ECOLOGICAL ANALYSIS OF AFROMONTANE GRASSLANDS IN THE EASTERN FREE STATE USING THE BIOTOPE QUALITY INDEX

by

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DECLARATION

I, Serero Abiot Modise (student no. 2005161829), declare that the thesis or publishable manuscripts that I herewith submit for the Doctoral Degree in Zoology at the University of the Free State are my original independent work (safe to the extent explicitly otherwise stated), and that I have not previously submitted it for a qualification at any other institution of higher education.

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ABSTRACT

Environmental disturbance poses threats to conservation efforts in highland montane grassland in temperate regions prone to land transformation, overgrazing, habitat fragmentation, soil erosion, uncontrolled fires and plant invasion. In this study, the Biotope Quality Index (*BQI*) was adopted and applied as a species surrogate to evaluate impact of disturbance on the Afromontane grassland vegetation types namely Sandy Eastern Free State, Basotho Montane Shrubland and North Drakensberg Highlands Drakensberg Grasslands in the Golden Gate Highlands National Park. In addition, the Biotope Conservation Status (*BCS*) was developed as an additional to *BQI* to indicate the impact of environmental disturbance on conservation efforts in protected areas. Chapter 1 begins with a brief history that highlights the importance of temperate grasslands and the role that protected areas play in the conservation of sensitive biomes. In Chapter 2, the most regularly used environmental quality indices, their application domain (air, water and terrestrial), arithmetic formulation, development and application process as well as suggestions into continuous usage and nomenclature of those environmental indices were reviewed. Chapter 3 indicates how the *BQI* was adopted into the BQI_{veg} for vegetation ecology analysis, using three common plant species abundance-cover scales (Domin, Braun-Blanquet and percentage density), and results suggested that, for *BQI*, species list must include most plants present in the vegetation type to allow variation within data points. Furthermore, BQI_{veg} was able to indicate variation in sites which were affected by relatively high degree of soil erosion, plant invasion / encroachment and long-term effects of historical agricultural practices. Moreover, the application of the *BQI* as a species surrogate was demonstrated, with inclusion of the *BCS* add-on as well as a ArcGIS Feature Analysis parameter, to illustrate the conservation status of the Sandy Eastern Free State in Chapter 4 and the Basotho Montane Shrubland grassland vegetation type in Chapter 5 using arthropod assemblages to evaluate as a unit of measure. Furthermore, the inclusion of the *BQI* as a species surrogate for rapid bioassessments and biomonitoring was supported using both plant communities and arthropod assemblages to highlight the impact of environmental disturbances in the Highlands Drakensberg Grassland vegetation unit in Chapter 6. In conclusion, *BQI* indicated the ability to use plants as a unit of measure and its flexibility for common vegetation abundance-cover scales. Again, *BQI* was able to highlight the impacts of environmental disturbances on plant communities and arthropod assemblages. Therefore, it is recommended that *BQI* and *BCS* be used as a surrogate for bioassessment and biomonitoring to assess impacts of disturbance toward and to indicate the conservation status of important ecosystems, bioregions, or biotopes.

Keywords

Biomonitoring, Biotope Conservation Status, Biotic Communities, Environmental Disturbance, Highveld Grassland Bioregion, Species Surrogate

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Introduction

The term 'grassland' is defined and used in different ways depending on perspective and application, but its simplest (ecology) refers to it as an environment dominated by grassland vegetation, with a grass layer cover of more than 25 % (Gibson, 2009). Descriptions and variation in definition are enhanced by defining floristic structural and appearance features (physiognomy[1]) as a plant community with a low-growing plant cover of non-woody species (Milner and Hughes, 1968), inclusion of other graminoids, herb and grass-like plants (sedges, restios and juncus) (Sims, 1988), an infrequent or a low abundance of scattered woody (trees or shrubs) vegetation (Gibson, 2009), occurring in both temperate and tropic climatic regions with seasonal variation of temperature and precipitation extremes (Mucina and Rutherford, 2006), naturally maintained by ecological disturbances such as fire, drought, grazing, erosion and frost and (or) freezing temperatures (White et al., 2001). The main characteristic of grasslands is an overwhelming prevalence of grasses (Poaceae) with other narrowed leaf monocots that include sedges (Cyperaceae), restios (Restionaceae), juncus (Juncaginaceae) and herbaceous plants (Dixon et al., 2014). Although there might be sporadic forest and shrub patches within a grassland, the degree in which we limit ecotope or biome is when trees or shrubs become co-dominant, with respect cover and being a diagnostic part of plant community (Lock, 2006; Bucini and Hanan, 2007), to being shrubland, savanna or woodlands.

Grasslands are among the largest terrestrial biomes[2] that occupy every continent except Antarctica, covering 31 - 43 % of the earth's land surface, with a total surface area of 41 - 56 x10^6 km^2 globally (Gibson, 2009), with a greater degree of variety in South America and sub-Saharan Africa where 16 and 12 dominant grassland divisions are respectively observed as shown in Figure 1.1 (Dixon et al., 2014). Grassland biomes play important ecological roles, including being sinks and storing relatively larger amounts of about 28 – 37 % global soil organic carbon deposition (Scurlock and Hall, 1998; Lal, 2004), provide functional landscape units for water filtration and regulation (Zhao et al., 2020), create basins for freshwater catchment areas (Gibson, 2009; Török et al., 2018), host a larger fraction of bovine, ungulate and other small grazing animals (Diaz et al., 1992; Delger and Tischew, 2018). Furthermore, grasslands contribute as a primary food source to the majority of fauna, with genetic diversity resources for resilience to disturbance (Abberton et al., 2010; Lei et al., 2026; Török et al., 2018), seed dispersal, and disease regulation (Harrison

[1] physiognomy: general appearance of floristic structural composition in a particular region (Schaminée et al., 2002).
[2] biome: is a broad ecological unit which represents a major life zone extending over a large natural area (Rutherford, 1997).

et al., 2010). Additionally, grasslands created space for agricultural practices (crop cultivated land and livestock rearing) for food security and medicine (CSIRO, 2012; Martín-López, 2014), mining activities for the extraction of natural resources for energy (MEA, 2005) and human settlement and development (de Groot et al., 2010) for human well-being and recreational benefits. Furthermore, grasslands host about 15 % of the World's Centres for Plant Endemism, that are also rich in biodiversity and endemic arthropods, birds, reptiles and mammals in uniquely distinct ecoregions worldwide (White et al., 2000; Egoh et al., 2011).

