass layer. Localised disturbance, such as erosion or road cuttings, may lead to AIP establishment in exposed microhabitats, despite the unit being otherwise less transformed (Mucina & Rutherford, 2006).

Ecologically, the Savanna vegetation types along the Bakwena N4 Toll Route differ in structure and in their response to disturbance. The more wooded vegetation types are generally more tolerant of grazing and browsing due to their greater woody biomass and deeper-rooted species (Scholes & Walker, 1993). However, under altered disturbance regimes, these areas may undergo shrub thickening or bush encroachment, particularly by species such as *Dichrostachys cinerea* and *Vachellia spp*., which can reduce grass layer cover and create favourable conditions for disturbance-adapted or shade-tolerant AIPs (Wigley *et al.*, 2009). By contrast, the more open, grass-dominated Marikana Thornveld may be more sensitive to reductions in grass cover caused by clearing, maintenance activities, or altered fire regimes, which may in turn increase vulnerability to invasion by aggressive alien herbaceous species or woody invaders (Bond *et al.*, 2003; Richardson & van Wilgen, 2004).

The transitional zone around Hartbeespoort and into the peri-urban matrix represents a dynamic interface where a range of AIP species may establish. These areas are characterised by stronger edge effects, more frequent disturbance, and spatially heterogeneous vegetation structure, all of which may increase susceptibility to invasion (Donaldson *et al.*, 2014; Foxcroft *et al.*, 2004). The combination of multiple vegetation units, differing disturbance regimes, and environmental gradients along the corridor provides a useful context for investigating AIP patterns in relation to vegetation structure and land-use history. These ecological contrasts are explored in subsequent chapters in relation to AIP diversity, density, and spatial persistence.

### 2.3 Land Use Zones (Urban, Peri-Urban, Rural)

To assess how human development intensity influences AIP patterns, the Bakwena N4 Toll Route was stratified into three primary land use zones: urban, peri-urban, and rural. These zones were defined based on the type and density of adjacent land cover, verified through Bakwena’s 200 m blue marker system and supplemented by satellite imagery and in-field landscape interpretation (Baard & Kraaij, 2019; Rouget *et al.*, 2003).

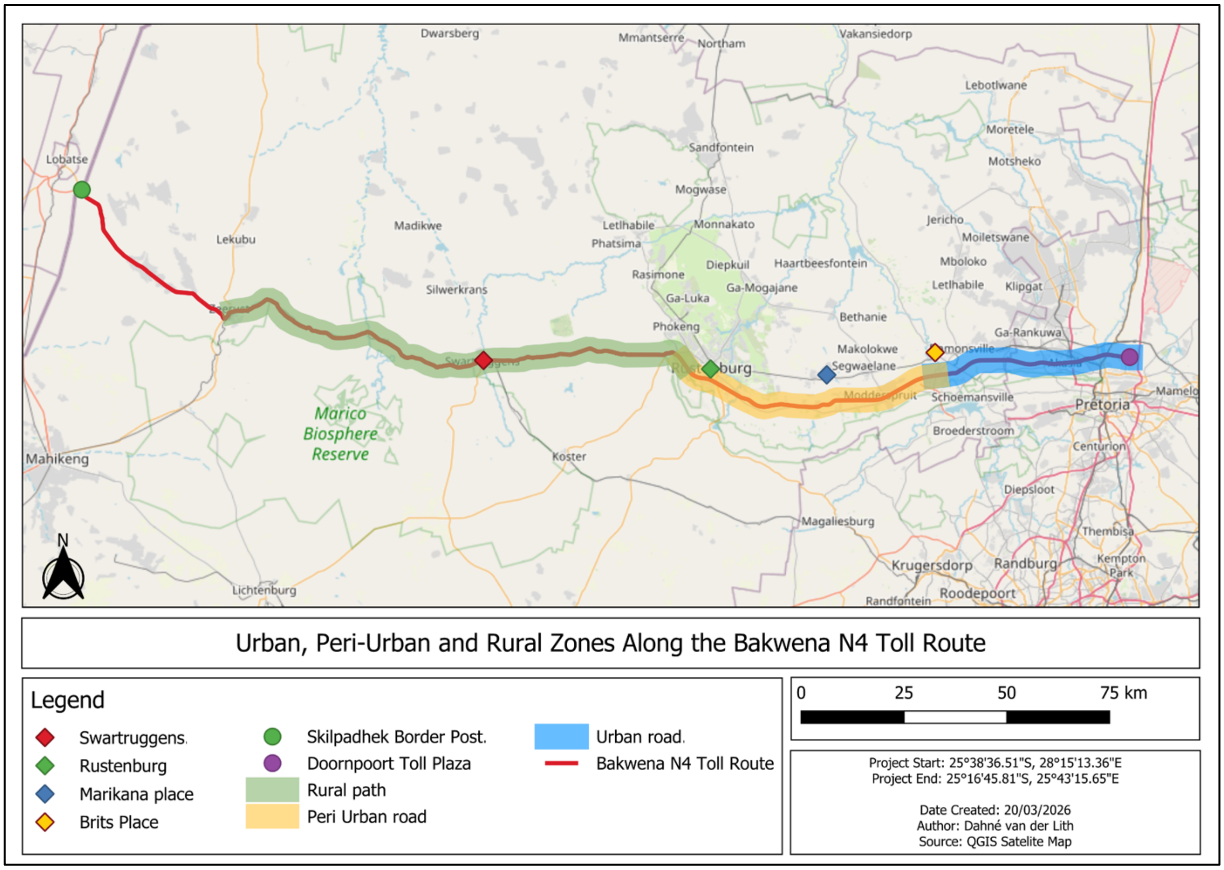


Figure 2.3 Urban, Peri-Urban and Rural Zones Along the Bakwena N4 Toll Route

The urban zone spans the eastern section of the route, from the Doornpoort Toll Plaza near Pretoria to the town of Brits. This area is characterised by dense residential development, sealed roads, commercial infrastructure, and high population pressure (van Huyssteen *et al.*, 2015). Notable urbanised areas along the route include Akasia, Onderstepoort, and other parts of the Tshwane metropolitan fringe. Road verges in this zone are heavily altered, with limited native vegetation cover and elevated propagule pressure from passing vehicles, informal dumping, garden escapees, and utility corridors (Meunier & Lavoie, 2012; von der Lippe & Kowarik, 2007). Frequent mowing, vegetation clearance, and maintenance-related disturbance further increase the establishment potential for ruderal, fast-growing AIPs adapted to urban edge conditions (van Wilgen *et al.*, 2020).

The peri-urban zone extends from Brits to Rustenburg. This section includes a patchwork of land uses, including smallholdings, rapidly expanding informal settlements, surface mining, and peri-industrial infrastructure. These zones are ecologically fragmented and often fall outside clearly defined municipal boundaries or conservation planning regimes, resulting in irregular or absent land management (Donaldson *et al.*, 2014). Locations such as Bapong, the Elands and Tharisa mine perimeters, and the edge of Rustenburg illustrate this mosaic of land cover. Peri-urban zones are frequently identified as invasion stepping stones, where overlapping disturbance types and fragmented governance allow both native and invasive species to coexist, compete, and expand (Foxcroft *et al.*, 2004; McConnachie *et al.*, 2012). Verge vegetation here often displays heterogeneity and persistent edge disturbance, which facilitates the long-term establishment and persistence of AIPs.

The rural zone covers the western portion of the route, from Rustenburg through Swartruggens, Groot Marico, and Zeerust, terminating at the Skilpadshek Border Post. This landscape is dominated by low-density commercial agriculture, extensive grazing areas, and patches of natural veld. Built infrastructure is limited, and road verges often remain partially intact or are adjacent to protected areas such as the Kgaswane Mountain Reserve and the Marico Biosphere Reserve (UNESCO, 2018; SANBI, 2017). Despite the lower frequency of direct disturbance, AIP spread remains a risk due to historical degradation (e.g., unrehabilitated road construction scars, disused borrow pits) and long-distance propagule dispersal by wind, vehicles, or livestock (Holmes *et al.*, 2008). While the rural verges generally exhibit higher vegetation cover, isolated invasion foci often go undetected or unmanaged.

This zone-based classification supports a comparative understanding of how AIPs respond to gradients in anthropogenic pressure and infrastructure intensity. It also aids in interpreting spatial differences in AIP species richness, abundance, and persistence patterns observed in later chapters. The defined zonings are urban (Doornpoort–Brits), peri-urban (Brits–Rustenburg), and rural (Rustenburg–Zeerust), which serve as a consistent framework for ecological analysis and targeted management recommendations along the corridor.

### 2.4 Climatic Conditions

The Bakwena N4 Toll Route traverses a broad climatic gradient associated with both longitudinal and altitudinal changes. The corridor falls within the summer-rainfall zone of South Africa, with precipitation concentrated between October and March (South African Weather Service [SAWS], 2021). This rainfall seasonality governs not only vegetation productivity and fire regimes, but also the establishment windows for AIPs, many of which rely on rainfall-triggered germination or dispersal (Foxcroft *et al.*, 2004; Le Maitre *et al.*, 2011).

Average annual rainfall generally increases from west to east along the route. The western sections of the corridor, particularly around Zeerust and Groot Marico, are relatively drier and more variable in rainfall, whereas the eastern sections receive higher annual rainfall and typically support greater plant biomass and denser vegetation cover (Midgley *et al.*, 2005; Mucina & Rutherford, 2006). This east-west rainfall gradient influences soil moisture availability and therefore shapes vegetation productivity, structure, and the competitive dynamics between indigenous and invasive species (Richardson & van Wilgen, 2004). These broad climatic differences are also relevant to the present study, as they help explain structural variation between the more open, grass-dominated Marikana Thornveld and the other, generally more woody or topographically complex vegetation types along the Bakwena N4 Toll Route.

In addition to spatial differences, the route is affected by inter-annual rainfall variability, with alternating years of drought and localised flooding (Boko *et al.*, 2007). This climatic instability plays a critical role in AIP establishment and persistence, as many invasive species exhibit opportunistic growth strategies, which include colonising disturbed or moisture-rich microsites in wet years and persisting through dry periods via resilient seed banks or rhizomes (Milton *et al.*, 2003; van Wilgen *et al.*, 2001).

Temperature gradients along the route broadly mirror this climatic variability. Maximum summer temperatures are generally higher in the lower-lying western regions of the corridor, whereas the eastern sections experience cooler winter conditions and are more frequently affected by low minimum temperatures and frost (SAWS, 2021). These temperature differences may also influence the distribution of certain AIP species along the route. Frost-sensitive species may be more likely to establish in the warmer western sections, while drought-tolerant or disturbance-adapted species may persist more widely depending on local vegetation structure, site conditions, and management history (Fynn *et al.*, 2005; Richardson *et al.*, 2000).

These climatic patterns also have direct implications for management timing and efficacy. Herbicide treatments are generally less effective during periods of heat stress or low soil moisture (Evans *et al.*, 2016). Similarly, mowing during drought years may stimulate compensatory growth or favour the recovery of deep-rooted perennials over shallow native species. Fire regimes, often manipulated for control, must also account for seasonal rainfall and fuel load conditions, which vary dramatically across the corridor (Bond *et al.*, 2003).

Given these gradients, climate served as a foundational variable in the spatial analysis of AIP dynamics. Interactions between rainfall, temperature, soil type, land use, and disturbance regimes were recognised as co-drivers of AIP abundance and vigour along the N4 and were integrated into the zonation and interpretation of patterns discussed in subsequent chapters.

### 2.5 Soil Types

The Bakwena N4 Toll Route spans a range of soil types that differ in fertility, structure, drainage, and ecological function. These soil forms influence local vegetation structure and site conditions, and may also affect the vulnerability of different parts of the route to invasion by AIP species (Freschet *et al.*, 2018; Milton *et al.*, 2003). Soils were identified by overlaying georeferenced sampling locations on the South African national soil classification map in QGIS (Figure 2.4). The dominant soil forms encountered included Ferric Luvisols, Rhodic Nitisols, Haplic Lixisols, Calcic Vertisols, and Lithic Leptosols, each presenting different conditions for plant growth, disturbance response, and invasion dynamics (Mucina & Rutherford, 2006; Soil Classification Working Group, 1991).

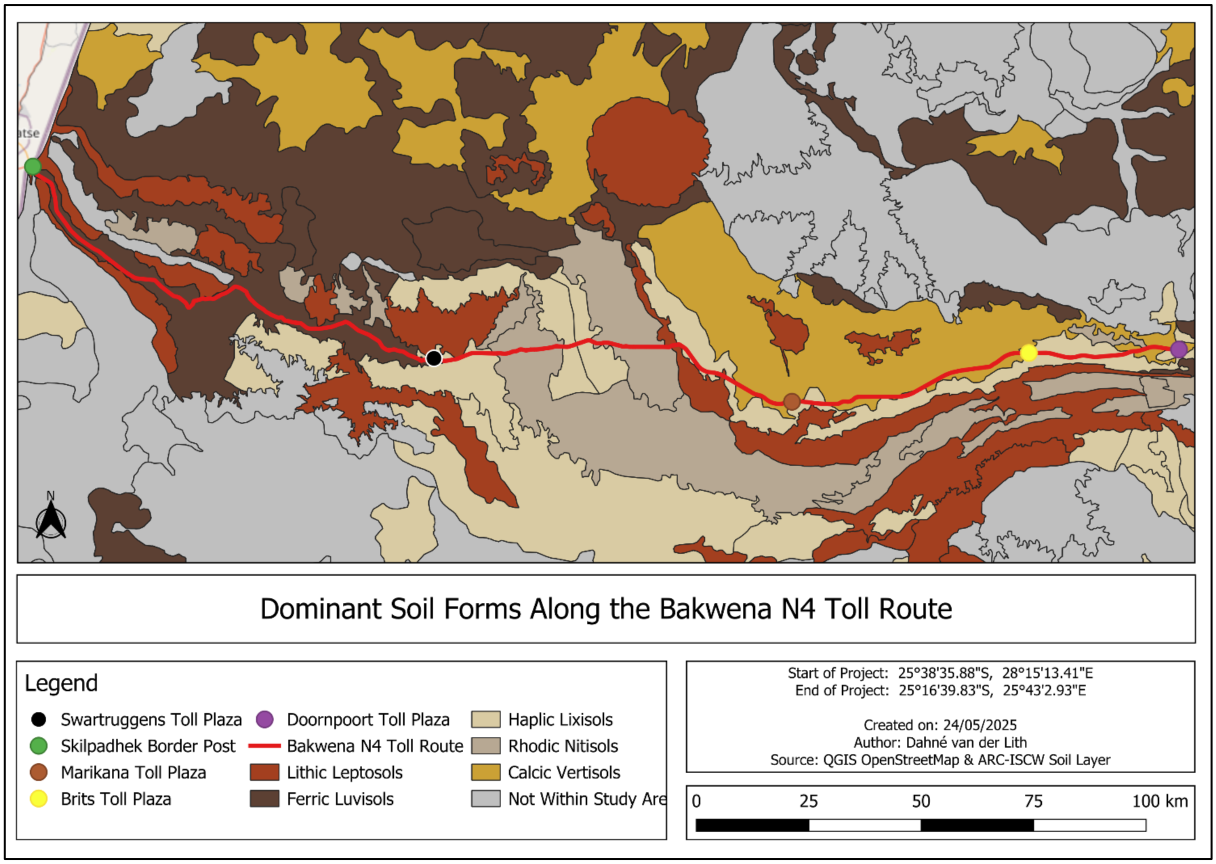


Figure 2.4 Dominant Soil Forms Along the Bakwena N4 Toll Route

Ferric Luvisols are moderately to highly fertile soils with well-developed structure and a characteristic subsurface accumulation of clay. They are typically found in the Bushveld regions, particularly on stable slopes and gently undulating uplands (Le Roux *et al.*, 2015). These soils are red to reddish-brown, well-drained, and rich in iron oxides, which support mixed woodland vegetation with good primary productivity. However, where vegetation cover is removed through road maintenance or compaction, the nutrient-rich surface layers provide ideal conditions for the establishment of indigenous pioneer grasses such as *Melinis repens* and fast-growing invasive alien weeds, such as *Parthenium hysterophorus* (Foxcroft *et al.*, 2004; Milton *et al.*, 2003). In degraded Luvisols, surface sealing may limit infiltration and exacerbate weed dominance by preventing natural seedling establishment.

Calcic Vertisols occur within parts of the Marikana Thornveld, with some portions also associated with Haplic Lixisols. Calcic Vertisols are generally heavy, base-rich clays characterised by shrink-swell behaviour, seasonal cracking, and high moisture-holding capacity, whereas Haplic Lixisols are more weathered and structured soils with moderate fertility and variable textural properties (Soil Classification Working Group, 1991). These differences in soil form are important because they influence drainage, water availability, rooting conditions, and vegetation structure, all of which may affect the establishment and persistence of alien invasive plants. In disturbed roadside settings, however, the influence of soil may be moderated by verge maintenance, vegetation clearing, compaction, and other anthropogenic disturbances that reduce indigenous cover and create opportunities for colonisation by disturbance-adapted AIP species (Le Maitre *et al.*, 2011).

Haplic Lixisols also occur along central parts of the corridor, including sections between Brits and Rustenburg, where they are associated with transitional and often fragmented land-use settings. These soils are generally moderately weathered, with variable texture and moderate fertility, and may support both grassy and woody vegetation depending on local topography, drainage, and disturbance history. In roadside and peri-urban environments, trampling, cultivation, traffic, and patchy maintenance may expose the soil surface and alter infiltration patterns, creating disturbed microsites that can favour adaptable AIP species (Henderson, 2007; Richardson *et al.*, 2000). Although these soils may be vulnerable under disturbed conditions, their role should be interpreted together with surrounding land use and vegetation condition rather than in isolation.

Further west along the route, Calcic Vertisols remain important, while other soil forms such as Ferric Luvisols, Lithic Leptosols, and Rhodic Nitisols also contribute to the edaphic complexity of the corridor. Calcic Vertisols are particularly notable because their high clay content, seasonal cracking, and tendency towards temporary waterlogging can create challenging rooting conditions and complicate rehabilitation after disturbance (Le Roux *et al.*, 2015). By contrast, Lithic Leptosols are generally shallower and more constrained by underlying rock, while Ferric Luvisols and Rhodic Nitisols may support somewhat deeper and more developed soil profiles under suitable conditions. These soil differences are likely to influence local vegetation structure, rooting depth, water retention, and plant competition, but their effects are filtered through broader environmental and management conditions along the route.

Taken together, the dominant soil forms along the Bakwena N4 Toll Route, namely Calcic Vertisols, Haplic Lixisols, Ferric Luvisols, Lithic Leptosols, and Rhodic Nitisols, contribute to a complex invasion setting. Invasion risk is therefore not solely a function of land use or vegetation structure, but also of local edaphic conditions that shape moisture availability, nutrient status, and rooting environment. At the same time, disturbed roadside conditions often override these natural constraints by creating open niches for colonisation. For this reason, soil is treated in this study as one of several interacting factors influencing AIP patterns, alongside vegetation structure, site condition, land use, and management history. These edaphic dynamics are considered further in the interpretation of species patterns, hotspots, and management implications in later chapters.

### 2.6 Known Environmental Pressures

Numerous environmental pressures along the Bakwena N4 corridor contribute to the establishment and persistence of AIPs, often interacting in complex and cumulative ways (Foxcroft *et al.*, 2004; van Wilgen *et al.*, 2020). One of the most consistent pressures is roadside disturbance, including routine maintenance activities such as mowing, slashing, and vegetation clearance. These operations typically remove competitive native cover, disturb the soil surface, and create open niches ideal for colonisation. In many cases, mowing schedules do not align with AIP growth cycles, resulting in the spread of viable seeds during peak reproductive periods (Baard & Kraaij, 2019; McConnachie *et al.*, 2012).

Stormwater runoff, especially where drainage infrastructure is lacking or poorly maintained, facilitates the physical movement of seeds and propagules across the corridor (Meunier & Lavoie, 2012). Accumulated water in road depressions or culverts also creates moist microhabitats, which are ideal for the germination of fast-establishing AIP species adapted to short-lived but fertile conditions (Holmes *et al.*, 2008).

Edge effects caused by adjacent land uses such as agriculture, mining, informal development, and industrial operations further compound invasion risk. These transitional edges typically support fragmented, simplified vegetation, frequent soil disturbance, and often elevated nutrient levels due to fertilisers or chemical runoff (Gaertner *et al.*, 2009; Rouget *et al.*, 2003). The boundary between natural and disturbed areas frequently shows a breakdown in native species resilience, allowing stress-tolerant and disturbance-adapted AIPs to proliferate.

In peri-urban areas, overlapping pressures such as unregulated land use, informal dumping, unauthorised access paths, and infrastructure encroachment create particularly unstable ecosystems (Donaldson *et al.*, 2014). These areas are often excluded from formal vegetation management and serve as reservoirs and corridors for AIP dispersal across vegetation groups and land-use types (Foxcroft *et al.*, 2004).

Vehicular movement and road construction activities further amplify these pressures. Vehicles act as mechanical dispersal agents via tyres, trailers, and wind turbulence (von der Lippe & Kowarik, 2007). Simultaneously, frequent verge compaction reduces porosity and plant-available water, limiting regeneration by native species and favouring AIPs adapted to compacted or degraded soils (Milton *et al.*, 2003). Roadworks and grading, especially when undertaken without follow-up restoration, often leave open soil patches that become focal points for colonisation.

Fire, whether planned, such as visibility burns, or accidental, such as ignitions from vehicles or power lines, is another major driver of disturbance along the route. Although fire plays an ecological role in Savanna systems, the timing, frequency, and intensity of burns can strongly influence vegetation recovery and invasion processes. Poorly timed or overly frequent burns may suppress indigenous regrowth, alter vegetation structure, and create conditions that favour fire-tolerant or short-lived invasive species (Bond *et al.*, 2003; van Wilgen *et al.*, 2010). In addition, post-fire nutrient flushes and exposed soil surfaces may create short-term establishment opportunities for alien herbaceous species.

Legacy disturbances such as disused borrow pits, decommissioned access roads, and erosion scars are frequently neglected in management planning but continue to support persistent AIP populations. These areas often function as unintended seed banks for recolonisation during wet years or after verge clearing (Milton *et al.*, 2003).

These pressures operate at multiple spatial and temporal scales, with cumulative effects that often outpace current verge management capacity. The absence of coordinated post-disturbance rehabilitation and the lack of integrated ecological maintenance strategies have enabled many road verges to evolve into self-sustaining invasion corridors. These dynamics were essential to interpreting the spatial distribution of AIP hotspots and changes in community composition across the route.

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# CHAPTER 3
# LITERATURE REVIEW

## 3.1 Introduction

Alien invasive plants (AIPs) introduced beyond their native ranges have become a major concern in South Africa, where they displace indigenous species, alter ecological processes such as fire regimes and nutrient cycling, and place a strain on management efforts across a variety of landscapes (Kumar Rai & Singh, 2020; Richardson *et al.*, 2000; Pyšek *et al.*, 2012; van Kleunen *et al.*, 2015). The spread of these species is closely associated with patterns of disturbance brought about by urban expansion, agriculture, mining, and the development of linear infrastructure (Potgieter *et al.*, 2020; Richardson & van Wilgen, 2004). Such activities modify the land in ways that create opportunities for AIPs to take hold, particularly in a country where high floristic diversity and a long history of land transformation increase vulnerability to invasion (Gaertner *et al.*, 2016).

It is important, however, to distinguish between alien, naturalised, and invasive species, as not all plants introduced beyond their native range become invasive. Some alien species remain dependent on continued cultivation, others become naturalised by forming self-sustaining populations without direct human assistance, and only a subset of these become invasive by spreading widely and causing ecological, economic, or social harm (Blackburn *et al.*, 2011; Pyšek *et al.*, 2004; Richardson *et al.*, 2000). The classification of invasive plants within national legislation reflects the range of threats these species present and the priorities that guide their control (Zengeya *et al.*, 2017).

The conditions that enable AIPs to establish are influenced by a combination of environmental characteristics and land use practices. Soil type, climate, water availability, and vegetation structure determine how receptive an area is to invasion, yet these characteristics are often reshaped by human activity (Dietz & Edwards, 2006; Rouget *et al.*, 2003). Roads, railways, and settlements alter the natural balance, creating fragmented and disturbed spaces that invasive plants are quick to exploit (Forman & Alexander, 1998; Novaes *et al.*, 2022). Recognising how these elements interact allows for a better understanding of where invasions are most likely to occur and how management responses can be designed to address both ecological conditions and human influence (El-Barougy *et al.*, 2021). This is particularly relevant in invasion studies because naturalised species may persist in transformed landscapes without necessarily becoming aggressive invaders, whereas invasive species are those that spread beyond the point of introduction and begin to alter ecosystem structure, functioning, or resource use (Blackburn et al., 2011; Pyšek et al., 2004; Richardson et al., 2000; Vilà et al., 2011).

Once established, AIPs bring a range of ecological and economic consequences that extend well beyond the areas they occupy. They disrupt fire patterns, reduce water availability, and interfere with natural regeneration, while also damaging infrastructure and adding to the cost of land and resource management (Görgens & van Wilgen, 2004; Tyser & Worley, 1992). These effects are often most visible along roads and other forms of infrastructure where the presence of invasive species complicates maintenance and amplifies the challenges associated with managing fragmented landscapes (Bobuľská *et al.*, 2025; Turner *et al.*, 2021). The link between disturbance and impact becomes especially clear in these environments, where human activity and biological invasion reinforce one another.

Linear infrastructure not only experiences the effects of invasion but also contributes to the spread of AIPs. By providing corridors for dispersal, introducing disturbance, and creating conditions that favour establishment, roads and railways play a key role in invasion dynamics (Ansong & Pickering, 2013; Gulzar *et al.*, 2024; Vittoz & Engler, 2007). The way in which these features are designed, maintained, and managed alongside surrounding land uses shapes how they either support or resist the spread of invasive plants. Strategies for managing invasions along these corridors must balance ecological concerns with the practicalities of working in disturbed environments (Esler & Archer, 2018).

AIP control efforts in these settings rely on a variety of techniques, including physical removal, chemical treatment, biological control, and combinations of these approaches (Kettenring & Adams, 2011; Shackleton *et al.*, 2015). The success of these methods depends on factors such as the intensity of disturbance, the species involved, and the nature of the surrounding landscape. Urban, peri-urban, and rural areas each present distinct conditions that influence both the risk of invasion and the options available for management (van Wilgen *et al.*, 2022b).

Monitoring and modelling play an increasingly important role in tracking how invasions unfold over time and in guiding decisions about where and how to intervene (Arnoldi *et al.*, 2022; van Wilgen *et al.*, 2022a). The ability to detect patterns of spread, anticipate future risks, and evaluate the outcomes of management actions is essential for reducing the long-term impact of AIPs. These tools provide a foundation for adapting strategies as conditions change and as new information becomes available.

In this review, we explore how AIPs establish and spread, the ecological and economic impacts they create, and the approaches that have been used to manage them along linear infrastructure. The review also distinguishes between alien, naturalised, and invasive species in order to clarify that not all exotic plants pose the same level of threat or require the same regulatory or management response. By drawing together current knowledge in these areas, the review identifies key gaps in understanding and suggests priorities for future research and practice.

## 3.2 Defining AIPs and Regulatory Categories in South Africa

AIPs are defined as species introduced beyond their native geographical range, whether through deliberate means such as horticulture, agriculture, erosion control, and forestry, or through unintentional pathways including global trade, contaminated materials, tourism, and land transformation (Blackburn *et al.*, 2011; Kacheche & Mzuza, 2021; Richardson *et al.*, 2000). In the South African context, many AIPs were introduced during colonial expansion, often for practical or aesthetic purposes. For instance, *Acacia mearnsii* was valued for its timber, *Jacaranda mimosifolia* for its ornamental appeal, and *Opuntia* species for fodder, yet all later spread beyond control as agricultural expansion, infrastructure development, and land disturbance intensified (Harris, 2004; Richardson *et al.*, 2020; Sipango *et al.*, 2022). This historical trajectory, marked by intentional introduction followed by unintentional dispersal, remains a consistent feature in the long-term persistence of many invasive populations (Skočajić & Nešić, 2020).

It is important to note, however, that alien or exotic status alone does not mean that a species is invasive. Alien species are simply those introduced outside their native range, whereas naturalised species are alien species that establish self-sustaining populations without continued direct human assistance. Only a subset of naturalised species become invasive, meaning that they spread beyond their point of introduction and cause ecological, economic, or social harm (Blackburn *et al.*, 2011; Pyšek *et al.*, 2004; Richardson *et al.*, 2000).

AIPs are distinguished from other alien or exotic species by their ecological behaviour (Blackburn et al., 2011; Richardson et al., 2000; Richardson & van Wilgen, 2004). These species typically form self-sustaining populations, spread quickly into natural or semi-natural habitats, and cause ecological, economic, or social harm (McLean *et al.*, 2018; Richardson *et al.*, 2000; Richardson & van Wilgen, 2004). What sets them apart is not their origin but their impact, as invasives are capable of altering ecosystem function, disrupting mutualisms, and degrading native biodiversity (Fridley *et al.*, 2007; Miehls *et al.*, 2009). Their success is often tied to a suite of traits that enhance competitiveness in disturbed or fragmented landscapes.

Traits such as prolific seed production, fast growth rates, vegetative reproduction, tolerance of wide environmental variation, and escape from native herbivores or pathogens give AIPs a clear advantage over indigenous species, especially in degraded ecosystems (Keane & Crawley, 2002; Richardson & Pyšek, 2006; van Kleunen *et al.*, 2010). These features allow them to exploit open ecological niches and destabilise native communities where disturbance has already weakened biotic resistance (Alpert *et al.*, 2000; Catford *et al.*, 2009; Drenovsky *et al.*, 2012). The consequences include reduced native species richness, shifts in community dominance, altered nutrient cycling and hydrology, and the breakdown of ecological interactions (Kull *et al.*, 2011; Traveset & Richardson, 2006).

Invasions often become self-reinforcing, where feedback loops involving fire, soil chemistry, or hydrological change maintain or even accelerate invasive dominance (D'Antonio & Vitousek, 1992; Euskirchen *et al.*, 2003; Gaertner *et al.*, 2014). These impacts have been observed throughout South Africa and differ based on land-use history, biome, and levels of disturbance (Mostert *et al.*, 2017). AIPs are particularly common along roads, in peri-urban edges, and in unmanaged spaces where propagule pressure and disturbance converge (Kalwij *et al.*, 2008b; Low & Rebelo, 1996). While coastal regions such as the Western Cape face critical threats to fynbos habitats, inland provinces like Gauteng and North-West are increasingly impacted due to expanding urbanisation and infrastructure (Driver & Atkinson, 2012; Rouget *et al.*, 2003; van Wilgen *et al.*, 2012). The scale and persistence of these invasions are closely tied not only to patterns of disturbance and land use, but also to the limited ability of many native species to compete with aggressive invaders.

South Africa's high endemism and ecological specialisation make many native species poorly equipped to resist aggressive invaders (Rouget *et al.*, 2006; van Wilgen *et al.*, 2008). The scale of invasion has become significant: over 10 million hectares were affected by AIPs by the mid-1990s (van Wilgen *et al.*, 2008; Versfeld *et al.*, 1998), doubling to nearly 20 million hectares by 2007 (Department of Water Affairs and Forestry, 2007; van Wilgen *et al.*, 2012). Current estimates indicate continued expansion at 5 to 10 percent annually (Kotzé *et al.*, 2010; Richardson *et al.*, 2020), underscoring the urgency for coordinated control.

South Africa now hosts over 500 alien plant species, of which at least 383 are considered invasive (Marti, 2017; SANBI, 2018). These species not only disrupt ecosystems but also impose significant economic costs by reducing agricultural output, threatening water security, and degrading ecosystem services (Kull *et al.*, 2011; van Wilgen *et al.*, 2008). In response, a formal regulatory framework was introduced under the National Environmental Management: Biodiversity Act (NEMBA, Act No. 10 of 2004), supported by the 2016 Alien and Invasive Species Regulations. This framework classifies alien plants into four regulatory categories based on risk, spread potential, and control needs (Department of Forestry, Fisheries and the Environment, 2020). The categorisation system is intended to guide landowner obligations, shape public compliance, and support national monitoring systems.

Alongside NEMBA, the Conservation of Agricultural Resources Act (CARA, Act No. 43 of 1983) historically used the terms weeds and invader plants, especially in agricultural and rural land management. In this regulatory context, a weed does not automatically refer to every alien species, but rather to plants regarded as undesirable in a particular land-use setting, while invader plants are those that spread aggressively and threaten agricultural productivity, biodiversity, or ecosystem functioning (Blackburn et al., 2011; Pyšek et al., 2004; Richardson et al., 2000; Vilà et al., 2011). The distinction is useful because it shows that alien, naturalised, invasive, and weed are related but not interchangeable terms within the South African management context.

Category 1a species, such as *Acacia adunca A.Cunn. ex G.Don*, must be eradicated entirely wherever found. Category 1b species, including *Lantana camara* and *Arundo donax*, may persist but require ongoing control and cannot be propagated or moved (Department of Forestry, Fisheries and the Environment, 2020; Raphela & Duffy, 2023). Category 2 species, like *Acacia mearnsii*, may be cultivated under regulated conditions in demarcated areas with valid permits (SANBI & CIB, 2024). Category 3 species, such as *Tipuana tipu*, are tolerated in certain regions but prohibited from new introductions, especially in sensitive ecosystems such as riparian zones (Potgieter *et* *al.*, 2020). These classifications support risk-based decision-making by linking ecological threats to permitted land-use practices. They also show that regulation is directed at specific listed taxa and risk categories rather than at all alien or naturalised plants indiscriminately. Nevertheless, the implementation of these regulations remains inconsistent. Although the framework offers structure, practical enforcement is hampered by overlapping mandates, limited budgets, and poor coordination between institutions (Nsikani & Geerts, 2024; Schelhas *et al.*, 2021; van Wilgen, 2018).

The effectiveness of control also depends heavily on local environmental conditions. Habitat characteristics such as soil type, land-use intensity, hydrology, and disturbance regimes influence both the vulnerability to invasion and the persistence of invasive species (Alston & Richardson, 2006; Chen *et al.*, 2017; Euskirchen *et al.*, 2003). Sites with chronic disturbance or weak enforcement, such as peri-urban road verges, can continue to act as propagule sources, undermining broader control efforts across the landscape (Pyšek *et al.*, 2020). This is particularly important in disturbed corridors, where naturalised alien species may persist at low levels for extended periods, but where repeated disturbance, soil movement, runoff, and propagule introduction can shift these populations towards invasive behaviour (Holmes et al., 2008; Meunier & Lavoie, 2012; Milton et al., 2003; von der Lippe & Kowarik, 2007).

In this context, the legal categorisation system remains a necessary but insufficient tool. It provides structure, sets priorities, and supports compliance (Richardson & van Wilgen, 2004; van Wilgen *et al.*, 2020), but its long-term success will rely on whether it can be meaningfully integrated with habitat-specific risk assessments, cross-sector coordination, and site-level implementation (Gaertner *et al.*, 2016; van Wilgen & Wannenburgh, 2016). Understanding how legal frameworks interact with ecological variability and land-use realities is key to strengthening AIP management, particularly in fragmented and dynamic spaces such as road infrastructure corridors like the Bakwena N4 Highway (Richardson & van Wilgen, 2004; Rouget, Cowling, *et al.*, 2003).

### 3.3 Environmental Filters: Habitat Conditions and Invasion Susceptibility

The establishment and spread of AIPs depend on multiple abiotic and biotic habitat characteristics, which jointly shape the conditions under which propagules can germinate, establish, and proliferate (Catford *et al.*, 2009; Dietz & Edwards, 2006). These environmental filters include soil composition and texture (Pergl *et al.*, 2023), moisture availability (Versfeld *et al.*, 1998), vegetation structure (van Wilgen *et al.*, 2008), disturbance history, and the underlying biological dynamics of the site (Rouget, Richardson, *et al.*, 2003). In road reserve environments, where ecological resistance is often reduced through physical disruption or habitat fragmentation, these habitat characteristics play an even more decisive role in shaping invasion trajectories (Meunier & Lavoie, 2012; Richardson & Pyšek, 2006).

Among abiotic factors, soil conditions may influence AIP establishment and persistence by shaping water availability, nutrient status, rooting depth, and the competitiveness of indigenous vegetation (Chiuffo *et al.*, 2022; Mostert *et al.*, 2017). Along road shoulders, however, these soil effects are often modified by disturbance associated with construction, maintenance activities, and heavy vehicle traffic, which can compact the soil, reduce porosity, and limit water infiltration (Awarri & Otto, 2023; Bassett *et al.*, 2005; Fu *et al.*, 2019). Such altered conditions may constrain native plant regeneration while creating opportunities for disturbance-tolerant AIPs to establish in exposed roadside environments (Gavrilescu, 2021; Pergl *et al.*, 2023; van Kleunen *et al.*, 2016). Along the Bakwena N4 Toll Route, variation in soil form is therefore relevant to invasion patterns, but it is best interpreted together with vegetation structure, land-use context, and site disturbance rather than as an isolated driver.

Soil chemistry also affects invasion outcomes. Native species often exhibit reduced tolerance to salinity or heavy metals, particularly near industrial areas or older highways with long disturbance histories (Eijsackers *et al.*, 2020; Ndhlovu *et al.*, 2024). On the other hand, many generalist AIPs not only tolerate but, in some cases, prefer such conditions, enabling them to dominate in edaphically constrained environments (Li *et* *al.*, 2022; van Kleunen *et al.*, 2016). This tolerance confers a long-term competitive advantage and allows invasive populations to expand their range into areas where indigenous vegetation is chemically excluded. In the context of the present study, however, these mechanisms are most relevant as general ecological processes rather than as dominant site conditions across the full Bakwena corridor, where invasion patterns are more directly shaped by disturbance, verge maintenance, adjacent land use, and vegetation structure.

Recent work has drawn attention to the importance of belowground biotic interactions, particularly the role of AIPs as ecosystem engineers that reshape microbial communities to reinforce their own persistence (Stefanowicz *et al.*, 2019; Wang *et al.*, 2022). Some nitrogen-fixing species have been shown to elevate soil nitrogen levels, thereby encouraging secondary invasions by nitrophilous weeds (Nsikani *et al.*, 2018). *Acacia saligna*, for example, modifies mycorrhizal associations in nutrient-poor ecosystems, inhibiting native plant recruitment and altering successional trajectories (Yelenik *et al.*, 2004). AIPs have also been shown to alter rhizosphere conditions in ways that suppress native microbial mutualists and promote microbial communities that support their own growth (Coats & Rumpho, 2014). This feedback loop creates ecological traps in which both abiotic and biotic soil properties become increasingly favourable to invaders, thereby reducing the likelihood of spontaneous native recovery without direct intervention (Chiuffo *et al.*, 2022; Soti *et al.*, 2020). While these belowground feedbacks are important for understanding invasion ecology more broadly, they are less directly observable in roadside monitoring studies and should be interpreted here as supporting mechanisms rather than primary variables measured in this study.

Vegetation structure is another important factor shaping invasion outcomes. More open, sun-exposed habitats, such as the grass-dominated Marikana Thornveld, may be more prone to invasion by fast-growing herbaceous species because the reduced woody cover allows invaders to occupy available niches with limited shading or above-ground competition (van Kleunen *et al.*, 2016). By contrast, the denser woody Savanna vegetation types along the route are generally more structurally complex, with greater woody cover, litter accumulation, and layered vegetation that may restrict light penetration and reduce space for seedling establishment (Bourlière & Hadley, 1970; Foxcroft *et al.*, 2010; Mhosisi Masocha, 2010). Even so, these more wooded systems are not resistant to invasion. Disturbance associated with roads, clearing, and edge effects may create canopy gaps or exposed patches that allow disturbance-adapted or shade-tolerant invaders to establish and gradually alter vegetation structure over time (Son *et al.*, 2024; van Kleunen *et al.*, 2016).

Invasion patterns are also likely to vary across the different vegetation settings along the route. The more open and transformed sections, particularly those fragmented by urban expansion, agriculture, or other land-use pressures, may experience higher propagule pressure and greater exposure to invasion along edges and disturbed verges (Lázaro-Lobo & Ervin, 2019; Mavimbela *et al.*, 2018). By contrast, the more wooded and structurally complex Savanna vegetation types may appear more intact in some areas, but they remain vulnerable to invasion along riparian corridors, old settlement footprints, and disturbed roadside margins, especially where fire regimes have been altered or nutrient conditions have been changed by leguminous invaders (du Plessis *et al.*, 2022; Foxcroft *et al.*, 2010; Gaertner *et al.*, 2014).

Historical land use adds further complexity to invasion patterns. Sites that appear similar in present-day structure may differ greatly in invasion resistance due to legacy effects of cultivation, grazing intensity, or fire suppression (Lebbink *et al.*, 2018). For example, roads that pass through previously farmed or overgrazed areas often exhibit reduced native seedbanks and simplified community structure, increasing vulnerability to colonisation (Ansong & Pickering, 2013; Lemke *et al.*, 2019; Rew *et al.*, 2017). Altered fire regimes, whether through suppression or increased burn frequency, can also shift competitive dynamics to favour AIPs that benefit from specific fire intervals (Tepley *et al.*, 2018; van Wilgen, 2010). These legacies complicate invasion management, especially when trying to restore ecological integrity in highly modified sites.

Given the interplay between soil properties, microbial interactions, vegetation structure, and historical land use, no single factor can reliably predict a site's susceptibility to AIP establishment. Invasion outcomes instead emerge from multiple interacting filters that vary in strength depending on spatial and temporal context (Rouget *et al.*, 2015; Soti *et al.*, 2020; Chapple *et al.*, 2022). This complexity underscores the importance of site-specific assessments, particularly along roads where microclimatic conditions, disturbance levels, and soil types can shift rapidly over short distances. For infrastructure environments like the Bakwena N4 corridor, understanding how AIPs interact with and modify these habitat filters is central to both predicting invasion risk and designing targeted control strategies (Chiuffo *et al.*, 2022).

### 3.4 Ecological, Infrastructural, and Socioeconomic Impacts of AIPs

AIPs have wide-ranging consequences that extend across ecosystems, infrastructure, and agricultural landscapes. Their impacts on biodiversity, ecosystem processes, and land productivity are particularly evident along roads and in other disturbed areas, where invasions often progress rapidly (Forman & Alexander, 1998; van Wilgen *et al.*, 2022b). The ecological consequences of AIPs are well documented across biomes, species groups, and ecological functions (Vilà *et al.*, 2011). By outcompeting native flora for essential resources such as light, water, nutrients, and space, AIPs reduce indigenous species richness and alter the structure of plant communities (Kumar Rai & Singh, 2020; van Kleunen *et al.*, 2016). These effects are often most pronounced in systems with high endemism or limited ecological resilience, where disturbance-sensitive native species are rapidly displaced (Gulzar *et al.*, 2024; Vilà *et al.*, 2011). As invasions progress, many AIPs form dense monocultures that simplify vegetation layers, reduce microhabitat variability, and lower overall habitat heterogeneity (Gallé *et al.*, 2023; Kumschick *et al.*, 2015). By simplifying vegetation structure and reducing habitat diversity, AIP monocultures set the stage for changes to ecological processes such as fire dynamics.

In fire-prone ecosystems such as Savannas, AIPs can fundamentally alter disturbance regimes. Species such as *Melia azedarach*, *Acacia mearnsii*, and *Arundo donax* may increase fuel loads or modify fire behaviour, resulting in fire frequencies or intensities that differ from natural patterns (O'Connor & van Wilgen, 2020; Brooks *et al.*, 2004). These shifts can disrupt indigenous species that are adapted to particular fire-return intervals and may contribute to the gradual degradation of fire-regulated systems (van Wilgen, 2010). Over time, the consequences may include reduced recruitment of sensitive species, altered successional pathways, and reinforcing feedback that favours continued invasive dominance.

Belowground processes are similarly disrupted. AIPs influence nutrient cycling by accelerating nutrient turnover or modifying nitrogen and phosphorus availability, often shifting soil conditions in ways that disadvantage native plant communities (Chamier *et al.*, 2012; Huntley, 2023). They also alter hydrological regimes by increasing water uptake and reducing water availability at both the site and catchment scales. Deep-rooted species such as *Eucalyptus* spp., *Populus* spp., and *Arundo donax* exhibit high transpiration rates, removing more water from the soil profile than comparable native species (Le Maitre *et al.*, 2015). These changes reduce streamflow, lower annual runoff, and heighten pressure on freshwater systems (Cullis *et al.*, 2009; Le Maitre *et al.*, 2020).

The magnitude of hydrological impacts depends on species traits such as rooting depth, seasonal activity, and growth form (Le Maitre *et al.*, 2020; van Kleunen *et al.*, 2016). According to Le Maitre *et al.* (2015), AIPs are responsible for the loss of between 1.44 and 2.44 billion cubic metres of surface water annually in South Africa, representing roughly 6% of national water resources (Le Maitre *et al.*, 2000). Without effective management, this figure could rise to as much as 16%, posing serious threats to aquatic systems and human water security (Le Maitre *et al.*, 2016). Invasion reduces wetland persistence, alters seasonal flooding patterns, and leads to the drying of ephemeral pans and stream corridors (Chamier *et al.*, 2012). Species that rely on these systems for germination, breeding, or feeding, such as amphibians, invertebrates, and water-dependent plants, are particularly vulnerable (Everson, 2016; Le Maitre *et al.*, 2002, 2020).

Species interactions are further shaped by hydrological change, which modifies habitat conditions and ecological linkages. As riparian vegetation declines, pollinator networks break down, food webs shift, and habitat corridors shrink, disrupting ecological functioning at multiple trophic levels (Gallagher & Campbell, 2017; Vilà *et al.*, 2011). Invasive plant dominance often correlates with declines in invertebrate abundance, reductions in vertebrate habitat quality, and losses in ecosystem services such as pollination and nutrient retention (Kumschick *et al.*, 2015; Vilà *et al.*, 2011). These declines in invertebrates are particularly concerning because they underpin key ecological functions, from nutrient cycling to providing food for higher trophic levels, and their loss can trigger cascading effects throughout ecosystems (Vilà *et al.*, 2011).

Beyond their ecological impacts, AIPs pose practical risks to physical infrastructure. Roads, culverts, and drainage systems are particularly vulnerable to root intrusion, soil destabilisation, and vegetation overgrowth (Booy *et al.*, 2017; Ruwanza & Mhlongo, 2020). Invasive species such as *Ricinus communis* and *Solanum mauritianum* establish rapidly in disturbed verges and embankments, where their root systems can undermine road foundations, obstruct water flow, and block driver sightlines (South African River Health Programme, 2001; Witkowski & Garner, 2008). These impacts result in higher maintenance costs and more frequent repair needs, especially for local governments managing under-resourced road networks (Forman & Alexander, 1998; Kalwij *et al.*, 2008b; Randrup *et al.*, 2001). In rural settings, unmanaged verges often become persistent AIP reservoirs. The lack of regular clearing or chemical control allows species to proliferate, creating roadside conditions that exacerbate erosion, waterlogging, and visibility hazards (Kotowska *et al.*, 2021). In this context, road infrastructure is not only vulnerable to invasion but also acts as a long-term vector for AIP spread.

Agricultural production is similarly affected by AIPs, especially on farms located near roads. Dense infestations limit the usable area of arable land, hinder routine operations such as ploughing and irrigation, and increase the costs of weed management (Eschen *et al.*, 2021; Kariyawasam *et al.*, 2021). Many invasive species possess underground storage organs or long-lived seedbanks, making them difficult to eliminate once established (Gioria *et al.*, 2014). For farmers, this translates into repeated clearing cycles, higher input costs, and declining yields (Kumar *et al.*, 2022). In extreme cases, land abandonment occurs when the financial and labour burdens of control become unsustainable (Nkambule *et al.*, 2017).

The combined effects of ecological degradation, infrastructure disruption, and agricultural constraints illustrate the broad and interconnected impacts of AIP invasions in South Africa (Kumschick *et al.*, 2015; Vilà *et al.*, 2011). These impacts tend to be particularly pronounced along roads, where consistent disturbance, altered drainage, and edge effects make conditions ideal for invasion (Nicolson, 2010; van Kleunen *et al.*, 2016). Vegetation clearing, soil compaction, and surface runoff during road construction reduce the ability of native species to re-establish, creating ecological openings that AIPs exploit during the early phases of post-construction succession (Bassett *et al.*, 2005; 2022; Son *et al.*, 2024). Without active management, these roadsides evolve into long-term invasion corridors, sustaining source populations that drive continued spread across surrounding landscapes (Nicolson, 2010; Son *et al.*, 2024).

### 3.5 Road Disturbance and Post-Construction Succession Dynamics

Road construction is among the most ecologically disruptive forms of land transformation, triggering physical and ecological changes that extend far beyond the road's physical footprint (Giunta, 2020). While road networks play an essential role in connecting urban centres, industries, and rural communities in South Africa (Maphela & Adanlawo, 2025), their development is accompanied by significant environmental trade-offs (Wang *et al.*, 2024). The processes of vegetation clearance, topsoil removal, excavation, grading, and compaction fundamentally alter surface topography, disrupt soil profiles, and interfere with nutrient cycling and hydrological flow (Giunta, 2020; Son *et al.*, 2024; Wang *et al.*, 2024). These disturbed conditions are characterised by exposed soil, low porosity, erosion risk, and altered water infiltration and create ideal niches for AIP establishment during and after construction.

Such environments foster strong edge effects, characterised by elevated light exposure, drying soils, and localised microclimatic shifts (Connor & McCoy, 2017; Kariyawasam *et al.*, 2021). These edges form a transitional interface between disturbed reserves and adjacent ecosystems, providing AIPs with entry points into previously uninvaded areas (Connor & McCoy, 2017; Holway, 2005). Once established, alien species such as *Arundo donax* can form dense stands, alter fire regimes, and modify litter and nutrient dynamics, further locking in disturbance at the site level (Brooks *et al.*, 2004; van Kleunen *et al.*, 2016). In many cases, this leads to long-term system shifts that inhibit native recovery and sustain an invaded state (Alston & Richardson, 2006; Witkowski & Garner, 2008).

These effects are particularly pronounced in South Africa's varied biomes, where roads frequently cut through intact ecosystems, fragmenting habitats and isolating ecological processes (Seiler, 2003). Fragmentation affects pollination, seed dispersal, predator-prey dynamics, and other interactions that maintain ecosystem function (Mullu, 2016). Exposed soil surfaces, left bare by vegetation stripping, become susceptible to wind, temperature extremes, and runoff, leading to declines in soil fertility, disruptions in microbial communities, and reduced regenerative capacity of native flora (DeVilleneuve *et al.*, 2023; Soti *et al.*, 2020; Stefanowicz *et al.*, 2019). These early conditions favour competitive species that exploit rapid nutrient pulses and lack of canopy competition. In linear infrastructure corridors, these effects are often intensified because disturbance is repeated along an extended strip of land rather than being confined to a single isolated site (Giunta, 2020; Meunier & Lavoie, 2012; Son et al., 2024; van Kleunen et al., 2016).

The recovery trajectory following road construction is largely determined by the intensity of disturbance and the legacy of native seed banks (Gioria *et al.*, 2014; Luo *et al.*, 2023). In early successional stages, vegetation is often dominated by pioneer annuals and other disturbance-responsive invaders such as *Parthenium hysterophorus, Ageratum conyzoides,* and *Echium plantagineum*, which exploit bare soil and early nutrient availability (Prach & Walker, 2019). If unmanaged, these assemblages may be replaced by persistent, large-statured invaders such as *Ricinus communis* or *Tithonia diversifolia*, which form closed canopies, suppress further regeneration, and maintain altered ecosystem conditions for decades (Chen & van Kleunen, 2025; Moran *et al.*, 2013; van Kleunen *et al.*, 2016; Wilson & Kumschick, 2024). The exact species composition will vary between regions and vegetation types, but the broader successional pattern of early colonisation followed by longer-term invasive persistence is widely recognised (Gioria et al., 2014; Moran et al., 2013; Prach & Walker, 2019; van Kleunen et al., 2016).

In road verge habitats like those along the Bakwena N4, maintenance practices may unintentionally reinforce these successional trends. While verge management typically aims to maintain visibility and prevent overgrowth, mistimed mowing (e.g., after flowering but before seed set) can facilitate seed dispersal rather than suppress seed dispersal (Jantunen *et al.*, 2007; Kalwij *et al.*, 2008a). Rhizomatous species such as *Arundo donax* are particularly responsive to physical disturbance, using mowing or slashing as cues for vegetative spread (Ceotto & Di Candilo, 2010; He *et al.*, 2012). Maintenance equipment may further contribute to invasion by unintentionally transporting viable seeds or root fragments along linear infrastructure routes (Kalwij *et al.*, 2008a; Turner *et al.*, 2021), accelerating spread across long distances (Bhatti, 2023; Meunier & Lavoie, 2012; Roberts *et al.*, 2018). To complement or replace mechanical approaches like mowing, chemical control is often used along road verges as a means of managing invasive growth.

Although chemical control is widely used to reduce aboveground biomass, it also poses risks. Broad herbicides can leave vegetation gaps that allow seedbank-stored AIPs to re-establish, especially when control is not followed by active restoration or reseeding with native species (D'Antonio *et al.*, 2016; Jubase *et al.*, 2019; McManamen *et al.*, 2018). Herbicides may also damage non-target species, shift soil microbial communities, or alter pH, further reducing the system's resistance to invasion over time (Rose & de Smidt, 2019; Weidenhamer & Callaway, 2010). Without coordinated post-treatment recovery efforts, chemical suppression may inadvertently reinforce long-term spread.

The recovery of natural vegetation in post-construction road verges is rarely spontaneous. Degraded soils, depleted microbial communities, and long-lived invasive seed banks limit the likelihood of passive re-establishment of native species (D'Antonio *et al.*, 2016; Kalwij *et al.*, 2008a). In many cases, this failure is amplified by ecological mismatches between native species and altered post-disturbance conditions. Indigenous plants often require specific soil, water, and biotic associations, while alien species are typically more generalist, thriving in nutrient-rich, open, and disturbed habitats (Pyšek *et al.*, 2020; Tartaglia & Aronson, 2024; van Kleunen *et al.*, 2016).

This non-specific nature of AIPs creates a reinforcing cycle: roads cause disturbance, which facilitates invasion; AIPs then alter ecosystem conditions, making it harder for native species to return; routine maintenance perpetuates the disturbance and enables continued spread (Kalwij *et al.*, 2008a; Son *et al.*, 2024; Wang *et al.*, 2024). Breaking this cycle requires long-term strategies that integrate locally adapted revegetation, targeted maintenance timing, and equipment hygiene protocols (Armstrong *et al.*, 2017; D'Antonio *et al.*, 2016; Turner *et al.*, 2021). In highly diverse and invasion-prone regions like South Africa, where road reserves are active zones of ecological transformation, recognising how post-construction succession influences long-term invasion risks is required (Giunta, 2020; Müllerová *et al.*, 2011; Wang *et al.*, 2024). This is especially relevant along corridors such as the Bakwena N4, where repeated disturbance, ongoing maintenance, and adjacent land-use pressures can sustain invasion long after the initial construction phase has ended.

### 3.6 Roads as Corridors: Spread and Establishment of AIPs

South Africa's growing road network supports national development, facilitating economic growth, enabling regional connectivity, and improving access to essential services (Maphela & Adanlawo, 2025). Strategic routes like the N4 are particularly vital in linking rural and urban zones, supporting commerce, and enabling mobility across a range of ecological contexts (Nkosi *et al.*, 2025). While these transport corridors deliver socio-economic benefits (Sakketa, 2023), they also impose considerable ecological costs. Roads are increasingly recognised as powerful drivers of environmental disruption, particularly concerning plant community dynamics (Forman & Alexander, 1998). The same infrastructure that supports development often contributes to habitat fragmentation, soil degradation, and the widespread establishment of AIPs (Holway, 2005; Kalwij *et al.*, 2008a).

Roads not only provide ideal conditions for AIPs, but they also act as active conduits for their spread. As linear structures, roads function as dispersal pathways that accelerate the movement of seeds and vegetative propagules across ecological boundaries and biogeographical zones (Christen & Matlack, 2009; Kalwij *et al.*, 2008a; Son *et al.*, 2024). This mobility is facilitated through various vectors, including vehicle tyres, construction and maintenance equipment, stormwater runoff, and wind turbulence generated by passing traffic (Anderson *et al.*, 2015; Ansong & Pickering, 2013). These processes allow propagules to bypass natural dispersal constraints and colonise distant or previously inaccessible sites, often at rates that far exceed spread in undisturbed landscapes (Christen & Matlack, 2009; Lemke *et al.*, 2019).

The physical design and condition of road reserves further enhance establishment opportunities for AIPs. Flat or concave verges with compacted, poorly draining soils accumulate moisture, nutrients, and sediment - conditions that favour seed germination and survival (Aranda *et al.*, 2021; Awarri & Otto, 2023; van Kleunen *et al.*, 2016). Structural factors such as road width, verge slope, and surface permeability also influence where and how seeds settle (Son *et al.*, 2024). In peri-urban areas, road reserves often intersect with disturbed sites, dumping grounds, and informal paths, creating high propagule pressure and acting as chronic seed sources for localised spread (Nelufule *et al.*, 2024; Tartaglia & Aronson, 2024). These interacting conditions make road reserves persistent invasion hotspots (Kalwij *et al.*, 2008a), where AIPs can both establish and expand over time.

Beyond site-level dynamics, roads alter landscape-scale connectivity by fragmenting habitats and increasing edge density relative to core habitat area (Holway, 2005; Marcantonio *et al.*, 2013; Mullu, 2016). This fragmentation disrupts species migration, seed dispersal networks, and spatial processes that underpin ecosystem resilience (Novaes *et al.*, 2022). Edge habitats, often drier and more exposed than interior zones, favour AIPs adapted to high-light, disturbed environments (van Kleunen *et al.*, 2016). Construction activities, especially those involving vegetation clearance and heavy machinery, compact soils and remove organic layers, reducing the ability of indigenous plants to recover while enhancing competitive advantages for disturbance-tolerant species (Bassett *et al.*, 2005; Mileusnić *et al.*, 2022; Müllerová *et al.*, 2011).

The road construction process itself can serve as a primary source of invasion. Seeds can be introduced inadvertently through construction materials, vehicles, worker clothing, and equipment (Ansong & Pickering, 2013; Rew *et al.*, 2017). Once introduced into these disturbed sites, AIPs often establish rapidly and outcompete native flora, especially where local species have limited disturbance tolerance or narrow ecological requirements (Gallé *et al.*, 2023; van Kleunen *et al.*, 2016). These roadside populations seldom remain isolated. Roads often connect otherwise fragmented invasive populations, fostering metapopulation dynamics through increased genetic exchange, range expansion, and population reinforcement (Perumal *et al.*, 2021; Schowalter, 2016). This connectivity enhances the long-term adaptability and persistence of AIPs, complicating control and increasing reinvasion risk (van Kleunen *et al.*, 2016; van Wilgen, 2018). In this way, roads function not only as sites of invasion but also as linkages between invaded patches across the wider landscape (Christen & Matlack, 2009; Meunier & Lavoie, 2012; Perumal et al., 2021; Schowalter, 2016).

The ecological consequences extend beyond plant communities. Vegetation clearing and grading during construction increase soil erosion and sedimentation (DeVilleneuve *et al.*, 2023), which in turn destabilise soils and alter surface runoff, raising flood risk (Seutloali, 2014). These hydrological changes, combined with soil nutrient shifts and exposure to fluctuating temperatures, further inhibit native species regeneration while creating niches exploited by AIPs (DeVilleneuve *et al.*, 2023; van Kleunen *et al.*, 2016). In areas where roads require raw material extraction, such as gravel, sand, or bitumen, long-term habitat alteration and topsoil removal can leave lasting ecological scars, often requiring decades to recover, if at all (D'Antonio *et al.*, 2016; Giunta, 2020).

The long-term effects of repeated disturbance and insufficient restoration along roadsides contribute to lasting changes in vegetation composition. In the South African context, the result is often the development of dense, alien-dominated assemblages within road reserves, which reduce native species richness and restructure plant communities (Kalwij *et al.*, 2008a; van Kleunen *et al.*, 2016). These areas, continually subjected to disturbance and lacking ecological buffering, become invasion reservoirs. Without targeted management, they act as ongoing propagule sources into neighbouring agricultural, urban, or semi-natural systems (Alpert *et al.*, 2000). This is especially relevant in road corridors that pass through multiple land-use types, where repeated edge disturbance and high propagule movement increase the likelihood of continued spread (Alston & Richardson, 2006; Ansong & Pickering, 2013; Foxcroft et al., 2004; Meunier & Lavoie, 2012).

While roads remain essential to development, their ecological footprint must be actively managed. The spread and persistence of AIPs in and around transport corridors are shaped by multiple interacting factors, including physical infrastructure design (Perumal *et al.*, 2021), soil and hydrological disturbance (Seutloali, 2014), dispersal vectors (Roberts *et al.*, 2018), and surrounding land use. As South Africa continues to expand its road infrastructure, integrating ecological considerations into design, planning, and maintenance will be essential for mitigating biological invasion and protecting ecosystem integrity (Giunta, 2020; Wang *et al.*, 2024). For corridors such as the Bakwena N4, this means recognising that road reserves are not passive landscape features, but active ecological pathways through which invasive plants can establish, persist, and spread across transformed and semi-natural environments.

### 3.7 Management Responses: Eradication Strategies Along Linear Infrastructure

Eradicating AIPs requires a multifaceted approach that combines mechanical, chemical, biological, and ecological restoration methods (van Wilgen, 2018). Mechanical techniques such as mowing, slashing, and hand-pulling are often used to suppress aboveground biomass and prevent seed production, especially when timed before flowering (Boy & Witt, 2013; van Wilgen *et al.*, 2012). Herbicides offer effective treatment for dense infestations and regenerating species, particularly in inaccessible areas or where rapid coverage is needed (Shackleton *et al.*, 2015; Üstüner *et al.*, 2020). Biological control has yielded long-term suppression of species like *Lantana camara*, *Opuntia ficus-indica*, and *Chromolaena odorata* through host-specific agents that reduce growth and reproductive output over time (Britannica, 2015; Moran *et al.*, 2013). In many cases, these direct interventions are complemented by ecological restoration measures such as replanting with indigenous species, mulching, or soil conditioning to promote recovery and increase biotic resistance to reinvasion (D'Antonio *et al.*, 2016). These approaches form the foundation of modern AIP control, though their success depends heavily on context, timing, and sustained implementation.

These generalised strategies, however, are difficult to apply consistently in road reserves, which present unique ecological and logistical constraints. In these settings, consistent and site-specific control is difficult to maintain. Invasive species such as *Arundo donax*, *Campuloclinium macrocephalum*, and *Tithonia rotundifolia* are particularly problematic in these contexts due to their tolerance to physical damage and their ability to regenerate subsequent to slashing or herbicide exposure (Meunier & Lavoie, 2012; Turner *et al.*, 2021; van Kleunen *et al.*, 2016). These species often resprout aggressively, form dense monocultures, and prevent natural succession, especially where disturbance is ongoing or management is inconsistent (Jo *et al.*, 2017; van Wilgen, 2018).

Jurisdictional fragmentation further complicates management. Road reserves often span multiple governance levels, from national concessionaires and provincial departments to local municipalities, without a shared strategy or coordinated schedules (Department of Transport, 2018). For example, mowing is commonly timed to align with road safety standards rather than ecological principles, frequently occurring after seed maturation, thereby accelerating AIP dispersal rather than suppressing it (Jantunen *et al.*, 2007). Maintenance machines often spread viable seeds and stem fragments over long distances, especially in disturbance-adapted species like *Ricinus communis* and *Solanum mauritianum* (Rew *et al.*, 2017; Wang *et al.*, 2024). This mismatch between operational maintenance cycles and ecological control timing is one of the main reasons why roadside AIP management often remains reactive rather than preventative (Jantunen et al., 2007; Rew et al., 2017; van Wilgen, 2018; Wang et al., 2024).

To address these challenges, researchers advocate for an Integrated Weed Management (IWM) model tailored to linear infrastructure. IWM integrates multiple control methods, mechanical, chemical, and biological, based on site conditions, seasonal windows, and practical constraints (Knezevic *et al.*, 2017; Paynter & Flanagan, 2004; van Wilgen, 2018). In road reserves, this allows control measures to be adapted to varying verge widths, slope stability, soil types, and access limitations. For example, mechanical slashing may be prioritised in high-visibility zones, while spot herbicide application can target regrowth in more sensitive or inaccessible areas (Boy & Witt, 2013; Moss, 2019; Scavo & Mauromicale, 2020).

Mechanical methods like mowing and brush-cutting remain widely used because of their compatibility with routine road maintenance. When applied strategically, particularly before flowering or seed set, they can significantly reduce propagule production (Scavo & Mauromicale, 2020; van Wilgen, 2018). However, when poorly timed or inconsistently applied, they risk stimulating vegetative regrowth or inadvertently spreading seeds (Kalwij *et al.*, 2008a; van Kleunen *et al.*, 2016). For woody AIPs such as *Melia azedarach* and *Solanum mauritianum*, combining cutting with immediate herbicide application on stumps yields more effective results (Witkowski & Garner, 2008). In erosion-prone or sloped terrain, low-impact techniques such as selective hand-pulling or chainsaw removal are often used to reduce soil disturbance (DeVilleneuve *et al.*, 2023; Scavo & Mauromicale, 2020).

Chemical control remains one of the most practical options in road contexts, especially for dense infestations or difficult-to-reach zones (Boy & Witt, 2013; The Nature Conservancy *et al.*, 2001). Targeted application techniques like rope wicking, stem injection, or spot spraying help minimise off-target impacts and are ideal for narrow verges or areas near wetlands (Johnson, 2008; Sorvig & Thompson, 2018). In areas under bridges, along guardrails, or on steep slopes, herbicides offer safe alternatives to mechanical removal (Johnson, 2008). Selective herbicides can be used to preserve native grasses and stabilise vegetation, while broad-spectrum spraying is generally avoided due to its tendency to strip ground cover and increase erosion risk (Paynter & Flanagan, 2004; Scavo & Mauromicale, 2020).

While chemical control offers immediate results, its limitations in reach and long-term sustainability have led to growing interest in biological alternatives. Biological control is slower to establish but provides long-term suppression in remote or low-priority zones where repeated visits are impractical (Britannica, 2015; Moran *et al.*, 2013). Once released, biocontrol agents can self-disperse along road networks, providing sustained impact and reducing reinvasion risk. Examples include the successful introduction of agents against *Chromolaena odorata* and *Lantana camara*, which have reduced plant vigour along roads in and around protected areas like the Kruger National Park (Foxcroft *et al.*, 2023; Vardien *et al.*, 2012).

Although full ecological restoration is rarely feasible along entire road networks, it can be strategically applied in high-risk locations such as interchanges, slopes, and bridge embankments. These sites, which experience frequent disturbance and seed input, benefit from stabilisation using fast-growing native species like *Themeda triandra*, *Panicum maximum*, or *Eragrostis curvula* (Brown & Bezuidenhout, 2020; Zama *et al.*, 2017). On new roads or widened corridors, pre-emptive actions such as topsoil replacement, mulching, and early planting may reduce initial AIP establishment, though implementation is often constrained by financial and engineering limitations (Cabral, 2023; Dai *et al.*, 2022; Wang *et al.*, 2024). Ultimately, effective AIP control along roads requires landscape-level coordination and forward planning. Management should be guided by local invasion risk, vegetation condition, and propagule pressure to prioritise vulnerable areas. When delivered through integrated and site-specific approaches, these strategies provide a more sustainable alternative to the reactive clearance cycles that currently dominate road verge management (Hierro *et al.*, 2006; Johnson, 2008).

### 3.8 Urbanisation, Land Use, and Management Challenges

Urbanisation associated with road infrastructure alters local environments in ways that make it easier for AIPs to establish (Nelufule *et al.*, 2024; Sakketa, 2023). The construction and expansion of roads disturb the environment by altering water flow, compacting soil, and increasing nutrient levels, which together reduce ecological resistance and create favourable conditions for disturbance-tolerant alien species (Müller *et al.*, 2013; Walker, 1999). These transformed environments are also characterised by artificial surfaces, fragmented green spaces, and unmanaged land that collectively create a mosaic of colonisation niches, ideal for invasive species to establish and persist (Hierro *et al.*, 2006; Potgieter *et al.*, 2020).

Beyond these unintentional environmental changes, many AIPs are also introduced through deliberate human activity, particularly in urban landscaping and greening initiatives. Some ornamental alien species planted for aesthetic appeal or shade provision may later escape cultivation and establish in open urban spaces such as road verges, stormwater channels, and buffer zones (Alpert *et al.*, 2000; Potgieter, 2019; Sumalatha *et al.*, 2024). The high frequency of disturbances in cities, combined with ongoing soil movement, compost use, and informal dumping, amplifies propagule pressure and aids in seed and fragment dispersal (Ansong & Pickering, 2013; Gaertner *et al.*, 2016; Rew *et al.*, 2017). Even where management is present, public resistance to removing attractive species such as *Jacaranda mimosifolia* adds a socio-political barrier to effective control (Booy *et al.*, 2017; Sumalatha *et al.*, 2024). This distinction is important because not all ornamental alien species that are introduced into urban landscapes become invasive, but repeated disturbance and propagule pressure increase the likelihood that some naturalised ornamentals will spread beyond cultivation and behave invasively (Ansong & Pickering, 2013; Gaertner et al., 2016; Potgieter, 2019; Sumalatha et al., 2024).

This pattern of invasion shifts notably in peri-urban areas, which serve as transitional zones between dense urban development and the surrounding rural areas. These landscapes typically support a combination of informal settlements, smallholdings, service infrastructure, and remnant natural vegetation, making them highly heterogeneous and difficult to manage uniformly (Chatterjee & Dewanji, 2024; Samat *et al.*, 2020). The irregular layout and rapid land use change introduce overlapping disturbance regimes from livestock grazing, dumping, trampling, and construction, all of which create continuous opportunities for AIP establishment (Alston & Richardson, 2006; Kapucu *et al.*, 2024). These disturbances are accompanied by high propagule pressure, driven by nearby cultivated alien species, legacy infestations from adjacent urban zones, and constant seed movement via vehicles, animals, and water (Ansong & Pickering, 2013; Ruas *et al.*, 2022). Fragmented vegetation further amplifies invasion risk, as widespread edge effects facilitate the spread of AIPs into semi-natural habitats (Müller *et al.*, 2013; van Kleunen *et al.*, 2016). Despite their proximity to cities, these zones often lack the institutional oversight and infrastructure found in urban areas, while also missing the formal conservation mandates that may exist further into rural regions (Eck & McGee, 2008; Jantunen *et al.*, 2007).

While rural landscapes may appear less invaded at first glance, they remain vulnerable due to a mix of ongoing disturbance and legacy effects. Agricultural activity, overgrazing, road construction, and altered fire regimes have contributed to the establishment of AIPs in many areas (Dai *et al.*, 2022; Marti, 2017). Invasions often occur along access roads, railway lines, degraded riparian zones, and abandoned pastures, where past land use has disrupted natural vegetation and created conditions that favour alien species (Alston & Richardson, 2006; Pretorius *et al.*, 2023). Some rural areas benefit from structured control through conservation or commercial farming programmes, but many communal and remote regions lack consistent management (Johnson, 2008; Kalwij *et al.*, 2008b). Fire suppression in protected areas can favour species like *Lantana camara*, and neglected infrastructure such as old roads and logging sites often continues to act as a source of propagules long after the disturbance has passed (Brooks *et al.*, 2004; van Wilgen, 2010; Foxcroft *et al.*, 2010; Potgieter *et al.*, 2020). These combined pressures make rural areas an important but often underestimated part of the broader invasion problem.

Despite differences in ecological conditions, urban, peri-urban and rural areas all face the common challenge of institutional fragmentation, which weakens long-term AIP control (Gallé *et al.*, 2023; Mullu, 2016). In urban settings, fragmented responsibilities across departments managing parks, servitudes or stormwater systems lead to inconsistent maintenance and poor coordination (Gaertner *et al.*, 2016; Sakketa, 2023). Peri-urban areas struggle with unclear land ownership and overlapping mandates, often resulting in neglect of high-risk sites (Eck & McGee, 2008; Jantunen *et al.*, 2007). In rural zones, management depends heavily on land tenure and access to resources, with many areas remaining unmanaged due to institutional neglect or landowner resistance (Johnson, 2008; van Wilgen, 2018). These structural gaps allow invasive species to persist and spread, even where control programmes are in place. Lasting success requires collaboration across agencies, sectors and landowners, supported by shared data and flexible strategies tailored to both ecological and governance realities (Boy & Witt, 2013; Zengeya *et al.*, 2017; Wilson *et al.*, 2009; Wilson & Kumschick, 2024). These governance and land-use contrasts are relevant to the Bakwena N4 Corridor because the route passes through urban, peri-urban, and rural settings, each with different sources of disturbance, propagule pressure, and management capacity.

### 3.9 Monitoring AIPs: Statistical Indices and Assessment Tools

Long-term monitoring of AIP dynamics requires analytical tools that can capture ecological change clearly and consistently across space and time (Collins *et al.*, 2000; Vilà *et al.*, 2011). Biodiversity indices are especially useful, as they help assess species richness, abundance, and community balance, and how these attributes shift in response to disturbance or management (Lamb *et al.*, 2009; Legendre, 2019; McCarthy & Magurran, 2004). Their strength lies in transforming complex ecological data into interpretable metrics, offering repeatable and standardised ways to track invasion processes and vegetation recovery over time (Anderson, 2003; Legendre, 2019).

The Shannon–Wiener Index (Shannon & Weaver, 1964) provides a composite measure of diversity by incorporating both species richness and evenness. It increases with the number of species and with the balance in their relative abundances (Shannon & Weaver, 1964). In invasion ecology, this index has proven useful in identifying critical thresholds of ecological degradation. For example, a study on *Tradescantia fluminensis* in New Zealand forests found that when invasive cover exceeded 30%, Shannon diversity declined by over 70%, revealing a tipping point beyond which native community structure collapsed (Standish *et al.*, 2001). In roadside systems, where disturbances often favour dominance by a few opportunistic species, this index helps distinguish between communities recovering toward a balanced state and those shifting toward homogenisation due to AIP persistence (Hierro *et al.*, 2006; McCarthy & Magurran, 2004).

Pielou's Evenness (Pielou, 1966) focuses specifically on how evenly individuals are distributed across species. This becomes particularly relevant in systems where richness remains stable, but species dominance shifts markedly. Studies of *Lantana* *camara* invasions in Indian Savannas found that evenness decreased significantly despite no apparent decline in species counts, indicating rising dominance by a single invasive species (Ramaswami & Sukumar, 2014). In roadside vegetation, this metric provides insight into subtle structural shifts that may otherwise be obscured by stable richness values, especially in habitats where a few AIPs can rapidly dominate open or disturbed microsites (Morris *et al.*, 2014; Whittaker, 1972).

Margalef's Richness Index, which adjusts species richness in relation to total abundance (Death, 2008), is particularly useful in fragmented, small-scale habitats such as road verges. It allows for comparisons across plots that differ in vegetation density or total abundance by standardising richness relative to sample size (Death, 2008). In fragmented vegetation systems, the index has been used to detect richness differences across disturbance gradients and successional stages, with lower values often associated with more degraded conditions (Briske, 2017). Along transport corridors, where microhabitat conditions and land-use intensity may vary considerably between sites, Margalef's index provides a useful metric for assessing changes in floristic richness in response to long-term disturbance.

To analyse how plant communities shift over time, compositional indices such as Bray–Curtis Dissimilarity and Sørensen Similarity are often employed. The Bray–Curtis index incorporates abundance data and is ideal for detecting turnover among dominant species, such as when an AIP is gradually replaced by native flora or, alternatively, by a new invader (Bray & Curtis, 1957; Anderson *et al.*, 2011). In Patagonia, Bray–Curtis was used to track recovery following the removal of non-native *Pinus* species, revealing partial return of native assemblages and replacement by secondary invaders in some sites (Gioria *et al.*, 2014). This index is particularly valuable in AIP monitoring as it captures changes in community composition that may be masked by unchanged species counts.

Sørensen Similarity, by contrast, uses presence-absence data and is better suited to early successional stages or to situations where abundance estimates may be less reliable (Sørensen, 2003; Legendre, 2019). Fleishman *et al.* (2006) showed its value in monitoring compositional change over time by detecting whether species assemblages reassembled after disturbance, regardless of changes in biomass or cover. Its application is particularly well-suited to contexts such as road verges, where sampling constraints may limit abundance estimates, but where community identity still provides important ecological insight.

Similarly, Yang *et al.* (2013) applied Sørensen's index to compare soil seed banks and above-ground vegetation across wetland successional stages, showing its usefulness for tracking compositional similarity during vegetation recovery where abundance alone would not fully capture community change. Its application is therefore well-suited to contexts such as road verges, where sampling constraints may limit abundance estimates, but where community identity still provides important ecological insight. This makes Sørensen particularly useful in studies where the presence or absence of species is more reliable than fine-scale abundance estimation.

Taken together, these indices offer complementary perspectives on invasion dynamics. Structural metrics like richness and evenness indicate how community balance is changing, while compositional metrics such as Bray–Curtis and Sørensen reveal shifts in species identity and assemblage similarity. Their combined use in similar published studies has proven effective in detecting invasion thresholds, dominance transitions, and patterns of ecological recovery or reassembly (Gaertner *et al.*, 2012; Latombe *et al.*, 2017). This literature-supported foundation strengthens their application in long-term road verge monitoring and provides a framework for modelling invasion trajectories, evaluating intervention outcomes, and prioritising management action along the Bakwena N4 Toll Route.

### 3.10 Modelling Invasion Trajectories: Long-Term Trends and Predictive Tools

Long-term ecological monitoring helps reveal how AIP invasions unfold over time, especially in highly disturbed systems such as road reserves, where short-term surveys often miss slow-developing or delayed invasion dynamics (Roura-Pascual *et al.*, 2009). Many AIPs display non-linear spread patterns, characterised by initial lag phases followed by sudden increases in abundance or dominance, often triggered by cumulative disturbance, crossing ecological thresholds, or gradual adaptation to novel conditions (Collins *et al.*, 2000; Crooks, 2005; van Kleunen *et al.*, 2016). Without sustained, multi-year data, these transitions can remain undetected, resulting in misinterpretation of both invasion severity and management outcomes (Gioria *et al.*, 2014; Hierro *et al.*, 2006; Suding *et al.*, 2004).

Despite their value, long-term datasets on AIPs in South Africa are relatively scarce, particularly within road reserves and linear infrastructure zones. Most available studies cover fewer than five years and rarely offer site-level resolution along disturbed corridors (Gaertner *et al.*, 2016; van Wilgen *et al.*, 2020). This is a missed opportunity, given the acknowledged role of roads in creating dispersal pathways, edge effects, and sustained habitat disturbance that favour AIP establishment (Meunier & Lavoie, 2012). Belaire *et al.* (2022) argue that high-frequency, plot-based monitoring is needed to identify consistent invasion hotspots and to understand why certain sites remain resistant or repeatedly invaded, even after intervention.

Beyond observation, predictive modelling has gained traction as a complementary approach for managing biological invasions more proactively (Rouget *et al.*, 2024). Modelling techniques such as generalised additive models, boosted regression trees, MaxEnt, and machine learning algorithms enable spatial prediction of invasion risk by combining species occurrence data with environmental variables like land use, elevation, climate, and distance to roads or rivers (Elith & Leathwick, 2009; Oppel *et al.*, 2012; Stanton *et al.*, 2012; Yu *et al.*, 2020). These models can generate habitat suitability maps and inform where future invasions are most likely to occur, even in areas not yet heavily infested.

In the South African context, predictive models have been used to forecast potential spread patterns for species such as *Lantana camara*, *Acacia dealbata*, and *Hakea* *sericea*, particularly in the Western Cape and within conservation areas like Kruger National Park (Mtyobila, 2023; Rouget *et al.*, 2004). Their application to road corridors remains underutilised, despite the fact that such environments exhibit predictable patterns of edge disturbance, connectivity, and propagule pressure (Giunta, 2020; Kalwij *et al.*, 2008). Benedetti and Morelli (2017) demonstrated that verges with strong links to urban and agricultural land uses tend to experience higher AIP pressure, emphasising the need to include landscape context in any spatial risk assessment.

When combined with long-term ecological data, predictive models become even more powerful. Calibrating models with empirical, plot-based monitoring improves accuracy, enables real-time validation, and allows for iterative refinement of control strategies (McCord & Pilliod, 2022). For example, remote sensing technologies can detect biomass changes, vegetation greenness, or shifts in canopy structure, providing early warnings of new invasions or resurgences of previously controlled species (Joshi *et al.*, 2004; Zaka & Samat, 2024). When integrated into a feedback loop with on-the-ground observations, these tools enable adaptive management - adjusting responses based on changing invasion dynamics (Foxcroft & McGeoch, 2011; Jarnevich *et al.*, 2023).

As land use intensifies and climate change alters environmental baselines across southern Africa, predictive frameworks are becoming indispensable. They can forecast future hotspots under future scenarios, offering a proactive lens through which managers can allocate resources more effectively and prepare for emerging threats (Jagarnath *et al.*, 2019; Rashid *et al.*, 2025; Ren *et al.*, 2025; Smolik *et al.*, 2010). Long-term, multi-scale studies remain essential to ground-truth these predictions and provide continuity in understanding plant community trajectories.

Invasion dynamics along transport corridors are strongly influenced by the spatial structure of road networks and the repeated disturbance associated with their construction and maintenance. Road verges often function as ecological transition zones where disturbed soils, edge effects, and frequent propagule introduction create favourable conditions for alien plant establishment (Holway, 2005; Smolik *et al.*, 2010).

Understanding how these processes interact across space and time requires approaches that move beyond short-term observations and incorporate both long-term monitoring and spatial analytical tools. Such approaches are increasingly recognised as important for identifying invasion patterns and improving management strategies within linear infrastructure environments. In the context of the Bakwena N4 Toll Route, this is especially important because invasion risk is shaped not only by species presence but also by changing land use, repeated verge disturbance, and spatial variation in vegetation condition along the corridor.

**Conclusion**

AIPs pose a persistent ecological and management challenge across South Africa. Their spread is linked to a range of drivers, including disturbance, land transformation, propagule pressure, and fragmented governance. The literature reveals that AIP success is shaped by both environmental filters, such as soil, vegetation structure, and fire regimes, and anthropogenic changes that alter ecological resistance. These species disrupt biodiversity, ecosystem functioning, and economic activities, particularly in already vulnerable systems like urban fringes, transport corridors, and degraded landscapes. At the same time, the literature also shows that not all alien species become invasive, and that invasion outcomes depend on the interaction between species traits, environmental conditions, and disturbance regimes.

Efforts to define and regulate AIPs in South Africa have evolved under national legislation, particularly the NEMBA framework and Alien and Invasive Species Regulations. While this system classifies species by risk and outlines landowner responsibilities, its effectiveness depends on implementation and context. The interaction between species traits and local environmental conditions complicates blanket approaches, reinforcing the need for context-specific interventions. This is especially relevant in fragmented environments like road verges, where enforcement is difficult and ecological variation is high. The distinction between alien, naturalised, and invasive species is therefore important, both ecologically and legally, because not all introduced species pose the same level of risk or require the same management response.

The impacts of AIPs are well-documented across multiple systems. Ecologically, they reduce native species richness, alter fire and nutrient cycles, and modify hydrology. Infrastructure and agriculture are also affected, as invasive plants damage physical structures and reduce land productivity. Roads, in particular, both experience and drive invasion through disturbance, edge effects, and propagule transport. Post-construction succession often favours disturbance-tolerant AIPs, especially when maintenance unintentionally reinforces their spread. Long-term recovery is rare without targeted restoration.

Management strategies must therefore be adapted to the realities of linear infrastructure. While mechanical, chemical, and biological methods offer tools for control, their success depends on timing, coordination, and context. Integrated Weed Management, supported by spatial planning and jurisdictional alignment, offers a pathway forward. Monitoring and biodiversity indices such as Shannon, Margalef, Bray–Curtis and Sørensen allow researchers and managers to track invasion dynamics and assess intervention outcomes. Predictive modelling adds an essential dimension to this work by anticipating risk and informing proactive management.

In the specific context of road infrastructure, the interaction between repeated disturbance, high propagule availability, and spatial connectivity makes predictive modelling and monitoring especially relevant. Road reserves are not static environments but constantly changing ecological interfaces where biotic and abiotic conditions shift with each maintenance cycle, weather event, or land-use change (Holway, 2005; Smolik *et al.*, 2010). Studies that combine long-term vegetation data with spatial models, like the Bakwena N4 Toll Route project, provide a model for adaptive, evidence-based management. This integrative approach not only supports more efficient control but also advances the development of scalable frameworks for managing biological invasions across other linear infrastructure systems in South Africa.

This study responds to the knowledge gaps identified in the literature by applying biodiversity indices and long-term spatial monitoring to assess AIP dynamics along a major national route. By linking vegetation composition with ecological, land use, and institutional variables, the research offers site-specific insight into invasion trajectories. These findings will support more targeted management strategies and contribute to the development of adaptive frameworks for road verge restoration in high-risk zones. In this way, the study aligns with the broader need for road corridor management approaches that are evidence-based, spatially explicit, and responsive to both ecological and operational realities.