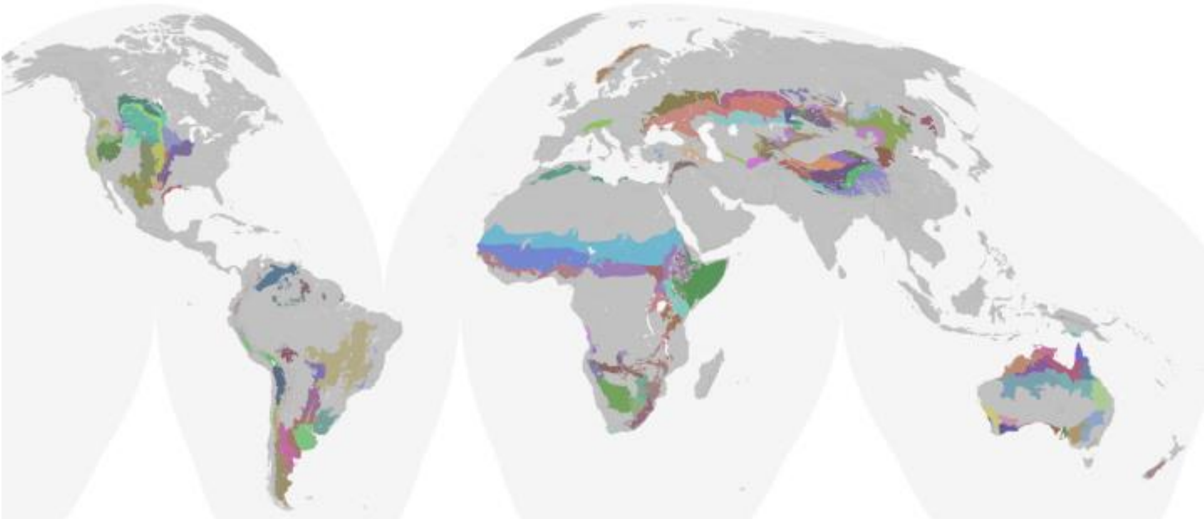

Figure 1.1: Terrestrial Ecoregions of the World (TEOW) illustrating ecoregions containing dominant grassland types as described by the International Vegetation Classification (IVC), adopted from Dixon et al. (2014).

Consequently, extended and intense exploitation practices for grasslands and natural resources led to global degradation on large scales for biomes sensitive to change (Steffen et al., 2015; Török et al., 2021). The main threats posed to global grassland biome conservation include habitat fragmentation and loss (Riecken et al., 2006; Zulka et al., 2014; Poschlod, 2015), plant community structure and vegetation type alteration (WRI, 200) and biodiversity loss (Bornand et al., 2016; Janssen et al., 2016). In addition to anthropogenic activities, natural disturbance threats such as frequent fires, prolonged drought periods, plant invasion and climate change also contribute to troubled sensitive biomes (Turner et al., 2007; Egoh et al., 2011). Earlier, MEA (2005) indicated that by year 1950, temperate grasslands had already lost about 70 % of their natural cover, and in addition, Reyers et al. (2005) reported that grassland biome in South Africa is most threatened with about 35 % cover prone to habitat transformation for cultivation, urbanisation, overgrazing and mining. Although there is research progress on grassland involvement in ecosystem processes and services, the majority of factors that influence the ecosystem are still not fully understood from organism, population, community, to ecosystem level in unimodal[3] or multimodal

[3] a model where response of a parameter is influenced by one factor (Hanan et al., 2014).

associations for ecological role and ecosystem functioning (van Jaarsveld et al., 2005). Some established initiatives for systematic conservation planning (SCP) are aimed to reduce and curb land degradation, promote biodiversity conservation, and ecosystem restoration of grassland ecosystems, namely United Nations Food and Agriculture Organization (FAO), UNESCO-MAB, the Conservation Science Program, World Wildlife Fund-Grassland Conservation Program, US Global 200 Programme, International Union for Conservation of Nature (IUCN), Convention on Biological Diversity (CBD), Central Grassland Conservation Initiative (CGCI), Temperate Grasslands Conservation Initiative (TGCI), Biodiversity and Protected Area Management (BIOPAMA), Communal Natural Reserves, Centers of Plant Diversity (CPD), Centers of Plant Endemism (CPE), SANBI South African Grasslands Programme, SANBI Botanic Gardens Conservation International (BGCI), RSA National Protected Areas (SANParks and World Heritage Sites) and Maloti-Drakensberg Transfrontier Park (Archibold, 1995; Mucina and Rutherford, 2006; Henwood, 2008; Carbutt et al., 2011). Another intention of these initiatives is to bridge the gap between grassland conservation stakeholders, i.e., society, ecologists, nature or environmental practitioners, and policy makers.

In this study, natural (or native) grasslands are described as Poaceae-dominated herbaceous vegetation with sporadic woody plants, where ecological processes primarily determine species and site characteristics with low or no human disturbance and influence, in that, vegetation composition (growth and life-form[4]) and co-occurrence of populations are driven by interactive processes of local geophysical, climatic and biotic parameters (Dixon et al., 2014), on which local flora and fauna are adapted to function successfully in their natural environment (Box and Fujiwara, 2014).

Biotic community and highlands grassland ecology

Southern Africa is located along the temperate grassland climate region, with unique geophysical factors that affect the co-occurring arthropod assemblages and plant taxa composition (Morin, 2011; Mittelbach and McGill, 2019). Biological (biotic) community ecology takes into consideration important roles that the abiotic component of ecosystem plays (i.e., chemical, physical and climatic units as well as influence on co-evolution and ecosystem processes) (Odum, 1969; Price et al., 2011). Temperate grasslands occur in regions with seasonal climatic conditions that have variation in precipitation (drier to moist, even temporal flooding) and temperature extremes (warm to cold, even partial freezing) for the annual weather fluctuation cycles (O’Connor and Bredenkamp, 1997; Bredenkamp et al., 2002). Biotic populations and communities

[4]growth-form: only considers plant structure and physiognomy (appearance).
life-form: considers plant structure and form as an adaptive response to environmental and geophysical conditions (Raunkiær, 1934).

inhabiting temperate grassland biomes have unique adaptations and are supported by a variety of microhabitats or biotopes[5] in a given region (New, 2019).