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# CHAPTER 4
# METHODOLOGY

## 4.1 Existing alien invasive plants (AIPs) data collected along Bakwena N4 Toll Route

This study aims to gather data regarding the frequency of AIPs along the Bakwena N4 Toll Route. To achieve its primary objectives, this study employs a quantitative research methodology, with a focus on the full evaluation of AIP plants along the Bakwena N4 Toll Route. The broader study area comprised the full Bakwena N4 Toll Route from Doornpoort to Skilpadshek, but the long-term Bakwena audit dataset analysed in Chapters 5 and 6 covered the monitored section between Pretoria/Tshwane and Zeerust, which was subdivided into standardised 4 km spatial units for analysis. The need for numerical data to allow a comprehensive quantification and analysis of the number of observed AIPs makes a quantitative research approach ideal for this study. The comprehensive evaluation of AIPs conducted in multiple locations along the Bakwena N4 Toll Route is intended to yield insights that can be applied to the toll route's larger context. The study drew on two complementary datasets: (1) annual roadside audit records collected by Bakwena from 2011 to 2024, and (2) paired plot-based vegetation surveys undertaken in road reserves and adjacent natural areas.

An annual toll route audit is required to assess the distribution, quantity, and ecological impact of AIPs on the Bakwena N4 Toll Route. Bakwena provided a substantial dataset spanning 14 years, from 2011 to 2024. This broad temporal scope allows for a detailed evaluation of trends and patterns (spatial and temporal) regarding the presence and dynamics of AIPs across time.

Visual surveys and inspections were conducted annually in April to coincide with the peak plant growth season. These surveys covered the entire Bakwena N4 Toll Route, including both road verges and the central median, to ensure comprehensive coverage of the road reserve. Due to the generally low density of alien invasive plants along the route, visual detection from the roadway was considered effective. Surveys were therefore undertaken by slowly driving along the yellow lane of the highway while visually scanning the road reserve for AIPs. The survey vehicle was equipped with appropriate safety measures, including a warning flag and flashing construction light, to ensure visibility to other road users during the inspection process.

To maintain consistency between years, the same general survey procedure was followed during each annual audit. The full route was traversed systematically, and the consultant visually scanned both sides of the road reserve and the median while travelling slowly enough to detect individual AIP occurrences. When an AIP was observed, the vehicle was stopped at the nearest safe location and the observation was verified on foot where necessary. Each individual plant or clearly distinguishable plant occurrence was recorded separately. For each record, the species identity, geographic position, and supporting field photograph were captured in Pic2Task. Where relevant, additional field notes such as plant height, surrounding vegetation condition, and other site characteristics were also recorded. This ensured that the monitoring dataset consisted of repeatable, georeferenced point observations rather than broad subjective site impressions.

When an alien invasive plant was observed within the road reserve, the consultant stopped the vehicle and conducted an on-site inspection. Each individual plant encountered was recorded as a separate observation in order to quantify abundance along the route. The species was identified using the consultant's plant identification expertise, and the observation was recorded using the mobile application Pic2Task. The application was used to capture a georeferenced photograph of the plant, automatically recording its GPS coordinates. The species name was entered into the application as part of the identification record, while additional information such as estimated plant height and other relevant field characteristics could also be captured. In addition to recording individual AIP occurrences, areas displaying poor vegetation condition or degradation were also documented during the surveys. The methodology used to assess vegetation quality is described later in this chapter.

Although the annual surveys were visual in nature, temporal change was assessed by comparing repeated observations collected in a consistent manner along the same road corridor over the full 2011 to 2024 period. Each observed AIP was recorded as a georeferenced field observation using the Pic2Task application, allowing observations from different years to be compared spatially and temporally. Annual totals and species records were then standardised as individuals per kilometre within defined route sections and 4 km spatial units, which made it possible to assess year-on-year changes in AIP abundance, distribution, and species composition despite the survey method being based on visual detection. Temporal change was therefore not inferred from subjective visual impressions alone, but from repeated annual records collected using the same monitoring approach and analysed quantitatively across time.

The data provided by Bakwena serves as the foundation for the categorisation of AIPs. Records collected over the past 14 years were used to classify alien invasive plants according to their respective regulatory categories, namely Category 1a, Category 1b, Category 2, and Category 3 species. In addition, the study assessed the spatial distribution of AIPs along the Bakwena N4 Toll Route relative to land use categories. These zones comprised urban (Pretoria to Brits), peri-urban (Brits to Rustenburg), and rural (Rustenburg to Zeerust) sections of the route.

The Pretoria to Brits section covers approximately 50 km, with road reserve widths ranging from approximately 15–25 m. The Brits to Rustenburg section extends for approximately 70 km, with road reserve widths ranging from approximately 15–40 m. The Rustenburg to Zeerust section covers approximately 45 km, with road reserve widths generally ranging from 10–20 m. Larger road reserve widths in parts of the Rustenburg section reflect previous provisioning for future road dualling. All monitoring data and plot locations used in this study were established prior to the commencement of the dualling project, and therefore reflect the original road reserve configuration.

Once the dualling project is completed, sections of the road reserve may be reduced in width.

The aim was to reveal patterns in these diverse zones, shedding light on the coexistence of multiple AIP categories within each geographical location. This research attempted to uncover potential links between the invasiveness of AIP categories and the driving variables in the relevant zones. By examining these patterns, the study aimed to uncover areas that require additional attention in the future. The invasiveness of the AIP category, as well as the unique characteristics of each zone, was critical in identifying locations that require targeted intervention and management methods. This analytical research, based on collaborative data classification and spatial examination, helped develop a more targeted and effective method for controlling and mitigating AIPs along the Bakwena N4 Toll Route.

## 4.2 Data analyses

A combination of descriptive and inferential statistical methods was used to explore patterns in AIP occurrence along the Bakwena N4 Toll Route between 2011 and 2024. The AIP dataset used in this chapter was derived from the annual Bakwena monitoring programme described in Chapter 5, Section 5.2.1. For analytical purposes, the route was subdivided into 4 km spatial segments based on the Bakwena blue marker system, allowing AIP records to be grouped into standardised spatial units for statistical analysis. As outlined in Chapter 5, Section 5.2.1, descriptive statistics, including annual means and standard deviations, were calculated to summarise AIP density across urban, peri-urban, and rural zones. To allow comparisons across different spatial extents, AIP counts were standardised as individuals per kilometre. These summaries supported the identification of overall trends and variation in AIP presence across space and time (Zar, 1999).

Regression analyses were used to assess temporal dynamics in annual AIP density and diversity metrics. Linear models were applied initially, and non-linear polynomial models were also explored, where they provided a better fit to observed trends (Shannon & Weaver, 1964), with each land-use zone treated separately. This approach, also detailed in Section 5.2.1, enabled the detection of long-term changes in species richness and abundance over the 14-year period and highlighted zone-specific patterns in response to management or environmental conditions.

Species composition changes were assessed using the Bray-Curtis Dissimilarity Index (Bray & Curtis, 1957) and the Sørensen Similarity Index (Sørensen, 1948), as described in Section 4.2.2. The Bray-Curtis index was used to compare year-on-year abundance data to reveal shifts in dominant species and overall community structure, while the Sørensen index was applied to presence-absence data to evaluate species turnover relative to the baseline year of 2011.

Hotspot identification followed a threshold-based method explained in Section 5.2.3. Quadrants were flagged as density hotspots if their AIP counts exceeded the 90th percentile. In addition, quadrants with Shannon-Wiener and Margalef diversity index values (Margalef, 1958; Death, 2008) greater than the mean plus two standard deviations were flagged as diversity hotspots. This provided a systematic basis for identifying areas with unusually high AIP concentrations or ecological complexity.

To examine potential drivers of AIP distribution, several Chi-square tests were performed (McHugh, 2013). These included tests of independence between vegetation quality categories ("good" vs "bad") and hotspot presence, as well as tests investigating the relationship between AIP occurrence and factors such as soil type, vegetation type, and land use zone (see Sections 5.2.3 and 5.3.3).

Finally, as explained in Section 6.3.2 of Chapter 6, plot-level AIP densities in natural areas and adjacent road reserves were compared using the Mann-Whitney U test (Mann & Whitney, 1947). This non-parametric approach was chosen because the AIP count data were not normally distributed, particularly in the denser woody Savanna vegetation types. The test allowed for an assessment of whether land-use position significantly influenced AIP abundance.

All statistical analyses were conducted using Microsoft Excel and Jamovi. This analytical approach supported both broad-scale landscape interpretation and fine-scale ecological insight, helping to identify management priorities and better understand AIP dynamics along the N4 corridor.

**4.3 Plot biodiversity data**

To investigate biodiversity variation and activities within distinct vegetation types affected by AIPs, a systematic plot placement strategy is required. This chapter's methodology outlines the selection and placement of various sampling plots, which is an essential step in revealing how AIPs affect biodiversity (Elzinga *et al.*, 1998; Kent, 2012).

The initial phase of the study involved surveying the entirety of the Bakwena N1/N4 Toll Route, spanning from Doornpoort to Zeerust. The objective of this survey was to identify the vegetation types present within the study area broadly. This preliminary step was essential, as different vegetation types necessitate distinct plot sizes and sampling methodologies to accurately characterize their composition, structure, and heterogeneity (Mueller-Dombois & Ellenberg, 1974).

As discussed in Chapter 2, the route traverses several types of savanna vegetation. For this study, these were grouped into the more open, grass-dominated Marikana Thornveld and the denser woody Savanna vegetation types along the route, which are generally characterised by greater woody cover, shrub density, or structural complexity.

The central objective of this study was to assess the prevalence of AIP species within the road reserve of the Bakwena N1/N4 Toll Route, and to compare it against the unmanaged natural areas immediately adjacent to the road reserve. This comparative approach aimed to provide insights into the effectiveness of AIP management strategies employed within the road reserve relative to uncontrolled natural areas. To achieve this objective, a strategic plot placement methodology was devised. Paired plots were established, with one plot positioned within the road reserve and a corresponding plot situated in the adjacent natural area, separated solely by the boundary fence. This pairing of plots allowed for a direct comparison of AIP species abundance and composition between the managed road reserve and the unmanaged natural areas (Barbour *et al.*, 1999; Kent, 2012).

The strategic placement of these paired plots was implemented across the Savanna vegetation settings identified along the route, with particular emphasis on the contrast between the more open, grass-dominated Marikana Thornveld and the other, generally more woody or structurally complex vegetation types. This approach ensured that the data collected represented contrasting conditions within both the road reserve and adjacent natural areas across the main vegetation structures encountered along the corridor. By applying this methodology, the study aimed to generate data that would support a comprehensive analysis of the effectiveness of AIP management practices within the road reserve, as reflected by the relative abundance and distribution of AIP species compared with adjacent unmanaged natural areas.

To ensure comprehensive data collection, a rigorous sampling approach was adopted. For the two broad vegetation structure groups used in this study, namely the more open, grass-dominated Marikana Thornveld and the remaining, generally more woody or structurally complex vegetation types, a total of 20 paired plots were established. Specifically, for each group, 10 plots were positioned within natural areas beyond the road reserve boundary fence, while 10 corresponding plots were established within the adjacent road reserve. This paired design, with plots separated only by the boundary fence, allowed for direct comparison of AIP prevalence between the managed road reserve and adjacent unmanaged natural areas. A diagrammatic representation of this plot placement strategy is provided (Figure 4.1), which illustrates the paired plot layout.

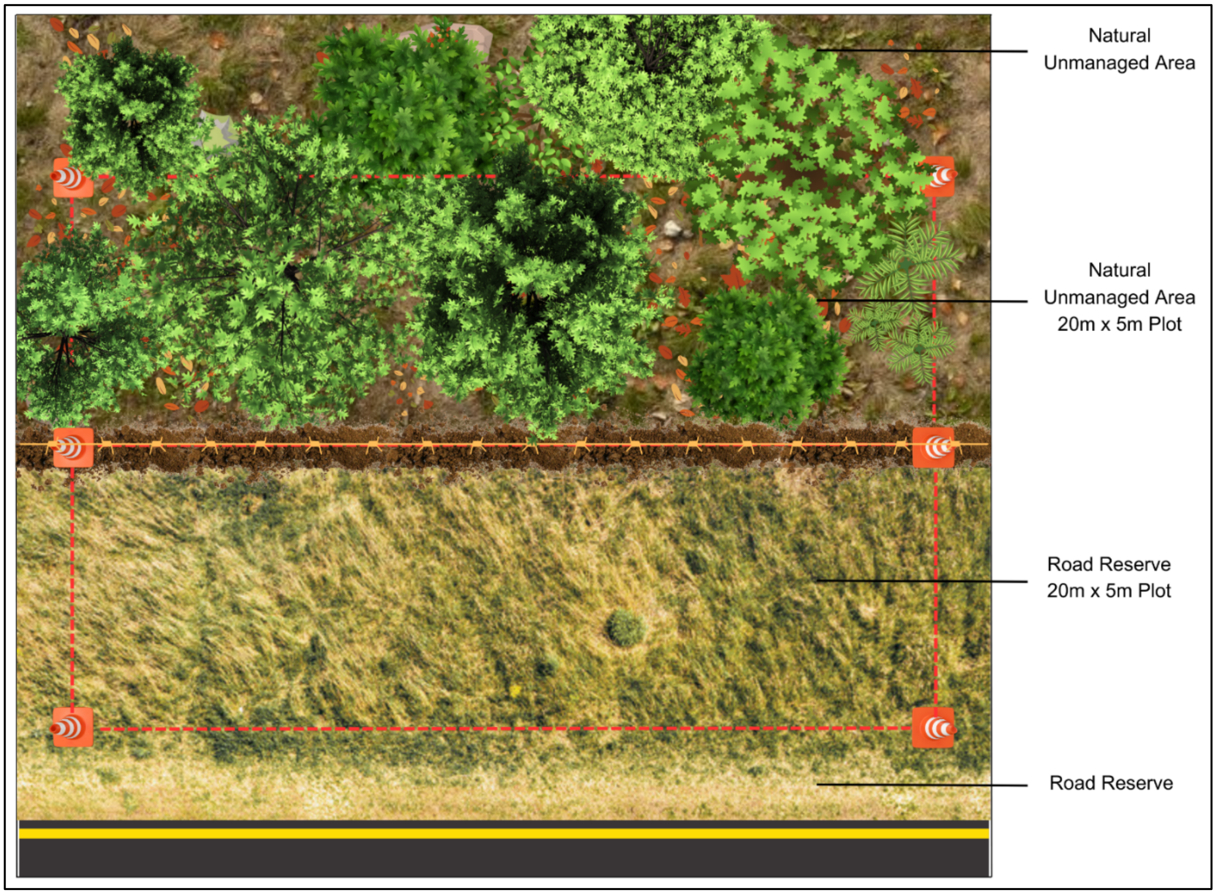

Figure 4.1 Visual representation of the paired plot layout.

Drawing on published vegetation sampling guidance and established plot-based survey methods, it was determined that twenty 100 m² plots would be used for the more woody and structurally complex vegetation types, while twenty 25 m² plots would be used for the more open, grass-dominated Marikana Thornveld. This difference in plot size was based on variation in vegetation structure across the study route. The more wooded vegetation types, which included larger shrubs and trees, required larger plots to capture vegetation composition and growth patterns more adequately. By contrast, the more open Marikana Thornveld, characterised by a stronger grass layer and less prominent woody cover, could be represented using smaller plots because of its comparatively simpler and more uniform vegetation structure (Mueller-Dombois & Ellenberg, 1974; Westhoff & van der Maarel, 1973).

The original sampling design proposed the establishment of 10 m x 10 m plots for the denser woody Savanna vegetation type. However, upon commencing fieldwork and assessing the site conditions, it became evident that the road reserve width, constrained by the boundary fence, would not accommodate plots of such dimensions. Consequently, an adaptive approach was adopted. To maintain the targeted sampling area of 100 $m^2$ per plot while accommodating the spatial constraints of the road reserve, plot dimensions were adjusted according to vegetation structure. In Marikana Thornveld, plots measured 10 m × 10 m, as the vegetation structure was sufficiently simple and open to allow full recording of species composition within the standard plot area, without the need for narrower subdimensions. For the denser woody Savanna vegetation types, where the road reserve imposed greater spatial constraints, plots were adjusted to 20 m × 5 m while still maintaining the same total sampling area of 100 $m^2$ per plot. This modification ensured that the paired plots, situated within the road reserve and the adjacent natural areas, respectively, would maintain consistent and comparable sampling areas.

The total length of the study route along the Bakwena N1/N4 Toll Road was calculated using Google Earth. The route traverses a range of Savanna vegetation types, and virtual markers were therefore used to identify the approximate extent of the main vegetation units along the corridor, including Marikana Thornveld and the remaining, generally more woody or structurally complex vegetation types further west. These divisions were used to support the interpretation of spatial patterns in AIP distribution along the route.

To ensure a representative and unbiased sampling strategy, a stratified random sampling approach was adopted. Sampling locations were distributed across the two broad vegetation structure groups used in this study, namely the more open, grass-dominated Marikana Thornveld and the remaining, generally more woody or structurally complex vegetation types along the route. Within each group, the route length associated with that vegetation setting was divided by the predetermined number of sampling locations to calculate approximate spacing intervals between sites.

This approach was used to achieve a spatially balanced distribution of plots along the study route while ensuring coverage of the main vegetation settings represented in the survey. By applying this stratified random methodology, the likelihood of excessive clustering was reduced, thereby improving the representativeness and statistical robustness of the dataset (Quinn & Keough, 2002).

Starting from the eastern section of the route, the path measurement tool in Google Earth was used to measure consecutive intervals along the corridor based on the spacing calculated for each vegetation structure group. Virtual markers, or pins, were then placed at each interval to identify the predetermined plot locations within the grass-dominated Marikana Thornveld and within the remaining, generally more woody or structurally complex vegetation types. This process enabled the planned distribution of sampling sites across the study route in a spatially balanced manner. A visual representation of the pin placement strategy, showing the distribution of selected plot locations along the route (Figure 4.2).

It is important to note that while this systematic approach guided the initial identification of plot locations, field-based assessments and adaptations were necessary to ensure the feasibility and accessibility of the sampling sites. Various logistical challenges and environmental constraints were encountered during the fieldwork phase, necessitating the relocation of certain plots to the nearest accessible areas.

One notable challenge arose in the Marikana Thornveld, where the presence of mining operations and associated electrical fencing precluded the establishment of plots within the intended natural areas. In such instances, the plots were repositioned to the closest viable location beyond the confines of the mining operations, maintaining the integrity of the sampling design while accommodating site-specific constraints.

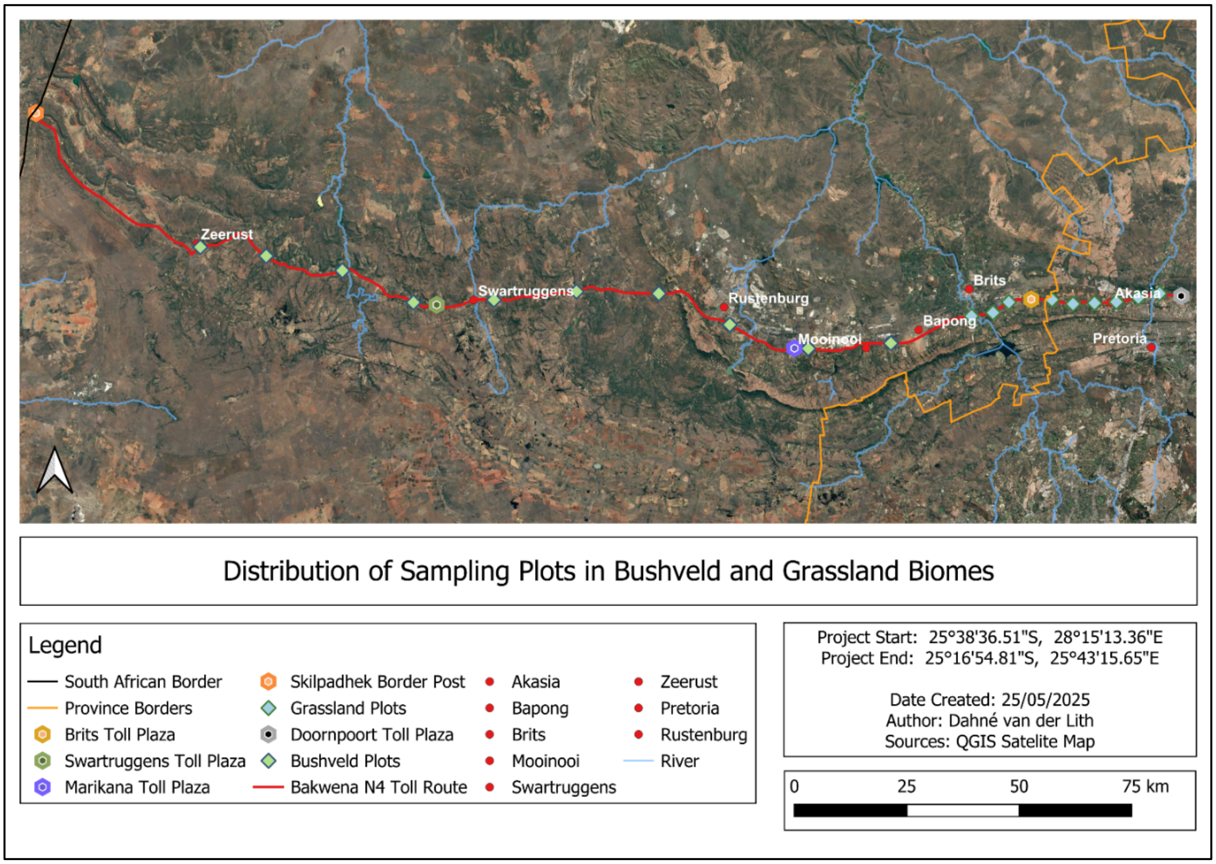


Figure 4.2 Visual representation of the distribution of the plots using the stratified sampling method.

To facilitate comprehensive data collection and documentation, the Pic2Task application was employed as a principal component of the fieldwork methodology. This versatile application enables the integration of multiple data streams, including high-resolution photographs, precise geographic coordinates, qualitative observations and vegetation characterisation parameters. The photographic capabilities of Pic2Task were used to capture detailed visual representations of the vegetation composition and structure at each sampling location. These photographic records will serve as an invaluable resource for later analysis and verification.

In addition to the visual documentation, Pic2Task allows for the annotation of each photograph with relevant metadata and field observations. Parameters such as vegetation height, qualitative assessments of vegetation quality, and the identification and enumeration of specific plant species can be recorded directly within the application, ensuring a comprehensive and organised dataset. Furthermore, Pic2Task incorporates advanced GPS tracking functionalities, enabling the precise geo-referencing of each photograph and associated data entry. This feature is particularly crucial in the context of this study, as it facilitates the accurate mapping and spatial analysis of the collected data, enabling the integration of the field observations with the predetermined sampling locations and the broader geographic context of the study area.

Careful quantification was conducted for every plant species that has been identified in every plot using the Pic2Task application. This covers the identification of species, counting, and centimetre-based height measuring. A modified Braun-Blanquet cover-abundance scale (see Table 4.1) was used to measure the cover abundance of each species, based on accepted techniques (Barbour *et al.*, 1987; Mueller-Dombois & Ellenberg, 1974).

Table 4.1. The modified Braun-Blanquet cover-abundance scale.

| Score | Modified Score | Description |
|---|---|---|
| N | 0.1 | Not many, 1-10 individuals |
| T | 0.5 | Sparsely or very sparsely present; cover very small (less than 5%) |
| 1 | 1 | Plentiful but of small cover (less than 5%) |
| 2 | 2 | Any number of individuals covering 5-25% of the area |
| 3 | 3 | Any number of individuals covering 25-50% of the area |
| 4 | 4 | Any number of individuals covering 50-75% of the area |
| 5 | 5 | Covering more than 75% of the area |

Since higher cover-abundance indicates dominance, this scale makes it possible to identify the dominant species in the environment. The assessment of the biodiversity of the plots was achieved by quantifying the cover-abundance of both native and AIPs, which offered a more detailed perspective on the ecological richness and complexity. Monitoring the cover-abundance of AIPs provided additional information about their possible impact on native vegetation and invasiveness. In addition, variations in the amount of cover over time functioned as markers of the health of the ecosystem, providing important details on disturbances and succession patterns.

One important indicator that provided information about the topographical dynamics of the research area along the Bakwena N4 Toll Route was the slope gradient measure within the plot data. Slope gradients affect soil moisture content and plant growth by changing water drainage patterns. Plant species differentially prefer microhabitat changes caused by varying slope angles. Relationships between topography and biodiversity were shown by analysing slope data in combination with plant diversity assessments. The methods suggested by Raghuvanshi *et al.* (2014) (Table 4.2) were used to measure slope gradients.

Table 4.2. Slope estimation classes

| **Class** | **Value Range** |
|---|---|
| Very gentle slope. | <15° |
| Gentle slope. | 16 - 25° |
| Moderately steep slope. | 26 - 35° |
| Steep slope. | 36 – 45° |
| Escarpment/cliff. | >45° |

Taxonomic consistency was given priority in this study due to the dynamic nature of plant taxonomic terminology. The South African Plant Names Database's terminology was followed by the plant names used in the analysis (SANBI, 2023).

### 4.4 Vegetation Quality as Measurement of Ecological Condition

In this study, vegetation quality was defined as the degree to which a site deviates from a benchmark condition, represented by natural plots that reflect undisturbed, mature vegetation communities (Parkes *et al.*, 2003). These reference plots were used to assess ecological integrity on the assumption that they represent the typical structure and composition of healthy native vegetation. The assessment accounted for both average conditions and the natural variation expected within the landscape, ensuring that comparisons were realistic and ecologically valid (Boyle *et al.*, 2018; Kent, 2012).

The aim of assessing vegetation quality was to evaluate whether areas with reduced ecological integrity were more likely to support AIP species. A visual evaluation method was applied across the N4 corridor, focusing on a set of ecological indicators that capture both the structure and function of vegetation. These included species composition, vegetation structure, canopy and ground cover, and evidence of disturbance (Elzinga *et al.*, 1998). In addition to this qualitative assessment, the Shannon–Wiener Index was used to compare species diversity between disturbed and undisturbed areas, providing a quantitative measure of variation in diversity (Shannon & Weaver, 1964).

The evaluation considered several key variables known to influence vegetation conditions. Density and basal cover were measured to provide insights into the resilience and stability of vegetation, recognising that declines in basal cover below critical thresholds often signal ecological stress (Bonham, 2013; Gao *et al.*, 2011; Mueller-Dombois & Ellenberg, 1974). Growth and vigour were assessed through plant vitality, leaf size, and shoot density, while successional composition was noted to capture the trajectory of recovery following disturbance, moving from pioneer to subclimax to climax stages (Elzinga *et al.*, 1998; Walker & del Moral, 2003). Further indicators such as bare patches, evidence of erosion, leaf condition, disease or pest symptoms, root development, and weed competition were also incorporated into the assessment to capture a complete picture of ecological health (Bonham, 2013; Brady & Weil, 2010; Elzinga *et al.*, 1998; Gaertner *et al.*, 2014; Holmes *et al.*, 2020; Richardson & Pyšek, 2006).

While these variables were not analysed separately, together they informed the overall vegetation quality rating. The strength of this approach is that it combines multiple ecological attributes into a single, practical, and repeatable measure of site condition. This enabled consistent comparisons across the full length of the corridor and provided a reliable basis for testing relationships between vegetation condition and the occurrence of invasive plants (Elzinga *et al.*, 1998; Kent, 2012).

For analysis, vegetation quality classes were further grouped into broader ecological condition categories. Sites classified as “good condition” were those falling within the green category, where vegetation cover was largely intact or recovering, visible erosion was limited or stable, and the site showed relatively lower levels of disturbance and invasive plant pressure. Sites classified as “bad condition” were those falling within the orange and red categories, where vegetation showed clear signs of degradation, including reduced cover, exposed soil, erosion, disturbance, and greater evidence of ecological stress. This grouping provided an objective basis for comparing AIP occurrence between relatively intact or recovering sites and more visibly degraded sites.

After the visual vegetation quality assessment is completed, the plants are categorised according to their individual statuses (Table 4.3). For analysis, red and orange zones were grouped as bad condition sites, while green and blue zones were grouped as good condition sites, and AIP occurrence was compared between these two broader ecological condition classes. The Shannon-Wiener Index is applied to make this comparison easier (Shannon & Weaver, 1964).

Table 4.3. Visual vegetation quality assessment status descriptions

| Status Colour | Status Description |
|---|---|
| Red (Status 1) | Requirement for urgent attention – Slippages, evidence of erosion inside the drainage line, etc. |
| Orange (Status 2) | Requirement for attention – Currently, no serious risk, erosion evident but stable. |
| Green (Status 3) | Remedial work has started – requirement for continued monitoring until sustainable results are achieved. |
| Blue (Status 4) | Construction work is currently underway – a requirement for continued monitoring until sustainable results are achieved. |

This analysis aimed to determine whether vegetation quality is associated with AIP abundance and distribution. By comparing vegetation condition categories with AIP presence, the study identified ecological patterns that may inform targeted management strategies. The vegetation quality assessment helped highlight areas where lower-quality vegetation coincides with higher AIP presence. For example, if AIP species were consistently more prevalent in red-status zones, this may indicate that degraded vegetation conditions contribute to invasion risk. Understanding these associations can support more effective prioritisation of management interventions in vulnerable areas.

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# CHAPTER 5

# LONG-TERM CHANGES IN ALIEN INVASIVE PLANTS (RATE OF CHANGE, FUTURE PREDICTIONS, PROBLEM AREAS)

## 5.1 Introduction

Alien invasive plants (AIPs) along highways present a major ecological challenge as transportation corridors facilitate their rapid spread (Baard & Kraaij, 2019; Meunier & Lavoie, 2012). The Bakwena N4 Toll Route illustrates this problem, as it spans urban, peri-urban, and rural areas where AIPs continue to disturb native ecosystems despite ongoing management and removal efforts. As with all plant communities, species composition changes naturally over time due to environmental conditions, human activities, climate change, and species interactions (Tylianakis *et al.*, 2008). Tracking these temporal changes in AIP communities provides valuable insight into the drivers of invasion, the resilience of ecosystems, and the effectiveness of ongoing control interventions.

Long-term monitoring is particularly important for assessing whether eradication programmes have achieved sustained reductions in invasive plant populations or whether declines are temporary and followed by reinvasion. Understanding these trends enables managers to distinguish between areas where interventions have had lasting impact and areas where invasions remain persistent. In addition, identifying hotspots of invasion and projecting future spread patterns can guide the prioritisation of resources and the refinement of management strategies along the corridor.

This chapter addresses Objective 1 by analysing long-term changes in AIP abundance, diversity, and distribution along the Bakwena N4 Toll Route between 2011 and 2024. It also addresses Objective 4 by applying a suite of biodiversity indices, including the Shannon–Wiener Diversity Index, Pielou's Evenness Index, Margalef's Richness Index, Bray–Curtis Dissimilarity, and Sørensen's Similarity, to evaluate diversity, evenness, and species turnover. Finally, it contributes to Objective 5 by identifying persistent hotspots and problem areas using spatial flagging and Chi-square testing, and by evaluating whether degraded vegetation is linked to higher invasion risk. Together, these analyses provide a clearer understanding of how AIP populations have responded to control measures over more than a decade and highlight areas where further management is required to support long-term ecological recovery along the route.

Trends in AIP occurrence were interpreted by examining changes in species abundance, density, and diversity across the sampled plots and across different environmental contexts along the corridor. Plots exhibiting consistently higher AIP densities or repeated occurrences of invasive species were interpreted as potential invasion hotspots. Predictions regarding potential future invasion risk were based on the observed relationships between alien plant occurrence, disturbance intensity, vegetation condition, and surrounding land-use characteristics recorded during the surveys. These interpretations, therefore, reflect the spatial and temporal patterns identified in the plot data and the associated environmental variables measured during field sampling.

## 5.2 Alien Invasive Plant Management Along the Corridor

Alien invasive plants occurring within the road reserve are actively managed through a chemical control programme implemented by environmental management consultants responsible for vegetation maintenance along the corridor. Control is primarily undertaken through the application of herbicides to targeted alien plant species occurring within the road reserve. Herbicide products are selected according to the species present, and application rates follow the dosage instructions provided for each specific product. In order to minimise the risk of herbicide resistance developing within invasive plant populations (Evans *et al.*, 2016), different herbicide formulations are rotated periodically as part of the management programme.

Monitoring of alien plant occurrences forms part of the consultants' routine environmental management responsibilities along the route. The corridor is inspected frequently, as the route is driven on an almost daily basis while undertaking other environmental management duties. During these inspections, new or previously recorded alien plant occurrences are noted and incorporated into the ongoing control programme. Herbicide treatments are then applied as soon as practically possible after identification.

Due to the length of the corridor and the time required to undertake chemical treatments effectively, the entire route is generally treated over a cycle of several months. This means that sufficient time may elapse between treatment cycles for new alien plants to establish and grow. As a result, new occurrences are often recorded during routine inspections and subsequently treated during follow-up control operations.

The management of alien invasive plants along the road reserve also takes place within a broader operational environment involving multiple contractors responsible for different aspects of corridor maintenance. For safety reasons, roadside vegetation must be maintained regularly to ensure adequate visibility for motorists and to prevent excessive vegetation growth along the road reserve. This typically involves routine mowing of roadside vegetation. Because different contractors are responsible for different maintenance activities, the environmental consultants responsible for alien plant control do not always have direct control over the timing of mowing operations.

In some instances, alien plants that have been recorded during inspections may be removed incidentally during routine mowing activities before chemical treatment can be applied. While this does not necessarily hinder the overall management objective, it does introduce an element of variability in the timing of control measures. Consequently, some alien plant occurrences recorded during surveys may no longer be present when follow-up chemical treatment is undertaken. The combined effects of routine monitoring, herbicide application, and general vegetation maintenance therefore result in a dynamic process of detection and control of alien invasive plants along the corridor.

### 5.3 Methods

An annual toll route audit is conducted to assess the distribution, quantity, and ecological impact of AIP on the Bakwena N4 Toll Route. Bakwena provided a comprehensive dataset spanning 14 years, from 2011 to 2024, enabling a detailed evaluation of trends and patterns in AIP presence and dynamics over time. The data collection process involves annual visual surveys and inspections, conducted every April to align with the plant growth season. These surveys encompass the entire Bakwena N4 Toll Route, including both sides of the road and the median, ensuring a thorough and representative analysis of the terrain. A consultant, equipped with expertise in plant identification, conducts the inspections using a technology-driven approach.

To facilitate spatial analysis of the long monitoring record, the Bakwena N4 Toll Route was divided into standardised spatial units. Bakwena uses a blue marker system along the highway, with markers placed at approximately 200 m intervals to assist with location identification and operational management. These markers were used as reference points to subdivide the route into 4 km segments, with each segment consisting of 20 consecutive blue markers. Given that the monitored section of the route is approximately 187 km long, this subdivision resulted in approximately 80 spatial segments. The use of these 4 km segments allowed AIP records to be grouped into manageable spatial units, enabling consistent comparison of invasion patterns along the entire route while maintaining sufficient spatial resolution to detect localised hotspots. These segments are referred to as quadrants in this study and were used as the primary spatial unit for calculating diversity indices, analysing abundance patterns, and identifying areas of persistent invasion. The full site-level dataset of alien invasive plant records across all quadrants and survey years is provided in Appendix A.

To evaluate the changes in AIP populations over the 14-year study period, this chapter employed various statistical analyses. Confidence intervals were examined for the relevant summary measures to indicate the degree of variation around the estimates and to support the interpretation of temporal and spatial patterns in AIP distribution.

### 5.3.1 Statistical Analyses of AIP Indices

To systematically assess the temporal and spatial variation in AIP populations, several ecological indices were applied. These indices provided a structured approach to evaluate species diversity, richness, evenness, and compositional stability over time and across different zones (urban, peri-urban, and rural) along the Bakwena N4 Toll Route. The zones correspond to approximately 50 km (Pretoria–Brits), 70 km (Brits–Rustenburg), and 45 km (Rustenburg–Zeerust) sections of the route, providing a spatial framework for comparing invasion dynamics across different land-use contexts.

The Shannon-Wiener Index was used to measure species diversity, incorporating both species richness (number of species) and evenness (distribution of species within a population). The index was calculated annually for each zone to track temporal trends in AIP diversity and assess the impact of management interventions. To assess whether diversity was increasing, decreasing, or remaining stable over time, regression analysis was conducted on the annual Shannon-Wiener Index values. This analysis was performed separately for each zone (urban, peri-urban, and rural), as well as at the 4 km quadrant level within each zone, allowing both large-scale and fine-scale temporal patterns in species diversity to be evaluated (Shannon & Weaver, 1964):

$$H' = -\sum_{S=1}^{S} p_S \, ln\,(p_S) \qquad \text{Equation 5.1}$$

The Pielou’s Evenness Index was applied to determine how evenly individual species were distributed within each area. A value closer to 1 indicates an even distribution of species, while values closer to 0 suggest dominance by one or a few species.

Evenness was calculated annually and analysed for each zone to detect whether management efforts influenced the distribution patterns of AIPs (Pielou, 1966):

$$J' = \frac{H\prime}{ln\,(S)} \qquad \text{Equation 5.2}$$

To evaluate species richness, the Margalef Richness Index was employed. This index considers the total number of species recorded each year in relation to the sample size, providing a standardized measure of richness across different environments. High richness values indicated areas with more diverse AIP populations, potentially complicating management efforts (Death, 2008):

$$D = \frac{S-1}{ln\,(N)} \qquad \text{Equation 5.3}$$

The Bray-Curtis Dissimilarity Index was used to assess changes in species composition over time by measuring differences in species abundance between years or zones. A high Bray-Curtis dissimilarity value between two years suggested substantial changes in species composition, such as the dominance of certain species and the decline of others. The index was calculated for each year by comparing the species composition to the baseline year (2011), reflecting shifts in community structure over time (Bray & Curtis, 1957):

$$C = \frac{\sum_{j=1}^{S} |x_{Uj} - x_{Vj}|}{\sum_{j=1}^{S} (x_{Uj} + x_{Vj})} \qquad \text{Equation 5.4}$$

Similarly, the Sørensen Similarity Index was applied to evaluate species composition stability by comparing the presence or absence of species over time. This index highlights areas where invasive species consistently appear, aiding in predicting future invasion patterns. Sørensen values were calculated for each year relative to the baseline year (2011), allowing for an assessment of compositional stability or turnover in different zones (Sørensen, 1948):

$$C = \frac{2c}{a+b} \qquad \text{Equation 5.5}$$

To analyse whether AIP distributions followed a random, uniform, or clustered pattern, a Poisson distribution model was applied. The Poisson distribution calculates the probability of a given number of occurrences within a fixed area, assuming events occur independently and at a constant rate. The formula used was (Krebs, 1999; Poisson, 1837):

$$P_x = e^{-\mu} \left(\frac{\mu^x}{x!}\right) \qquad \text{Equation 5.6}$$

To statistically assess whether the observed AIP distribution significantly deviated from the expected Poisson distribution, a Chi-square goodness-of-fit test was performed. This test determines if the observed distribution of AIPs fits an expected random pattern. The Chi-square formula used was (Krebs, 1999; Pearson, 1900):

$$x^2 = \sum \frac{(O_i - E_i)^2}{E_i} \qquad \text{Equation 5.7}$$

If the calculated Chi-square value exceeded the critical threshold, the null hypothesis, that AIP occurrences followed a Poisson distribution, was rejected.

To support comparisons across time and space, the mean formula was used to calculate the average number of AIPs recorded per sample area (4 km road quadrants). The formula for the mean is (Fisher, 1992):

Threshold = Mean + (2 x Standard Deviation). Equation 5.8

This formula was applied to determine the average AIP abundance per section along the Bakwena N4 Toll Route.

### 5.3.2 Historical Trends in AIP Abundance

To analyse trends in the abundance of AIP species along the Bakwena N4 Toll Route from 2011 to 2024, a standardised metric was used: AIP counts per kilometre (km). For each year and zone (urban, peri-urban, rural), the number of AIP individuals recorded was divided by the total length of the respective road section. This provided a consistent measure of AIP density (individuals/km), allowing comparisons across zones of different sizes.

Due to the typical skewness and variability of ecological count data, the AIP density values were natural log-transformed (ln) before statistical analysis. This transformation helped stabilise the variance, reduce the influence of extreme values, and improve the linearity of the relationship between AIP density and time. It also enabled more accurate interpretation of the rate of change as a proportional rather than an absolute difference.

Regression analysis was conducted to assess temporal trends in AIP density over the 2011 - 2024 period. Both linear and non-linear polynomial regression models were applied to identify the nature of any directional trends. Analyses were performed separately for urban, peri-urban, and rural zones to account for environmental and management differences across the landscape. The coefficient of determination ($R^2$) and p-values were used to evaluate the strength and significance of the observed relationships.

All statistical analyses were conducted using Jamovi and Microsoft Excel. These platforms enabled the calculation of trendlines, fit statistics, and the visualisation of annual AIP density patterns over time. Where model assumptions were not met (e.g., homoscedasticity or normal residuals), polynomial models were used to better capture observed trends in the data.

### 5.3.3 Historical Trends in AIP Diversity and Composition

This section examines historical trends in the diversity and composition of AIP species along the Bakwena N4 Toll Route from 2011 to 2024. The aim is to evaluate how AIP communities have changed over time and to assess their response to ongoing management interventions. Identifying these temporal patterns supports the prediction of future invasion dynamics and enables more focused, zone-specific planning. To capture these changes, a combination of diversity and composition indices was applied across urban, peri-urban, and rural zones, allowing for comparisons at multiple ecological scales.

Species diversity was assessed using the Shannon-Wiener Index and the Margalef Richness Index, both of which were calculated annually for each zone (urban, peri-urban, and rural), as well as for each 4 km quadrant within those zones. The Shannon-Wiener Index provided a measure of both species' richness and evenness, offering insight into how evenly individuals were distributed among AIP species. This index was particularly useful for identifying whether diversity was increasing due to the suppression of dominant species or decreasing due to the spread of a few high-impact AIPs.

In contrast, the Margalef Index focused on richness while adjusting for total AIP abundance, making it more suitable than raw species counts, which can be misleading when rare species are present but not ecologically dominant. Both indices were visualised using mean ± standard deviation (SD) plots, which helped illustrate interannual variability and the potential impact of management efforts. Where trends were observed, regression analysis was conducted using linear or polynomial models, depending on the pattern in the data.

Species composition was assessed using the Bray-Curtis Dissimilarity Index and the Sørensen Similarity Index, with each year compared to the values in 2011 as the baseline. The Bray-Curtis Index was calculated using abundance data to evaluate how species assemblages shifted in terms of dominance and presence. This index identified not only changes in the number of species, but also in the relative abundance of dominant versus less common AIPs. The Sørensen Index, by contrast, was calculated using presence/absence data, highlighting species turnover and the retention or loss of species across years.

Both the Bray-Curtis and Sørensen indices were visualised with time-series graphs that included mean ± standard deviation (SD) lines to assess year-to-year variation and detect whether compositional changes remained within expected bounds. A general mean value was calculated across all years for each zone (urban, peri-urban, and rural), providing a central reference point for evaluating similarity trends. The standard deviation for each year was calculated based on the Sørensen index and Bray-Curtis Dissimilarity Index values obtained from all individual 4 km quadrants that made up the respective zone, ensuring that spatial variability within each landscape type was accurately reflected.

Using these values, upper and lower thresholds were constructed for each year by adding and subtracting the standard deviation from the mean (i.e., upper threshold = mean + SD; lower threshold = mean − SD). These thresholds were then visualised as green lines flanking the mean, and error bars were applied to each annual index point to show the standard deviation derived from the 4 km quadrant-level data. This approach allowed the graph to highlight both temporal and spatial variability and to assess whether changes in similarity remained within expected natural fluctuation or indicated more significant compositional shifts. Regression analysis was applied where visual patterns suggested directional change, with polynomial models used to better capture any non-linear trends in species turnover over time.

All analyses were performed using Jamovi and Excel, which supported the calculation of indices, transformation of data, and statistical testing of temporal patterns. The use of visualisation and regression models helped clarify whether diversity and composition were shifting in response to management or broader ecological dynamics, and whether specific zones showed more pronounced trends over time.

### 5.3.4 Identifying Problem Areas

To identify problem areas along the Bakwena N4 Toll Route, a variety of statistical and ecological analyses were undertaken to pinpoint zones where AIP species are particularly persistent or problematic. This process firstly involved calculating key biodiversity indices mentioned above, followed by an assessment of temporal trends, and subsequently to that, species distributions, and performing statistical tests to assess patterns and trends. Problem areas were defined as zones that exhibited high initial (2011) diversity, richness, or evenness of AIP species, or areas where species composition remained stable and invasive populations re-established consistently, i.e., where eradication had little impact.

The Shannon-Wiener Index and Pielou's Evenness Index (Equation 4.2) mentioned above were calculated to assess species diversity and the distribution of individuals among species, respectively. Pielou's Evenness Index measures how evenly individuals are distributed across species. A value closer to 1 indicates even distribution, while a value closer to 0 suggests dominance by one or a few species. High evenness and diversity values were considered indicative of hotspots where multiple invasive species coexist, potentially complicating management efforts.

The Margalef Richness Index (Equation 4.3) is another important measure used to assess species richness within a given area. This index emphasizes species richness by accounting for the number of species observed relative to the total population size, offering insights into the complexity of the ecosystem. Margalef's Index provides a straightforward way to evaluate biodiversity by focusing on species count, which forms the foundation for calculating the Shannon-Wiener Index. A high value of the Margalef Index indicates greater richness, which can signal areas of high ecological importance or areas where invasive species management might face additional challenges due to the diversity of species present.

The composition of invasive species within and between zones was assessed using the Bray-Curtis and Sørensen similarity indices. The indices were calculated from the historical data to determine problem areas.

The study area was divided into 4-kilometre quadrants. Each quadrant's data, collected annually, included the number of AIPs and the Shannon-Wiener, Pielou, and Margalef indices calculations. This subdivision allowed for fine-scale spatial analysis to identify specific hotspots. To further investigate plant distribution, histograms of AIP plant counts across quadrants were generated for each year. These histograms revealed whether the distribution of AIPs was aggregated, random, or uniform. To statistically validate the observed distributions, a Poisson distribution (Equation 4.6) was calculated.

The expected frequencies from the Poisson distribution were then scaled by multiplying by the total number of quadrants (80). These values were compared to the observed frequencies using a chi-square test for goodness of fit. To further identify specific hotspots, a 90$^{th}$ percentile threshold was calculated for alien plant counts. The 90$^{th}$ percentile represents the value below which 90% of the data falls, highlighting the top 10% of quadrants with the highest counts. In Excel, the Percentile formula was used to calculate the 90$^{th}$ percentile value, and conditional formatting was applied to flag cells exceeding the threshold. Similarly, thresholds for the Shannon-Wiener and Margalef indices were determined by adding two standard deviations to the mean.

Quadrants exceeding these thresholds were flagged as having exceptionally high diversity or richness, identifying them as potential problem areas. These thresholds provided a robust method for distinguishing quadrants with unusually high ecological activity.

For Pielou’s Evenness Index, raw values were examined directly. Values closer to 1 indicated a highly even distribution of species, which, when combined with high Shannon or Margalef values, suggested areas where multiple invasive species thrived simultaneously. Such areas were marked as significant management challenges because they reflected ecosystems where invasive species successfully coexisted and persisted over time. By integrating these various analyses, the study was able to pinpoint specific problem areas along the Bakwena N4 Toll Route, guiding targeted interventions to mitigate the spread and impact of AIP species.

In addition to species-based indices, vegetation quality was assessed during each annual audit as a supporting indicator of ecological condition. Data provided by Bakwena included annual vegetation condition ratings for the Bakwena N4 Toll Route from 2011 to 2024, categorised into four colour-coded status levels. These ranged from urgent erosion issues (red) to stable or actively managed conditions (blue). To enable consistent year-on-year comparison and spatial analysis, these qualitative ratings were translated into a numeric system (1 to 4) corresponding to their original categories. The ratings were then mapped onto the existing 4 km quadrants used for AIP monitoring, with each quadrant assigned a yearly status based on the dominant condition reported within its extent.

These ratings captured aspects such as plant health, erosion, ground cover, and vegetation structure. While not a quantitative ecological index, this classification offered contextual insight into areas where vegetation degradation might influence AIP persistence. Quadrants identified as severely degraded were cross-referenced with AIP abundance and diversity thresholds to explore potential ecological feedback loops in which poor vegetation quality may promote the establishment or re-establishment of alien species.

To examine the statistical association between vegetation condition and invasive plant hotspots, the vegetation quality values were first standardised across years into binary categories: “Bad” (ratings 1 or 2) and “Good” (ratings 3 or 4). These were then matched to quadrant-level flags for AIP problem areas based on whether the quadrant exceeded the 90th percentile for species count, diversity indices (Shannon-Wiener, Margalef), or combined thresholds. A binary flag was created for each year to indicate whether a quadrant was considered a hotspot (flagged = 1) or not (flagged = 0).

For each year, the number of flagged and non-flagged quadrants was tabulated by vegetation category (“Bad” vs “Good”), resulting in a set of 2×2 contingency tables. These tables were then subjected to Chi-square tests of independence in Excel to determine whether vegetation quality was significantly associated with hotspot status. The analysis enabled year-by-year evaluation of whether degraded vegetation conditions corresponded with statistically higher levels of AIP invasion. This integration of qualitative vegetation data into the quantitative hotspot framework supported a more holistic assessment of environmental drivers influencing alien species dynamics along the route.

## 5.4 Results and Analysis

5.4.1 Historical Trends in AIP Abundance.

From 2011 to 2024, the counts of AIP per kilometre in urban areas fluctuated, starting at 1.6 in 2011 and decreasing to a low of 0.0 in 2013 before gradually increasing again to 1.3 in 2023. The overall mean count for AIPs per kilometre in urban areas during this period was 0.7 (Appendix A; Table S5.8). Regression analysis indicated a slight positive trend in AIP counts over time; however, the result was not statistically significant. The low coefficients of determination values of the linear and polynomial regression ($R^2$ = 0.1813 and 0.189) suggest that no meaningful trend can be inferred. The observed fluctuations appear to be random rather than following a clear increasing or decreasing pattern. Given the weak fit of both the linear and polynomial regression models, these results do not provide strong evidence for a consistent temporal change in AIP densities in urban areas.

From 2011 to 2024, the counts of AIP per kilometre in peri-urban areas varied, starting at 2.1 in 2011 and reaching a low of 0.0 in 2015 before stabilizing around 0.9 in the later years (Appendix A; Table S5.8). The overall mean count for AIPs per kilometre in peri-urban areas during this period was 0.8. Linear regression analysis indicated no clear trend in AIP densities over time in peri-urban areas. The relationship was not statistically significant, and the extremely low $R^2$ = 0.044, suggests that changes in AIP counts are likely due to natural fluctuations rather than a consistent trend. A polynomial regression model was applied to better represent the observed variability in AIP densities but also returned a low $R^2$ value.

From 2011 to 2024, the counts of AIP per kilometre in rural areas ranged from a high of 1.6 in 2011 to a low of 0.2 in multiple years, with a mean count of 0.5 over the study period (Appendix A; Table S5.8). The polynomial regression model (Figure 5.1) provided a strong fit to the observed data, ($R^2$ = 0.6158), indicating that a non-linear pattern better explains the changes in AIP densities over time. The trend suggests that

AIP densities initially declined sharply in rural areas, reaching a low point around 2018, before stabilising and showing a slight increase after 2020. This pattern implies that while management efforts may have successfully reduced AIP densities in the earlier years, the recent plateau and upward trend suggest a possible resurgence or stabilisation rather than a continued decline. The improved model fit highlights the complexity of AIP population dynamics and the potential influence of external environmental or management factors. All regression results can be found in Chapter 5, Supplementary Material.

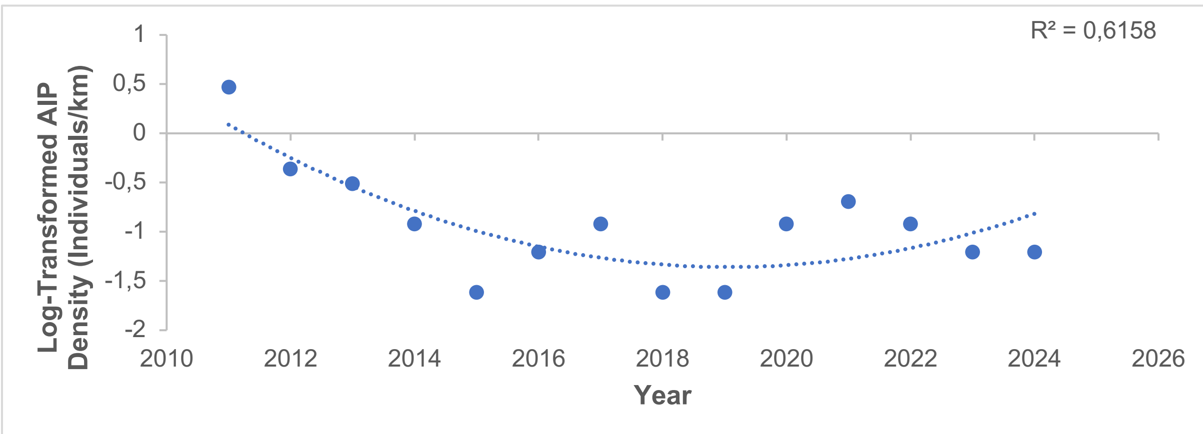



Figure 5.1 Non-Linear Polynomial Regression Model for Log-Transformed AIP Counts per Kilometre in Rural Areas (2011–2024)

### 5.4.2 Historical Trends in AIP Diversity and Composition

**Urban Areas**

In urban areas, the Shannon-Wiener Index peaked at 2.37 in 2011 before declining sharply in 2013 and 2015, where values approached 0. From 2016 onward, the index gradually increased and stabilised between 1.2 and 1.8 for the rest of the study period. A non-linear polynomial regression model ($R^2 = 0.2809$) was fitted to the data, capturing the initial decline and later recovery in diversity. While the regression suggests some temporal structure, the relatively low $R^2$ indicates that time alone does not strongly explain changes in diversity.

A second graph, displaying yearly Shannon-Wiener values with mean ± standard deviation (Figure 5.2), revealed high variability between 2011 and 2015, shown by wide error bars and sharp year-to-year shifts. This pattern indicates a period of ecological instability, likely influenced by the initial stages of invasion and early management responses. From 2016 onward, the index values became more consistent, and the error bars narrowed considerably. This suggests not only a recovery in diversity but also increasing uniformity in species composition across plots, which may indicate a stabilising effect following sustained management interventions.

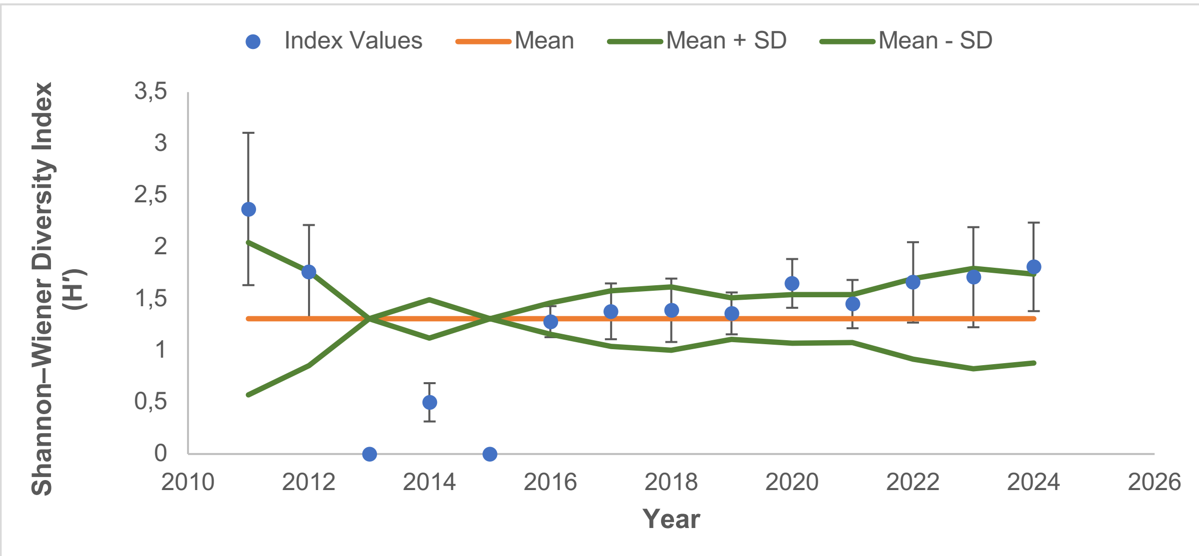



Figure 5.2 Shannon-Wiener Diversity Index for Urban areas with Mean ± SD (2011–2024)

The Margalef richness index in urban areas also fluctuated, ranging from near 0 to above 3. A polynomial regression model ($R^2 = 0.2907$) captured an initial decline in richness to a minimum around 2014–2015, followed by a steady increase from 2016 onward. As with the Shannon-Wiener Index, the relatively low $R^2$ value suggests that time alone only partially explains the observed patterns, with local disturbances or management practices likely contributing to richness trends.

The Sørensen Index, calculated relative to the 2011 baseline, showed fluctuations in species composition from 2012 to 2024 (Figure 5.3). The highest similarity was recorded in 2012 (±0.57), while the lowest occurred in 2013, dropping to near 0. In the years that followed, the index stabilised around a mean of 0.27. All annual values fell within the expected range (mean ± one standard deviation), indicating that observed changes in species presence were within normal variation. This suggests a relatively consistent assemblage of AIPs over time, with turnover occurring gradually.

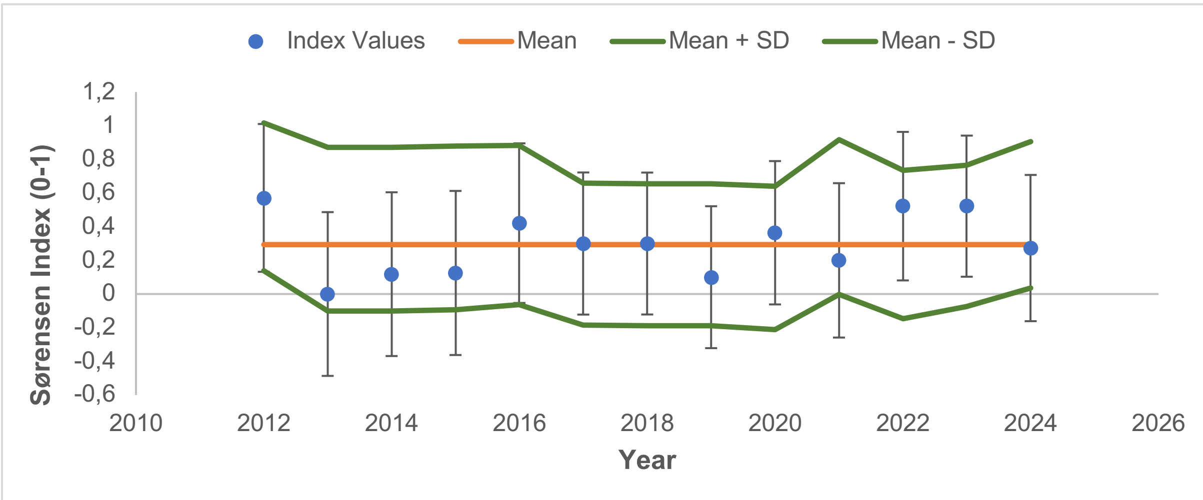



Figure 5.3 Temporal Trends of Urban Sørensen Indices with Mean ± SD (2012-2024 vs. 2011).

The Bray-Curtis Index for urban areas remained consistently high, indicating persistent compositional change relative to 2011 (Figure 5.4). The lowest dissimilarity occurred in 2012 (0.50), while the highest was in 2013, reaching 1.0. After this peak, the index stabilised around the mean of 0.76, with reduced variability observed in 2022–2024. The trend suggests a slight decline in compositional turnover over time, possibly reflecting more stable or recurring species structures in urban zones.

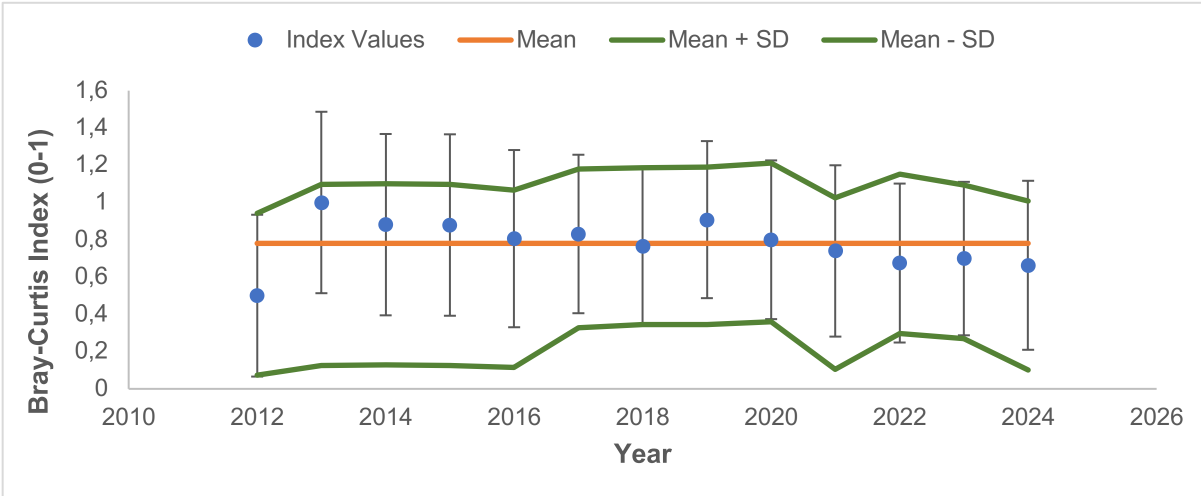



Figure 5.4 Temporal Trends of Urban Bray-Curtis Index with Mean ± SD (2012-2024 vs. 2011)

**Peri-Urban Areas**

In peri-urban areas, the Shannon-Wiener Index varied over time, peaking at 2.44 in 2018 and reaching its lowest value

s in 2014 and 2015. From 2016 onwards, the index stabilised around a mean of approximately 1.8. A polynomial regression model ($R^2$ = 0.155) showed a weak temporal trend, suggesting that the changes in species diversity were not strongly time-dependent but rather influenced by external or localised factors.

The second graph, which plotted annual diversity with mean ± standard deviation, showed considerable interannual variability during the early part of the study period (Figure 5.5). Wide error bars and year-on-year shifts in index values between 2011 and 2015 reflect inconsistent management outcomes and varying levels of disturbance or species colonisation. After 2016, the graph indicated a period of stabilisation. Narrower standard deviation ranges and smaller changes between consecutive years suggest that species diversity in peri-urban areas became more predictable and evenly distributed, possibly due to increased control effectiveness or ecological recovery.

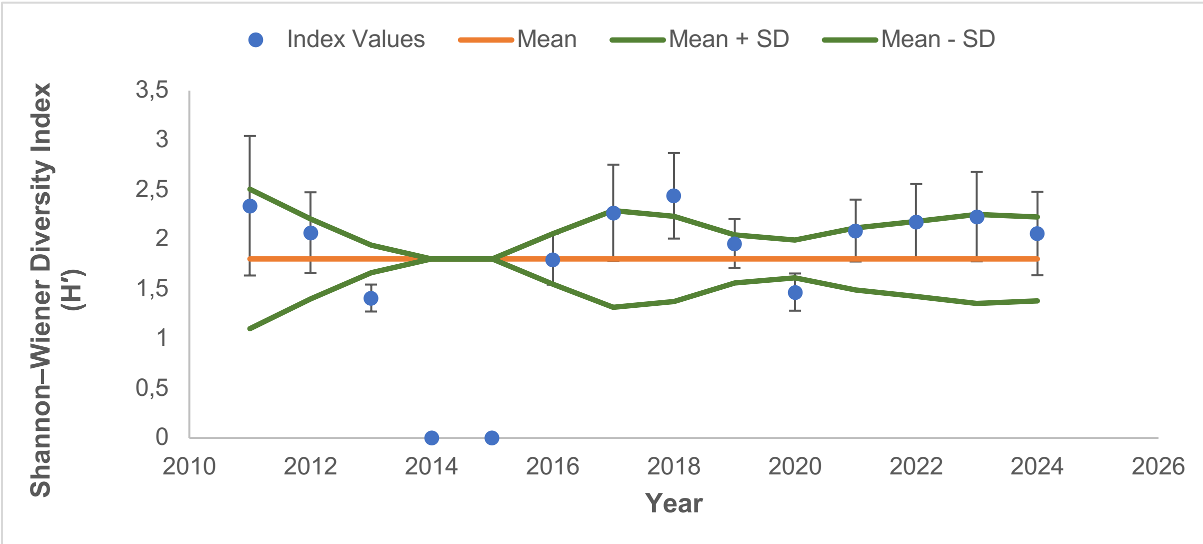



Figure 5.5 Shannon-Wiener Diversity Index for Peri-Urban areas with Mean ± SD (2011–2024)

The Margalef richness index also showed considerable variability, ranging from near 0 to above 3.0. The regression model ($R^2 = 0.1176$) suggests an initial decline in richness from 2011 to 2015, followed by a gradual recovery. However, the weak model fit again indicates that time was not the primary factor shaping AIP richness in peri-urban areas.

The Sørensen Index for peri-urban zones showed a notable decline from 2012 (±0.67) to a low of 0.00 in 2015. The index then gradually recovered and stabilised around 0.45 in later years. Deviations in 2015 and 2020, as well as elevated variability in 2012, 2015, 2016, and 2022, suggest inconsistent patterns of species presence across quadrants, likely linked to differences in disturbance, recolonisation, or management effectiveness (Figure 5.6).

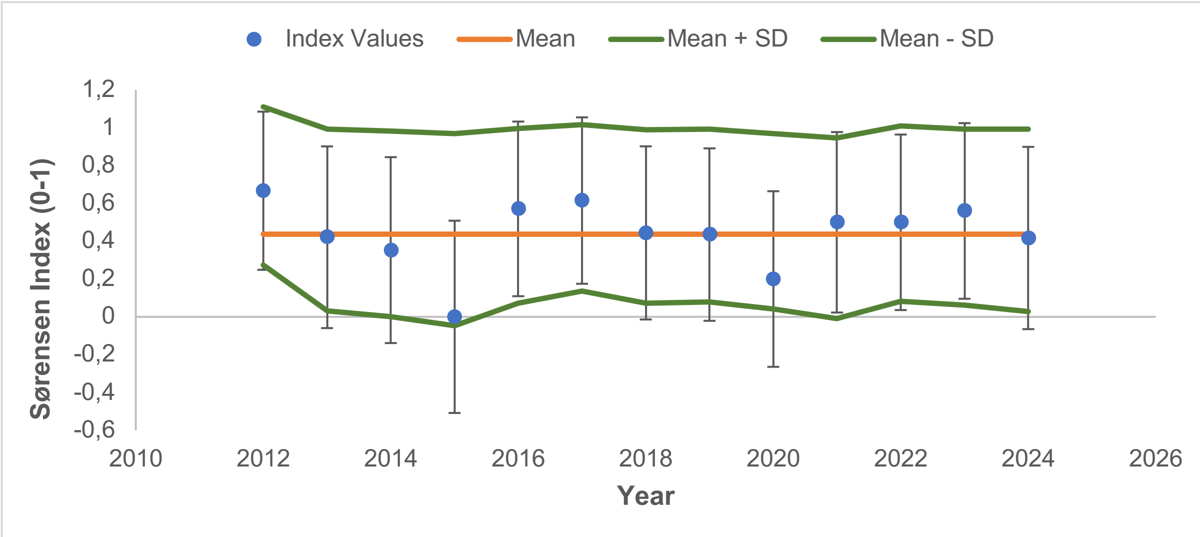



Figure 5.6 Temporal Trends of Peri-Urban Sørensen Index with Mean ± SD (2012-2024 vs. 2011)

The Bray-Curtis Index fluctuated moderately over time, peaking at 1.0 in 2015 and reaching its lowest value (±0.52) in 2012. After 2015, values declined and stabilised near the mean of 0.71. Error bars indicated high variability in 2013–2015, but in more recent years, such as 2018, 2020, and 2024, showed reduced spread, suggesting improved compositional stability across peri-urban zones (Figure 5.7).

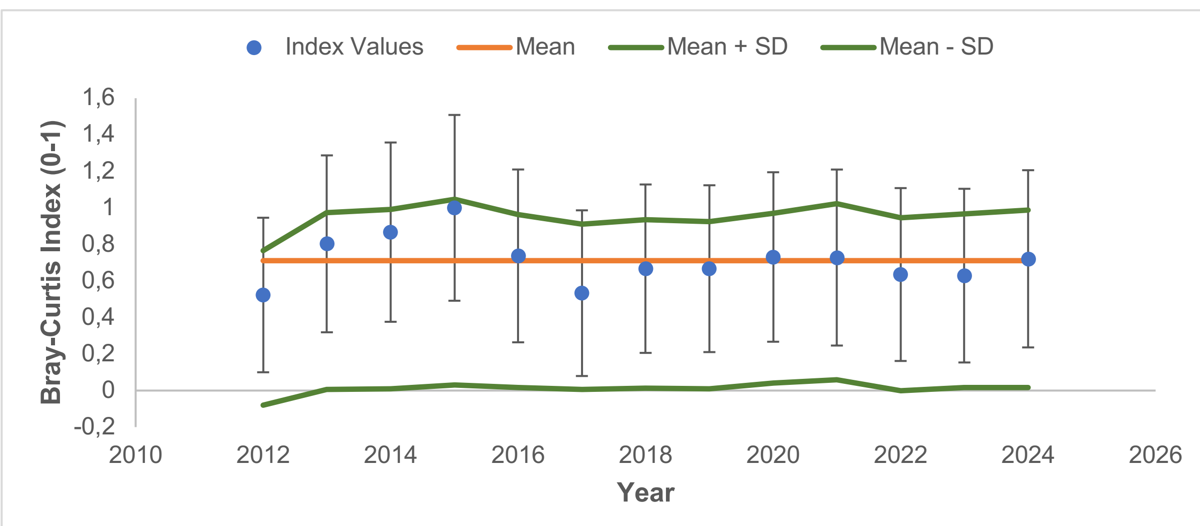



Figure 5.7 Temporal Trends of Peri-Urban Bray-Curtis Index with Mean ± SD (2012-2024 vs. 2011)

**Rural Areas**

In rural areas, the Shannon-Wiener Index fluctuated throughout the study period, with values exceeding 2.1 in 2011 and 2023 and dropping below 1.2 in 2020. The polynomial regression model ($R^2 = 0.1543$) indicated an initial decline followed by stabilisation and a slight upward trend in more recent years, although the model explained little of the overall variation.

Annual diversity values with mean ± standard deviation provided clearer insight into these dynamics (Figure 5.8). Early and mid-study years showed broad error bars and uneven trends, highlighting inconsistency in species diversity across rural plots. This likely reflects the complexity of rural environments, where variation in land use, disturbance, and edge effects can influence diversity at fine spatial scales. From 2016 onward, the graph revealed narrower error margins and more consistent yearly values, suggesting that diversity levels not only recovered but also became more evenly distributed across sampling sites, indicating an emerging ecological equilibrium or improved management coverage.

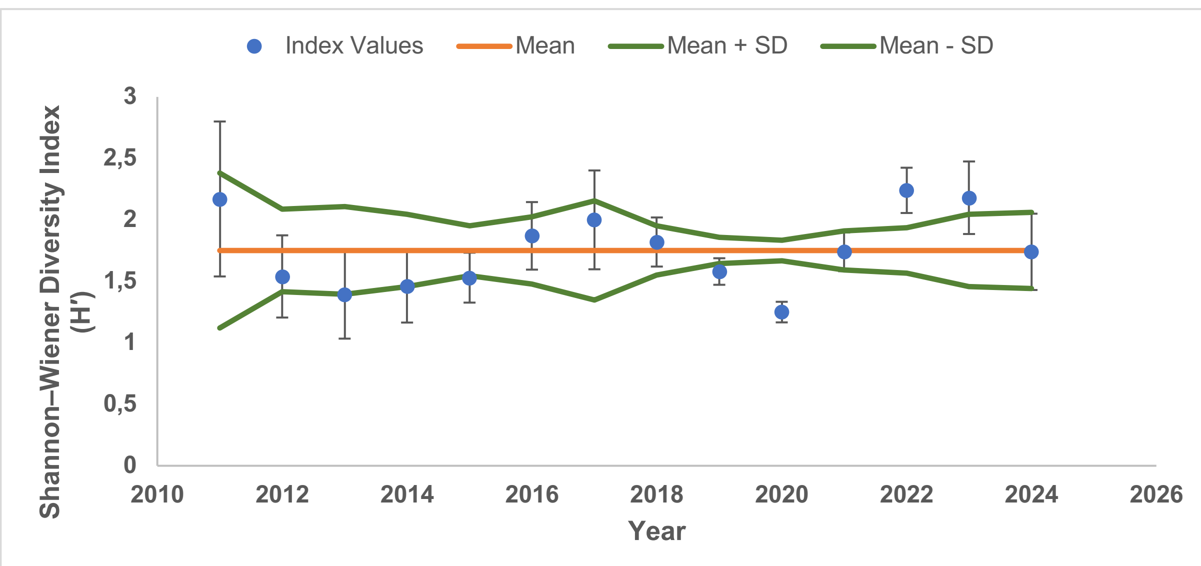



Figure 5.8 Shannon-Wiener Diversity Index for Rural areas with Mean ± SD (2011–2024)

The Margalef richness index ranged from just under 1.0 to over 2.5. The polynomial regression model ($R^2$ = 0.0943) suggested a shallow U-shaped trend, with a minor decrease in richness early in the period, followed by modest increases toward 2024. Again, the low $R^2$ indicates that richness variation was likely driven by localised environmental factors rather than time alone.

The Sørensen Index for rural areas was relatively stable over time, with values consistently around the mean of 0.51. The highest value (±0.70) occurred in 2012, and the lowest (±0.36) in 2021. Despite moderate year-to-year variation, all values fell within expected bounds, pointing to stable assemblages of AIPs in rural zones (Figure 5.9). However, a statistically significant regression ($b = -0.0207$, $R^2 = 0.511$, $p = 0.009$) confirmed a meaningful decline in similarity over time. A polynomial model ($R^2 = 0.647$) further improved the fit, suggesting a non-linear divergence in species presence relative to the 2011 baseline.

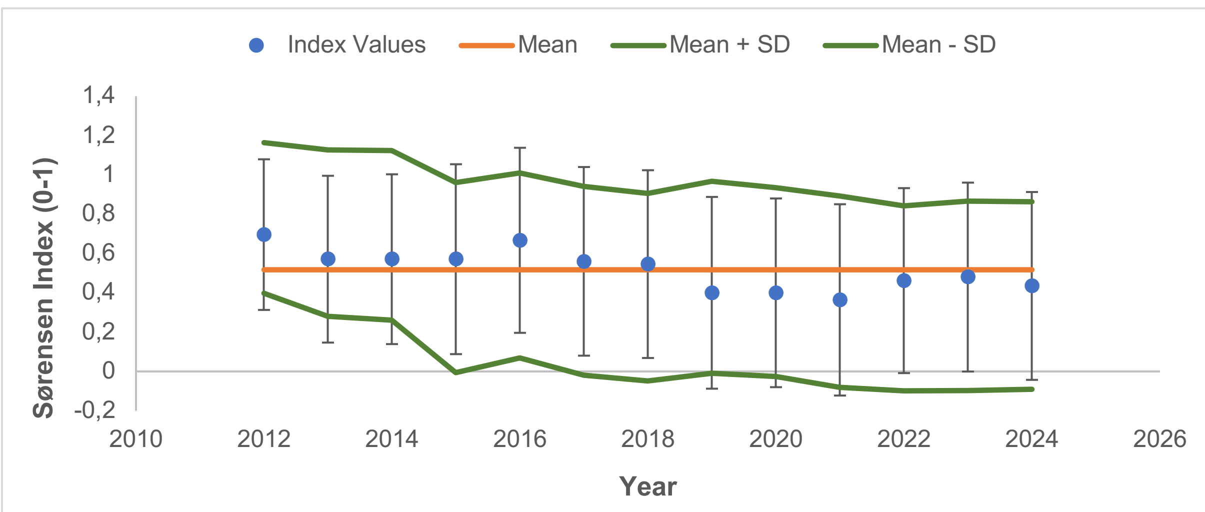



Figure 5.9 Temporal Trends of Rural Sørensen Index with Mean ± SD (2012-2024 vs. 2011)

The Bray-Curtis Index for rural zones showed a gradual upward trend, increasing from 0.42 in 2012 to over 0.87 in 2024. After 2015, values stabilised, and from 2021 onward, index values clustered closely around the mean of 0.75. This suggests increasing but eventually stabilising dissimilarity in species composition (Figure 5.10). The regression model confirmed a significant positive trend ($b = 0.0317$, $R^2 = 0.628$, $p = 0.0021$), with a polynomial fit ($R^2 = 0.829$) indicating a non-linear pattern of change, potentially driven by long-term ecological succession or changes in management intensity.

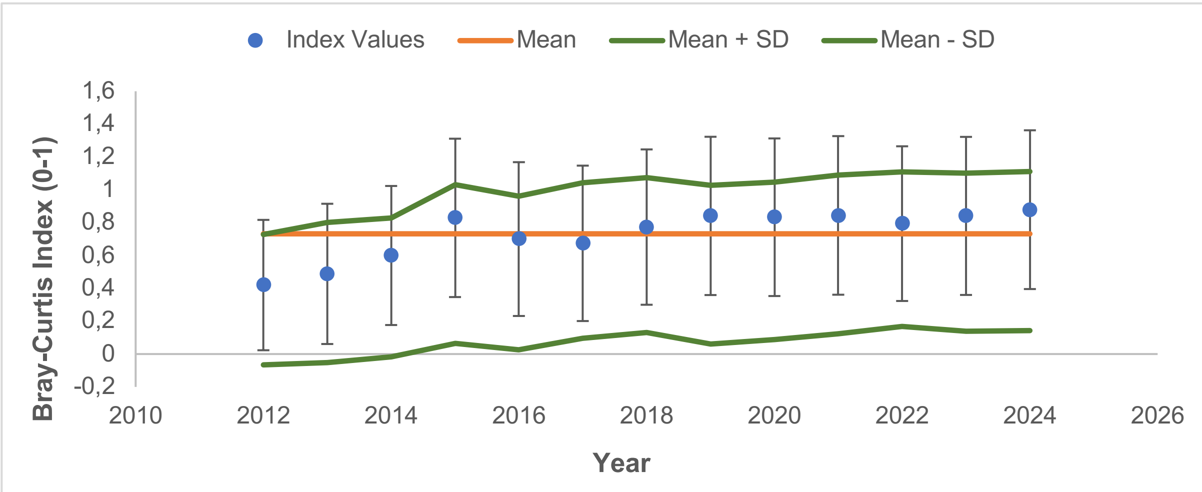



Figure 5.10 Temporal Trends of Rural Bray-Curtis Index with Mean ± SD (2012-2024 vs. 2011)

### 5.4.3 Identifying Problem Areas

Pielou’s Evenness Index for urban areas showed only a modest upward trend, with both the linear and polynomial models producing low $R^2$ values, indicating a poor fit. While the polynomial regression model performed slightly better than the linear model ($R^2 = 0.1234$), it still explained very little of the variation in evenness over time. Given the lack of statistical significance and the weak model fit, the observed trend in urban evenness is likely influenced by external or site-specific factors and does not contribute substantially to the study's objectives.

Pielou’s Evenness Index for peri-urban areas showed a weak upward trend, but regression results indicated no statistically significant relationship with time. Both the linear and polynomial models had low $R^2$ values, with the polynomial regression providing a slightly improved fit ($R^2 = 0.0772$). While the curved trendline suggests minor changes in evenness over time, the overall explanatory power remains low.

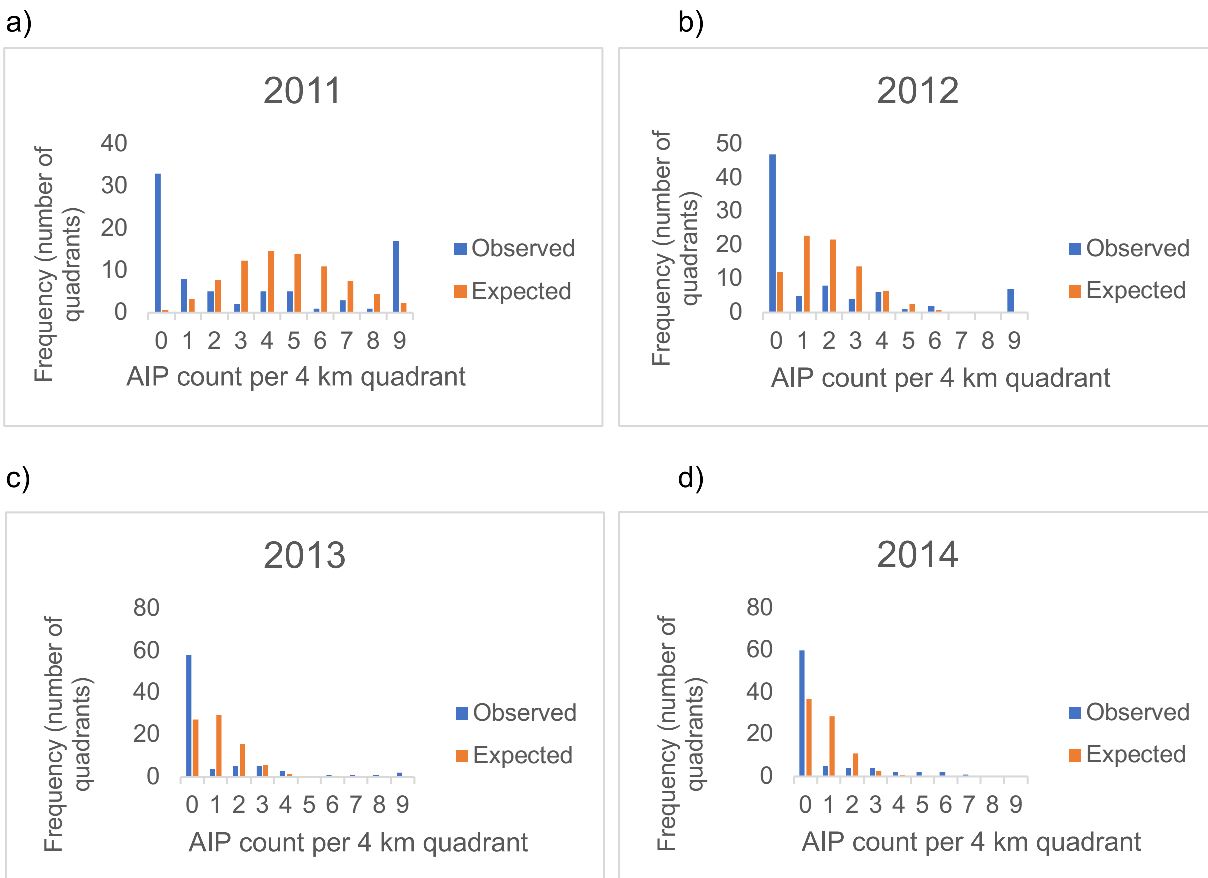
Pielou’s Evenness Index for rural areas showed a more noticeable upward trend than in urban and peri-urban zones. Although the regression results approached statistical significance at the 10% level, the relatively low $R^2$ values from both the linear and polynomial models ($R^2 = 0.248$ and $0.3317$, respectively) indicate that time alone does not explain much of the variation in evenness. The polynomial model suggested a modest improvement in capturing the trend, showing an initial decline followed by an increase in evenness. All regressions can be found under Chapter 5, Supplementary Material.

#### 5.4.3.1 Spatial distribution of AIP

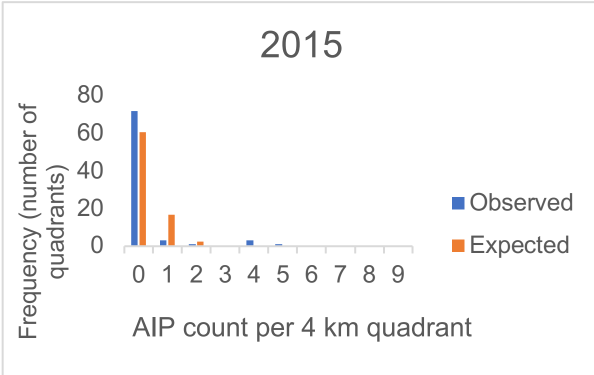
By examining the spatial distribution of AIPs across all zones, it is evident that the distributions across the various years deviate significantly from a random distribution (Poisson). The following histograms (Figure 5.11) illustrate the observed frequency distribution of AIP counts (blue) compared to Poisson’s expected distribution (orange) for each year. All the histograms exhibit a variance that is larger than the mean, indicating overdispersion.

If the calculated Chi-square value exceeded the critical value from statistical tables, the null hypothesis, that the observed distribution matched the expected distribution, was rejected. This result indicated a non-random distribution, suggesting aggregated hotspots of infestation requiring focused management.