There are fundamental concepts in ecology that help understand the role of ecosystems processes and function to maintain biotic community and abiotic integrity over time, e.g. nested hierarchical of living systems (Odum, 1969; Kay, 1999), continuous organisms' interaction with their abiotic and biotic counterparts (Scheiner and Willig, 2008), flow of energy and material (Jørgensen et al., 2000; O'Connor et al., 2019), resilience and stability (Mulder et al., 2001; Pfisterer and Schmid, 2002), species ability to evolve adaptive and plasticity traits (Melo and Marriog, 2015; de Andreazzi et al., 2018), complex and heterogenous distributed communities () (;). that may also apply to temperate highlands grassland biomes. For this study, we begin by assuming that a natural grassland that is matured in succession is able to sustain an optimal number of species with optimal population size (including biological complexity) (Scheiner and Willig, 2008), as well as sustain their ecological processes, to provide optimal heterogeneous sustainable system with sufficient resources, again with adequate resistance (or avoidance) when under perturbational stress while still allowing natural succession to take place (Norton, 2009; Bredenhand, 2014). In addition, biotic communities acquire adaptive capabilities suitable for survival in natural temperate grasslands and its fluctuating seasonal and daily environmental conditions. Communities cohabiting the same biotope might follow a non-equilibrium theories approaches, in that temporal and fluctuating environmental conditions allow interactive processes to balance species representation in assemblages even though ratios might fluctuate over time depending on season (Odum, 1969; Lahav, 2004). The dependency of grassland communities to share common resources and continue to evolve with each other, relies on the ability of vegetation types to sustain optimum diversity or complexity, to populations and ecosystems maintain stability and resilience over time in ever-changing and fluctuating highlands environmental conditions (Damiani, 2002; New, 2019).

Brief history of grassland biome in Southern Africa

The global grassland distribution finds the northern hemisphere represented by prairies with elevation range from 200 m to 1200 m for North America, steppes with elevation range from 1000 m to 2000 m for Eurasia with most grasslands within mid-elevations. In comparison, the southern hemisphere grasslands are represented by pampas with elevation ranges from 600 m to 1300 m for Argentina, Brazil, and Uruguay, higher grassveld plateaus with elevation ranges of 500 m to 3800 m for southern Africa, and at lower elevations of 200 m to 1500 m for tussocks in New Zealand and alps of Australia characterised by smaller drier areas with occasional grass patches (Archibold, 1995; Mucina and Rutherford, 2006). Evolutionarily, primary grasslands are hypothesised to originate toward the end of the Eocene in the

[5] biotope: smallest geographical unit of the biosphere or a habitat that can be delimited by convenient boundaries and is characterised by its biota (Lincoln et al., 1998).

Paleogene period due to climatic changes towards drier terrestrial conditions for both the northern and southern hemispheres. Recent or modern vegetation types within grassland can be traced back using fossil pollen evidence to the Oligocene (Scott et al., 1997). In South Africa, the shift in grassland composition and outlining of biome boundaries most likely occurred during glacial and interglacial periods, allowing lowering of vegetation zones forced frost-controlled temperate grassland units to spread over much wider ranges from high-altitude units of Drakensberg, extending northward to current Highveld regions, westward into interior drier units, then later to moist units in eastward and southward regions (Mucina and Rutherford, 2006).

Southern Africa went through a considerate change in climatic conditions during the Neogene period and also saw major tectonic uplift on central highveld raising interior plateaux to form high-altitude Great Escapement Mountain chain folding on eastern side of the subcontinent with elevation range from 1400 m to 3000 m, which form a greater part of interior land mass (Partridge, 1997). The formation of the Great Escarpment Mountain chain changed the topography of the landscape, soil profile, and influenced climate conditions (Bredenkamp et al., 2002). Overtime, gradual interior spread of grasslands and related vegetation types (Figure 1.2) i.e., savanna and shrublands saw the coevolution of biotic herbivorous communities in the form of grazing mammals (especially Artiodactyls) and plant feeding arthropod guilds radiated (Cowling et al., 1997; Brown and Bezuidenhout, 2020). Currently, southern African grasslands are well adapted to grazing herbivory pressure, which now helps to remove extra foliage biomass, and together with other ecological factors such as fire and seasonal variation of temperature and rainfall, with inclusion of mist, frost, and snow, all play a role in limiting and restricting woody plant species to establish, thus maintaining grassland floristic structure (O'Connor and Bredenkamp, 1997; WWF, 2001; Bredenkamp et al., 2002; Mucina and Rutherford, 2006). Grasslands in southern Africa are not uniform in floristic structure and composition but have variation depending on topography, soil profile and climate conditions, creating biotopes with distinct mosaic interchange propositions between C_3, C_4 and sporadic CAM plants. Again, grasslands differ according to their location, elevation, plateau and aspect regions with high plant endemism, i.e., Drakensberg Alpine and Soutpansberg Centers of Plant Endemism with 34 grass and 161 orchid taxa endemic to grassland biome (van Wyk and Smith, 2001; Steenkamp et al., 2002), and endemic herbaceous plants are concentrated in low and northern parts of Drakensberg Alpine Centre of Endemism escapement (Mucina and Rutherford, 2006).