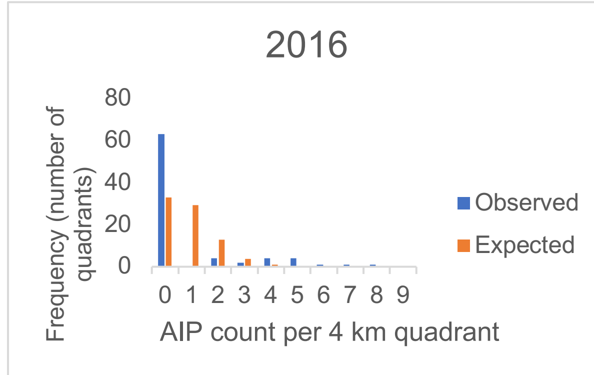
Additionally, the Chi-Square test results for all years exceed the critical value of 15.507, leading to the rejection of the null hypothesis of a random distribution.



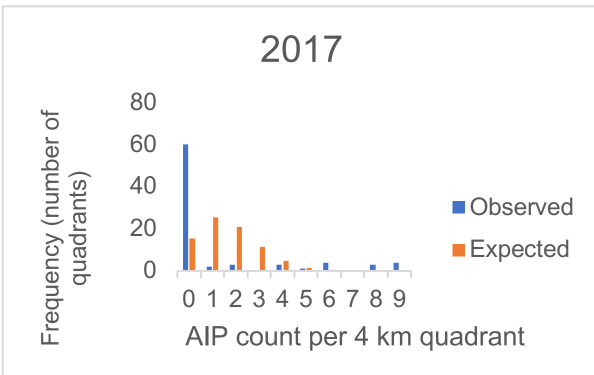
e)


2015
Frequency (number of quadrants)
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

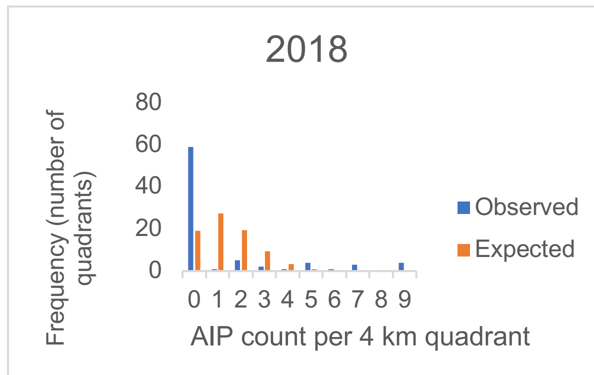
f)


2016
Frequency (number of quadrants)
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

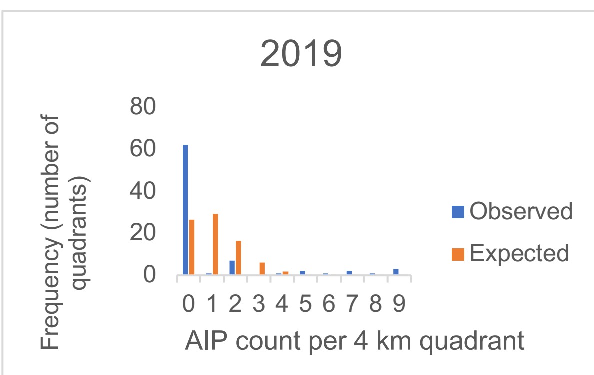
g)


2017
Frequency (number of quadrants)
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

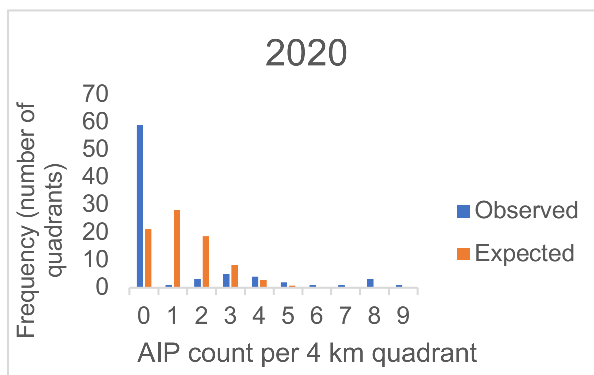
h)


2018
Frequency (number of quadrants)
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

i)


2019
Frequency (number of quadrants)
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

j)


2020
Frequency (number of quadrants)
70
60
50
40
30
20
10
0
0 1 2 3 4 5 6 7 8 9
AIP count per 4 km quadrant
Observed
Expected

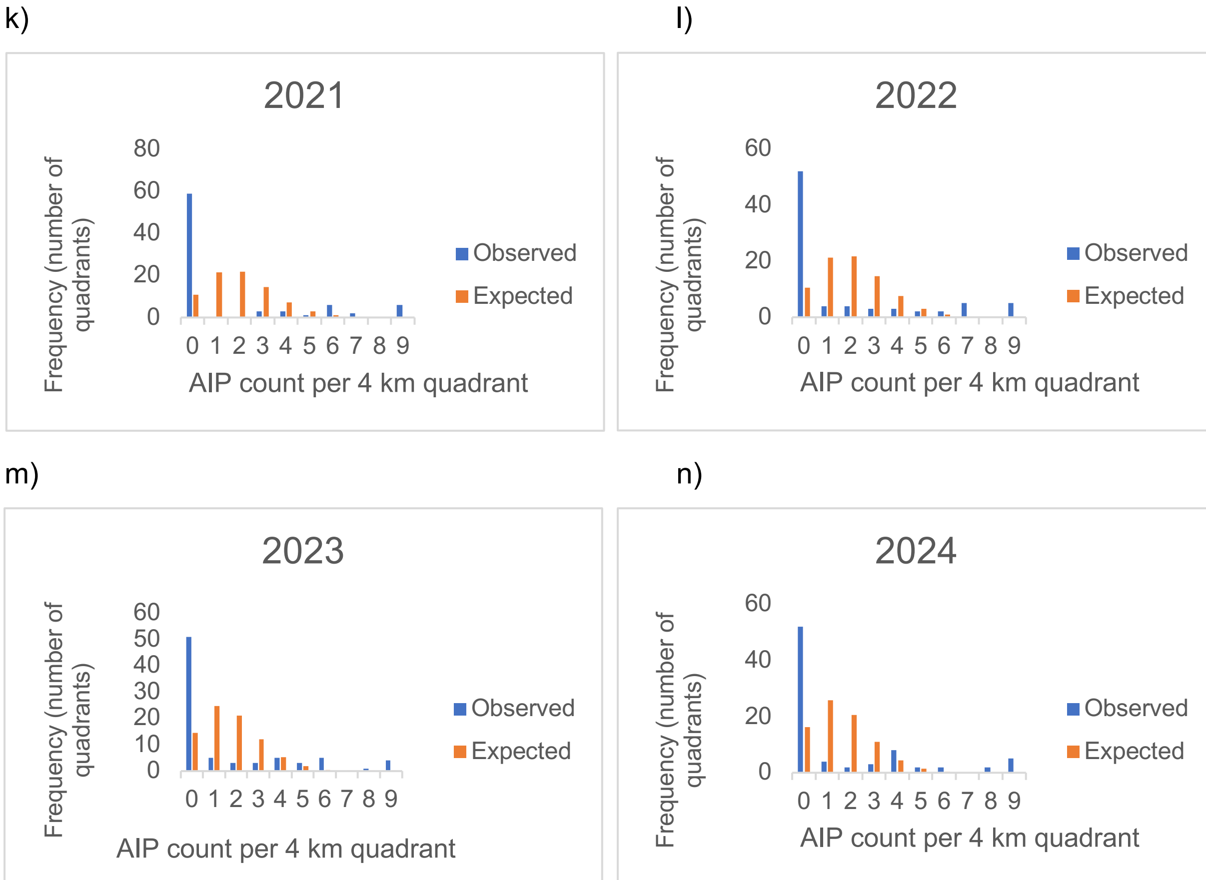



Figure 5.11 Expected vs Observed Frequency of AIPs per Quadrant from 2011 to 2024.

The number of quadrants exceeding the 90th percentile threshold for AIP counts (Table 5.1) varied across urban, peri-urban, and rural areas from 2011 to 2024. The 90th percentile represents the cut-off point above which the top 10% of quadrants, in terms of AIP abundance, are flagged as potential hotspots. This method allows for the consistent identification of areas with unusually high AIP densities each year. In the urban area, which consists of 14 quadrants, the number of flagged quadrants ranged from 0 (in 2016) to 3 (in 2013 and 2023), indicating occasional problem areas but generally low levels of concern relative to other zones.

The peri-urban area, with 26 quadrants, displayed more fluctuation. No quadrants were flagged in 2015, while up to 4 quadrants were flagged in multiple years (2017, 2018, 2019, 2021, and 2024), suggesting the presence of recurring high-density patches. In the rural area, which contains 41 quadrants, the number of flagged quadrants was consistently higher, ranging from none in 2023 to a peak of 7 in 2015. This pattern indicates that rural zones have experienced more persistent and widespread AIP invasions over the years.

The number of quadrants with Shannon indices exceeding the upper threshold varied across urban, peri-urban, and rural areas from 2011 to 2024. In urban areas, the flagged quadrants ranged from 0 (2013, 2015, and 2016) to 2 (2012, 2020, 2021, 2023, and 2024), indicating sporadic increases in problem areas. Peri-urban areas exhibited fluctuations, with flagged quadrants ranging from 0 in 2014 and 2015 to a peak of 5 in 2017. Rural areas consistently showed the highest numbers of flagged quadrants, peaking at 6 in 2013 and remaining elevated throughout the study period, with a minimum of 0 in 2022.

The number of quadrants with Margalef indices exceeding the upper threshold varied across the urban, peri-urban, and rural areas from 2011 to 2024. Urban areas consistently exhibited low numbers, with flagged quadrants ranging from 0 (2015) to 3 (2011). Peri-urban areas showed moderate variation, with flagged quadrants peaking at 4 in 2017 and remaining relatively stable in other years, fluctuating between 0 and 3. In rural areas, the flagged quadrants were higher, peaking at 6 in 2013 and showing a general decline over time, with fewer problem areas in later years, such as 1 in 2024.

The table summarises the flagged quadrants per year, highlighting the distribution of problem areas based on counts and diversity indices. Each year had between 8 and 11 quadrants flagged for having counts in the 90th percentile. Among these, some quadrants were flagged solely for high counts, while others were flagged for additional diversity indices such as Shannon and Margalef. Quadrants flagged for all indices consistently had high Pielou's Evenness Index values exceeding 0.9, indicating a more uniform distribution of species within these areas. The results show that problem quadrants varied yearly, with certain years like 2012 and 2014 showing more quadrants flagged for all indices, suggesting potential hotspots requiring further investigation. The dataset can be found under Chapter 5 Supplementary Material.

Table 5.1 Summary of the Flagged Quadrants.

| Year | Total Quadrants Flagged (90th Percentile Counts) | Urban | Peri-Urban | Rural | Quadrants Flagged for Counts Only | Quadrants Flagged for Counts and Shannon | Quadrants Flagged for All Indices (Counts, Shannon, Margalef) |
|---|---|---|---|---|---|---|---|
| 2011 | 8 | 1 | 2 | 5 | 6 | 1 | 1 |
| 2012 | 9 | 1 | 2 | 6 | 1 | 2 | 5 |
| 2013 | 8 | 2 | 1 | 5 | 3 | 3 | 2 |
| 2014 | 11 | 1 | 2 | 8 | 3 | 1 | 3 |
| 2015 | 8 | 1 | 0 | 7 | 6 | 0 | 2 |
| 2016 | 11 | 2 | 4 | 5 | 4 | 2 | 2 |
| 2017 | 11 | 2 | 5 | 4 | 2 | 2 | 4 |
| 2018 | 8 | 2 | 4 | 2 | 5 | 0 | 3 |
| 2019 | 8 | 2 | 4 | 2 | 4 | 0 | 4 |
| 2020 | 8 | 2 | 2 | 4 | 5 | 0 | 3 |
| 2021 | 8 | 1 | 4 | 3 | 5 | 0 | 3 |
| 2022 | 10 | 3 | 3 | 4 | 7 | 0 | 1 |
| 2023 | 10 | 4 | 5 | 1 | 5 | 1 | 2 |
| 2024 | 8 | 2 | 4 | 2 | 3 | 3 | 2 |

#### 5.4.3.2 Vegetation Quality Trends and Associations with Problem Areas

From 2011 to 2024, the number of quadrants classified as severely degraded (Status 1) remained relatively low but showed important spatial and temporal signals relevant to AIP problem areas. Status 1 ratings peaked in 2020 and 2021, with 12 and 11 quadrants, respectively, requiring urgent attention due to slippages, erosion, or poor vegetation structure (Table S5.6). These years also coincided with elevated AIP counts and high Shannon diversity values in multiple zones. Between 2016 and 2024, the number of Status 4 quadrants, which indicate construction activity or stable conditions, dropped to nearly zero, while the proportion of quadrants rated Status 2 or 3 increased. This trend suggests a broad shift away from highly stable vegetation conditions, with more areas showing signs of ongoing degradation or incomplete recovery. The consistent presence of degraded vegetation in the same quadrants flagged for AIP hotspots points to a feedback loop in which structural disturbance may facilitate alien plant persistence or reinvasion. As such, vegetation quality status, particularly Status 1, provides a useful indicator for identifying ecologically vulnerable areas that require combined intervention for erosion control and invasive species management.

From 2011 to 2024, Status 1 vegetation ratings, which indicate severely degraded conditions such as erosion, slippages, or structural collapse, were recorded across all three zones along the Bakwena N4 Toll Route. Rural areas showed the highest and most consistent occurrence of Status 1 quadrants, with early records in 2011 and 2013 (2 quadrants each), followed by a marked increase from 2018 onward. The peak occurred in 2020 and 2021, when 10 rural quadrants were flagged each year, suggesting a period of widespread ecological instability (Table S5.6). This was followed by a gradual decline to 3 quadrants in 2024. These degraded rural sites strongly overlapped with previously identified AIP hotspots, supporting the relationship between poor vegetation conditions and alien plant persistence.

Peri-urban areas also displayed recurring Status 1 ratings, particularly during 2016, 2017, and in later years from 2019 to 2024 (Table S5.6). While less frequent than in rural zones, their presence in up to 3 quadrants (e.g. 2017) indicates persistent structural vulnerability likely linked to edge effects or repeated disturbance in transitional areas. In urban zones, Status 1 ratings were rare but not absent, with one quadrant affected in both 2020 and 2022 (Table S5.6). These instances may reflect isolated erosion or drainage issues associated with infrastructure activities.

Chi-square tests were conducted annually from 2011 to 2024 to assess whether vegetation quality (“Bad” vs “Good”) was associated with AIP hotspot status (flagged vs not flagged) (Table S5.7). The Chi-square values across the 14 years varied considerably, reflecting differences in the strength of association between vegetation quality and AIP hotspot status. In three specific years (2017 ($p = 0.0231$), 2022 ($p = 0.0025$), and 2024 ($p = 0.006$)), the p-values were well below the standard 0.05 threshold, indicating a statistically significant relationship between degraded vegetation and the presence of AIP hotspots. These years returned the strongest evidence that hotspot distribution was not independent of vegetation quality, with flagged quadrants disproportionately occurring in areas classified as “Bad” vegetation.

In addition to these three years, 2014 ($p \approx 0.056$) and 2011 and 2012 (both $p \approx 0.085$) produced Chi-square values close to the 0.05 threshold. While not strictly significant at the 95% confidence level, these values indicate marginal significance and suggest a possible underlying relationship between vegetation degradation and AIP hotspots in those years. These borderline results may reflect weaker or more spatially dispersed associations, possibly moderated by other site-specific factors.

The remaining years yielded higher p-values and low Chi-square values, indicating no significant association between vegetation quality and hotspot status during those periods. In those years, flagged quadrants were more evenly distributed between “Bad” and “Good” vegetation categories, and no meaningful pattern emerged. Overall, the results highlight temporal variation in the strength of association between vegetation condition and AIP presence, with clear significance in some years and weaker, less consistent patterns in others.

## 5.5 Discussion

### 5.5.1 Historical Trends in AIP Abundance

The trends observed in AIP abundance across the study area speak directly to the objective of evaluating long-term changes in invasive plant populations. In urban areas, AIP counts per kilometre showed no consistent directional trend from 2011 to 2024. Although a dip was recorded in 2013, followed by a gradual rise through to 2023, regression analyses produced very low explanatory power. The lack of statistical significance suggests that invasion trajectories in urban zones are influenced more by short-term, localised disturbances, reinvasion pressure, or variability in maintenance intensity than by time. This finding is consistent with Foxcroft *et al.* (2004), who demonstrated that persistent propagule pressure from human-dominated sites sustains long-term invasions in Kruger National Park.

In peri-urban areas, AIP counts showed similar instability with no clear directional change. Variability across the time series reflects the transitional character of these zones, where mixed land uses and edge effects create complex invasion dynamics. The low coefficients of determination confirm that these invasions remain unpredictable, echoing Alston and Richardson (2006), who showed that disturbance and proximity to propagule sources drive unstable invasion patterns along Cape Town’s urban–wildland gradients. An unexpected outcome here was the apparent stabilisation observed after 2015, which may suggest temporary limits to invasion caused by ecological saturation or local management interventions.

In contrast, rural areas showed a more structured trajectory, with densities declining steadily from 2011 to 2018 before levelling off and increasing slightly after 2020. The stronger polynomial regression fit suggests that these landscapes exhibit more predictable patterns of change. This supports Kalwij *et al.* (2008), who found that invasion patterns along road verges in less-disturbed areas can be more spatially structured and predictable. However, the plateau and slight rebound after 2020 were unexpected, as continued decline had been anticipated. This suggests possible reinvasion from untreated areas or reduced management intensity in recent years, aligning with concerns raised by van Wilgen *et al.* (2012) that large-scale control programmes in South Africa face sustainability challenges when follow-up control is reduced.

Overall, abundance trajectories demonstrate that invasion dynamics are not governed by simple linear decline or recovery. Instead, outcomes vary according to land use: urban and peri-urban zones remain unstable, while rural areas initially decline before showing signs of resurgence.

### 5.5.2 Historical Trends in AIP Diversity and Composition

Patterns of diversity and species composition across the Bakwena N4 Toll Route address the objective of applying biodiversity indices to monitor invasive assemblages. In urban areas, diversity declined sharply in the early years before gradually recovering. The Shannon–Wiener and Margalef indices suggest that communities are reassembling, but not with the same species. This aligns with Nelufule *et al.* (2024), who documented continual species turnover in urban environments and highlighted how assemblages shift under urban pressures. Composition indices support this interpretation, with Bray–Curtis and Sørensen values showing ongoing turnover, pointing to the gradual establishment of new assemblages. An unexpected finding was the apparent stability in richness despite these shifts, suggesting that species replacement is occurring rather than net species loss.

In peri-urban areas, consistently higher diversity and fluctuating richness indicate a balance between species persistence and new invasions. The lack of great directional change in either diversity or composition reflects the transitional nature of these landscapes, which often act as reservoirs of both native and invasive species. Similar patterns have been observed in Cape Town by Alston & Richardson (2006), who showed that disturbance and proximity to propagule sources drive unstable yet species-rich assemblages along urban–wildland gradients. Recurring dips in Sørensen similarity confirm that communities remain in flux despite stable diversity, echoing Rouget *et al.* (2004), who highlighted how fragmented landscapes can support persistent but shifting invasion hotspots. An unexpected outcome was that diversity did not decline substantially over time, suggesting coexistence between native and invasive species rather than complete displacement.

In rural areas, diversity patterns indicated early instability followed by convergence in later years. Narrower error margins in recent years suggested stabilisation, yet composition indices revealed continued divergence from the 2011 baseline. Increasing Bray–Curtis dissimilarity and declining Sørensen similarity highlight ongoing turnover, reflecting a broader trend described by Richardson *et al.* (2007) in riparian systems, where reductions in density do not prevent shifts toward new dominant invaders. The emergence of novel invasive assemblages in rural areas was unexpected, as suppression of original dominants was anticipated to stabilise communities rather than open niches for secondary invaders, a phenomenon also noted by Gaertner *et al.* (2012) in restoration contexts.

These findings confirm that focusing solely on diversity or abundance can be misleading, as stable metrics may conceal substantial compositional change. The use of multiple indices was therefore essential in capturing turnover and shifts in community structure.

### 5.5.3 Identifying Problem Areas

The identification of hotspots and problem areas relates to the objective of pinpointing priority areas for management. The analysis confirmed that AIP populations are not randomly distributed but cluster in distinct ecological and infrastructural hotspots. Flagged quadrants and diversity indices provided a clearer picture of where management efforts should be prioritised. Variations across years and land-use zones suggest that invasion dynamics are influenced by ecological conditions, disturbance history, and management intensity. The statistical rejection of the null hypothesis in several Chi-square tests, together with evidence of overdispersion, supports the presence of clustered invasion behaviour, reinforcing the importance of spatially targeted control. This finding is consistent with Rouget *et al.* (2004), who mapped clustered hotspots of invasion across South Africa.

In urban areas, flagged quadrants were relatively few and inconsistent, reflecting sporadic cycles of reinvasion. Short-lived increases in AIP counts were followed by declines, likely due to localised interventions. However, the absence of a clear temporal trend in diversity or evenness indicates that these improvements may be temporary. The recurrence of problem areas in later years points to reinvasion driven by edge effects and small-scale disturbances. This pattern is consistent with Foxcroft *et al.* (2004), who showed that propagule pressure sustains reinvasion despite repeated control interventions.

Peri-urban areas displayed more erratic behaviour, with repeated peaks in flagged quadrants throughout the study period. These fluctuations reflect the transitional nature of peri-urban landscapes, where fragmented habitats and intermittent disturbance create favourable conditions for invasive plants' establishment and spread. High richness combined with moderate diversity suggests that invasives coexist with native flora, complicating ecological interpretation. Similar patterns have been noted in South African road-verge studies by Kalwij *et al.* (2008), who found that mixed land uses sustain unstable assemblages. An unexpected outcome here was the persistence of moderate evenness, indicating that invasives are spreading without yet fully dominating these areas.

Rural areas showed a clearer trajectory, with high invasion levels early in the study followed by a gradual decline in flagged quadrants. This suggests that management in these zones has been relatively effective. However, persistent hotspots remained in certain sites, functioning as refuges or reinvasion points. This finding mirrors Gelbard & Belnap (2003) and Boon *et al.* (2023), who reported that refuges and inadequate follow-up often allow invasives to persist despite control efforts. The rising dissimilarity in species composition and increasing evenness in rural areas were unexpected, suggesting either ecological recovery or redistribution among invasive species.

Chi-square analyses reinforced these spatial findings by showing significant associations between degraded vegetation and hotspot presence in specific years. In 2017, 2022, and 2024, p-values fell well below the 0.05 threshold, indicating strong links between poor vegetation quality and AIP clustering. Other years, such as 2011, 2012, and 2014, showed borderline significance, suggesting weaker but still relevant relationships. These results align with van Wilgen *et al.* (2012), who emphasised that disturbance and management gaps strongly influence invasion clustering in South Africa.

Taken together, the findings show that invasion dynamics along the N4 are shaped by land use, vegetation condition, and disturbance history. The results highlight both expected and unexpected outcomes. The instability of urban and peri-urban invasions is consistent with earlier South African studies, which found that frequent disturbance and high propagule pressure create fluctuating invasion patterns. In contrast, rural sections of the corridor showed clearer management responses, with declining invasion levels over much of the study period, although persistent hotspots remained in certain locations. These patterns suggest that while management efforts can reduce overall invasion pressure, localised ecological conditions and reinvasion sources continue to influence AIP persistence. This underscores the need for adaptive management strategies that target persistent hotspots while also accounting for reinvasion risks from surrounding landscapes.

## 5.6 Conclusion

This study assessed long-term trends in alien invasive plant (AIP) populations along the Bakwena N4 Toll Route using data collected between 2011 and 2024. By analysing changes in AIP abundance, diversity, and spatial distribution across urban, peri-urban, and rural zones, the study evaluated the effectiveness of the existing eradication programme and identified patterns in invasion dynamics over time. The integration of flagged quadrants, biodiversity indices, vegetation condition assessments, and statistical testing provided a comprehensive framework for analysing how AIP populations respond to management interventions and environmental conditions.

The results indicate that while AIP densities have declined in some areas, particularly in rural zones, progress has been uneven across the corridor. Rural sections showed the most consistent reductions in flagged quadrants and improvements in species evenness, suggesting relatively effective control. In contrast, peri-urban areas exhibited greater instability, with intermittent peaks in flagged quadrants and sustained species richness, indicating ongoing reinvasion. Urban zones showed weaker temporal patterns and more sporadic problem areas, reflecting the influence of continuous disturbance and complex land-use pressures.

Spatial analysis demonstrated that AIP hotspots were not randomly distributed but clustered in specific quadrants that were repeatedly flagged across multiple years. These areas often exhibited high diversity and evenness values, indicating persistent invasion pressure. Statistical testing further showed that degraded vegetation conditions were significantly associated with AIP presence in several years, highlighting the role of ecological conditions in shaping invasion dynamics. Overall, the results show that while management interventions have reduced AIP abundance in certain sections of the route, invasion dynamics vary considerably across zones, emphasising the importance of continued monitoring and spatially targeted management.

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# CHAPTER 6

# SPATIAL DISTRIBUTION OF ALIEN INVASIVE PLANTS ACROSS VEGETATION TYPES, LAND-USE ZONES, AND SITE CONDITIONS

## 6.1 Introduction

The proliferation of alien invasive plants (AIPs) presents a significant ecological challenge, particularly in environments where human activity alters natural landscapes. Transportation corridors, including highways and road reserves, act as important conduits for the rapid spread of invasive species (Forman & Alexander, 1998). While road reserves are subject to regular intervention to control AIPs, adjacent natural areas often remain unmanaged (Foxcroft *et al.*, 2010). Despite these efforts, AIPs continue to persist within road reserves, raising the question of whether reinvasion from surrounding natural ecosystems undermines eradication measures. Understanding how AIPs are distributed between these land-use categories is therefore critical for developing effective, landscape-scale management strategies (Gaertner *et al.*, 2012).

Vegetation structure and land-use context play a central role in shaping patterns of invasion. This includes whether road reserves occur in urban, peri-urban, or rural landscapes, as well as their proximity to natural or disturbed vegetation. Along the Bakwena N4 Toll Route, the more open, grass-dominated Marikana Thornveld and the other, generally more woody or structurally complex vegetation types may differ in their susceptibility to invasion because of variation in vegetation structure, disturbance response, and resource availability (Gaertner *et al.*, 2012; Richardson *et al.*, 2000). The more open vegetation settings may be more exposed to colonisation by fast-establishing herbaceous invaders, while the more wooded vegetation types may experience invasion in disturbed gaps, edges, and roadside clearings (Foxcroft *et al.*, 2010; Milton & Dean, 1998). These risks may be amplified in rural and peri-urban settings, where road reserves often adjoin fragmented natural vegetation, transformed land, or disturbed edges (Gelbard & Belnap, 2003; Hansen & Clevenger, 2005; Meunier & Lavoie, 2012). The interaction between vegetation structure, land-use context, and AIP spread is therefore important for assessing reinvasion risk and guiding targeted management (Gaertner *et al.*, 2009).

A further consideration is the potential correlation between AIP densities in natural areas and those in adjacent road reserves. A strong association would suggest that reinvasion from unmanaged natural areas limits the long-term success of road reserve management (Foxcroft *et al.*, 2004). In contrast, if road reserves maintain lower AIP densities despite proximity to infested natural areas, this would indicate that existing control measures are effective in preventing establishment (van Wilgen *et al.*, 2012). Assessing this relationship is critical for evaluating the sustainability of current approaches.

Beyond invasive species dynamics, the broader composition and diversity of native vegetation within road reserves and natural areas are important indicators of ecosystem integrity. Native species richness and functional composition may vary with land use, disturbance, and soil conditions, providing insights into resilience to invasion and ecological connectivity (Pauchard & Alaback, 2004). Differences between road reserves and natural areas can highlight how invasives interact with native communities and influence overall stability (D'Antonio & Vitousek, 1992; Davies & Sheley, 2007; Milton *et al.*, 2007).

Accordingly, this chapter addresses Objective 2 by comparing invasion patterns between the more open, grass-dominated Marikana Thornveld and the other, generally more woody or structurally complex vegetation types along the route, as well as across urban, peri-urban, and rural land-use zones, and Objective 3 by testing whether vegetation quality and land-use position (road reserve vs. natural area) influence AIP presence. It also examines whether soil type contributes to differences in invasion patterns and compares native species richness and composition between land-use categories. Together, these analyses provide insight into the spatial drivers of AIP distribution and inform more effective, landscape-scale management strategies along the Bakwena N4 Toll Route.

## 6.2 Methods

This chapter investigates the distribution of AIPs across different vegetation settings and land-use categories along the Bakwena N4 Toll Route. The primary objectives were to test whether AIP densities differ between natural areas and adjacent road reserves in the open, grass-dominated Marikana Thornveld and the other, generally more woody or structurally complex vegetation types along the route; to assess whether vegetation setting and land-use position (natural versus road reserve) are significantly associated with AIP presence; to evaluate whether soil type influences AIP occurrence; and to compare indigenous species richness and composition between land-use types.

A stratified plot-based sampling approach was used to collect data on the presence and abundance of AIPs across contrasting vegetation settings and land-use categories along the route (Elzinga *et al.*, 1998; McCune & Grace, 2002). The study compared the open, grass-dominated Marikana Thornveld with the generally more woody or structurally complex vegetation types, with specific attention to AIP distribution in natural areas and adjacent road reserves. A total of 38 plots were surveyed, of which 20 were located within the broader woody vegetation group and 18 within Marikana Thornveld. The initial intention was to distribute 40 plots evenly, but mining activities in parts of the Marikana Thornveld prevented the placement of two paired plots. Plots were established in pairs to enable comparison between road reserve vegetation and adjacent natural vegetation while accounting for edge effects associated with the road corridor. Within each pair, one plot was located inside the road reserve and the second was placed in adjacent natural vegetation beyond the road reserve boundary. Plot pairs were distributed at regular intervals within each vegetation group to achieve a spatially balanced sampling design along the route.

Due to differences in vegetation structure, plots located in the more woody and structurally complex vegetation types were designed to cover 100 m² (20 m × 5 m) to accommodate denser tree and shrub cover, whereas plots in the more open, grass-dominated Marikana Thornveld were smaller, covering 25 m² (5 m × 5 m), as

the vegetation was dominated by grasses and smaller plants that required less space for adequate documentation (Mueller-Dombois & Ellenberg, 1974).
In the more woody and structurally complex natural vegetation types, where denser vegetation could obscure smaller invasive species, additional data were collected by means of five randomly placed 1 m² subplots within each 100 m² plot. This allowed for a more detailed assessment of smaller AIPs that might otherwise have been overlooked in the broader survey (Barbour *et al.*, 1999). By contrast, in road reserve plots and in the more open, grass-dominated Marikana Thornveld plots, where vegetation was less dense and visibility was generally higher, AIPs were recorded directly from the full plot without the need for subplots.

Data collection in each plot involved recording the name of every plant species present, noting their abundance, and documenting additional ecological characteristics such as soil type, plant height, plant growth form (tree, shrub, or grass), aspect, and slope (Kent, 2011). Species were identified using a combination of field guides (Oudtshoorn, 2012; van Wyk & van Wyk, 1997; van Wyk & van Wyk, 2014) and expert consultation. In cases where species could not be conclusively identified, an expert was consulted. Each plant species encountered in a plot was recorded, regardless of whether indigenous or alien. To facilitate direct comparisons, the same data recording criteria were applied to both alien and indigenous species, with an additional column used to indicate whether a species was an AIP (Pyšek *et al.*, 2004).

The cover of all plants was estimated using the Braun-Blanquet cover scale, which provides a reliable measure of dominance within a given plot (Braun-Blanquet, 1932). This scale was also applied to estimate vegetation cover for other plant groups, allowing for comparisons of relative abundance between different species and across land-use types.

Data collection took place during the rainy season (March 2024) to maximise the likelihood of recording all species present in each plot. Conducting surveys in this season ensured that annuals, forbs, and species with ephemeral growth cycles were included in the dataset, thereby providing a more comprehensive representation of biodiversity (Gibson, 2015). Seasonal variation was not directly analysed in this chapter, but by standardising data collection to the peak growing season, the study minimised the potential for underestimating species richness due to temporary dormancy (Magurran, 2004).

To estimate plant density within the study plots, Braun-Blanquet cover-abundance values were transformed into numerical plant counts through a structured methodology incorporating plant growth form and height. This transformation follows established ecological principles for converting categorical cover data into density estimates (van der Maarel, 1979, 2005; Dengler *et al.*, 2011). Given that the Braun-Blanquet scale provides cover-abundance categories rather than absolute plant numbers, the approach involved using cover class midpoints and estimating the average ground area occupied by individual plants. This allowed for estimation of plant density per 100 m², ensuring consistency across species with varying growth habits (Wikum & Shanholtzer, 1978).

The first step in the process was classifying each recorded plant into a refined functional type based on its growth form (grass, shrub, tree, or succulent) and measured height. This classification was necessary to distinguish between low-growing vegetation, such as grasses and small succulents, and larger plants, such as shrubs and trees occupying more ground area per individual (Scholes & Walker, 2009). Height thresholds were determined based on plant structural characteristics (Westhoff & van der Maarel, 1973). Grasses were divided into short grasses (≤0.3 m) and tall grasses (>0.3 m), while succulents were categorised as small (≤0.3 m) or large (>0.3 m). Similarly, shrubs were classified into small shrubs (≤2.0 m) and large shrubs (>2.0 m), while trees were divided into small trees (≤7.0 m) and large trees (>7.0 m). This categorisation ensured that functionally similar plants were grouped for density estimation while allowing flexibility for species that exceeded predefined height ranges (Dengler *et al.*, 2011).

Once plants were assigned to their respective categories, an estimated ground area per individual was determined based on literature values for typical canopy spread in Savanna vegetation types (Scholes & Walker, 2009; White, 1983). These values were critical for estimating plant density from cover data, as they accounted for the spatial footprint of each plant type (Chytrý & Tichý, 2003). The estimated ground areas per individual were as follows: short grasses were assigned 0.02 m² per plant, tall grasses 0.15 m², small succulents 0.03 m², large succulents 0.3 m², small shrubs 0.5 m², large shrubs 2.0 m², small trees 10.0 m², and large trees 40.0 m². By incorporating both plant height and functional type, this approach ensured that plant counts reflected realistic spacing and growth dynamics (Maarel, 2005).

The next step involved transforming Braun-Blanquet abundance scores into per cent cover midpoints, following the methodology recommended by van der Maarel (2005). Since Braun-Blanquet classes represent cover ranges rather than exact percentages, midpoints were assigned to each category (Dengler *et al.*, 2011). The midpoints were defined as follows: cover very small (<5%) was assigned 2.5%, plentiful but small cover (<5%) was assigned 2.5%, not many individuals (1-10) was assigned 5%, covering 5-25% was assigned 15%, covering 25-50% was assigned 37.5%, covering 50-75% was assigned 62.5%, and covering 75%+ was assigned 87.5% (Westhoff & van der Maarel, 1973). These midpoint values provided a standardised basis for converting qualitative Braun-Blanquet data into a continuous numerical scale (Dengler *et al.*, 2011). To estimate the number of plants per 100m², a numerical transformation was applied using the formula:

$$Estimated\ Plant\ Cover = \left(\frac{Cover\ Midpoint}{100}\right) \times \frac{100}{Ground\ Area\ per\ Plant\ (m^2)} \quad \text{Equation 6.9}$$

The cover midpoint (%) was the assigned Braun-Blanquet midpoint, and the ground area per plant (m²) was the estimated space occupied by a single individual of that plant type (van der Maarel, 2005). This calculation effectively distributed the total estimated cover over the plot area, adjusting for species-specific growth patterns (Dengler *et al.*, 2011; Wikum & Shanholtzer, 1978). By incorporating height-based plant categorisation and species-specific ground area estimates, this approach ensured realistic density estimates across structurally diverse plant communities (Chytrý & Tichý, 2003).

Since AIPs in the denser woody Savanna vegetation type natural areas were recorded within subplots, an extrapolation process was required to estimate densities at the full 100m² scale for meaningful comparisons. This was done by calculating the mean AIP count per subplot using the following formula:

$$\text{Mean Density} = \frac{\sum Xi}{n} \qquad \text{Equation 6.10}$$

$Xi$ represents the number of AIPs recorded in each subplot, and $n$ is the total number of subplots (five per plot). The estimated AIP density for the entire 100m² plot was then determined by multiplying the mean subplot density by 100:

$$\text{Extrapolated Density} = \text{Mean Density} \times 100 \qquad \text{Equation 6.11}$$

Testing for differences in AIP density between road reserves and natural vegetation The above method ensured that AIP densities in natural areas within the denser woody Savanna vegetation types were standardised to match those recorded in road reserves, which did not require extrapolation because AIPs were recorded directly across the entire plot (Barbour *et al.*, 1999).

To account for variability in AIP densities within subplots, additional statistical measures were applied. Variance was calculated to determine the spread of values using the formula:

$$\text{Variance} = \frac{\sum(Xi - \bar{X})^2}{n-1} \qquad \text{Equation 6.12}$$

where $\bar{X}$ represents the mean density. Standard deviation was then derived from the variance to measure the degree of variation in AIP counts across subplots:

$$\text{Standard Deviation} = \sqrt{Variance} \qquad \text{Equation 6.13}$$

In order to assess the precision of the estimated AIP densities, standard error was calculated using the formula:

$$\text{Standard Error} = \frac{Standard\ Deviation}{\sqrt{n}} \qquad \text{Equation 6.14}$$

Confidence intervals were determined to provide a range within which the true AIP density was likely to fall. Using a 95% confidence level, the confidence interval was calculated as:

$$\text{CI} = \bar{X} \pm (t \times Standard\ Error) \qquad \text{Equation 6.15}$$

With t representing the critical value from the t-distribution table. These calculations helped assess the reliability of the estimated AIP densities and allowed for more accurate comparisons between different land-use types and vegetation categories (McCune & Grace, 2002; Zar, 1999).

Before conducting formal statistical tests, data normality was assessed to determine whether parametric or non-parametric methods were appropriate. Normality was examined using histograms to visualise the distribution of AIP densities, along with descriptive statistics such as skewness values. The analysis revealed that AIP densities exhibited high skewness, particularly in the denser woody Savanna vegetation types natural areas, where some plots contained extremely high densities while others recorded very low or zero values. Given this non-normal distribution, parametric tests such as t-tests were deemed inappropriate, and non-parametric alternatives were used instead.

To compare AIP densities between natural areas and road reserves, a t-test was initially conducted. However, due to the skewed distribution of the data, a Mann-Whitney U test was performed as a more suitable alternative. The Mann-Whitney U test ranks values from both groups and calculates the sum of ranks. The test statistic U was computed using the formula (Mann & Whitney, 1947; McKnight & Najab, 2010):

$$U = n_1 n_2 + \frac{n_1(n_1+1)}{2} - R_1 \qquad \text{Equation 6.16}$$

where $n_1$ and $n_2$ are the sample sizes of the two groups, and $R_1$ represents the sum of ranks for one of the groups. This test was used to assess whether AIP densities in natural areas were significantly different from those in road reserves. The results indicated a statistically significant difference between natural areas and road reserves in the more woody and structurally complex vegetation group, whereas no significant difference was found between natural areas and road reserves in the grass-dominated Marikana Thornveld.

To evaluate whether AIP presence was associated with vegetation groups and land-use position, a Chi-Square test was conducted. This test compares the observed frequencies of AIP presence against expected frequencies under the assumption of independence. The Chi-Square test was calculated using the formula (McHugh, 2013):

$$X^2 = \sum \frac{(O-E)^2}{E} \qquad \text{Equation 6.17}$$

$O$ represents the observed frequency of AIP presence or absence in each category, and $E$ is the expected frequency, determined using:

$$E = \frac{Row\ Total \times Column\ Total}{Grand\ Total} \qquad \text{Equation 6.18}$$

Separate Chi-Square tests were conducted to assess whether AIP presence varied significantly across vegetation types (Marikana Thornveld vs the denser woody Savanna vegetation types) and across land-use categories (natural area vs road reserve), treating each factor independently. By grouping the number of plots where AIPs were present or absent within each category, it was possible to determine whether certain vegetation types or land-use positions were more prone to AIP invasion.

A final analysis was conducted to assess whether soil type played a role in AIP distribution. AIPs were grouped by soil type, and a Chi-square test was used to determine whether AIP presence was significantly associated with specific soil types (Legendre & Legendre, 2012; McHugh, 2013). This analysis was conducted across all plots, without separating by vegetation group or land-use category, which assumes that the effect of soil type on AIP presence is independent of other potential drivers. Although this is a common exploratory approach where sample sizes are limited (Quinn & Keough, 2002; Zar, 1999), it does not account for possible interactions or confounding variables such as vegetation structure or land-use history (Gotelli & Ellison, 2013). Given the limitations of the dataset and the lack of sufficient replication across all combinations of factors, each variable, namely soil type, land use, and vegetation group, was tested independently. Results from the soil analysis should therefore be interpreted with caution and regarded as indicative rather than definitive, acknowledging that other environmental and anthropogenic factors may also influence AIP distribution (Gaertner *et al.*, 2012; Richardson *et al.*, 2000).

Each statistical test was selected to address specific aspects of AIP distribution. The Mann-Whitney U test was used to compare AIP densities between natural areas and road reserves for the Marikana Thornveld and denser woody Savanna vegetation types separately, ensuring that the impact of land-use type on AIP abundance was properly evaluated (Mann & Whitney, 1947; McKnight & Najab, 2010). The Chi-Square test for vegetation group and land-use position provided insights into whether AIP presence was influenced by broader landscape characteristics. Additionally, the Chi-Square test for soil type examined whether specific environmental conditions played a role in AIP establishment (Zar, 1999).

This structured methodology provided a comprehensive approach to understanding AIP distribution patterns across different vegetation groups and land-use categories. By incorporating subplot extrapolation, variance calculations, non-parametric statistical tests, and contingency table analyses, the study ensured a robust framework for evaluating AIP invasion dynamics (Barbour *et al.*, 1999; Quinn & Keough, 2002). The selection of appropriate statistical methods accounted for the non-normal distribution of data (Gotelli & Ellison, 2013), while the examination of soil type added a layer of insight into potential environmental drivers of AIP establishment. These analyses contribute to the broader understanding of how AIP control strategies should be adapted to account for landscape-scale invasion processes (Legendre & Legendre, 2012).

## 6.3 Results

### 6.3.1 General Plant Diversity

AIP presence varied significantly across the two dominant vegetation groups surveyed, namely Marikana Thornveld and the denser woody Savanna vegetation types, indicating a clear vegetation-related trend in invasion patterns. Plots in the denser woody Savanna vegetation types consistently exhibited higher AIP densities and frequencies, particularly in natural areas, where densities ranged from 0 to 583 AIP/100 $m^2$, with a mean of 78.4 (SD = 178.91). In contrast, Marikana Thornveld plots recorded much lower AIP densities, with natural areas having a maximum of 201 AIP/100 $m^2$ and a mean of 23 (SD = 66.78). Road reserves had a maximum of 5 AIP/100 $m^2$, with a mean of only 0.56 (SD = 1.67), and many plots showed no AIP presence at all (Table 6.1; Fig. 6.1b).

Vegetation composition varied across Marikana Thornveld and the denser woody Savanna vegetation types within the study area, with noticeable differences between plots located in natural areas and those in the road reserve. The denser woody Savanna vegetation types exhibited greater species richness than Marikana Thornveld, with natural areas in the denser woody Savanna vegetation types containing a mean of 12.4 species per plot (SD = 4.58), whereas natural areas in Marikana Thornveld had significantly lower species richness (mean = 4.89; SD = 2.03) (Fig. 6.1a).

Grass was the dominant plant type across all plots, particularly in Marikana Thornveld, where it accounted for most recorded species. Shrubs were more abundant in the denser woody Savanna vegetation types, especially in natural areas, while trees were sparsely distributed, with most recorded trees being large, mature individuals. Succulents were the least common plant type, with a slightly higher prevalence in natural areas than in road reserves.

Vegetation height varied significantly across land-use types. In natural areas within the denser woody Savanna vegetation types, tall grasses, shrubs, and large trees were common, whereas vegetation in road reserves was significantly shorter due to periodic clearing. Grass in the road reserves is mowed regularly to maintain visibility and reduce wildlife-related hazards, which further restricts plant height and overall biomass accumulation (Forman & Alexander, 1998). Similarly, in Marikana Thornveld, road reserves contained shorter grasses, while natural areas supported taller and less disturbed vegetation. Road reserves, being subject to regular maintenance and vegetation clearing, consistently supported fewer plant species than adjacent natural areas. In the road reserves of the denser woody Savanna vegetation types, the mean species richness was 4.9 (SD = 2.02), while Marikana Thornveld road reserves showed even lower richness (mean = 4.11; SD = 1.27) (Fig. 6.1a).

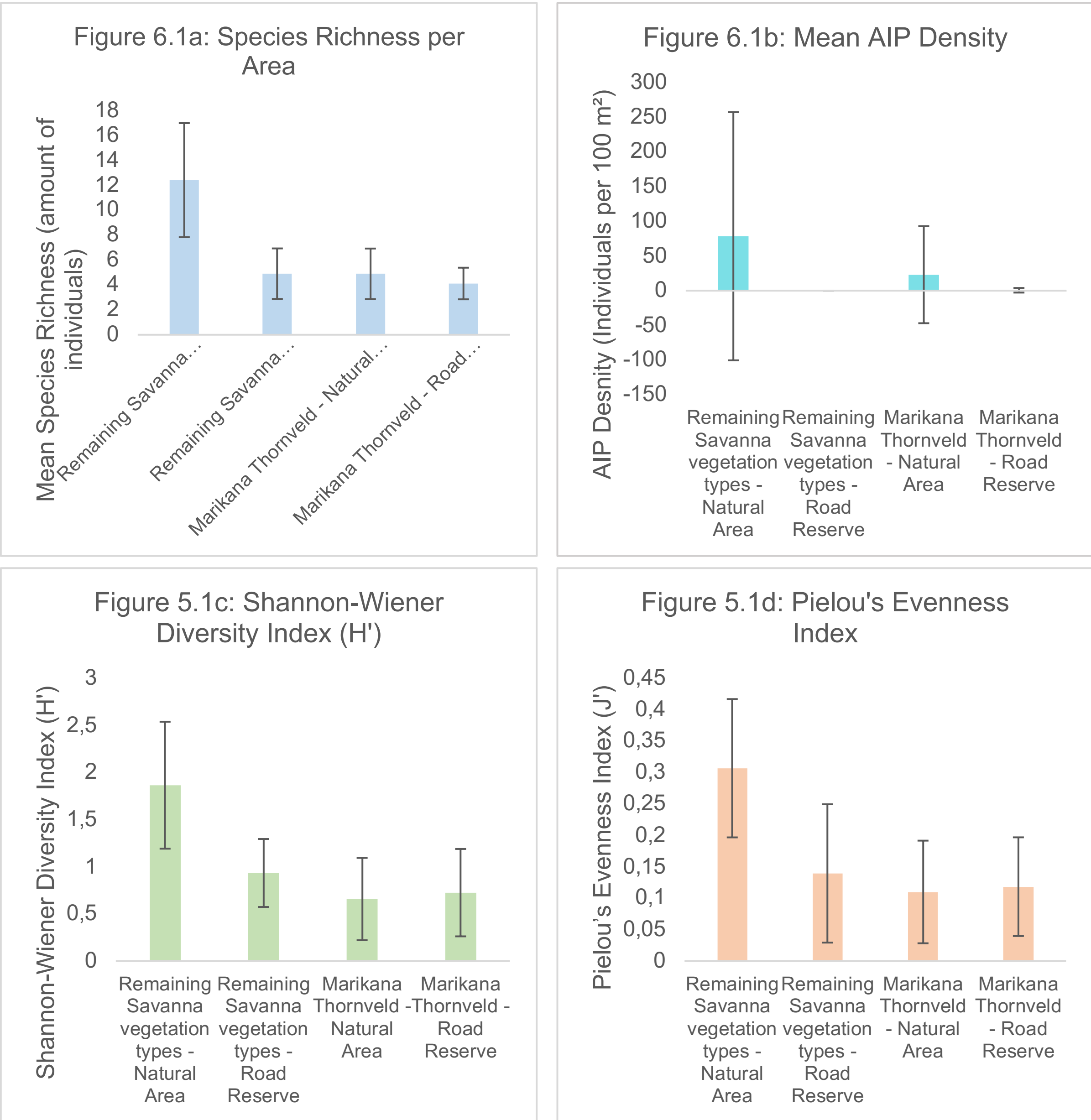


Figure 6.1 Mean species richness (a), AIP density (b), Shannon diversity index (c), and Pielou's evenness index (d) across vegetation group and land-use categories along the Bakwena N4 Toll Route. Error bars represent 95% confidence intervals.

Diversity was assessed using several ecological indices, including the Shannon-Wiener Diversity Index, Pielou's Evenness Index, and species richness. The Shannon Index showed higher values in the denser woody Savanna vegetation types than in Marikana Thornveld. Natural areas in the denser woody Savanna vegetation types had the highest diversity (mean H = 1.87; SD = 0.67), while road reserves in the denser woody Savanna vegetation types had lower diversity (mean

H = 0.93; SD = 0.36). In Marikana Thornveld, natural areas had lower diversity (mean H = 0.66; SD = 0.44), while road reserves displayed slightly higher values (mean H = 0.73; SD = 0.46) (Fig. 6.1c).

Evenness values, as measured by Pielou's Index, ranged from 0.06 to 0.39, indicating how uniformly species were distributed across plots. Natural areas in the denser woody Savanna vegetation types had the highest evenness (mean = 0.31; SD = 0.11), suggesting a more even species composition with no single species dominating. In contrast, road reserves exhibited lower evenness values (denser woody Savanna vegetation types: mean = 0.14; SD = 0.11; Marikana Thornveld: mean = 0.12; SD = 0.08), indicating that fewer species dominated these plots (Fig. 6.1d).

### 6.3.2 AIP Densities in Natural Areas vs Road Reserves

AIPs were recorded in 10 of the 38 plots, with presence strongly concentrated in natural areas and largely absent from road reserves. All ten road reserve plots in the denser woody Savanna vegetation types recorded zero AIP individuals, indicating either effective management, unsuitable growing conditions, or a combination of both. In Marikana Thornveld, only one road reserve plot showed any AIP presence, with a modest count of 5 individuals, made up of one low-cover species (*Convolvulus arvensis*), which was recorded in early-stage establishment (see Table 6.1; Table S6.1).

Table 6.1: Summary of AIP Density and Abundance by Vegetation Group and Land Use Type

| Vegetation Group | Land Use | Sample Size | Plots with AIP presence | Number of Plants | Number of AIPs | AIP Density (per 100m$^2$) Mean | AIP Density Standard Deviation |
|---|---|---|---|---|---|---|---|
| Denser woody Savanna vegetation types | Natural Area | 10 | 6 | 4595 | 784 | 78.4 | 178.91 |
| Denser woody Savanna vegetation types | Road Reserve | 10 | 0 | 8219 | 0 | 0 | 0 |
| Marikana Thornveld | Natural Area | 9 | 3 | 6112 | 207 | 23 | 66.77 |
| Marikana Thornveld | Road Reserve | 9 | 1 | 4958 | 5 | 0.56 | 1.67 |

In total, 1,047 AIP individuals were recorded across the full dataset. The vast majority of approximately 94.7% occurred in natural areas. This pattern held across both vegetation groups. Road reserves accounted for 5.3% of the total AIP count, despite covering almost half of the total plots surveyed (Table 6.1; Table S6.1).

Within the denser woody Savanna vegetation types natural area, AIPs were present in 6 of the 10 plots, with densities ranging from 20 to 583 individuals per 100 m² (See Table 6.1; Table S6.1). One site recorded the highest count with 583 individuals, dominated by *Arundo donax*, which formed dense stands with high cover. Other sites showed moderate invasion levels (Table S6.1), with 40–60 individuals each, often involving *Lantana camara* or *Parthenium hysterophorus*. Lighter invasions of around 20 individuals occurred in a few additional plots. AIPs were completely absent from the denser woody Savanna vegetation types road reserve plots (Table S6.1).

In Marikana Thornveld natural areas, AIPs were detected in three of nine plots, although densities were lower than in the denser woody Savanna vegetation types (Table 6.1). One plot stood out with 201 individuals, composed mainly of *Tithonia rotundifolia* and *Ipomoea indica*, both of which formed semi-dense upper layers over the indigenous vegetation (Table S6.1). The remaining four invaded plots showed only scattered AIP presence. In Marikana Thornveld, invasion was therefore more dispersed, with AIPs occurring in small pockets or along isolated edges of natural vegetation rather than forming dense, continuous invasions.

The Marikana Thornveld road reserves reflected a similar pattern of low invasion pressure. Of the nine plots surveyed, only one contained any AIPs, with 5 individuals recorded. These were observed in low cover classes and were notably immature, suggesting recent establishment or failed expansion. All other road reserve plots in both Marikana Thornveld and the denser woody Savanna vegetation types were entirely free of AIPs, reinforcing the overall pattern that 18 of the 19 road reserve plots were uninvaded (Table 6.1; Table S6.1).

Species richness varied across invaded plots (Figure 6.1a). *Lantana camara* occurred in multiple sites, often at low cover levels (<5–25%). The most heavily invaded site was structurally dominated by *Arundo donax*, a tall grass species with more than 75% cover, and this site recorded the highest AIP count in the denser woody Savanna vegetation types. Other species included *Parthenium hysterophorus*, which occurred as a small shrub with a distinct upright growth form (Table S6.2).

The vertical structure in the natural area was diverse. Shrubs such as *Lantana camara* and *Parthenium hysterophorus* ranged between 0.2 m and 1.25 m in height. *Arundo donax* contributed strongly to the ground layer, occurring in dense stands in some areas. Spatial distribution also varied between species: *Arundo donax* formed near-monotypic dense stands, while *Lantana camara* occurred in scattered small clumps (Table S6.1; Table S6.2).

All invaded plots occurred on gentle slopes (<15°), with both north- and south-facing aspects. Soils were consistently Ferric Luvisols or Nitosols. Notably, two of the most invaded plots had Rhodic Nitosols or Lixisols and also showed signs of other disturbance indicators such as high basal cover and altered shrub layers (Table S6.1).

AIP presence in Marikana Thornveld was more fragmented than in the denser woody Savanna vegetation types, with lower overall densities but a wider mix of species. One Marikana Thornveld natural area recorded the highest AIP count for this vegetation group (201 individuals), dominated by *Tithonia rotundifolia* and *Ipomoea indica* (Table S6.1). *Tithonia rotundifolia* had a cover ranging between 50–75% and contributed most of the vertical structure in this plot, with a shrub height of 1.5 m. *Ipomoea indica* occurred at 25–50% cover and at a similar height. These dense shrub layers indicate an established invasion front.

AIP presence in the remaining Marikana Thornveld natural areas was minimal. One plot recorded five individuals of *Parthenium hysterophorus*, a low-growing shrub approximately 0.4 m in height with sparse cover below 5%. Another site contained a single Lantana camara shrub reaching 2.5 m in height. These areas showed little vertical development, and the observed AIPs were consistent with early-stage invasion. In one case, an isolated *Tipuana tipu* tree (4 m tall) was recorded. Full species listings and structural data are provided in Table S6.1.

All Marikana Thornveld natural plots were located on very gentle, south-facing slopes and were underlain by Calcic Vertisols or Haplic Lixisols. While most sites showed low invasion pressure, the more heavily invaded plot exhibited visible structural transformation and altered ground cover. Species like *Tithonia* and *Ipomoea*, occurred in semi-dense clusters, in contrast to the more scattered AIP patterns observed elsewhere in the Marikana Thornveld (Table S6.1).

In the road reserve plots, only one plot showed AIP presence. The only species recorded was *Convolvulus arvensis*, represented by five individuals. It formed a low shrub layer of less than 1.1 m in height and occurred at less than 5% cover. As in the natural plots, this road reserve plot was located on a south-facing, gently sloping Calcic Vertisol (Table S6.1).

Environmental conditions varied widely across the 38 surveyed plots. In the denser woody Savanna vegetation types, several natural plots were located in relatively undisturbed areas with dense vegetation, while others showed signs of historic land clearing or erosion. One plot was positioned near the Klein-Maricopoort Dam and featured strong, dense vegetation with little evidence of disturbance. In contrast, its paired road reserve plot was highly disturbed due to regular use by heavy vehicles pulling off the main road. A different plot was situated close to a residential property with cleared vegetation and a planted tree line along the boundary, while the adjacent road reserve remained undisturbed. Plots showcasing dense tree cover were in good condition (Table S6.1 under Further Comments).

In the denser woody Savanna vegetation types, several natural plots showed varying levels of disturbance. One plot had an open vegetation structure, likely due to past clearing, with farmland to the southwest and a residential property nearby. Another site displayed visible erosion and many bare soil patches, indicating a disturbed condition, although the adjoining road reserve was well maintained. Some plots were located between dense vegetation to the east and more open areas to the west, and the adjacent road reserves in these areas were in good condition. In other cases, the surrounding environment included built structures that may have influenced vegetation structure and disturbance. One of the sites recorded as a hotspot was situated along the edge of private land and showed a large invasion of *Arundo donax*. Another natural area on private property had dense tree cover that may have been planted, while a separate plot had sparse vegetation and low tree density. All associated road reserves in these cases were intact and generally well maintained. Full descriptions are provided in Table S6.1.

In the Marikana Thornveld vegetation type, the surrounding land was more heavily influenced by human activity. One plot was located in the middle of cultivated land where natural vegetation was absent, although the adjacent road reserve was intact. Another site at the edge of agricultural land showed poor vegetation cover and large areas of exposed soil. A nearby plot showed signs of early vegetation recovery following previous clearing. The road reserves paired with these plots were in good condition. One natural area was located near a crocodile farm and river, where the surrounding vegetation had been cleared, likely for access purposes. Other sites were in healthier condition, positioned next to streams or low-density housing, and these supported dense vegetation and well-kept road reserves.

One disturbed Marikana Thornveld plot was located near a landfill and had almost no remaining vegetation. The road reserve in this area also appeared poorly managed. Another plot, previously described as an AIP hotspot, was positioned next to Bon Accord Dam and surrounded by disturbed land uses such as agricultural buildings and institutional grounds. Vegetation in the plot was limited, likely due to clearing or cultivation, although the road reserve remained intact. The final Marikana Thornveld plot was located inside a private nature reserve. Although no clear signs of disturbance were visible, the vegetation was in good condition, and the associated road reserve was also well maintained. Further details are available in Table S6.1.

Two plots recorded AIP densities exceeding 100 individuals and were identified as invasion hotspots, one located in the denser woody Savanna vegetation types and the other in the Marikana Thornveld vegetation type. These sites stood out not only for their high AIP counts but also for their visibly altered vegetation structure and dominance by a small number of invasive species (Table S6.1).

The denser woody Savanna vegetation types hotspot recorded the highest AIP count in the study, with 583 individuals. This site was positioned near the edge of private property, where additional *Arundo donax* clumps were visible approximately 20 to 30 metres to the north and northeast, suggesting a larger, continuous invasion front. A dirt road separated the property from the adjacent road reserve. Vegetation structure at the site was significantly altered, with *Arundo* forming dense, monodominant patches taller than three metres and covering more than 75 per cent of the ground surface. These plants were spatially clustered, forming a thick core that expanded outward into surrounding areas.

The Marikana Thornveld hotspot recorded 201 AIP individuals. This plot was located next to Bon Accord Dam and was surrounded by high levels of human disturbance, including agricultural facilities and institutional buildings. Vegetation in and around the site was sparse, likely due to previous clearing or cultivation. The dominant invasive species were *Tithonia rotundifolia* and *Ipomoea indica*, both of which formed a dense, low shrub layer reaching up to 1.5 metres in height and covering up to 75 per cent of the area. The invasion was semi-clustered, with denser patches along the plot edges that gradually opened toward the centre. Additional structural and floristic details are provided in Table S6.1.

In both hotspot plots, vertical structure was visibly altered. In Marikana Thornveld, grass and shrub layers were dominated by alien species, suppressing indigenous growth. Ground cover was extensive in both cases, with clear competition for space and light. These hotspots were associated with recent or historic disturbance, proximity to infrastructure, and neighbouring propagule sources, suggesting a link between edge effects, land-use history, and invasion intensity.

No AIPs were recorded in any of the denser woody Savanna vegetation types road reserve plots. Vegetation in these areas was typically dominated by indigenous grasses and shrubs, with limited visible disturbance. Canopy structure was open, with sparse shrub cover and minimal evidence of trampling, soil exposure, or previous invasion events.

### 6.3.3 Diversity Metrics

Shannon diversity varied across plots (Figure 6.1c), with natural areas in the denser woody Savanna vegetation types generally recording the highest values. Several of these sites exceeded a Shannon Index of 2.0, indicating high species richness and relatively balanced community composition. Road reserves within the denser woody Savanna vegetation types showed moderate diversity in some cases, with at least one site reaching a Shannon score above 1.3. In contrast, Marikana Thornveld plots had lower and more variable diversity scores. Some sites recorded very low values, including one road reserve and one natural area where diversity fell below 0.16. A few Marikana Thornveld plots exceeded 1.0, although these were less common and were typically associated with higher indigenous species richness. Further detail is provided in Table S6.2.

Pielou's Evenness scores revealed additional patterns in species distribution (Figure 6.1d). The most even plots tended to have moderate plant densities and high richness, particularly in denser woody Savanna vegetation types, natural areas where evenness exceeded 0.80 in some sites. At the opposite end of the spectrum, several plots showed extremely low evenness, reflecting dominance by one or two species. One heavily invaded denser woody Savanna vegetation types plot recorded complete dominance by a single alien species, with both Shannon and Evenness values equal to zero. A similar trend occurred in Marikana Thornveld natural areas, where some plots were skewed by single-species dominance. One road reserve plot in Marikana Thornveld showed a surprisingly high evenness value above 0.80, despite having only moderate species richness. Complete plot-level evenness scores are listed in Table S6.2.

Alien species presence was also associated with variation in diversity indices. In the denser woody Savanna vegetation types, one site with 583 individuals of a single AIP showed no diversity, while another site with 60 AIP individuals and 11 species had a Shannon Index of 1.9. In Marikana Thornveld, the most heavily invaded plot had over 200 AIPs and a Shannon Index below 0.9, indicating moderate structural change. Another Marikana Thornveld plot recorded 1,974 plants but only one AIP species, resulting in a Shannon value of only 0.295 due to strong species dominance. Full details on AIP numbers, species richness, and diversity indices are presented in Table S6.2.

### 6.3.4 Statistical Analyses

A comparison between the denser woody Savanna vegetation types, natural areas and road reserves revealed a statistically significant difference in AIP densities ($U = 20.0$, $p = 0.0059$) (see Table 6.2). All road reserve plots in the denser woody Savanna vegetation types showed a complete absence of AIPs, while more than half of the natural plots contained alien individuals, with densities ranging from 20 to 583 per 100 m². Although one natural site recorded an unusually high density of 583 individuals, it was confirmed as an outlier, and its removal did not alter the outcome of the test. The mean AIP density in the denser woody Savanna vegetation types natural areas was 78.4 individuals per 100 m², compared to a consistent zero in road reserves. Rank-based comparisons supported this result, with natural plots ranked significantly higher than road reserves. Summary statistics and group distributions are provided in Table 6.1 and Table 6.2, which together show that natural areas, particularly in the denser woody Savanna vegetation types, are more heavily invaded than road reserves.

Table 6.2 Mann–Whitney U test results comparing AIP densities between Marikana Thornveld and the denser woody Savanna vegetation types, and between natural areas and road reserves.

| Comparison | Group 1 (n) | Group 2 (n) | U-value | p-value | Significant? | Notes |
|---|---|---|---|---|---|---|
| Denser woody Savanna vegetation types: Natural vs Road Reserve | 10 | 10 | 20 | 0.0059 | Yes | All road reserves = 0 AIPs. |
| Marikana Thornveld: Natural vs Road Reserve | 9 | 9 | 31.5 | 0.3023 | No | Low densities in both groups. |
| All Natural vs All Road Reserves | 19 | 19 | 101.5 | 0.0031 | Yes | Natural areas more invaded overall. |
| Denser woody Savanna vegetation types vs Marikana Thornveld (all plots) | 19 | 19 | 201 | 0.439 | No | Denser woody Savanna vegetation types showed a slightly higher trend. |

In Marikana Thornveld, differences in AIP density between natural and road reserve plots were not statistically significant ($U = 31.5$, $p = 0.3023$). AIPs were recorded in three of the nine natural plots, with densities ranging from one to 201 individuals per 100 $m^2$. Among road reserves, only one plot recorded any AIPs, with five individuals present, while the rest were uninvaded. Despite this variation, mean densities in both categories remained below 25 individuals per plot. The single high-density Marikana Thornveld site influenced the range but not the significance of the result. These findings suggest that, in the Marikana Thornveld setting, land use alone does not strongly determine AIP density. Full group-level values and distribution summaries are presented in Table 6.2.

The Chi-square test of independence was used to examine the association between AIP presence and vegetation group or land-use type (Table 6.3). The result was statistically significant ($p < 0.05$), indicating that AIP presence was not randomly distributed across categories. Natural plots within the denser woody Savanna vegetation types had the highest number of invaded sites, while all road reserve plots within this group remained free of AIPs. In Marikana Thornveld, AIP presence was low across both natural and managed sites. This pattern indicates that vegetation group and land-use position significantly influenced AIP distribution, while soil type did not appear to structure presence across the study area. Chi-square outputs and contingency data are provided in Table 6.3.

Table 6.3 Chi-square test results assessing the association between AIP presence, vegetation group, and land-use type.

| Test | $\chi^2$ | df | p-value | Significant? | Notes |
|---|---|---|---|---|---|
| Vegetation Group*Land Use | 10.724 | 3 | 0.0133 | Yes | AIPs homogeneously distributed. |
| Soil Type | 3.33 | 4 | 0.504 | No | No significant soil-AIP relationship. |

When all natural area plots ($n = 19$) and road reserve plots ($n = 19$) were grouped together, irrespective of vegetation group, a further Mann-Whitney U test confirmed a significant difference in AIP densities ($U = 101.5$, $p < 0.05$). This suggests that, overall, natural areas were more frequently and more heavily invaded than adjacent road reserves. However, when Marikana Thornveld and the denser woody Savanna vegetation types were compared irrespective of land use, the result was not statistically significant ($U = 201$, $p > 0.05$), although the pattern still suggested higher invasion in the denser woody Savanna vegetation types. Variation within the vegetation groups, particularly at a few high-density sites, may have reduced statistical power. Detailed test statistics are reported in Table 6.2.

A final Chi-square test assessed the association between AIP presence and soil type (Table 6.4). Across all plots, five major soil types were recorded: Ferric Luvisols, Rhodic Nitisols, Haplic Lixisols, Calcic Vertisols, and Lithic Leptosols. Expected values were calculated under the assumption of independence between soil type and AIP presence. AIPs occurred across several soil classes, although Ferric Luvisols and Rhodic Nitisols were most frequently associated with invaded plots in the denser woody Savanna vegetation types. Two of the most heavily invaded sites in this group were located on Rhodic Nitisols. In Marikana Thornveld, most AIP-containing plots occurred on Calcic Vertisols or Haplic Lixisols. Despite these patterns, the test result was not statistically significant ($p > 0.05$), indicating no strong overall association between AIP presence and soil type. A full breakdown of soil classifications and AIP distributions is provided in Table 6.4.

Table 6.4 Observed and expected frequencies of AIP presence across soil types with Chi-square test results.

| Soil Type | AIP Present | AIP Absent | Total | Expected Present | Expected Absent |
|---|---|---|---|---|---|
| Ferric Luvisols | 2 | 6 | 8 | 1.89 | 6.11 |
| Calcic Vertisols | 2 | 10 | 12 | 2.84 | 9.16 |
| Rhodic Nitosols | 3 | 3 | 6 | 1.42 | 4.58 |
| Haplic Lixisols | 2 | 8 | 10 | 2.37 | 7.63 |
| Calcic Lixisols | 0 | 2 | 2 | 0.47 | 1.53 |
| **Total** | **9** | **29** | **38** | - | - |

**Chi-square test statistic ($\chi^2$) = 3.33**

**Degrees of freedom (df) = (5 – 1) × (2 – 1) = 4**

**p-value ≈ 0.504**

## 6.4 Discussion

The findings of this study indicate that AIP presence along the Bakwena N4 Toll Route is strongly shaped by vegetation structure, land use, and local disturbance history. Across all surveyed plots, the denser woody Savanna vegetation types appeared more prone to invasion than Marikana Thornveld, particularly in natural areas where several plots recorded high AIP densities and visibly altered vegetation structure. This difference appears to be linked to variation in vegetation complexity and proximity to human-altered landscapes, which is consistent with patterns described by Rouget *et al.* (2004) and Richardson *et al.* (2007), who found that structurally and ecologically complex systems may sustain more intense invasions under suitable disturbance conditions.

The denser woody Savanna vegetation types' natural plots supported greater species richness and taller, more structurally complex vegetation, which created multiple canopy layers and heterogeneous microhabitats. Such diversity increases the number of ecological niches that can be exploited by invaders. Shrubs and trees, especially in partially disturbed plots, facilitated shaded understories and accumulated litter, creating conditions favourable to shade-tolerant species like *Lantana camara* and persistent colonisers like *Arundo donax*. Similar patterns were recorded in riparian systems where vertical complexity enhanced invasion success (Gaertner *et al.*, 2012; Richardson *et al.*, 2007). The presence of these species in dense stands shows how vertical complexity and disturbance can interact to support dominant invasions that displace indigenous vegetation.

Disturbance history played a critical role in enabling AIP establishment. Several invaded plots of the denser woody Savanna vegetation types were situated in areas with evidence of historic clearing, erosion, or adjacent land use changes. These disturbances reduce native cover and create open microsites for AIP germination and expansion (Davis *et al.*, 2000). Once established, species such as *Parthenium hysterophorus* further disrupt native assemblages by altering soil structure and shading out ground flora (van Kleunen *et al.*, 2016). In contrast, the denser woody Savanna vegetation types and road reserve plots remained completely free of AIPs despite their proximity to the same propagule sources. This points to the importance of routine maintenance interventions, such as mowing and clearing, in preventing early establishment. These findings align with previous work showing that frequent vegetation removal in road verges limits biomass accumulation and propagule retention (Forman & Alexander, 1998).

While Marikana Thornveld showed some AIP presence, particularly in natural areas, overall densities were lower, and invasions were more fragmented. This lower invasion intensity may be related to the vegetation structure of Marikana Thornveld, which is more strongly herbaceous and has less woody cover than the denser woody Savanna vegetation types (Low & Rebelo, 1996). Under these conditions, competition for ground-level resources and the limited availability of sheltered establishment sites may restrict the spread of certain invaders, particularly alien shrubs and vines that are less suited to exposed conditions (Foxcroft *et al.*, 2010; Milton & Dean, 1998). Marikana Thornveld road reserves showed a similar pattern, with only one invaded site, where AIP presence was minor, and the recorded individuals were still immature.

The outlier presents a different scenario. Despite the generally lower invasion levels recorded in Marikana Thornveld, this plot was heavily invaded by *Tithonia rotundifolia* and *Ipomoea indica*, which formed dense shrub layers and altered the vertical structure of the vegetation. The site's proximity to Bon Accord Dam and nearby municipal infrastructure, together with evidence of prior mechanical clearing, suggests that localised disturbance and high propagule pressure can override the broader vegetation-related pattern observed in this study. Similar edge-effect dynamics have been noted by Alston and Richardson (2006) and, more recently, by Nelufule *et al.* (2024) in Tshwane, where urban pressure and prior disturbance created invasion hotspots in otherwise less heavily invaded systems.

Land use emerged as a critical factor in determining AIP density and distribution. Across both vegetation groups, natural areas were more heavily invaded than road reserves. The Mann-Whitney U test showed significant differences in AIP density when comparing all road reserves against natural areas, regardless of vegetation group. The denser woody Savanna vegetation types natural areas, in particular, showed consistent and often dense AIP presence, while the complete absence of

AIPs in the denser woody Savanna vegetation types road reserves highlights the effectiveness of ongoing maintenance. However, this protection does not extend beyond the managed corridor. Many invaded natural plots were situated directly adjacent to road reserves, underscoring that management within the road reserve cannot compensate for unmanaged, invaded areas nearby. These border plots, such as plot 08RemVegNat, were found within 30 m of a clean road reserve yet hosted hundreds of AIP individuals, suggesting that reinvasion is likely unless neighbouring land is also addressed. Similar spillover effects from adjacent unmanaged areas have been noted by van Wilgen *et al.* (2012).

In Marikana Thornveld areas, the effect of land use was less pronounced. Although natural areas did show slightly higher AIP counts, the difference between them and road reserves was not statistically significant. This suggests that in structurally simple systems, natural ecological barriers may reduce the reliance on active management. Still, hotspot cases indicate that such defences can be breached under the right combination of propagule availability and disturbance (Rouget *et al.*, 2004).

Urbanisation and associated human activity were consistently linked to increased invasion pressure, though not in a uniform way. The denser woody Savanna vegetation plots near homesteads, farmyards, or previously cleared areas showed higher AIP densities. The use of ornamental species like *Jacaranda mimosifolia* in residential areas creates a pathway for species to escape cultivation and establish in surrounding vegetation. Similarly, roads and driveways increase physical disturbance, aid seed dispersal, and fragment vegetation, all of which support AIP spread (Foxcroft *et al.*, 2013; Gelbard & Belnap, 2003).

In the Marikana Thornveld vegetation type, urban and peri-urban edges likewise facilitated invasion, though at a smaller scale. One plot adjacent to schools, churches, and cultivated land recorded the second-highest AIP density. Despite being within a vegetation structure that generally resists invasion, the proximity of intense human activity, along with historic disturbance, likely created conditions that promoted colonisation. Interestingly, not all urban-adjacent plots were invaded, suggesting that susceptibility depends on more than proximity alone. Disturbance intensity, vegetation recovery status, and the type of alien species present all appear to interact (Meyer *et al.*, 2021).

The role of soil type in explaining these patterns was less clear. Although certain high-invasion plots occurred on Rhodic Nitosols, the overall Chi-Square test did not detect a significant association between soil class and AIP presence. This suggests that while soil properties might contribute to invasion risk in individual cases, they do not exert a strong, consistent influence across the entire study area. Rather, the combined effects of disturbance, vegetation structure, and management seem to play a more dominant role (Ł Aska, 2001).

Invasion hotspots that are characterised by high AIP density and low species diversity were most evident in plots where a single species dominated. In the denser woody Savanna vegetation types, one plot was completely overrun by *Arundo donax*, with no other species present. This resulted in both Shannon diversity and Pielou's Evenness values of zero. Similarly, in the Marikana Thornveld, another plot showed very low evenness and moderate diversity, reflecting semi-monodominant conditions. These findings echo previous research showing that once dominance is established, diversity tends to collapse, making recovery more difficult without direct intervention (Kumar Rai & Singh, 2020; van Kleunen *et al.*, 2016).

Altogether, the results demonstrate that AIP occurrence is driven by a combination of ecological context and human influence. The denser woody Savanna vegetation types' structural complexity and greater historical disturbance leave it more vulnerable to invasion, particularly in unmanaged natural areas. The Marikana Thornveld vegetation type remains more resistant overall, but this resistance can be breached under high propagule pressure and intense land use. Management interventions in road reserves appear effective in limiting invasions but are insufficient if adjacent land remains untreated. Targeted control beyond road margins, especially in hotspots and peri-urban zones, will be critical for long-term success in preventing AIP spread along the Bakwena N4.

## 6.5 Conclusion

This study demonstrates that AIP patterns across the landscape are shaped by a combination of vegetation structure, land-use history, and management intensity. The denser woody Savanna vegetation types were more prone to invasion, with greater vertical complexity, legacy disturbances, and fragmented land use creating favourable conditions for alien species establishment. In contrast, Marikana Thornveld displayed lower invasion levels, with relatively low AIP densities even in unmanaged plots, likely linked to its more open vegetation structure and less sheltered establishment conditions.

Despite the varying levels of susceptibility, one outcome remained consistent: road reserves, particularly in the denser woody Savanna vegetation types, were almost entirely AIP-free. This provides strong, empirical evidence that Bakwena's eradication and management programme is highly effective. Even in cases where road reserves were adjacent to heavily invaded natural plots, invasive species were absent from the managed margins. This suggests that active, routine interventions such as clearing, mowing, and maintenance can override the expected influence of surrounding land use or vegetation type-specific risk factors.

Land use did play a notable role, especially in natural areas that were previously cleared or disturbed. These sites were more likely to host high AIP densities, particularly in the denser woody Savanna vegetation types. Yet, the absence of AIPs in similarly exposed road reserves highlights that with proper and sustained intervention, the influence of land use on invasion can be mitigated.

The findings highlight that long-term success in AIP control depends not only on understanding ecological risk factors but also on implementing consistent, well-targeted management strategies. Bakwena's programme offers a practical model for maintaining road reserves as functional barriers, even in high-risk zones. While this study could not directly quantify the contribution of adjacent natural areas to road reserve invasions due to the success of the eradication programme, the absence of AIPs in maintained reserves is itself evidence of effective long-term management. This effectiveness, however, also limits the ability to detect broader associations or directional spread. In theory, the programme could become even more cost-effective if interventions were extended into adjacent natural areas. Future research comparing invasion dynamics between managed and unmanaged zones could provide valuable insight into reinvasion rates and long-term containment potential.

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# CHAPTER 7
# DISCUSSION AND CONCLUSION

## 7.1 Discussion

The long-term dataset from 2011 to 2024 provides a rare opportunity to evaluate the effectiveness of a sustained alien invasive plant (AIP) removal strategy along a major South African transport corridor. This dissertation assessed changes in AIP abundance and diversity over time, examined differences between Marikana Thornveld and the denser woody Savanna vegetation types, compared road reserves with adjacent natural areas, and explored how land use, disturbance history, soil type, and vegetation quality influenced invasion outcomes. Taken together, the findings show that invasion dynamics are strongly shaped by ecological context and human activity, and that although the control programme has achieved measurable progress, challenges remain in maintaining long-term success.

At a broad temporal scale, AIP densities declined significantly in rural denser woody Savanna vegetation areas, confirming that repeated removal has reduced populations over time. This outcome is important given the large tracts of intact vegetation in these zones, where unchecked invasions could have long-term ecological impacts. However, diversity indices revealed that eradication has not fully stabilised ecological dynamics. As Richardson and Pyšek (2006) observed, the removal of dominant species can create opportunities for secondary invaders to establish, leading to compositional shifts rather than full recovery. In contrast, urban and peri-urban areas showed weaker or inconsistent declines. Urban plots remained subject to persistent reinvasion pressure, consistent with Foxcroft *et al.* (2013), who noted that high levels of disturbance and propagule pressure characterise these environments. Peri-urban areas proved most resistant to long-term suppression, with densities stabilising rather than declining and diversity levels remaining high. These landscapes, shaped by agriculture, development pressure, and edge effects, continue to act as reservoirs and reinvasion sources for surrounding zones, echoing the challenges reported by Gaertner *et al.* (2012) in transitional environments.

Comparisons between vegetation types provided further insight. Natural areas within the denser woody Savanna vegetation types consistently supported higher AIP densities than sites in Marikana Thornveld, reflecting the influence of vegetation structure, resource availability, and disturbance regimes. The greater vertical complexity of the denser woody Savanna vegetation types likely created niches more favourable to certain invaders, including shade-tolerant or structurally opportunistic species such as *Lantana camara* and *Arundo donax*, which is consistent with Richardson *et al.* (2007) and Gaertner *et al.* (2012), who showed that vertical structure can facilitate invasion. Marikana Thornveld, by contrast, was generally less heavily invaded, with its more strongly herbaceous cover likely constraining establishment opportunities for some shrubs and climbers through competition at ground level, in line with Milton and Dean (1998). This lower invasion level was not absolute, however. Localised hotspots in Marikana Thornveld natural areas, including those invaded by *Tithonia rotundifolia* or *Ipomoea indica*, showed that propagule pressure and disturbance can override the broader vegetation-related pattern, a finding echoed by Alston and Richardson (2006) and Nelufule *et al.* (2024) in urban-wildland systems.

One of the more unexpected findings was the near absence of AIPs in road reserves within the denser woody Savanna vegetation types, despite high densities in adjacent natural plots. This suggests that routine road reserve maintenance, including mowing and clearing, has been highly effective in limiting establishment, a conclusion supported by Kalwij *et al.* (2008). However, the persistence of invasions directly beyond the road verge indicates that control within managed corridors is not sufficient unless neighbouring land is also considered. This spillover effect reinforces the need for cross-boundary coordination, as noted by van Wilgen *et al.* (2012). In Marikana Thornveld, differences between road reserves and natural sites were less pronounced, suggesting that vegetation structure may already limit invasion to some extent, although hotspot plots show that these constraints can still be overcome under high propagule pressure and local disturbance.

The role of soil type proved less important than initially expected. Although some high-invasion plots were associated with particular soils, statistical tests showed no consistent link between edaphic conditions and AIP presence at the corridor scale. Instead, disturbance and land use emerged as stronger drivers, supporting the conclusions of Holmes *et al.* (2020) that invasion outcomes in South Africa are shaped more by disturbance history than by inherent site properties.

Vegetation quality provided another lens through which to understand invasion risk. Degraded sites were more likely to coincide with AIP hotspots in years such as 2017, 2022, and 2024, showing that loss of ecological integrity creates opportunities for invasive dominance. This pattern is consistent with Holmes *et al.* (2020), who found that disturbance and degradation facilitate invasions in fynbos systems, and with Afonso *et al.* (2020), who emphasised that degraded urban and peri-urban sites sustain unstable assemblages.

While these findings provide important insights, several limitations must be acknowledged. The analysis relied on annual audit data collected from 2011 to 2024, which, while unique in its temporal scope, limited the study to observational trends without direct experimental testing. Soil data were relatively coarse and not always linked to plot-scale vegetation dynamics, reducing the ability to detect edaphic drivers. Vegetation quality assessments captured broad disturbance patterns but may not fully reflect ecological processes such as soil biota or seedbank dynamics.

These limitations highlight the need for further research. Future studies should investigate the mechanisms behind the resistance of denser woody Savanna vegetation types to road reserves, exploring whether routine maintenance or ecological characteristics provide protection. More detailed analysis of soil–invasion interactions at smaller spatial scales is needed, as well as long-term tracking of species turnover following removal to determine whether secondary invasions replace initial dominants. Additional work on socio-ecological dimensions, such as landowner practices and public perceptions of AIPs, would also strengthen the understanding of invasion persistence in urban and peri-urban settings.

Based on the findings, several recommendations emerge for management along the Bakwena N4. Control in rural Bushveld areas has been effective but should be coupled with ecological restoration to prevent secondary invasions. In peri-urban zones, management must expand beyond road reserves to include adjacent private and communal lands that act as reinvasion sources. Routine audits should incorporate vegetation quality metrics to identify emerging hotspots earlier. Finally, closer collaboration with municipalities and landowners is necessary to reduce ornamental escapes and disturbance-driven reinvasion near urban centres.

### 7.2 Conclusion

This dissertation set out to evaluate the effectiveness of a long-term AIP removal programme along the Bakwena N4 Toll Route and to assess how vegetation type characteristics, land use, disturbance, soil type, and vegetation quality shape invasion outcomes. Using a unique dataset collected annually from 2011 to 2024, the study provided a rare opportunity to analyse temporal and spatial trends across an entire transport corridor.

The aim of the study was achieved. Analyses confirmed that the removal programme has curbed the worst impacts of AIPs and reduced densities in several settings, especially in rural Bushveld areas, but that it has not eradicated invasions. Urban areas maintained low but persistent populations due to continual propagule pressure, while peri-urban landscapes proved the most resistant to long-term suppression, acting as reinvasion sources for surrounding zones. These results show that the programme can be considered a qualified success: it has delivered measurable ecological benefits and slowed spread, but its long-term effectiveness depends on further refinement and sustained effort.

The objectives were also met. Comparisons between Marikana Thornveld and the denser woody Savanna vegetation types showed that the denser woody Savanna vegetation types, with their greater structural complexity and history of disturbance, were more prone to invasion than Marikana Thornveld, which generally showed lower invasion levels. Analyses of land use confirmed significant contrasts, with peri-urban areas remaining consistently unstable, urban zones subject to reinvasion, and rural areas showing the strongest progress under control. The evaluation of vegetation quality further showed that degraded sites coincided with invasion hotspots in specific years, reinforcing the role of ecological condition in shaping invasion dynamics. Soil type, by contrast, was not a significant driver, highlighting disturbance and land use as the more decisive factors.

Taken together, the findings demonstrate that AIP dynamics along the N4 are structured by the interaction of ecological context and human influence. The long-term dataset confirmed that control interventions can be effective in slowing the spread and reducing densities, but it also showed that outcomes are uneven and site-specific. These insights reinforce the importance of context-sensitive and spatially explicit management. Long-term success will depend on maintaining intensive control in rural Bushveld, strengthening rapid response capacity in urban areas, and developing integrated approaches for peri-urban landscapes where structural drivers of invasion remain strong.

Beyond its case study, this research contributes to broader invasion ecology and management in South Africa. It demonstrates the value of long-term monitoring for understanding invasion trajectories, provides evidence that control programmes can succeed under certain conditions, and highlights the persistent risks of reinvasion in transitional and urban contexts. By integrating ecological patterns with land use and management practices, the study offers a framework for adaptive, landscape-scale strategies that can be applied not only along the Bakwena N4 but also across other transport corridors and disturbed ecosystems facing similar invasion pressures.