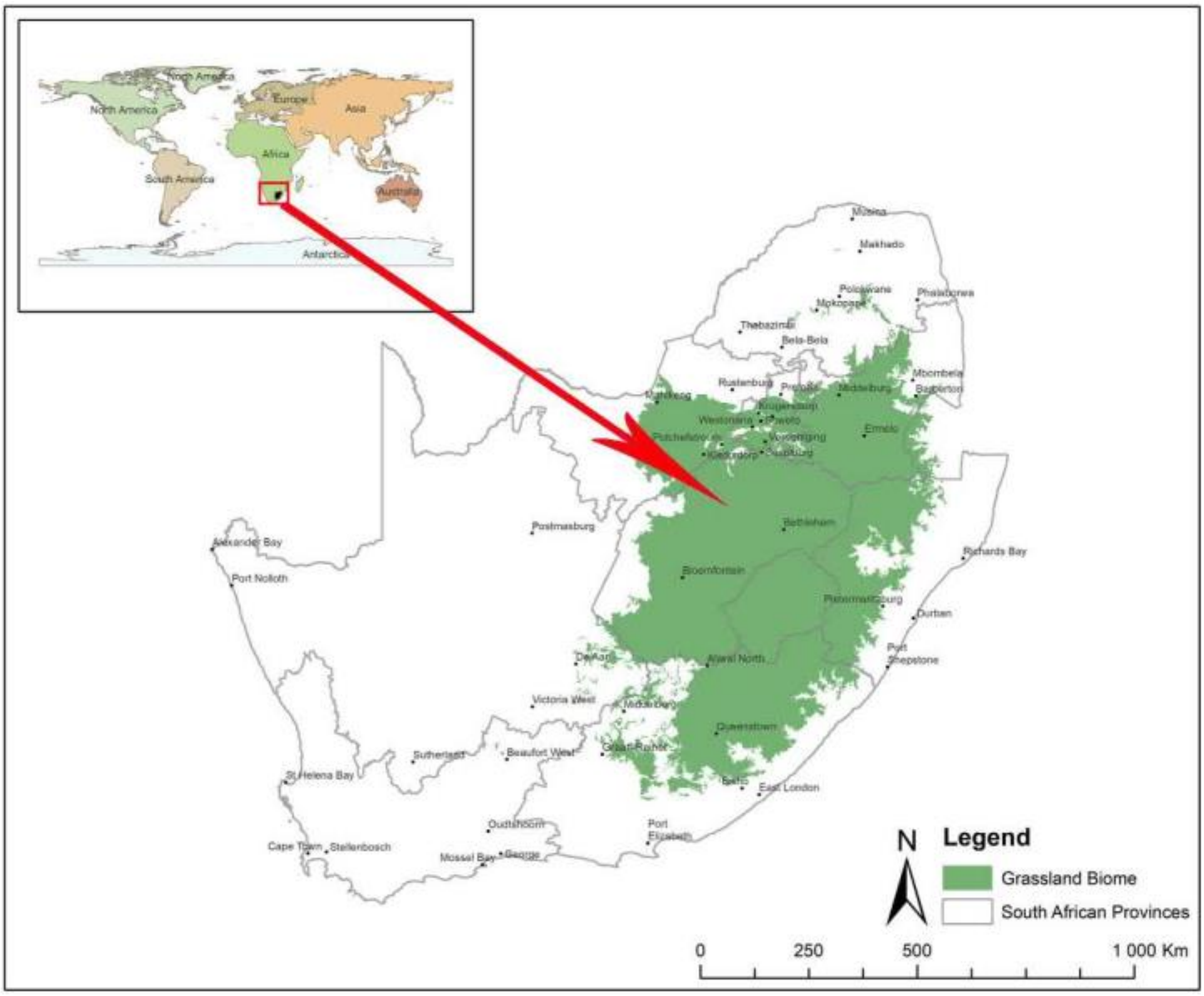

Figure 1.2: Grassland biome vegetation regions in southern Africa including Lesotho, South Africa and Swaziland (adapted from Mucina and Rutherford, 2006)

The local communal and agricultural vocabulary refers to rangeland as 'grassveld' for the types of indigenous grassland that provide most of the foraging material for grazing and browsing animals (Trollope et al., 1990), as a way to separate the jargon of southern African grassland ecology from the rest of the world. Furthermore, the distinction of main grassveld units into 'sour' and 'sweet' veld divisions takes into account the palatability of grassveld, landscape and soil profile, as well as the microclimate conditions of a region, in that sweet grasses are finer in leaf structure with high nitrogen ratio and remain palatable in the winter season, for lower more arid regions with annual average rainfall below 600 mm with low leaching and high soil nutrients, which make 'sweetveld', while comparatively, sour grasses are tall and tufted poorly grazed due to a low nitrogen ratio, occur in highly leached or well-drained soil of low nutrients, annual rainfall above 600 mm, but the dry shoots above ground tend to accumulate high biomass in winter season and hence fires are very common in 'sourveld' highland regions (Ellery et al., 1995; Maltitz, 2018). Therefore, in this study, the usage of term 'veld', 'sweetveld', 'sourveld', 'grassveld' and 'highveld' is acknowledged and adopted, but for the purpose of this study and consistency, vegetation types and bioregion units are referred to according to Cowling et al. (1997) as well as Mucina and Rutherford (2006).

Mountain grassland vegetation types

The main ecological factors that form the basis of grassland biome subdivisions include the altitude gradient, annual average rainfall range (i.e., drier 500 mm to moist 700 mm) and soil profile that correlate with the floristic composition and structure (O'Connor and Bredenkamp, 1997). Elevation and topography have a stronger influence on most climatic variables especially in mountainous highlands grassland regions affecting temperature, rainfall, frost, mist snow, wind action, and soil deposition (Bredenkamp, et al., 2002). Mucina and Rutherford (2006) categorised the grassland biome into four major subdivisions namely 1. Drakensberg Grassland (Gd) found primarily in the Great Escarpment portion of Drakensberg/Maloti/ uKhahlamba that include Lesotho (annual rainfall above 600 mm), 2. Dry Highveld Grassland (Gh) found in the central Free State, western parts of KwaZulu-Natal, eastern parts of the Northern Cape and southern parts of the Western Cape provinces on relative flat topography with undulating small mountains and small woody shrub interruptions (annual rainfall below 600 mm), 3. Mesic Highveld Grassland (Gm) found in the Free State, Eastern Cape, KwaZulu-Natal, Gauteng, margins of Mpumalanga, Limpopo provinces, Lesotho and Swaziland with nutrient rich soils, mixed floristic compositions of sour-sweetveld and sporadic shrublands (annual rainfall ranges between 500-700 mm) and lastly, 4. sub-Escarpment Grassland (Gs) found along rolling hills and foothills of the Drakensberg and Northern Escarpment with varying degrees of ecological variables (annual rainfall ranges between 500-700 mm) especially for the Eastern Cape, KwaZulu-Natal, Mpumalanga, Free State, margins of Northern Cape provinces and Lesotho, as respective vegetation bioregion units.

Highlands and mountain grasslands are characteristic of rugged topography, steep terrain, discontinuous gradients, shallow soil ground cover (depth below 1 m) at times with freezing daily intervals, rocky and exposed bedrock (Elias, 2020). In concept, ecological biotope or vegetation bioregion units of African mountain systems are categorised at various elevations (Figure 1.3) namely 1. uppermost belt (not found in southern Africa); 2. Afroalpine[6] belt above 3600 m with higher plant endemism consisting of bryophytes, tussock grasses (*Festuca* spp.), giant lobelias (*Lobelia* spp.), and senecios (*Dendrosenecio* spp.), 3. mid-subalpine belt between 2800 m and 3600 m characterised by sclerophyllous montane-highlands fynbos genera such as *Erica* spp., *Philippia* spp., *Cliffortia* spp., *Protea* spp., and *Passerina* spp., while most are evergreen woody plants; and then, 4. the lower undifferentiated Afromontane[7] belt mosaic between 1800 m and 2800 m consisting of moist grasslands, wetlands, indigenous forests and shrublands, along the Great Escarpment Mountain Chain

[6] alpine: bioregion above the climatic tree line in mountains > 3600m with variation in aspect and daily freezing of soil (Elias, 2020).
[7] afromontane: mountainous bioregion between 1800-3600m characterised by mosaic of biotope units e.g., moist grasslands, wetlands, indigenous forests and shrublands Elias, 2020).

(Carbutt and Edwards, 2015; de Deus Vidal and Clark, 2019; Brown and du Preez, 2020). The highlands montane grasslands ecoregion is home to a range of animals adapted to terrain and climate created by escarpments that include distinctive bird, mammal, reptile, amphibian and arthropod communities.

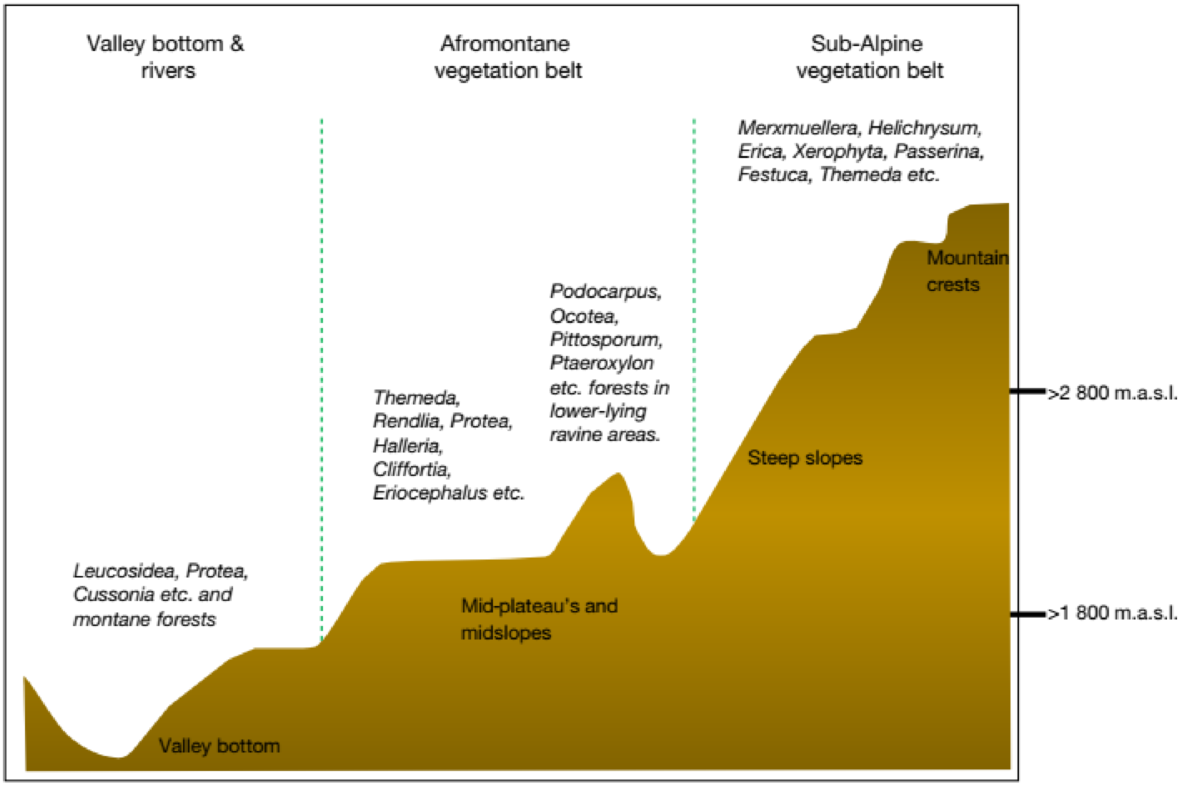

Figure 1.3: Typical illustration of broad vegetation belts of the higher-lying mountains of southern Africa, (adopted from Brown and du Preez, 2020)

Adaptations of biotic communities in highlands grasslands

The majority of ecological studies were trying to understand patterns and processes concerning origin, maintenance and consequences of species assemblages, biodiversity, population dynamics and community clustering within local bioregion (Morin, 2011), as well as dynamics associated with evolutionary adaptations between interacting biotic communities and their environment (Mittelbach and McGill, 2019). Biotic communities experience different living conditions created by the microclimate that is influenced by terrain and elevation levels in southern African mountain systems. Some factors that affect adaptation depend on association of species and communities with mountainous environment, in that some are permanent residents (complete life cycle and foraging), regular affiliate (some part of life cycle and daily duties), and while others are occasional visitors (mainly for foraging) (Elias, 2020). Some species with higher mobility capabilities are able to move around different altitude regions or levels, while others with limited or restricted mobility are confined to a specific altitude region of a mountain (Caccianiga et al. 2006). Species experience and respond to the gradient effect of environmental factors differently in individual or (and) communal shared capacity (Pierce et al., 2005). Another driving factor for trait adaptation beside environment and microclimate, are co-evolved biotic interactions between and among species as well as their respective communities (Ridley, 2004; Price et al., 2011). The influence that species within a community through interaction such as mutualism, competition for resources (food, space, mates), predator-prey and parasite or (and) parasitoid-host relations, also play a role in behavioural, population dynamics and community structuring that assist to facilitate important ecological functions and processes (Chang and Snyder, 2004; Speight et al., 2008; New, 2009). Below is a brief overview and examples of some ecological adaptation that biotic communities might acquire while inhabiting southern African mountain systems, that include terrain (substrate surface), ambient temperature fluctuations, moisture and rainfall availability, veld fire, herbivory, and soil erosion.

Highlands terrain

Generally, biotic endemism increases with elevation being greater in the Afroalpine and sub-Alpine ecoregions who are isolated and with extreme conditions. In contrast, biodiversity and species richness are greater down slope in the heterogeneous Afromontane ecoregion with mosaic of different biotopes (Töpfer and Gedeon, 2020). The uppermost highlands plateaus and slopes have little soil cover on the rocky and bedrock substrate, in that shrub species tend to grow between eroded rocky outcrops (Heenan and Mitchell, 2003), grasses at the summit are tuft and scattered following were soil aggregates (O’Connor and Bredenkamp, 1997). Birds avoid nesting on ground to build nests on shrubs, forest patches and in caves (Töpfer and Gedeon, 2020), small mammals seek shelter in shrubs and rocky outcrops, also borrow shallow chambers underground (du Preez and Brown, 2011). On the other hand, girdle lizards (Cordylidae) vary in morphology and behaviours along mountain altitude gradient depending on biotope they inhabit, in that serpentine girdle lizard (*Chamaesaura* spp.) has reduced limbs and greatly elongated occupying lower grassy Afromontane ecoregion, while flat girdle lizard (*Platysaurus* spp.) are highly flatten and bauplan allow them to use narrowest cliff cracks to retreat occupying the higher sub-Alpine ecoregion (Loveridge, 1944; Lang, 1991). Most ground-dwelling arthropods reside under rocks and stones forming semi-social groupings (Patrick and Patrick, 2012), while other orthopteran and dipteran insects have secondarily lost wings due to strong winds, high atmospheric pressure, and cold environment (Korner, 2007; Pratt et al., 2008), and lastly, ability of arthropod lepidoptera species to exist in different colour variations along escarpment elevation ecozones gradient (Brehm et al., 2003; Trewick and Morris, 2008).