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# APPENDIX A

## FULL SITE-LEVEL DATASET OF ALIEN INVASIVE PLANT RECORDS ACROSS ALL QUADRANTS (2011–2024)

Please see Appendix A for the full site-level table of alien invasive plant records across all quadrants and survey years.

## CHAPTER 5 SUPPLEMENTARY MATERIAL

Table S5.1 Regression results for log-transformed alien invasive plant (AIP) counts per km across urban zones (2011–2024).

| Area | Year | Counts per sq/km | Log Transformed |
|---|---|---|---|
| Urban | 2011 | 1.6 | 0.470003629 |
| Urban | 2012 | 0.6 | -0.510825624 |
| Urban | 2013 | 0.0001 | -9.210340372 |
| Urban | 2014 | 0.1 | -2.302585093 |
| Urban | 2015 | 0.1 | -2.302585093 |
| Urban | 2016 | 0.3 | -1.203972804 |
| Urban | 2017 | 0.5 | -0.693147181 |
| Urban | 2018 | 0.8 | -0.223143551 |
| Urban | 2019 | 0.6 | -0.510825624 |
| Urban | 2020 | 0.7 | -0.356674944 |
| Urban | 2021 | 1 | 0 |
| Urban | 2022 | 1.5 | 0.405465108 |
| Urban | 2023 | 1.3 | 0.262364264 |
| Urban | 2024 | 0.8 | -0.223143551 |

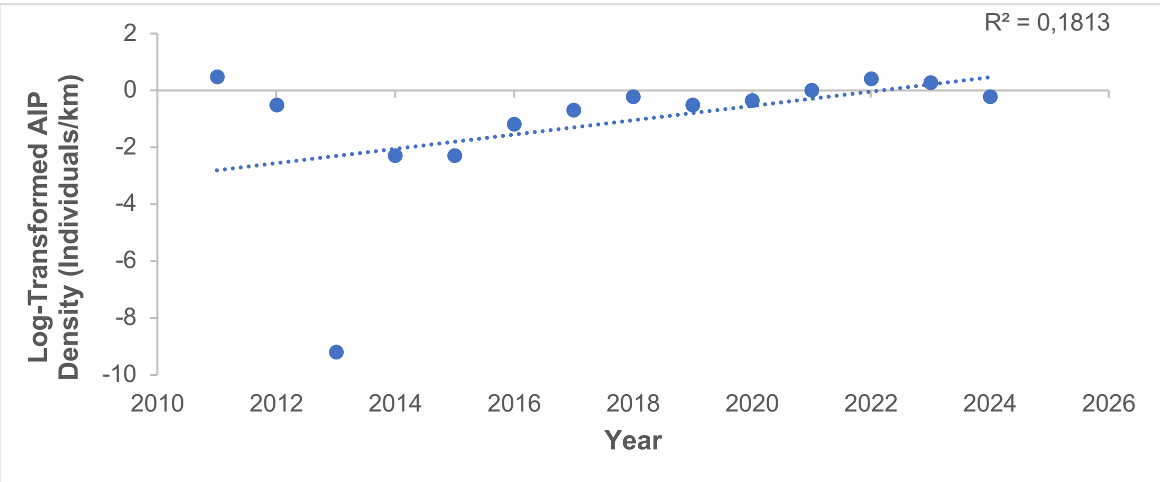


Figure S5.1 Linear regression model of log-transformed alien invasive plant (AIP) counts per km in urban areas (2011–2024).

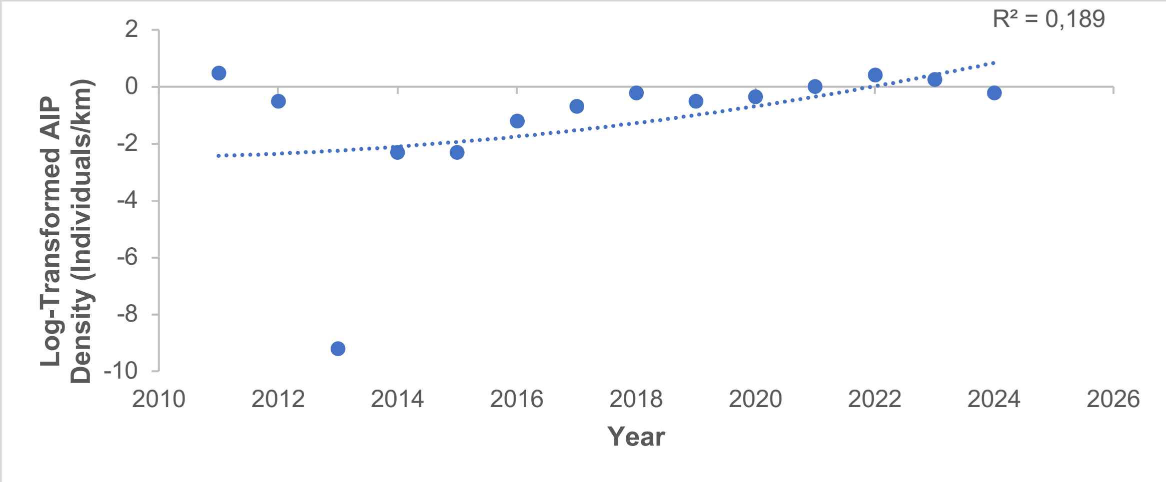


Figure S5.2. Polynomial regression model of log-transformed alien invasive plant (AIP) counts per km in urban areas (2011–2024).

Table S5.2 Regression results for log-transformed alien invasive plant (AIP) counts per km across peri-urban zones (2011–2024).

| Area | Year | Counts per sq/km | Log Transformed |
|---|---|---|---|
| Peri-Urban | 2011 | 2.1 | 0.741937345 |
| Peri-Urban | 2012 | 0.8 | -0.223143551 |
| Peri-Urban | 2013 | 0.3 | -1.203972804 |
| Peri-Urban | 2014 | 0.2 | -1.609437912 |
| Peri-Urban | 2015 | 0.0001 | -9.210340372 |
| Peri-Urban | 2016 | 0.4 | -0.916290732 |
| Peri-Urban | 2017 | 1 | 0 |
| Peri-Urban | 2018 | 0.9 | -0.105360516 |
| Peri-Urban | 2019 | 0.7 | -0.356674944 |
| Peri-Urban | 2020 | 0.5 | -0.693147181 |
| Peri-Urban | 2021 | 1 | 0 |
| Peri-Urban | 2022 | 0.9 | -0.105360516 |
| Peri-Urban | 2023 | 0.9 | -0.105360516 |
| Peri-Urban | 2024 | 0.9 | -0.105360516 |

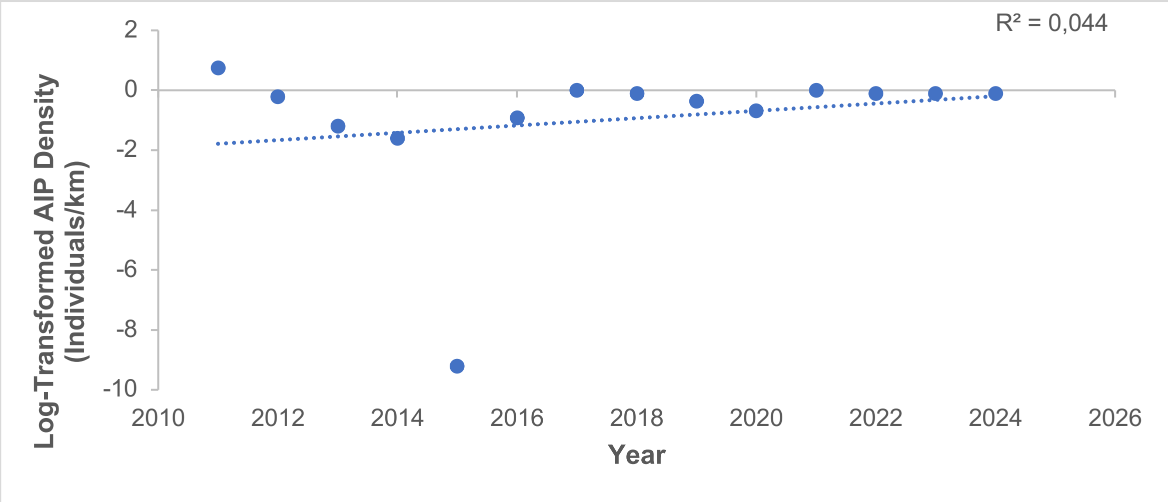


Figure S5.3 Linear regression model of log-transformed alien invasive plant (AIP) counts per km in peri-urban areas (2011–2024).

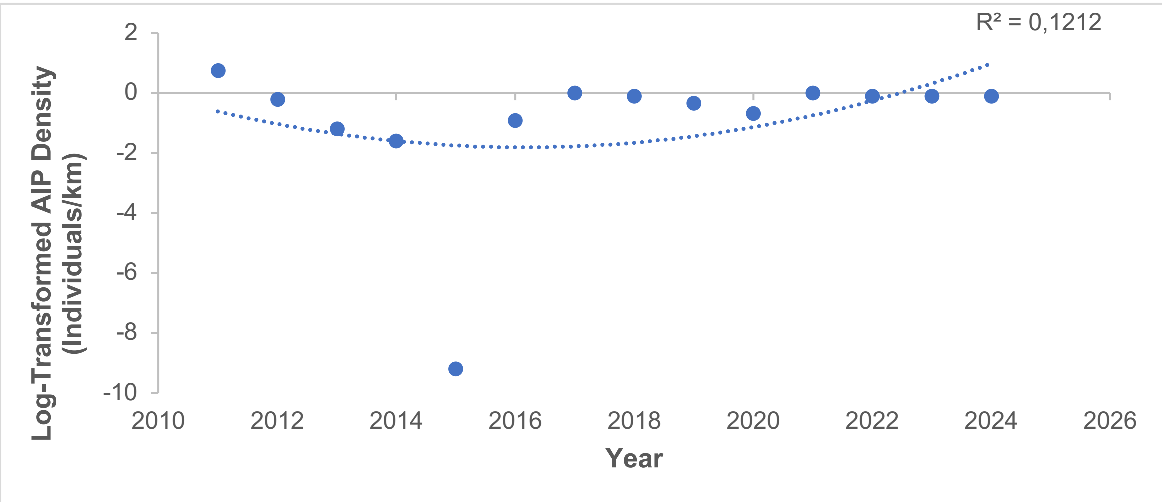


Figure S5.4 Polynomial regression model of log-transformed alien invasive plant (AIP) counts per km in peri-urban areas (2011–2024).

Table S5.3 Regression results for log-transformed alien invasive plant (AIP) counts per km across rural zones (2011–2024).

| **Area** | **Year** | **Counts per sq/km** | **Log Transformed** |
|---|---|---|---|
| Rural | 2011 | 1.6 | 0.470003629 |
| Rural | 2012 | 0.7 | -0.356674944 |
| Rural | 2013 | 0.6 | -0.510825624 |
| Rural | 2014 | 0.4 | -0.916290732 |
| Rural | 2015 | 0.2 | -1.609437912 |
| Rural | 2016 | 0.3 | -1.203972804 |
| Rural | 2017 | 0.4 | -0.916290732 |
| Rural | 2018 | 0.2 | -1.609437912 |
| Rural | 2019 | 0.2 | -1.609437912 |
| Rural | 2020 | 0.4 | -0.916290732 |
| Rural | 2021 | 0.5 | -0.693147181 |
| Rural | 2022 | 0.4 | -0.916290732 |
| Rural | 2023 | 0.3 | -1.203972804 |
| Rural | 2024 | 0.3 | -1.203972804 |

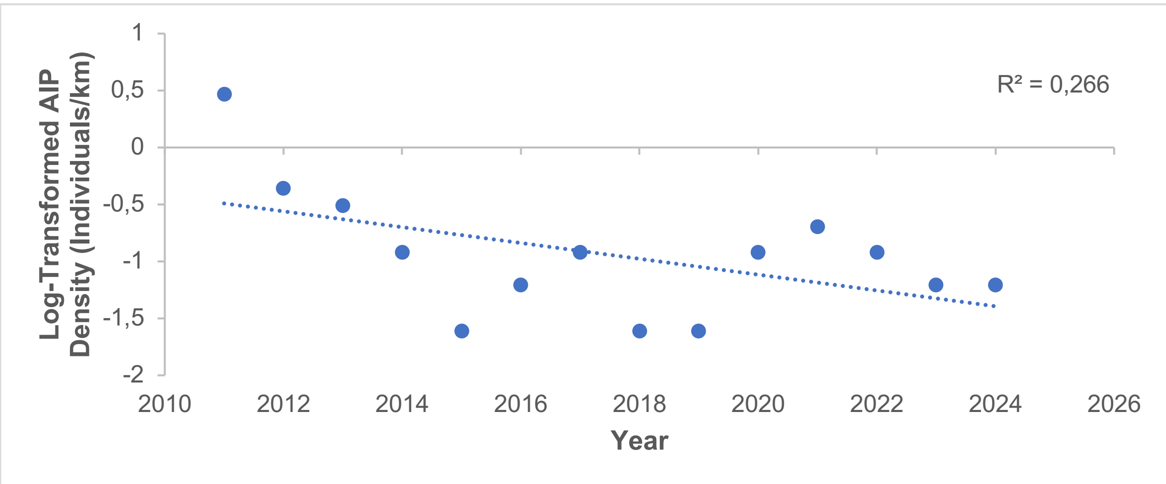


Figure S5.5 Linear regression model of log-transformed alien invasive plant (AIP) counts per km in rural areas (2011–2024).

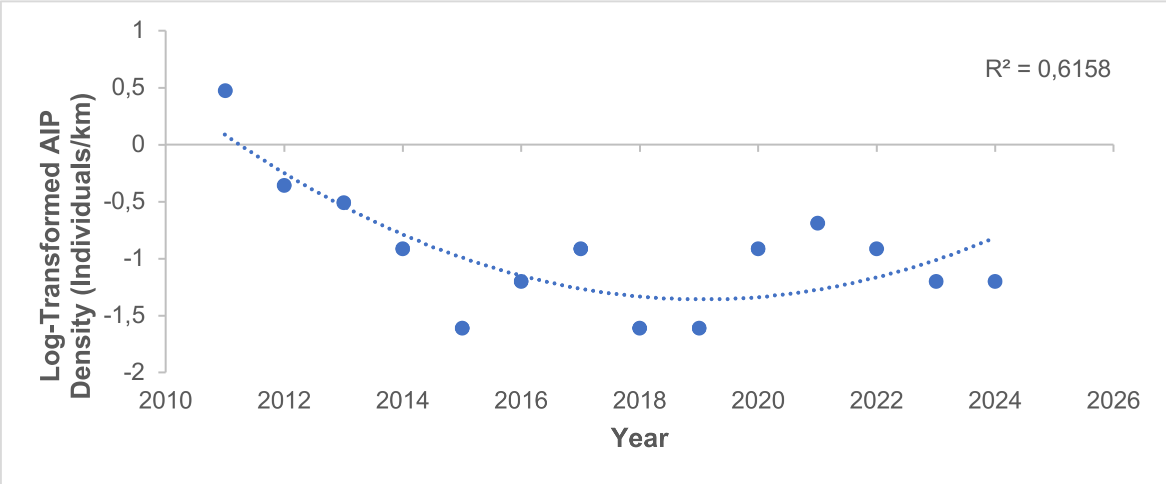


Figure S5.6 Polynomial regression model of log-transformed alien invasive plant (AIP) counts per km in rural areas (2011–2024).

Table S5.4 Regression results for Pielou's Evenness Index across urban, peri-urban, and rural zones (2011–2024).

| Year | Urban Pielou Index | Peri-Urban Pielou Index | Rural Pielou Index |
|---|---|---|---|
| 2011 | 0.875 | 0.886 | 0.802 |
| 2012 | 0.98 | 0.9 | 0.793 |
| 2013 | 0 | 0.875 | 0.863 |
| 2014 | 0.722 | 0.853 | 0.814 |
| 2015 | 0 | 0 | 0.852 |
| 2016 | 0.923 | 0.926 | 0.853 |
| 2017 | 0.858 | 0.914 | 0.867 |
| 2018 | 0.865 | 0.952 | 0.933 |
| 2019 | 0.843 | 0.893 | 0.983 |
| 2020 | 0.846 | 0.818 | 0.778 |
| 2021 | 0.898 | 0.906 | 0.894 |
| 2022 | 0.797 | 0.945 | 0.933 |
| 2023 | 0.82 | 0.931 | 0.947 |
| 2024 | 0.929 | 0.896 | 0.835 |

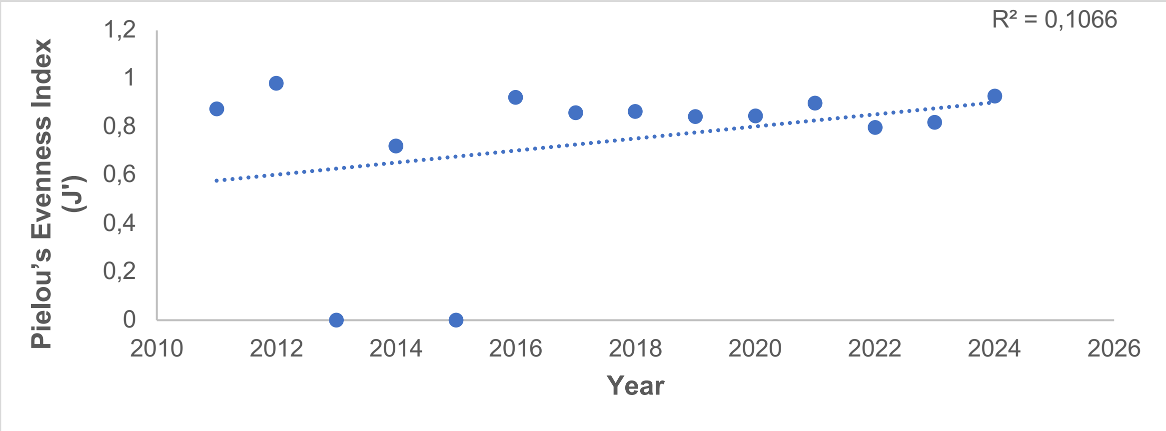


Figure S5.7 Linear regression model of Pielou's Evenness Index in urban areas (2011–2024).

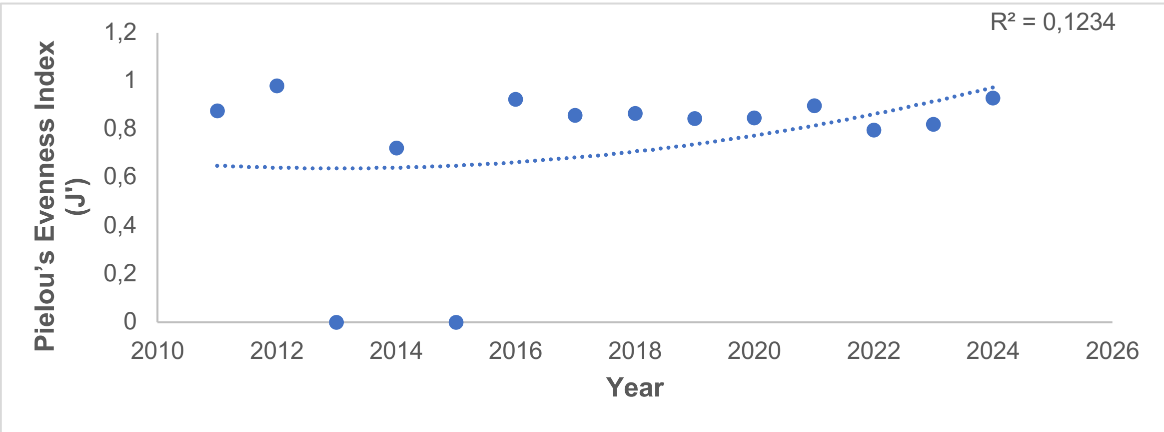


Figure S5.8 Polynomial regression model of Pielou's Evenness Index in urban areas (2011–2024).

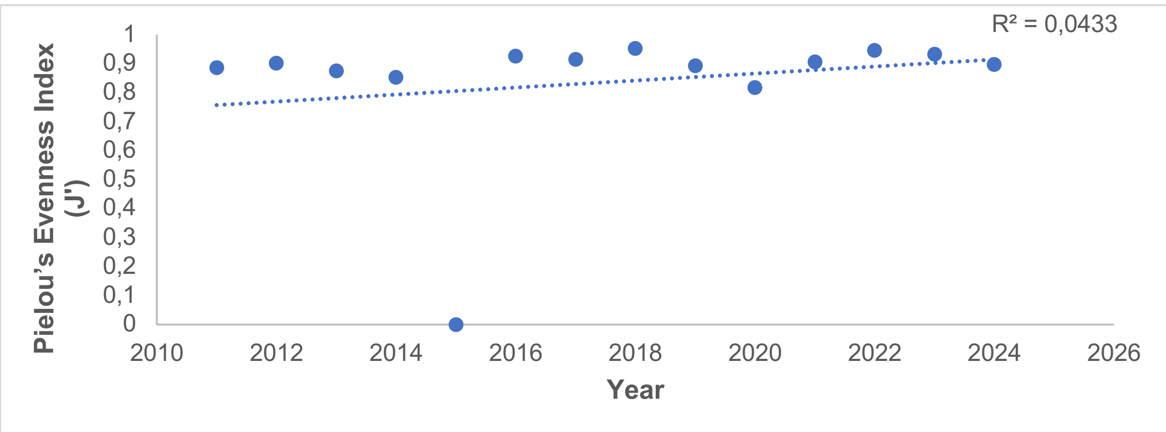


Figure S5.9 Linear regression model of Pielou's Evenness Index in peri-urban areas (2011–2024).

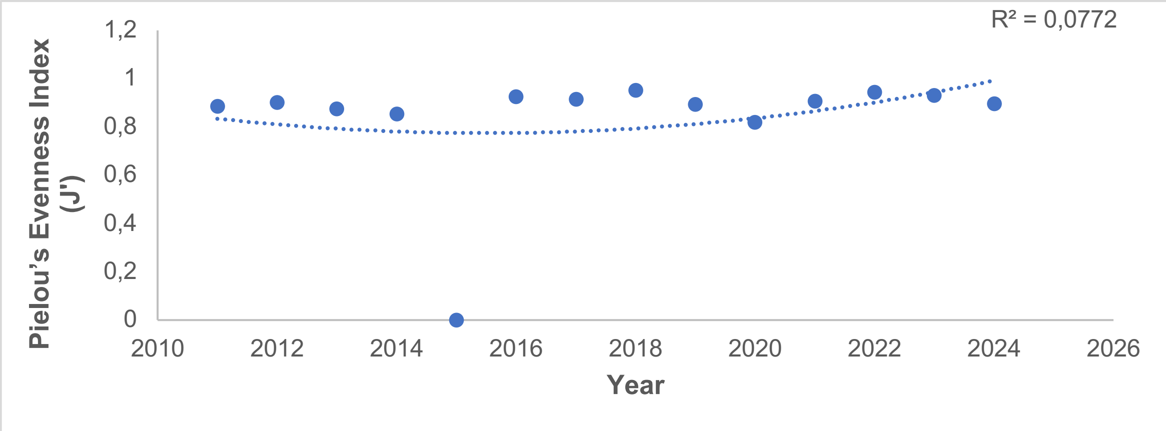


Figure S5.10 Polynomial regression model of Pielou's Evenness Index in peri-urban areas (2011–2024).

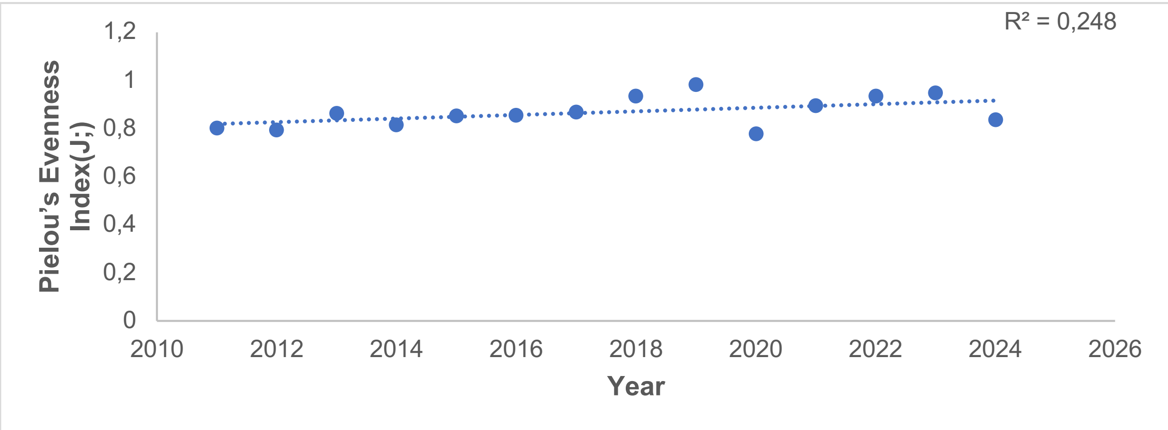


Figure S5.11 Linear regression model of Pielou's Evenness Index in rural areas (2011–2024).

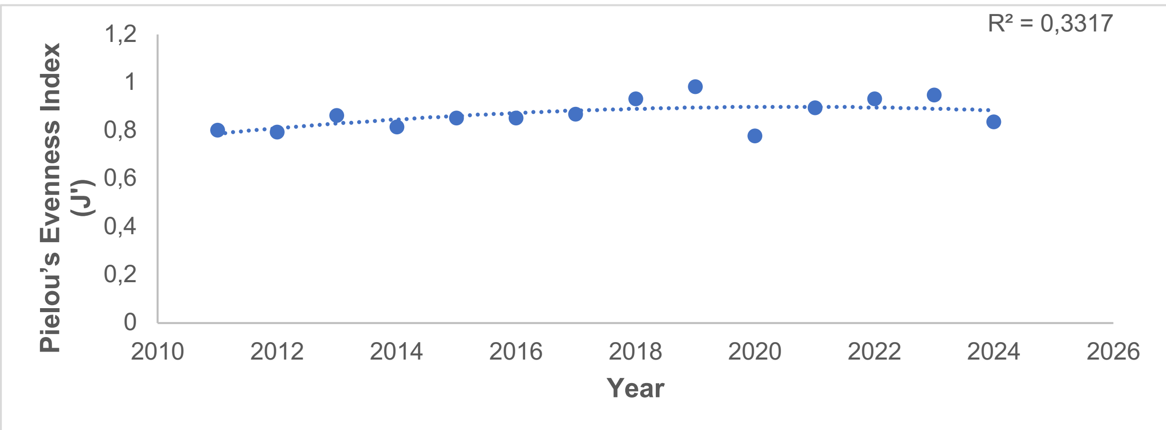


Figure S5.12 Polynomial regression model of Pielou's Evenness Index in rural areas (2011–2024).

Table S5.5 Counts of AIPs found in each quadrant from 2011-2024

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13-1 | 19 | 12 | 8 | 5 | 0 | 2 | 2 | 9 | 5 | 0 | 3 | 5 | 0 | 0 |
| 13-21 | 10 | 0 | 0 | 0 | 0 | 4 | 2 | 6 | 2 | 4 | 0 | 0 | 0 | 0 |
| 13-25 | 14 | 0 | 0 | 0 | 0 | 0 | 8 | 5 | 14 | 7 | 18 | 13 | 6 | 8 |
| 13-29 | 13 | 3 | 0 | 0 | 0 | 2 | 5 | 0 | 0 | 0 | 0 | 3 | 6 | 6 |
| 13-41 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 |
| 13-5 | 4 | 4 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 |
| 14-1 | 1 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-21 | 3 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-25 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 5 | 0 | 0 | 0 | 0 |
| 14-29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-45 | 18 | 9 | 3 | 6 | 4 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-57 | 1 | 4 | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-61 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 |
| 15-1 | 5 | 4 | 2 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-17 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 3 | 3 |
| 15-21 | 8 | 4 | 2 | 2 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-25 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 |
| 15-29 | 4 | 1 | 3 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 4 |
| 15-45 | 23 | 2 | 1 | 0 | 5 | 6 | 6 | 7 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-49 | 20 | 13 | 14 | 6 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 1 | 1 | 4 |
| 15-5 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 |
| 15-9 | 23 | 11 | 10 | 7 | 0 | 3 | 0 | 1 | 4 | 4 | 20 | 7 | 2 | 8 |
| 16-1 | 4 | 4 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 6 | 2 | 1 | 1 |
| 16-13 | 17 | 1 | 4 | 2 | 1 | 4 | 4 | 0 | 5 | 0 | 4 | 4 | 4 | 0 |
| 16-17 | 4 | 6 | 3 | 3 | 2 | 3 | 8 | 0 | 0 | 0 | 0 | 9 | 5 | |
| 16-21 | 13 | 10 | 6 | 4 | 4 | 7 | 24 | 4 | 7 | 21 | 12 | 3 | 6 | 5 |
| 16-29 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 8 | 0 | 7 | 4 | 4 |
| 16-41 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 |
| 16-5 | 4 | 3 | 3 | 2 | 1 | 0 | 1 | 0 | 2 | 6 | 0 | 0 | 0 | 0 |
| 16-25 | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-13 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 2 | 0 | 7 | 3 | 10 |
| 14-53 | 2 | 6 | 2 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-13 | 14 | 1 | 2 | 4 | 1 | 0 | 6 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-9 | 1 | 2 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 |
| 13-17 | 26 | 9 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 9 | 0 | 0 | 0 |
| 13-13 | 10 | 4 | 1 | 0 | 0 | 5 | 6 | 3 | 2 | 3 | 0 | 2 | 6 | 4 |
| 13-9 | 11 | 2 | 0 | 0 | 0 | 0 | 11 | 5 | 9 | 5 | 3 | 13 | 9 | 9 |
| 12-57 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 3 |
| 12-53 | 5 | 3 | 0 | 0 | 0 | 5 | 9 | 9 | 0 | 0 | 7 | 2 | 2 | 2 |
| 12-49 | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 3 | 9 | 4 | 4 | 4 |
| 12-45 | 7 | 5 | 3 | 3 | 0 | 8 | 15 | 12 | 0 | 0 | 3 | 5 | 5 | 5 |
| 12-41 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 4 | 0 | 7 | 11 | 11 |
| 10-21 | 1 | 2 | 0 | 0 | 0 | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10-17 | 7 | 2 | 0 | 0 | 0 | 4 | 2 | 7 | 9 | 4 | 0 | 9 | 15 | 4 |
| 10-13 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 2 | 8 | 2 |
| 9-9 | 13 | 3 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 17 | 7 | 9 | 9 |
| 9-25 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 0 | 3 | 4 | 1 | 1 | 1 |
| 9-17 | 5 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 |
| 9-13 | 18 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 |
| 9-5 | 5 | 1 | 4 | 5 | 4 | 4 | 8 | 0 | 8 | 2 | 6 | 3 | 3 | 3 |
| 9-1 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 5 | 4 | 4 | 4 |
| 14-41 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10-5 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10-1 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | | 8 | 0 | 0 | 0 | 0 |
| 10-9 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 2 | 0 | 0 | 6 | 5 | 0 |
| 9-21 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 29 | 6 | 0 |
| 12-17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 6 | 8 | 6 | 0 | 0 | 0 |
| 16-9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 3 | 4 | 0 | 0 | 0 |
| 14-5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14-49 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 15-33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15-41 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 16-33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 16-37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-13 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-21 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-25 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12-37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 9-29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 13-33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 13-37 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | 380 | 152 | 86 | 62 | 22 | 71 | 132 | 114 | 89 | 106 | 161 | 163 | 137 | 126 |

Table S5.6 Vegetation quality status rankings for each quadrant along the Bakwena N4 Toll Route (2011–2024)

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13-1 | 4 | 4 | 4 | 4 | 4 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 |
| 13-21 | 2 | 2 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 | 2 | 3 | 3 |
| 13-25 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 1 | 1 | 1 | 3 | 2 | 1 |
| 13-29 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 3 |
| 13-41 | 4 | 4 | 4 | 4 | 4 | 1 | 1 | 2 | 3 | 3 | 3 | 3 | 3 | 3 |
| 13-5 | 3 | 3 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 14-1 | 3 | 3 | 3 | 3 | 3 | 1 | 3 | 1 | 2 | 2 | 2 | 3 | 3 | 3 |
| 14-17 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 3 | 3 | 2 | 2 | 2 |
| 14-21 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 |
| 14-25 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 3 | 3 | 3 |
| 14-29 | 2 | 2 | 3 | 3 | 3 | 2 | 2 | 1 | 1 | 2 | 1 | 2 | 2 | 2 |
| 14-45 | 3 | 3 | 2 | 2 | 2 | 2 | 4 | 3 | 2 | 3 | 3 | 3 | 3 | 3 |
| 14-57 | 4 | 4 | 4 | 4 | 4 | 3 | 4 | 3 | 1 | 1 | 1 | 1 | 2 | 3 |
| 14-61 | 4 | 4 | 4 | 4 | 4 | 3 | 4 | 2 | 4 | 1 | 1 | 1 | 2 | 3 |
| 15-1 | 1 | 2 | 1 | 2 | 2 | 2 | 2 | 2 | 4 | 1 | 1 | 1 | 1 | 2 |
| 15-17 | 3 | 3 | 3 | 3 | 3 | 4 | 2 | 2 | 4 | 3 | 3 | 1 | 1 | 3 |
| 15-21 | 3 | 3 | 3 | 3 | 3 | 4 | 2 | 2 | 2 | 3 | 1 | 1 | 2 | 3 |
| 15-25 | 2 | 2 | 2 | 2 | 2 | 4 | 2 | 3 | 2 | 2 | 1 | 2 | 2 | 1 |
| 15-29 | 2 | 2 | 2 | 2 | 2 | 4 | 2 | 3 | 2 | 3 | 3 | 3 | 3 | 3 |
| 15-45 | 2 | 2 | 2 | 2 | 2 | 4 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| 15-49 | 2 | 2 | 2 | 2 | 2 | 4 | 2 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 15-5 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 4 | 1 | 2 | 2 | 2 | 3 |
| 15-9 | 1 | 2 | 1 | 2 | 1 | 4 | 2 | 2 | 4 | 1 | 2 | 2 | 2 | 2 |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 16-1 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| 16-13 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 2 | 1 | 3 | 3 | 3 | 3 |
| 16-17 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 3 | 3 | 3 | 2 | 2 | 2 |
| 16-21 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 2 | 3 | 3 | 1 | 1 | 2 |
| 16-29 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 3 | 1 | 2 | 2 | 2 | 2 | 2 |
| 16-41 | 4 | 4 | 4 | 4 | 4 | 2 | 2 | 2 | 2 | 2 | 1 | 1 | 2 | 2 |
| 16-5 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 |
| 16-25 | 4 | 4 | 4 | 4 | 4 | 1 | 2 | 2 | 3 | 3 | 3 | 3 | 3 | 3 |
| 15-13 | 3 | 3 | 2 | 2 | 2 | 4 | 2 | 2 | 4 | 1 | 2 | 3 | 3 | 3 |
| 14-53 | 3 | 2 | 2 | 2 | 2 | 2 | 4 | 3 | 2 | 2 | 1 | 1 | 1 | 1 |
| 14-13 | 2 | 2 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| 14-9 | 3 | 3 | 3 | 3 | 3 | 2 | 3 | 2 | 2 | 2 | 2 | 3 | 3 | 3 |
| 13-17 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 2 | 2 | 3 | 3 | 3 | 2 | 2 |
| 13-13 | 2 | 2 | 2 | 2 | 2 | 3 | 1 | 2 | 2 | 2 | 3 | 2 | 1 | 1 |
| 13-9 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 2 | 3 | 3 | 2 | 2 | 2 | 2 |
| 12-57 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 2 | 3 | 2 | 3 | 3 | 3 |
| 12-53 | 4 | 4 | 4 | 4 | 4 | 2 | 2 | 2 | 2 | 2 | 3 | 2 | 2 | 2 |
| 12-49 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 2 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-45 | 4 | 4 | 4 | 4 | 4 | 2 | 2 | 2 | 2 | 3 | 2 | 3 | 2 | 2 |
| 12-41 | 4 | 4 | 4 | 4 | 4 | 2 | 1 | 2 | 2 | 2 | 2 | 3 | 3 | 3 |
| 10-21 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 2 | 3 | 4 | 4 | 2 | 2 | 2 |
| 10-17 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 2 | 3 | 4 | 4 | 2 | 3 | 3 |
| 10-13 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 4 | 4 | 3 | 3 | 3 |
| 9-9 | 2 | 3 | 2 | 2 | 3 | 2 | 3 | 3 | 2 | 2 | 3 | 3 | 3 | 3 |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 9-25 | 3 | 3 | 2 | 2 | 3 | 2 | 2 | 3 | 3 | 4 | 4 | 2 | 3 | 3 |
| 9-17 | 2 | 2 | 2 | 2 | 3 | 2 | 2 | 2 | 2 | 1 | 2 | 2 | 3 | 3 |
| 9-13 | 2 | 2 | 2 | 2 | 3 | 2 | 2 | 2 | 2 | 2 | 3 | 3 | 2 | 2 |
| 9-5 | 2 | 2 | 2 | 2 | 3 | 2 | 3 | 3 | 2 | 2 | 2 | 1 | 2 | 3 |
| 9-1 | 2 | 2 | 2 | 2 | 3 | 2 | 2 | 3 | 3 | 2 | 2 | 3 | 3 | 3 |
| 14-41 | 3 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| 10-5 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 4 | 4 | 3 | 3 | 3 |
| 10-1 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 3 | 4 | 4 | 2 | 2 | 2 |
| 10-9 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 4 | 4 | 3 | 2 | 2 |
| 9-21 | 3 | 3 | 3 | 2 | 3 | 2 | 2 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-17 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 16-9 | 3 | 3 | 3 | 3 | 3 | 3 | 2 | 2 | 3 | 1 | 3 | 3 | 3 | 3 |
| 14-5 | 3 | 3 | 3 | 3 | 3 | 2 | 3 | 2 | 2 | 3 | 3 | 3 | 3 | 3 |
| 14-33 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 1 | 1 | 1 | 2 | 2 | 2 | 1 |
| 14-37 | 3 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| 14-49 | 3 | 3 | 3 | 3 | 3 | 2 | 4 | 3 | 1 | 1 | 1 | 3 | 3 | 3 |
| 15-33 | 3 | 3 | 3 | 2 | 2 | 4 | 2 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 15-37 | 3 | 3 | 3 | 3 | 3 | 4 | 2 | 3 | 2 | 2 | 1 | 2 | 2 | 2 |
| 15-41 | 2 | 2 | 2 | 2 | 2 | 4 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 |
| 16-33 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 |
| 16-37 | 4 | 4 | 4 | 4 | 4 | 3 | 2 | 3 | 2 | 2 | 3 | 2 | 3 | 3 |
| 12-1 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-5 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-9 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 12-13 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-21 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-25 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-29 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-33 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 |
| 12-37 | 4 | 4 | 4 | 4 | 4 | 3 | 3 | 3 | 2 | 2 | 2 | 2 | 2 | 3 |
| 9-29 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 3 | 4 | 3 | 2 | 2 |
| 13-33 | 2 | 2 | 2 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 | 3 | 2 | 2 |
| 13-37 | 2 | 1 | 2 | 2 | 2 | 3 | 2 | 2 | 2 | 2 | 3 | 3 | 3 | 3 |

Table S5.7 Classification of quadrants into good and bad vegetation condition categories along the Bakwena N4 Toll Route

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 13-1 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Good | Good | Good |
| 13-21 | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good | Good | Good | Good | Bad | Good | Good |
| 13-25 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad |
| 13-29 | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good |
| 13-41 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Good | Good | Good | Good | Good | Good |
| 13-5 | Good | Good | Bad | Bad | Bad | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 14-1 | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Bad | Bad | Good | Good | Good |
| 14-17 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Bad | Bad | Bad |
| 14-21 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Good | Good | Good |
| 14-25 | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good | Good |
| 14-29 | Bad | Bad | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 14-45 | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Bad | Good | Good | Good | Good | Good |
| 14-57 | Good | Good | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Good |
| 14-61 | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Bad | Bad | Good |
| 15-1 | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Bad |
| 15-17 | Good | Good | Good | Good | Good | Good | Bad | Bad | Good | Good | Good | Bad | Bad | Good |
| 15-21 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Good | Bad | Bad | Bad | Good |
| 15-25 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Good | Bad | Bad | Bad | Bad | Bad | Bad |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 15-29 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Good | Bad | Good | Good | Good | Good | Good |
| 15-45 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 15-49 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Good | Good | Good | Good | Good | Good | Good |
| 15-5 | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Good |
| 15-9 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Good | Bad | Bad | Bad | Bad | Bad |
| 16-1 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 16-13 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Good | Good |
| 16-17 | Good | Good | Good | Good | Good | Good | Bad | Bad | Good | Good | Good | Bad | Bad | Bad |
| 16-21 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Good | Good | Bad | Bad | Bad |
| 16-29 | Good | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Bad | Bad | Bad | Bad |
| 16-41 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 16-5 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Good | Good | Good | Good | Good |
| 16-25 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Good | Good | Good | Good | Good | Good |
| 15-13 | Good | Good | Bad | Bad | Bad | Good | Bad | Bad | Good | Bad | Bad | Good | Good | Good |
| 14-53 | Good | Bad | Bad | Bad | Bad | Bad | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad |
| 14-13 | Bad | Bad | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 14-9 | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Bad | Bad | Good | Good | Good |
| 13-17 | Good | Good | Good | Good | Good | Good | Good | Bad | Bad | Good | Good | Good | Bad | Bad |
| 13-13 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad |
| 13-9 | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Good | Bad | Bad | Bad | Bad |
| 12-57 | Good | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Bad | Good | Good | Good |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 12-53 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad |
| 12-49 | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Good | Good | Good | Good | Good |
| 12-45 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Bad | Good | Bad | Bad |
| 12-41 | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good | Good |
| 10-21 | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Good | Good | Bad | Bad | Bad |
| 10-17 | Good | Good | Good | Good | Good | Good | Good | Bad | Good | Good | Good | Bad | Good | Good |
| 10-13 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 9-9 | Bad | Good | Bad | Bad | Good | Bad | Good | Good | Bad | Bad | Good | Good | Good | Good |
| 9-25 | Good | Good | Bad | Bad | Good | Bad | Bad | Good | Good | Good | Good | Bad | Good | Good |
| 9-17 | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good |
| 9-13 | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Bad | Good | Good | Bad | Bad |
| 9-5 | Bad | Bad | Bad | Bad | Good | Bad | Good | Good | Bad | Bad | Bad | Bad | Bad | Good |
| 9-1 | Bad | Bad | Bad | Bad | Good | Bad | Bad | Good | Good | Bad | Bad | Good | Good | Good |
| 14-41 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 10-5 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 10-1 | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good | Good | Bad | Bad | Bad |
| 10-9 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Bad | Bad |
| 9-21 | Good | Good | Good | Bad | Good | Bad | Bad | Good | Good | Good | Good | Good | Good | Good |
| 12-17 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 16-9 | Good | Good | Good | Good | Good | Good | Bad | Bad | Good | Bad | Good | Good | Good | Good |
| 14-5 | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Good | Good | Good | Good | Good |

| Quadrant | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 14-33 | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 14-37 | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 14-49 | Good | Good | Good | Good | Good | Bad | Good | Good | Bad | Bad | Bad | Good | Good | Good |
| 15-33 | Good | Good | Good | Bad | Bad | Good | Bad | Good | Good | Good | Good | Good | Good | Good |
| 15-37 | Good | Good | Good | Good | Good | Good | Bad | Good | Bad | Bad | Bad | Bad | Bad | Bad |
| 15-41 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Good | Good | Good | Good | Good |
| 16-33 | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Bad |
| 16-37 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-1 | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Good | Good | Good | Good |
| 12-5 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-9 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-13 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-21 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-25 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-29 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-33 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good |
| 12-37 | Good | Good | Good | Good | Good | Good | Good | Good | Bad | Bad | Bad | Bad | Bad | Good |
| 9-29 | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Good | Bad | Bad |
| 13-33 | Bad | Bad | Bad | Bad | Bad | Bad | Bad | Good | Good | Good | Good | Good | Bad | Bad |
| 13-37 | Bad | Bad | Bad | Bad | Bad | Good | Bad | Bad | Bad | Bad | Good | Good | Good | Good |