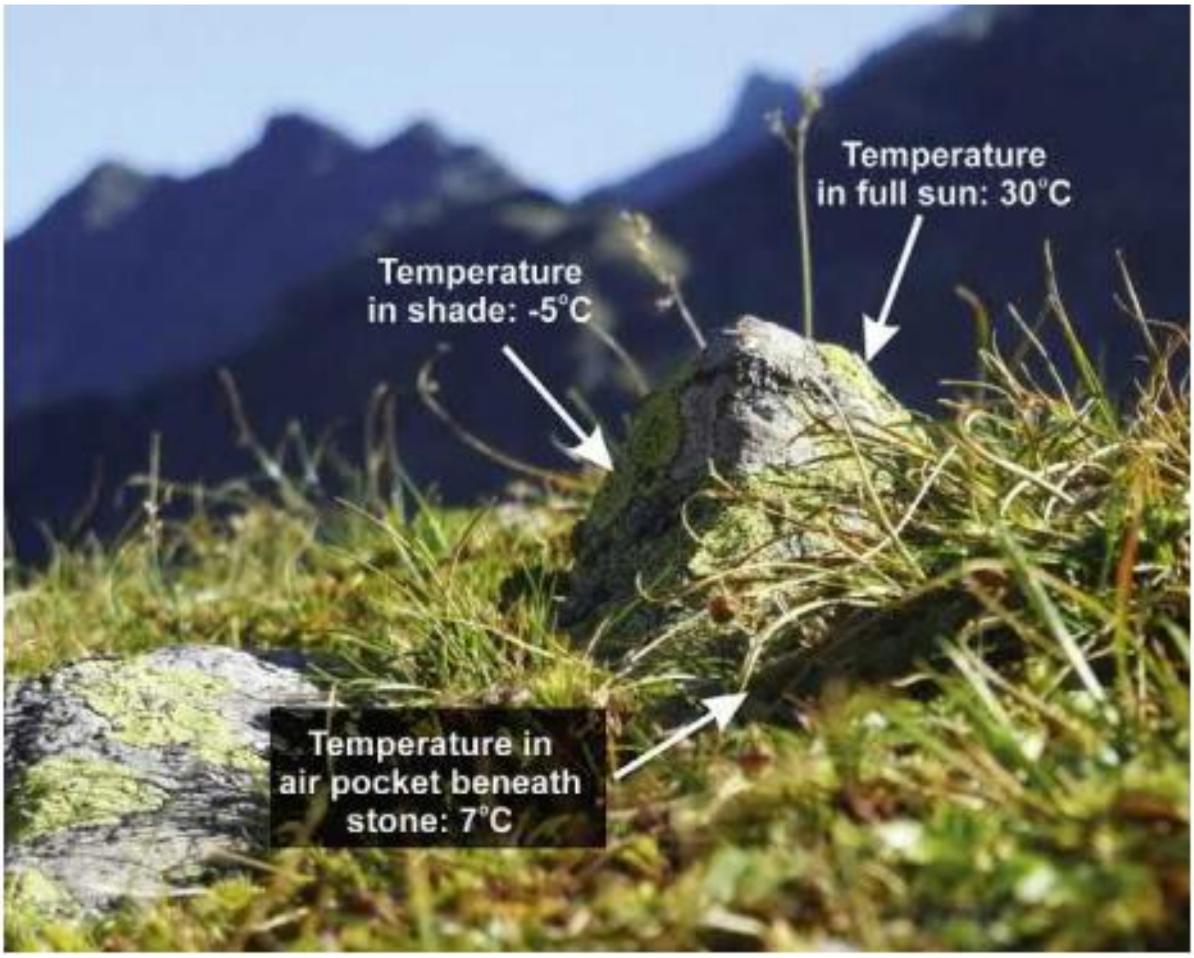

Figure 1.4: Illustration of differences in temperature regimes created in microhabitat shown for ground surface, rock shaded and sun exposed sides (adopted from Elias, 2020).

Ambient temperature fluctuation

Southern African mountain systems are affected by temperate seasonal periodic regional climate differences of summer or winter, and daily microclimate fluctuations for day or night, as well as slope aspect and rock boulder geometry. Cold tolerance responses vary among species depending on taxa and cold-inducing environmental factors (wind, snow, frost, frozen soil and water pools). Except for annual plants that lose their vegetative state and die in the winter period, other plants that overwinter have narrower leaves to reduce surface area, limiting tissue damage and oxidative stress, while those at higher elevation have a foliar freezing resistant cryoprotectant mechanism for low night temperature, during frost and snow cover (Gilbert et al., 2018). Most animal taxa have synchronised life cycles with active stages (feeding juvenile and adult) in spring and summer (Walsh et al., 2018), and overwintering will be an inactive egg or pupae stage during the cold periods, especially seen within frog and insect species (Storey and Storey, 1988). Again, most girdle lizards of *Cordylus* spp. and *Pseudocordylus* spp. genera (Cordylidae) occupying colder upper Afromontane and sub-Alpine ecoregion are viviparous (live birth) to avoid incubating eggs, which is a rare phenomenon in squamates (Lang, 1991). Small mammals like ice rats (*Otomys sloggetti*) huddle together in burrows for warmth, thus protecting from cold days or nights and strong winds (du Preez and Brown, 2011). Arthropod tolerance to low temperature is a dynamic phenomenon in that body temperature varies depending on their ambient surrounding conditions, and hence are considered poikilothermic. So, in order for arthropods to maintain a more or less stable internal temperature, the insect must continually select suitable microhabitats (Figure 1.4). Arthropods that tolerate freezing temperatures avoid freezing by either structural morphology barriers (harden tan exoskeleton) e.g. in adults of Carabidae (Coleoptera), physiological (super cooling ability or chill tolerant) e.g., in adult of Tenebrionidae (Coleoptera), and biochemical (metabolic synthesis of haemolymph polyhydric compounds or antifreeze proteins, for cryoprotection) e.g., in aquatic nymphs of Libellulidae (Odonata) adaptations (Rivers, 2008; Toxopeus and Sinclair, 2018).

Moisture and rainfall availability

Another important limiting factor for high-altitude biotic communities is the moisture and seasonal availability of water in various elevation ecoregions. Generally, plants show a range of strategies to deal with drought periods (and edaphic moisture loss) in that perennial grass (*Themeda triandra*, *Eragrostis* spp., *Andropogon* spp, *Aristida* spp.) avoids the low rainfall period by state of dormancy, whereas the vegetative shoot ceases growth and the above-ground tissues die (reducing water usage) but below-ground rhizoids or stolons reduce metabolic activity rates to conserve water usage; some annual herbaceous plants (*Oxalis* spp.: Oxalidaceae, *Monopsis* spp.: Lobelioideae) synchronise life cycle with drought period by hardened seed stages to avoid dedication (Archibold, 1995; Moffett, 1996). Drought-tolerant grass (*Festuca* spp, *Merxmuellera* spp.) has shortened narrowed leaf blades, sunken stomata, and thick cuticles that allow it to retain vegetative tissues during drought the period that occupies the sub-Alpine ecoregion (Archibold, 1995; Bentley and O'Connor, 2018; Sylvester et al., 2020). Amphibians are probably the most sensitive taxa to changes in temperature and water availability in the environment (du Preez and Carruthers, 2009). Although African river stream frogs (Prxyicephalidae) occupy sub-Alpine ecoregion plateaux, they are associated with permanent water bodies (Channing, 2001; Bates, 2002), the Phofung stream frog *Amietia hymenopus* prefers to lay eggs under ice cold winter water, while the winter active Maloti river frog *A. vertebralis* tadpole and adult synthesis polyhydric protein compounds with slow metabolic rates to avoid tissue damage by freezing (Lambiris, 1989; Bates, 2004). Most aquatic insects enter into drought resistance stages as aerial adult or juvenile diapause stage for Trichoptera and Plecoptera occupying sub-Alpine ecoregion, while species of Odonata and Hemiptera synchronise life stages to avoid drought season of the Afromontane ecoregion (Brock et al., 2003; Lytle, 2008).