Table S5.8 Species-by-year matrix of alien invasive plant abundance (2011–2024)

| **Species** | **2011** | **2012** | **2013** | **2014** | **2015** | **2016** | **2017** | **2018** | **2019** | **2020** | **2021** | **2022** | **2023** | **2024** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Total count** | **380** | **152** | **86** | **62** | **22** | **71** | **132** | **114** | **89** | **106** | **161** | **163** | **137** | **126** |
| *Acacia mearnsii* | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Agave sisalana* | 22 | 26 | 19 | 13 | 8 | 9 | 6 | 6 | 7 | 1 | 0 | 2 | 2 | 0 |
| *Ailanthus altissima* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 2 | 0 |
| *Argemone mexicana* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 7 | 0 |
| *Arundo donax* | 58 | 17 | 14 | 10 | 7 | 11 | 11 | 13 | 12 | 16 | 29 | 25 | 13 | 28 |
| *Campuloclinium macrocephalum* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 21 | 13 | 5 | 25 | 5 | 7 | 16 |
| *Casuarina equisetifolia* | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cereus jamacaru* | 0 | 0 | 0 | 0 | 0 | 1 | 4 | 3 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cestrum elegans* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 4 | 0 | 0 | 4 |
| *Datura ferox* | 5 | 0 | 0 | 0 | 0 | 0 | 13 | 3 | 4 | 26 | 12 | 11 | 6 | 1 |
| *Echium plantagineum* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Eucalyptus globulus* | 26 | 26 | 6 | 5 | 1 | 2 | 4 | 2 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Flaveria bidentis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 23 | 36 | 15 | 6 | 29 |
| *Gleditsia triacanthos* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 3 | 0 |
| *Grevillea robusta* | 0 | 0 | 0 | 0 | 0 | 6 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| Species | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Ipomoea indica* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 0 | 2 | 0 | 0 | 0 | 0 |
| *Lantana camara* | 4 | 0 | 0 | 0 | 0 | 2 | 5 | 6 | 3 | 1 | 8 | 4 | 4 | 5 |
| *Ligustrum lucidum* | 16 | 10 | 8 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 0 |
| *Melia azedarach* | 108 | 43 | 27 | 24 | 2 | 17 | 26 | 22 | 7 | 9 | 7 | 3 | 3 | 7 |
| *Morus alba* | 16 | 7 | 7 | 3 | 2 | 3 | 1 | 2 | 0 | 2 | 0 | 2 | 2 | 0 |
| *Nicotiana glauca* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 |
| *Opuntia ficus-indica* | 5 | 7 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 |
| *Pennisetum setaceum* | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 3 | 11 | 0 | 18 | 4 | 4 | 11 |
| *Populus simonii* | 12 | 3 | 4 | 0 | 0 | 5 | 7 | 3 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ricinus communis* | 12 | 4 | 0 | 0 | 0 | 4 | 6 | 3 | 2 | 3 | 5 | 9 | 13 | 5 |
| *Schinus terebinthifolius* | 0 | 0 | 1 | 1 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Solanum elaeagnifolium* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 6 | 1 | 0 | 0 | 0 | 0 | 0 |
| *Solanum mauritianum* | 9 | 2 | 0 | 0 | 2 | 3 | 2 | 0 | 0 | 0 | 0 | 3 | 3 | 0 |
| *Sorghum halepense* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tecoma stans* | 34 | 0 | 0 | 0 | 0 | 5 | 16 | 7 | 13 | 13 | 4 | 14 | 15 | 6 |
| *Tipuana tipu* | 9 | 6 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tithonia rotundifolia* | 22 | 0 | 0 | 0 | 0 | 0 | 11 | 7 | 5 | 1 | 12 | 33 | 29 | 9 |

| Species | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Xanthium strumarium* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 4 | 5 | 18 | 15 | 5 |

# CHAPTER 6 SUPPLEMENTARY MATERIAL

Table S6.1 Plot-Level Summary of Alien Invasive Plant (AIP) Presence and Environmental Conditions

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot01RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Cenchrus ciliaris* | Plentiful but small cover, <5% | 0.6 | Grass | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | Disturbed by heavy vehicle traffic |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.6 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Chloris gayana* | Covering 25-50% | 0.6 | Grass | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 0.15 | 250.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Hyparrhenia hirta* | Cover very small, <5% | 1.5 | Grass | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Setaria sphacelata* | Covering 50-75% | 1.2 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 416.67 | |
| Plot02RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Cenchrus ciliaris* | Covering 50-75% | 0.5 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 416.67 | Undisturbed |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.45 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Digitaria eriantha* | Covering 5-25% | 1.5 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 100.00 | |
| Plot03RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 75%+ | 0.45 | Grass | Very Gentle <15° | Ferric Luvisols | No | 87.5 | 0.15 | 583.33 | Good condition |
| | Denser woody Savanna vegetation types | Road Reserve | *Senegalia burkei* | Cover very small, <5% | 0.4 | Shrub | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.5 | 5.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Senegalia burkei* | Cover very small, <5% | 0.3 | Shrub | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.5 | 5.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Road Reserve | *Senegalia burkei* | Cover very small, <5% | 0.4 | Shrub | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.5 | 5.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Senegalia burkei* | Cover very small, <5% | 0.35 | Shrub | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.5 | 5.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cymbopogon plurinodis* | Plentiful but small cover, <5% | 1.8 | Grass | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Eragrostis tef* | Covering 5-25% | 0.45 | Grass | Gentle Slope | Ferric Luvisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Urochloa mosambicensis* | Not many individuals, 1-10 | 0.5 | Grass | Gentle Slope | Ferric Luvisols | No | 5 | 0.15 | 33.33 | |
| Plot04RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Chloris gayana* | Covering 25-50% | 0.7 | Grass | Gentle Slope | Ferric Luvisols | No | 37.5 | 0.15 | 250.00 | Intact |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 25-50% | 0.5 | Grass | Gentle Slope | Ferric Luvisols | No | 37.5 | 0.15 | 250.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Senegalia burkei* | Not many individuals, 1-10 | 0.2 | Shrub | Gentle Slope | Ferric Luvisols | No | 5 | 0.5 | 10.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Digitaria eriantha* | Cover very small, <5% | 1.2 | Grass | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cymbopogon plurinodis* | Cover very small, <5% | 1 | Grass | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Setaria sphacelate var. torta* | Plentiful but small cover, <5% | 0.6 | Grass | Gentle Slope | Ferric Luvisols | No | 2.5 | 0.15 | 16.67 | |
| Plot05RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Vachellia karroo* | Not many individuals, 1-10 | 0.7 | Tree | Gentle Slope | Rhodic Nitosols | No | 5 | 10 | 0.50 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Digitaria eriantha* | Covering 25-50% | 1.4 | Grass | Gentle Slope | Rhodic Nitosols | No | 37.5 | 0.15 | 250.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Road Reserve | *Eragrostis tef* | Plentiful but small cover, <5% | 0.6 | Grass | Gentle Slope | Rhodic Nitosols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cenchrus ciliaris* | Covering 25-50% | 0.5 | Grass | Gentle Slope | Rhodic Nitosols | No | 37.5 | 0.15 | 250.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 25-50% | 0.8 | Grass | Gentle Slope | Rhodic Nitosols | No | 37.5 | 0.15 | 250.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cynodon dactylon* | Covering 5-25% | 0.15 | Grass | Gentle Slope | Rhodic Nitosols | No | 15 | 0.02 | 750.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cymbopogon plurinodis* | Cover very small, <5% | 1.5 | Grass | Gentle Slope | Rhodic Nitosols | No | 2.5 | 0.15 | 16.67 | |
| Plot06RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Cymbopogon plurinodis* | Covering 25-50% | 1 | Grass | Very Gentle <15° | Haplic Lixisols | No | 37.5 | 0.15 | 250.00 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Hyparrhenia hirta* | Plentiful but small cover, <5% | 1.6 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cynodon dactylon* | Covering 5-25% | 0.2 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.02 | 750.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Urochloa oligotricha* | Cover very small, <5% | 1 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.4 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Cenchrus ciliaris* | Plentiful but small cover, <5% | 0.7 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| Plot07RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Cenchrus ciliaris* | Covering 75%+ | 0.7 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 87.5 | 0.15 | 583.33 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Plentiful but small cover, <5% | 0.9 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.15 | 16.67 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot08RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Hyparrhenia hirta* | Covering 50-75% | 1.7 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 0.15 | 416.67 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 25-50% | 0.8 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 37.5 | 0.15 | 250.00 | |
| Plot09RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Digitaria eriantha* | Plentiful but small cover, <5% | 1.2 | Grass | Very Gentle <15° | Calcic Lixisols | No | 2.5 | 0.15 | 16.67 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Themeda triandra* | Covering 5-25% | 0.8 | Grass | Very Gentle <15° | Calcic Lixisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Setaria sphacelate var. torta* | Covering 25-50% | 0.6 | Grass | Very Gentle <15° | Calcic Lixisols | No | 37.5 | 0.15 | 250.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.59 | Grass | Very Gentle <15° | Calcic Lixisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Eragrostis rigidior* | Covering 25-50% | 0.75 | Grass | Very Gentle <15° | Calcic Lixisols | No | 37.5 | 0.15 | 250.00 | |
| Plot10RemVegRR | Denser woody Savanna vegetation types | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.6 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 100.00 | Well maintained |
| | Denser woody Savanna vegetation types | Road Reserve | *Eragrostis rigidior* | Covering 75%+ | 0.7 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Setaria sphacelate var. torta* | Covering 5-25% | 0.55 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Themeda triandra* | Plentiful but small cover, <5% | 0.6 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Denser woody Savanna vegetation types | Road Reserve | *Hyparrhenia hirta* | Cover very small, <5% | 1.4 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| Plot01RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Covering 50-75% | 5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 40 | 1.56 | Dense vegetation, |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Not many individuals, 1-10 | 7 | Tree | Very Gentle <15° | Ferric Luvisols | No | 5 | 40 | 1.00 | near dam, undisturbed |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Plentiful but small cover, <5% | 4 | Tree | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Covering 25-50% | 3.5 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 2 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Not many individuals, 1-10 | 3.5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cymbopogon plurinodis* | Not many individuals, 1-10 | 0.45 | Grass | Very Gentle <15° | Ferric Luvisols | No | 5 | 0.15 | 90.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Heteropogon contortus* | Covering 50-75% | 1 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 120.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Not many individuals, 1-10 | 0.17 | Grass | Very Gentle <15° | Ferric Luvisols | No | 5 | 0.02 | 27.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia zeyheri* | Covering 5-25% | 0.3 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.5 | 53.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia burkei* | Covering 5-25% | 0.6 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.5 | 30.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Urochloa oligotricha* | Covering 5-25% | 0.3 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.02 | 40.00 | |
| Plot02RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Combretum molle* | Covering 5-25% | 5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 15 | 40 | 1.00 | Previously cleared near residence |
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Covering 25-50% | 4 | Tree | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 40 | 0.94 | |
| | Denser woody Savanna vegetation types | Natural Area | *Jacaranda mimosifolia* | Covering 5-25% | 5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 15 | 40 | 1.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia caffra* | Covering 5-25% | 4 | Tree | Very Gentle <15° | Ferric Luvisols | No | 15 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Not many individuals, 1-10 | 3 | Tree | Very Gentle <15° | Ferric Luvisols | No | 5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Dombeya rotundifolia* | Not many individuals, 1-10 | 5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Gymnosporia buxifolia* | Not many individuals, 1-10 | 1 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 5 | 0.5 | 10.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Chloris gayana* | Covering 5-25% | 1.23 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Commelina africana* | Cover very small, <5% | 0.06 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cymbopogon plurinodis* | Covering 50-75% | 1.3 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Eragrostis rigidior* | Covering 50-75% | 1.5 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 120.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Euclea crispa* | Cover very small, <5% | 0.17 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 26.67 | |
| | Denser woody Savanna vegetation types | Natural Area | *Lantana camara* | Cover very small, <5% | 0.27 | Shrub | Very Gentle <15° | Ferric Luvisols | Yes Cat 1b | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 25-50% | 0.37 | Grass | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 0.15 | 85.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Pellaea calomelanos* | Covering 5-25% | 0.1 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.5 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Scolopia zeyheri* | Cover very small, <5% | 0.52 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Searsia zeyheri* | Cover very small, <5% | 0.47 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia caffra* | Cover very small, <5% | 0.4 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| Plot03RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Vachellia tortilis* | Plentiful but small cover, <5% | 3 | Tree | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 40 | 1.00 | Dense tree cover |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Covering 25-50% | 8 | Tree | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 40 | 0.94 | |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Plentiful but small cover, <5% | 7 | Tree | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia burkei* | Covering 25-50% | 3.5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 40 | 0.94 | |
| | Denser woody Savanna vegetation types | Natural Area | *Grewia flava* | Cover very small, <5% | 1 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 5.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Plentiful but small cover, <5% | 1.7 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 2 | 30.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Combretum molle* | Cover very small, <5% | 0.29 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cymbopogon plurinodis* | Covering 5-25% | 1 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Eragrostis tef* | Covering 5-25% | 0.56 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 70.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Euclea crispa* | Cover very small, <5% | 0.43 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 5-25% | 0.65 | Grass | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.15 | 55.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia burkei* | Covering 25-50% | 0.73 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 0.5 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia mellifera* | Cover very small, <5% | 0.15 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 20.00 | |
| Plot04RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Cover very small, <5% | 3 | Tree | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 40 | 1.00 | Open vegetation, past clearing |
| | Denser woody Savanna vegetation types | Natural Area | *Grewia flava* | Not many individuals, 1-10 | 2 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 5 | 2 | 2.50 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Covering 50-75% | 5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 40 | 1.56 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia caffra* | Covering 5-25% | 9 | Tree | Very Gentle <15° | Ferric Luvisols | No | 15 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Gymnosporia buxifolia* | Plentiful but small cover, <5% | 3 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 2 | 1.25 | |
| | Denser woody Savanna vegetation types | Natural Area | *Aloe maculata* | Covering 5-25% | 0.2 | Succulent | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.03 | 53.33 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Plentiful but small cover, <5% | 0.35 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.5 | 28.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Combretum zeyheri* | Covering 5-25% | 0.28 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 15 | 0.5 | 25.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cymbopogon plurinodis* | Covering 50-75% | 1.25 | Grass | Very Gentle <15° | Ferric Luvisols | No | 62.5 | 0.15 | 100.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Eragrostis Superba* | Plentiful but small cover, <5% | 0.4 | Grass | Very Gentle <15° | Ferric Luvisols | No | 2.5 | 0.15 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Covering 25-50% | 1.5 | Tree | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 10 | 20.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 25-50% | 0.3 | Grass | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 0.02 | 76.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Parthenium hysterophorus* | Covering 25-50% | 0.39 | Shrub | Very Gentle <15° | Ferric Luvisols | Yes Cat 1b | 37.5 | 0.5 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Rhoicissus tridentata* | Covering 25-50% | 0.45 | Shrub | Very Gentle <15° | Ferric Luvisols | No | 37.5 | 0.5 | 40.00 | |
| Plot05RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Covering 50-75% | 6.5 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 40 | 1.56 | Disturbed, erosion, bare patches |
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Covering 25-50% | 7.5 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 37.5 | 40 | 0.94 | |
| | Denser woody Savanna vegetation types | Natural Area | *Grewia flava* | Not many individuals, 1-10 | 1 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 5 | 0.5 | 10.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Covering 25-50% | 0.55 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 37.5 | 0.5 | 53.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cenchrus ciliaris* | Covering 25-50% | 0.59 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 37.5 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Cymbopogon plurinodis* | Covering 50-75% | 1.4 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 0.15 | 190.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Dittrichia graveolens* | Covering 5-25% | 0.85 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 15 | 0.5 | 30.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Lantana camara* | Covering 5-25% | 0.6 | Shrub | Very Gentle <15° | Rhodic Nitosols | Yes Cat 1b | 15 | 0.5 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Plentiful but small cover, <5% | 0.26 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 5-25% | 0.56 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 15 | 0.15 | 40.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Setaria sphacelata* | Covering 50-75% | 0.75 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 0.15 | 120.00 | |
| Plot06RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Covering 50-75% | 7 | Tree | Very Gentle <15° | Haplic Lixisols | No | 62.5 | 40 | 1.56 | Vegetation denser to east |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Plentiful but small cover, <5% | 9 | Tree | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Lantana camara* | Cover very small, <5% | 1.25 | Tree | Very Gentle <15° | Haplic Lixisols | Yes Cat 1b | 2.5 | 10 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Carissa macrocarpa* | Plentiful but small cover, <5% | 1.75 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 2 | 1.25 | |
| | Denser woody Savanna vegetation types | Natural Area | *Celtis africana* | Covering 25-50% | 7 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 37.5 | 2 | 18.75 | |
| | Denser woody Savanna vegetation types | Natural Area | *Aloe maculata* | Covering 5-25% | 0.24 | Succulent | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.03 | 70.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Cover very small, <5% | 0.55 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Carissa macrocarpa* | Cover very small, <5% | 0.35 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.5 | 26.67 | |
| | Denser woody Savanna vegetation types | Natural Area | *Combretum zeyheri* | Covering 5-25% | 0.55 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.5 | 50.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Hyparrhenia hirta* | Covering 5-25% | 0.68 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Lantana camara* | Cover very small, <5% | 0.2 | Shrub | Very Gentle <15° | Haplic Lixisols | Yes Cat 1b | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 50-75% | 0.9 | Grass | Very Gentle <15° | Haplic Lixisols | No | 62.5 | 0.15 | 108.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Searsia zeyheri* | Covering 5-25% | 0.95 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.5 | 60.00 | |
| Plot07RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Dichrostachys cinerea* | Covering 50-75% | 9 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 40 | 1.56 | Dense vegetation with nearby buildings |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Plentiful but small cover, <5% | 10 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Plentiful but small cover, <5% | 8 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Euclea crispa* | Not many individuals, 1-10 | 2 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 5 | 2 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Gymnosporia buxifolia* | Not many individuals, 1-10 | 1.5 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 5 | 0.5 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Olea europaea* | Cover very small, <5% | 1.5 | Tree | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 10 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Aloe maculata* | Covering 5-25% | 0.2 | Succulent | Very Gentle <15° | Rhodic Nitosols | No | 15 | 0.03 | 70.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Cover very small, <5% | 0.16 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Combretum zeyheri* | Cover very small, <5% | 0.15 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Commelina africana* | Cover very small, <5% | 0.36 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Kalanchoe lanceolata* | Covering 5-25% | 0.18 | Succulent | Very Gentle <15° | Rhodic Nitosols | No | 15 | 0.03 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Lantana camara* | Covering 5-25% | 0.2 | Shrub | Very Gentle <15° | Rhodic Nitosols | Yes Cat 1b | 15 | 0.5 | 40.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Panicum coloratum* | Covering 50-75% | 0.3 | Grass | Very Gentle <15° | Rhodic Nitosols | No | 62.5 | 0.02 | 120.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senecio pterophorus* | Cover very small, <5% | 1 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Cover very small, <5% | 0.13 | Shrub | Very Gentle <15° | Rhodic Nitosols | No | 2.5 | 0.5 | 20.00 | |
| Plot08RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Arundo donax* | Covering 75%+ | 10 | Grass | Very Gentle <15° | Rhodic Lixisols | Yes Cat 1b | 87.5 | 0.15 | 583.33 | Heavy *Arundo* invasion, near property |
| Plot09RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Covering 75%+ | 6 | Tree | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 40 | 2.19 | Dense tree cover, likely planted |
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Cover very small, <5% | 3 | Tree | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Dichrostachys cinerea* | Cover very small, <5% | 2.75 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 2 | 1.25 | |
| | Denser woody Savanna vegetation types | Natural Area | *Euclea crispa* | Plentiful but small cover, <5% | 2.5 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 2 | 1.25 | |
| | Denser woody Savanna vegetation types | Natural Area | *Vachellia nilotica* | Cover very small, <5% | 7 | Tree | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Aloe maculata* | Covering 5-25% | 0.12 | Succulent | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.03 | 45.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Covering 5-25% | 0.45 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.5 | 50.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Diospyros lycioides* | Cover very small, <5% | 0.75 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 24.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Eragrostis rigidior* | Covering 50-75% | 0.85 | Grass | Very Gentle <15° | Calcic Vertisols | No | 62.5 | 0.15 | 96.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Covering 5-25% | 0.25 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.5 | 26.67 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia zeyheri* | Covering 5-25% | 0.8 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.5 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Senegalia burkei* | Cover very small, <5% | 0.56 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 20.00 | |
| Plot10RemVegNat | Denser woody Savanna vegetation types | Natural Area | *Vachellia karroo* | Cover very small, <5% | 3 | Tree | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 40 | 1.00 | Sparse vegetation |
| | Denser woody Savanna vegetation types | Natural Area | *Dichrostachys cinerea* | Covering 25-50% | 3.5 | Tree | Very Gentle <15° | Calcic Vertisols | No | 37.5 | 40 | 0.94 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia lancea* | Plentiful but small cover, <5% | 4 | Tree | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Gymnosporia buxifolia* | Cover very small, <5% | 1.5 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Ziziphus mucronata* | Not many individuals, 1-10 | 0.5 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.5 | 10.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia leptodictya* | Cover very small, <5% | 4 | Tree | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 40 | 1.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Searsia pyroides* | Not many individuals, 1-10 | 2 | Tree | Very Gentle <15° | Calcic Vertisols | No | 5 | 10 | 0.50 | |
| | Denser woody Savanna vegetation types | Natural Area | *Aloe maculata* | Covering 5-25% | 0.27 | Succulent | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.03 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Asparagus laricinus* | Covering 5-25% | 1.1 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.5 | 30.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Denser woody Savanna vegetation types | Natural Area | *Dichrostachys cinerea* | Covering 5-25% | 1.3 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.5 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Digitaria tricholaenoides* | Covering 5-25% | 0.48 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Digitaria eriantha* | Covering 5-25% | 1.4 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Eragrostis rigidior* | Covering 50-75% | 0.4 | Grass | Very Gentle <15° | Calcic Vertisols | No | 62.5 | 0.15 | 108.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Setaria sphacelate var. torta* | Covering 5-25% | 0.85 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 40.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Themeda triandra* | Covering 5-25% | 0.76 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 60.00 | |
| | Denser woody Savanna vegetation types | Natural Area | *Urtica ferox* | Covering 5-25% | 0.02 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.02 | 60.00 | |
| Plot01MarThrnNat | Marikana Thornveld | Natural Area | *Andropogon gayanus* | Covering 75%+ | 1.6 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | No vegetation remains, inside cultivated area |
| | Marikana Thornveld | Natural Area | *Sorghum Halepense* | Cover very small, <5% | 2.8 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Aeschynomene indica L.* | Cover very small, <5% | 1.95 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| Plot02MarThrnNat | Marikana Thornveld | Natural Area | *Cenchrus ciliaris* | Covering 75%+ | 0.57 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | Poor cover, bare ground |
| | Marikana Thornveld | Natural Area | *Dichrostachys cinerea* | Cover very small, <5% | 0.7 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Natural Area | *Parthenium hysterophorus* | Cover very small, <5% | 0.4 | Shrub | Very Gentle <15° | Calcic Vertisols | Yes Cat 1b | 2.5 | 0.5 | 5.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot03MarThrnNat | Marikana Thornveld | Natural Area | *Themeda triandra* | Cover very small, <5% | 1.2 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | Recovering after clearing |
| | Marikana Thornveld | Natural Area | *Pentzia incana* | Cover very small, <5% | 0.61 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Natural Area | *Digitaria eriantha* | Not many individuals, 1-10 | 1.3 | Grass | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.15 | 33.33 | |
| | Marikana Thornveld | Natural Area | *Digitaria tricholaenoides* | Covering 75%+ | 0.7 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | |
| Plot04MarThrnNat | Marikana Thornveld | Natural Area | *Heteropogon contortus* | Plentiful but small cover, <5% | 1 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | Cleared for vehicle access |
| | Marikana Thornveld | Natural Area | *Themeda triandra* | Covering 5-25% | 1.5 | Grass | Moderately Steep Slope 26-35° | Haplic Lixisols | No | 15 | 0.15 | 100.00 | |
| | Marikana Thornveld | Natural Area | *Digitaria tricholaenoides* | Plentiful but small cover, <5% | 1.2 | Grass | Moderately Steep Slope 26-35° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Digitaria eriantha* | Plentiful but small cover, <5% | 1 | Grass | Moderately Steep Slope 26-35° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Setaria sphacelata* | Plentiful but small cover, <5% | 1.1 | Grass | Moderately Steep Slope 26-35° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Dichrostachys cinerea* | Not many individuals, 1-10 | 0.69 | Shrub | Moderately Steep Slope 26-35° | Haplic Lixisols | No | 5 | 0.5 | 10.00 | |
| Plot05MarThrnNat | Marikana Thornveld | Natural Area | *Melinis Repens* | Cover very small, <5% | 0.8 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | Dense natural vegetation, near stream |
| | Marikana Thornveld | Natural Area | *Lantana camara* | Covering 5-25% | 2.5 | Tree | Very Gentle <15° | Haplic Lixisols | Yes Cat 1b | 15 | 40 | 1.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Marikana Thornveld | Natural Area | *Senegalia burkei* | Cover very small, <5% | 1.35 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Natural Area | *Digitaria eriantha* | Plentiful but small cover, <5% | 1.4 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Cynodon dactylon* | Covering 25-50% | 0.2 | Grass | Very Gentle <15° | Haplic Lixisols | No | 37.5 | 0.02 | 1875.00 | |
| | Marikana Thornveld | Natural Area | *Heteropogon contortus* | Cover very small, <5% | 0.56 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Aloe maculata* | Cover very small, <5% | 0.39 | Succulent | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.3 | 8.33 | |
| | Marikana Thornveld | Natural Area | *Sporobolus africanus* | Cover very small, <5% | 1.4 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Melinis Repens* | Cover very small, <5% | 1.38 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| Plot06MarThrnNat | Marikana Thornveld | Natural Area | *Aloe maculata* | Not many individuals, 1-10 | 0.22 | Succulent | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.03 | 166.67 | Dense vegetation, near house |
| | Marikana Thornveld | Natural Area | *Sporobolus africanus* | Covering 50-75% | 1.45 | Grass | Very Gentle <15° | Haplic Lixisols | No | 62.5 | 0.15 | 416.67 | |
| | Marikana Thornveld | Natural Area | *Dichrostachys cinerea* | Not many individuals, 1-10 | 0.35 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.5 | 10.00 | |
| Plot07MarThrnNat | Marikana Thornveld | Natural Area | *Heteropogon contortus* | Covering 5-25% | 0.7 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.15 | 100.00 | Disturbed area near landfill |
| | Marikana Thornveld | Natural Area | *Digitaria eriantha* | Covering 50-75% | 1.4 | Grass | Very Gentle <15° | Haplic Lixisols | No | 62.5 | 0.15 | 416.67 | |
| | Marikana Thornveld | Natural Area | *Ziziphus mucronata* | Not many individuals, 1-10 | 1.3 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.5 | 10.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Marikana Thornveld | Natural Area | *Themeda triandra* | Plentiful but small cover, <5% | 1.5 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Cynodon dactylon* | Cover very small, <5% | 0.2 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.02 | 125.00 | |
| | Marikana Thornveld | Natural Area | *Eragrostis rigidior* | Cover very small, <5% | 0.45 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| Plot08MarThrnNat | Marikana Thornveld | Natural Area | *Tithonia rotundifolia* | Covering 50-75% | 1.5 | Shrub | Very Gentle <15° | Calcic Vertisols | Yes Cat 1b | 62.5 | 0.5 | 125.00 | Sparse, previous cultivation |
| | Marikana Thornveld | Natural Area | *Ipomoea indica* | Covering 25-50% | 1.5 | Shrub | Very Gentle <15° | Calcic Vertisols | Yes Cat 1b | 37.5 | 0.5 | 75.00 | |
| | Marikana Thornveld | Natural Area | *Tagetes minuta* | Not many individuals, 1-10 | 1.1 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.5 | 10.00 | |
| | Marikana Thornveld | Natural Area | *Tipuana tipu (Benth.) Kuntze* | Covering 5-25% | 4 | Tree | Very Gentle <15° | Calcic Vertisols | Yes Cat 3 | 15 | 40 | 1.00 | |
| Plot09MarThrnNat | Marikana Thornveld | Natural Area | *Ziziphus mucronata* | Not many individuals, 1-10 | 1.2 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.5 | 10.00 | Private reserve, well vegetated |
| | Marikana Thornveld | Natural Area | *Melinis Repens* | Covering 50-75% | 1.3 | Grass | Very Gentle <15° | Calcic Vertisols | No | 62.5 | 0.15 | 416.67 | |
| | Marikana Thornveld | Natural Area | *Aloe maculata* | Not many individuals, 1-10 | 0.25 | Succulent | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.03 | 166.67 | |
| | Marikana Thornveld | Natural Area | *Searsia lancea* | Plentiful but small cover, <5% | 0.75 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Natural Area | *Digitaria eriantha* | Plentiful but small cover, <5% | 1.3 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Natural Area | *Conyza sumatrensis (Retz.) E. Walker var. sumatrensis* | Plentiful but small cover, <5% | 1.25 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot01MarThrnRR | Marikana Thornveld | Road Reserve | *Andropogon gayanus* | Covering 75%+ | 1 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Sorghum Halepense* | Plentiful but small cover, <5% | 1.8 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| Plot02MarThrnRR | Marikana Thornveld | Road Reserve | *Cenchrus ciliaris* | Covering 75%+ | 0.5 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Eragrostis rigidior* | Cover very small, <5% | 0.3 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.02 | 125.00 | |
| | Marikana Thornveld | Road Reserve | *Asparagus laricinus* | Not many individuals, 1-10 | 0.74 | Grass | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.15 | 33.33 | |
| Plot03MarThrnRR | Marikana Thornveld | Road Reserve | *Vachellia karroo* | Not many individuals, 1-10 | 0.15 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.5 | 10.00 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Themeda triandra* | Covering 75%+ | 1 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | |
| | Marikana Thornveld | Road Reserve | *Pentzia incana* | Not many individuals, 1-10 | 0.45 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.5 | 10.00 | |
| Plot04MarThrnRR | Marikana Thornveld | Road Reserve | *Chloris gayana* | Plentiful but small cover, <5% | 1.1 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Siphocodon debilis* | Plentiful but small cover, <5% | 0.3 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Road Reserve | *Heteropogon contortus* | Not many individuals, 1-10 | 0.47 | Grass | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.15 | 33.33 | |
| | Marikana Thornveld | Road Reserve | *Urochloa Mosambicensis* | Covering 25-50% | 0.72 | Grass | Very Gentle <15° | Haplic Lixisols | No | 37.5 | 0.15 | 250.00 | |
| | Marikana Thornveld | Road Reserve | *Melinis Repens* | Not many individuals, 1-10 | 0.54 | Grass | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.15 | 33.33 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot05MarThrnRR | Marikana Thornveld | Road Reserve | *Chloris gayana* | Covering 25-50% | 1.2 | Grass | Very Gentle <15° | Haplic Lixisols | No | 37.5 | 0.15 | 250.00 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Urochloa Mosambicensis* | Plentiful but small cover, <5% | 0.3 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.02 | 125.00 | |
| | Marikana Thornveld | Road Reserve | *Dactyloctenium aegyptium (L.) Willd.* | Cover very small, <5% | 0.45 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Road Reserve | *Asparagus laricinus* | Cover very small, <5% | 0.52 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.5 | 5.00 | |
| Plot06MarThrnRR | Marikana Thornveld | Road Reserve | *Digitaria eriantha* | Covering 5-25% | 1.5 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.15 | 100.00 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Chloris gayana* | Covering 5-25% | 1.2 | Grass | Very Gentle <15° | Haplic Lixisols | No | 15 | 0.15 | 100.00 | |
| | Marikana Thornveld | Road Reserve | *Sporobolus africanus* | Plentiful but small cover, <5% | 1.4 | Grass | Very Gentle <15° | Haplic Lixisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Road Reserve | *Setaria sphacelata* | Not many individuals, 1-10 | 0.55 | Grass | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.15 | 33.33 | |
| | Marikana Thornveld | Road Reserve | *Asparagus laricinus* | Not many individuals, 1-10 | 0.45 | Shrub | Very Gentle <15° | Haplic Lixisols | No | 5 | 0.5 | 10.00 | |
| Plot07MarThrnRR | Marikana Thornveld | Road Reserve | *Heteropogon contortus* | Covering 75%+ | 0.35 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | Poorly Managed |
| | Marikana Thornveld | Road Reserve | *Digitaria eriantha* | Cover very small, <5% | 1 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Road Reserve | *Conyza sumatrensis (Retz.) E. Walker var. sumatrensis* | Cover very small, <5% | 1.1 | Shrub | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.5 | 5.00 | |
| | Marikana Thornveld | Road Reserve | *Cynodon dactylon* | Cover very small, <5% | 0.21 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.02 | 125.00 | |

| PlotID | Vegetation Type | Position | PlantID | Abundance Scale | Height | Type of Plant | Slope | Soil Type | Alien | Cover Midpoint | Plant Area | Estimated Plants per 100m2 | Further Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Marikana Thornveld | Road Reserve | *Convolvulus arvensis* | Cover very small, <5% | 0.25 | Shrub | Very Gentle <15° | Calcic Vertisols | Yes Cat 1b | 2.5 | 0.5 | 5.00 | |
| Plot08MarThrnRR | Marikana Thornveld | Road Reserve | *Cenchrus ciliaris* | Covering 75%+ | 0.65 | Grass | Very Gentle <15° | Calcic Vertisols | No | 87.5 | 0.15 | 583.33 | Well Maintained |
| | Marikana Thornveld | Road Reserve | *Digitaria eriantha* | Plentiful but small cover, <5% | 0.84 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Road Reserve | *Andropogon gayanus* | Cover very small, <5% | 0.59 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |
| | Marikana Thornveld | Road Reserve | *Setaria sphacelata* | Not many individuals, 1-10 | 0.67 | Grass | Very Gentle <15° | Calcic Vertisols | No | 5 | 0.15 | 33.33 | |
| Plot09MarThrnRR | Marikana Thornveld | Road Reserve | *Digitaria eriantha* | Plentiful but small cover, <5% | 0.57 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | Well maintained |
| | Marikana Thornveld | Road Reserve | *Heteropogon contortus* | Covering 5-25% | 0.59 | Grass | Very Gentle <15° | Calcic Vertisols | No | 15 | 0.15 | 100.00 | |
| | Marikana Thornveld | Road Reserve | *Andropogon gayanus* | Covering 25-50% | 0.62 | Grass | Very Gentle <15° | Calcic Vertisols | No | 37.5 | 0.15 | 250.00 | |
| | Marikana Thornveld | Road Reserve | *Cynodon dactylon* | Plentiful but small cover, <5% | 0.25 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.02 | 125.00 | |
| | Marikana Thornveld | Road Reserve | *Setaria sphacelata* | Cover very small, <5% | 0.29 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.02 | 125.00 | |
| | Marikana Thornveld | Road Reserve | *Urochloa Mosambicensis* | Cover very small, <5% | 0.31 | Grass | Very Gentle <15° | Calcic Vertisols | No | 2.5 | 0.15 | 16.67 | |

Table S6.2 Plot-Level Diversity Indices

| Plot ID | Vegetation Type | Position | Shannon Index (H') | Pielou's Evenness (J') | Evenness | Species Richness (S) | Total Number of Plants | Alien Species Count |
|---|---|---|---|---|---|---|---|---|
| Plot01RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 1.13 | 0.169013386 | 0.7 | 5 | 801 | 0 |
| Plot02RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 0.855 | 0.133076643 | 0.778 | 3 | 617 | 0 |
| Plot03RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 0.822 | 0.124092981 | 0.395 | 8 | 753 | 0 |
| Plot04RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 1.11 | 0.175363182 | 0.62 | 6 | 561 | 0 |
| Plot05RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 1.34 | 0.182653738 | 0.689 | 7 | 1535 | 0 |
| Plot06RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 1.01 | 0.143295208 | 0.564 | 6 | 1151 | 0 |

| Plot ID | Vegetation Type | Position | Shannon Index (H') | Pielou's Evenness (J') | Evenness | Species Richness (S) | Total Number of Plants | Alien Species Count |
|---|---|---|---|---|---|---|---|---|
| Plot07RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 0.129 | 0.020165924 | 1.186 | 2 | 600 | 0 |
| Plot08RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 0.661 | 0.101648676 | 0.954 | 2 | 667 | 0 |
| Plot09RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 1.37 | 0.208362616 | 0.853 | 5 | 717 | 0 |
| Plot10RemVegRR | Denser woody Savanna vegetation types | Road Reserve | 0.916 | 0.136601446 | 0.569 | 5 | 817 | 0 |
| Plot01RemVegNat | Denser woody Savanna vegetation types | Natural Area | 1.82 | 0.305715708 | 0.761 | 11 | 385 | 0 |
| Plot02RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.27 | 0.357924399 | 0.784 | 18 | 568 | 20 |

| Plot ID | Vegetation Type | Position | Shannon Index (H') | Pielou's Evenness (J') | Evenness | Species Richness (S) | Total Number of Plants | Alien Species Count |
|---|---|---|---|---|---|---|---|---|
| Plot03RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.06 | 0.352700974 | 0.805 | 13 | 344 | 0 |
| Plot04RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.13 | 0.35126574 | 0.807 | 14 | 430 | 60 |
| Plot05RemVegNat | Denser woody Savanna vegetation types | Natural Area | 1.92 | 0.30125587 | 0.803 | 11 | 586 | 60 |
| Plot06RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.08 | 0.34185228 | 0.813 | 13 | 439 | 21 |
| Plot07RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.22 | 0.36272747 | 0.821 | 15 | 455 | 40 |
| Plot08RemVegNat | Denser woody Savanna vegetation types | Natural Area | 0 | 0 | 0 | 1 | 583 | 583 |

| Plot ID | Vegetation Type | Position | Shannon Index (H') | Pielou's Evenness (J') | Evenness | Species Richness (S) | Total Number of Plants | Alien Species Count |
|---|---|---|---|---|---|---|---|---|
| Plot09RemVegNat | Denser woody Savanna vegetation types | Natural Area | 1.9 | 0.331582358 | 0.765 | 12 | 308 | 0 |
| Plot10RemVegNat | Denser woody Savanna vegetation types | Natural Area | 2.26 | 0.363960214 | 0.816 | 16 | 497 | 0 |
| Plot01MarThrnNat | Marikana Thornveld | Natural Area | 0.251 | 0.039066944 | 0.229 | 3 | 617 | 0 |
| Plot02MarThrnNat | Marikana Thornveld | Natural Area | 0.0973 | 0.015238377 | 0.0885 | 3 | 593 | 5 |
| Plot03MarThrnNat | Marikana Thornveld | Natural Area | 0.37 | 0.057290279 | 0.267 | 4 | 638 | 0 |
| Plot04MarThrnNat | Marikana Thornveld | Natural Area | 1.38 | 0.266317569 | 0.772 | 6 | 178 | 0 |
| Plot05MarThrnNat | Marikana Thornveld | Natural Area | 0.295 | 0.038878111 | 0.134 | 9 | 1974 | 1 |
| Plot06MarThrnNat | Marikana Thornveld | Natural Area | 0.674 | 0.105528845 | 0.613 | 3 | 594 | 0 |
| Plot07MarThrnNat | Marikana Thornveld | Natural Area | 1.14 | 0.174555407 | 0.635 | 6 | 686 | 0 |
| Plot08MarThrnNat | Marikana Thornveld | Natural Area | 0.848 | 0.158449641 | 0.611 | 4 | 211 | 201 |
| Plot09MarThrnNat | Marikana Thornveld | Natural Area | 0.863 | 0.134186841 | 0.482 | 6 | 621 | 0 |
| Plot01MarThrnRR | Marikana Thornveld | Road Reserve | 0.129 | 0.020165924 | 0.186 | 2 | 600 | 0 |
| Plot02MarThrnRR | Marikana Thornveld | Road Reserve | 0.627 | 0.094884979 | 0.571 | 3 | 741 | 0 |
| Plot03MarThrnRR | Marikana Thornveld | Road Reserve | 0.169 | 0.026398342 | 0.153 | 3 | 603 | 0 |
| Plot04MarThrnRR | Marikana Thornveld | Road Reserve | 0.89 | 0.152840973 | 0.553 | 5 | 338 | 0 |

| Plot ID | Vegetation Type | Position | Shannon Index (H') | Pielou's Evenness (J') | Evenness | Species Richness (S) | Total Number of Plants | Alien Species Count |
|---|---|---|---|---|---|---|---|---|
| Plot05MarThrnRR | Marikana Thornveld | Road Reserve | 0.845 | 0.141211397 | 0.61 | 4 | 397 | 0 |
| Plot06MarThrnRR | Marikana Thornveld | Road Reserve | 1.3 | 0.233784289 | 0.808 | 5 | 260 | 0 |
| Plot07MarThrnRR | Marikana Thornveld | Road Reserve | 0.64 | 0.0969716 | 0.398 | 5 | 735 | 5 |
| Plot08MarThrnRR | Marikana Thornveld | Road Reserve | 0.439 | 0.067778582 | 0.317 | 4 | 650 | 0 |
| Plot09MarThrnRR | Marikana Thornveld | Road Reserve | 1.49 | 0.230934392 | 0.833 | 6 | 634 | 0 |