Grassland wildfire (veld fire)

Some ecological factors that have a paradox effect on ecosystem patterns and processes are fire and herbivory, on which the majority of sour grasses build up fuel in dried biomass and use veld fires to clear previous biomass to give way to new growth in lower Afromontane and valley bottom ecoregions, and while a sparse pattern of tuft or tussock grass makes natural fire buffers to restrict blazing in sub-Alpine ecoregion (Archibold, 1995). Some angiosperms occurring in fire-prone environments have fire-related traits that as a result of direct fire (flames) or indirect effect such as heat, smoke and charing (Paula et al., 2008). Plants with fire related traits have evolutionary adaptive properties that include fire-stimulated re-sprouting (Poaceae), fire-stimulated flowering (Amaryllidaceae), fire-stimulated seed release (Fabaceae) and fire-stimulated germination (Orchidaceae) response action (Snyman, 2003; Lamont et al., 2013). There are some special woody shrub plants with a thick fire-resistant bark, and distant apical foliage growth to avoid intense flames that include *Luecosidea sericea*, *Protea caffra*. and *Cussonia paniculata* shrubs (Mucina and Rutherford, 2006; He et al., 2016). Most mobile animals flee relatively slow burning fires and take refuge underground or in neighbouring moist biotopes (Little et al., 2013; Pausas, 2019). The changes seen in arthropod assemblages after fire events include the increase of scavenger groups such as Staphylinidae and Tenebrionidae in grasslands and savanna biomes (Hartley et al., 2007; Certini et al., 2021).

The high-altitude mountain ecosystem contains indigenous highlands biotopes with unique and sensitive biotic communities, biodiversity, landscapes or physical features and ecosystem processes and services that are worth studying and conserving, to develop biotic assessments and monitoring tools for better management and sustainability of temperate grasslands in southern Africa.

Choice of study area and sites

Protected areas, biodiversity and ecosystems conservation

Temperate grassland bioregions remain in a dilemma of how to maintain livelihoods and conservation, with approximately 60 % irreversibly degraded, 31.78 % active for land usage (cultivation, forestry plantations, urban and industrial areas, mines and quarries) and only 2.2 % are formally protected for conservation (Table 1.1) (Carbutt et al., 2011; New, 2019) in southern Africa, and while there is an estimated 3.4 to 5.5 % of the world's temperate grassland biome is protected (Bertzky et al., 2012). The grassland biome in South Africa is currently one of the most threatened biomes (Kaiser et al., 2008). Furthermore, grassland conservation is currently biased towards highland regions mostly in Drakensberg and a small percentage for Mesic Highveld, Dry Highveld and sub-Escarpment grassland vegetation bioregions with least protected areas (Carbutt et al., 2011). Although there are plans to increase the number of grassland biome protected areas, there is still a lot of uneven conservation efforts by both government and private programs (Dudley, 2008; Borrini-Feyerabend et al., 2013).

Table 1.1: The conservation status of temperate grasslands in southern Africa (adopted from Carbutt et al., 2011).

Broad vegetation unit (bioregion)	Total area (km^2)	Area in Pas (km^2)	Protected area (km^2)	Transformed (km^2)	Transformed (%)
Drakensberg Grassland	42,177	2,477.48	5.87	8.222	22.30
Dry Highveld Grassland	117,753	1,785.57	1.52	32.717	31.51
Mesic HIghveld Grassland	125,044	1,996.70	1.60	51.689	42.91
sub-Escarpment Grassland	75,615	1,080.19	1.43	27.547	39.60
Summation for Grassland biome	**360,589**	**7,339.94**	**2.04**	**120,175**	**33.33**

The IUCN definition of protected areas was adopted as 'a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve long-term conservation of nature with associated ecosystem services and cultural values' (IUCN, 2008; Boitani et al., 2008). Protected areas are further expanded into six management categories as follows in Table 1.2;

Table 1.2: Six management categories and descriptions for types of protected areas (adopted from IUCN, 2008)

Category	Description
Ia	Strict nature reserve: Strictly protected for biodiversity and also possibly geological/ geomorphological features, where human visitation, use and impacts are controlled and limited to ensure protection of the conservation values.
Ib	Wilderness area: Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, protected and managed to preserve their natural condition.
II	National park: Large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III	Natural monument or feature: Areas set aside to protect a specific natural monument, which can be a landform, sea mount, marine cavern, geological feature such as a cave, or a living feature such as an ancient grove.
IV	Habitat/species management area: Areas to protect particular species or habitats, where management reflects this priority. Many will need regular and active interventions to meet the needs of particular species or habitats, but this is not a requirement of the category.
V	Protected landscape or seascape: Where the interaction of people and nature over time has produced a distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI	Protected areas with sustainable use of natural resources: Areas that conserve ecosystems, together with associated cultural values and traditional natural resource management systems. Generally large, mainly in a natural condition, with a proportion under sustainable natural resource management and where low-level non-industrial natural resource use compatible with nature conservation is seen as one of the main aims.

A recent report indicated that Africa has more than 1,812 national parks covering a total surface area of 3,112,027 km^2 of the continent, and while in sub-Saharan Africa alone, more than 1 million km^2 of land out of 23 million km^2 surface area cover, representing approximately 4 %, has been reserved as national parks (Muhumuza and Balkwill, 2013). In southern Africa, four grassland vegetation bioregions, namely Drakensberg Grassland (Gd), Dry Highveld Grassland (Gh), Mesic Highveld Grassland (Gm), and sub-Escarpment Grassland (Gs) are protected with this formally established the Golden Gate Highlands National Park, uKhahlamba Drakensberg Park World Heritage Site and Sehlabathebe National Park, that are part of a joint Maloti-Drakensberg Transfrontier Conservation Area between South Africa and Lesotho (Carbutt et al., 2011).

Golden Gate Highlands National Park (GGHNPark*)*

South African National Parks (SANParks) are the custodian that was formed by the National Park Bill of 1926 with legislated policies that formally recognised the establishment of national parks endorsed by the National Park Act of 1976 (SANParks, 2013a); which according to IUCN protected area management category II 'are large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities' (Dudley, 2008). Current National Environmental Management: The Protected Areas Act (57 of 2003) recognises 19 national parks that comprise SANParks in eight main terrestrial biomes and at sea, in which the Golden Gate Highlands National Park (GGHNPark) is the only park exclusively representing the grassland biome (SANParks, 2013b).

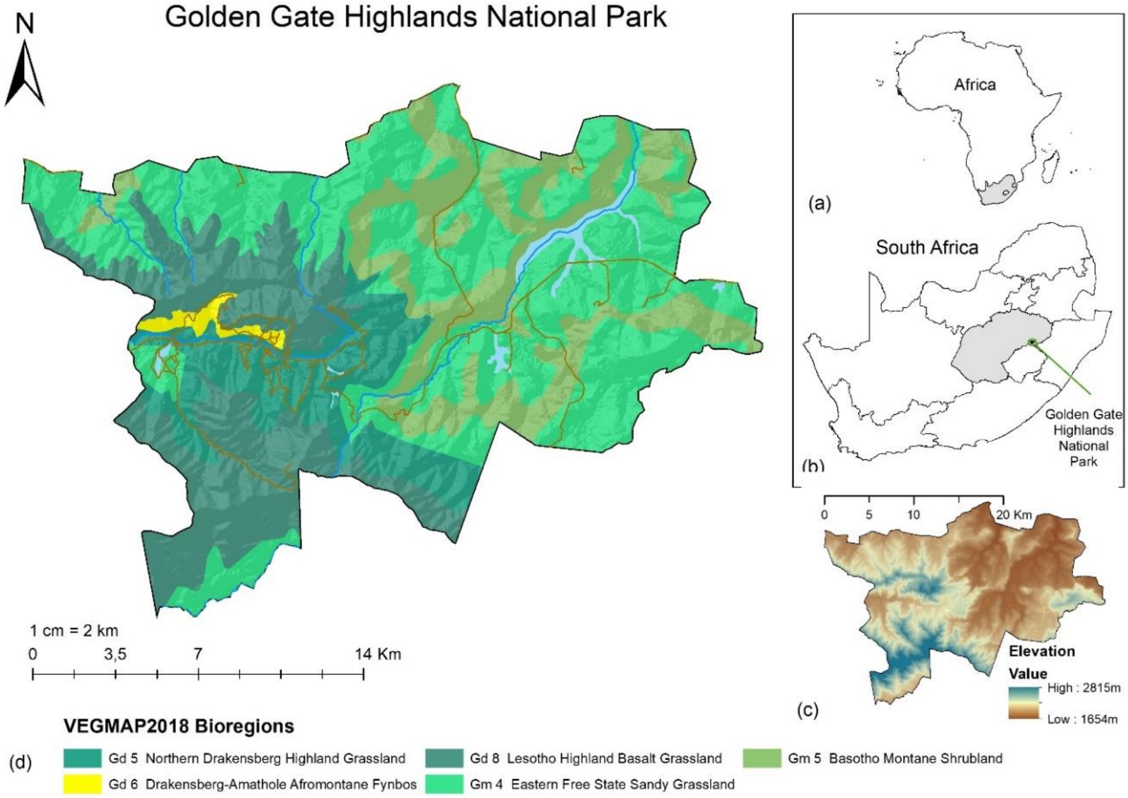

Figure 1.5: Location of the study area illustrating (a) the African Continent, (b) the eastern section of the Free State Province highlighted in South Africa outline, (c) the elevation and hillshade of GGHNPark created using the DEM image from USGS Explore Earth, and (d) highlighting the vegetation types present in GGHNPark (adopted from SANBI VEGMAP 2018).

GGHNPark is located on the north-eastern border of Lesotho and the eastern Free State Province of South Africa (Figure 1.5), along foothills of the Rooiberg Mountain Range which forms part of northern part of the Maloti-Drakensberg Mountains at an elevation between 1892 m and 2837 m (Wessels and Wessels, 1991; Mol and Viles, 2010). The park covers a surface area of about 342 km^2 (32758 ha) and according to Grab et al. (2011), GGHNPark falls under the geomorphology of the Karoo Basin Supergroup comprising of the volcanic Drakensberg Group with sedimentary sills and dykes at high-altitude, and the lower Stormberg Group which is made up of dynamic assortment of sandstones, namely;

- <u>Molteno Formation</u>: thickness range 10 m - 100 m inner layer, with fine to very coarse grains, primarily quartz-rich feldspathic wackes sandstones (Eriksson 1984).
- <u>Elliot Formation</u>: thickness range 28 m - 150 m mid-layer, with red-coloured, argillaceous fine to coarse sandstone grains. well known for GGHNPark for preserving vertebrate fossil content (late Triassic (200 - 230 m.y.a.) dinosaur fossil eggs with foetal skeletons) (Eriksson, 1984).
- <u>Clarens Formations</u>: thickness range 115 m -195 m outer layer, with light-coloured, fine-grained sandstones, sandy siltstones and mudstones, and

subordinate coarser-grained components. The layer is famous for spectacular variety geometry of landforms and extensive cliffs (Eriksson, 1984).

The geomorphology of the landscape at GGHNPark is complemented by the temperate sub-Alpine and Afromontane grassland bioregions, making the ecology and ecosystem processes of the park interesting to study and conserve. The GGHNPark falls into the weather and climate zone of the eastern Free State province, which is characterised by a higher summer rainfall receiving about 700 mm to 1630 mm annual precipitation, with a warmer mean annual temperature ranging from 13°C to 30°C, and winter rainfall of between 500 mm and 700 mm annual precipitation, and with a colder annual temperature ranging from 1°C to 26°C (Grab et al., 2011). The highland bioregion is within winter frost, moist cold and snow region that is mainly characterised by graminoids, forbs, herbs, woody shrubs and trees (Mucina and Rutherford, 2006; Telfer et al., 2012).

In this study, we recognise the biotope units as vegetation type categories described by Mucina and Rutherfiord (2006), with Figure 1.5. that illustrates GGHNPark is represented by two main grassland biome subdivisions that include;

1. Drakensberg Grassland vegetation bioregion
    - Northern Drakensberg Highland Grassland (Gd5)
    - Drakensberg-Amathole Afromontane Fynbos (Gd6)
    - Lesotho Highlands Basalt Grassland (Gd8)
2. Mesic HighIveld Grassland bioregion
    - Eastern Free State Sandy Grassland (Gm4)
    - Basotho Montane Shrubland (Gm5)

All represented vegetation types were initially considered for sampling, but species data from only three, namely Northern Drakensberg Highland Grassland, Eastern Free State Sandy Grassland and Basotho Montane Shrubland, were reliable and analysed for the study. Data from Drakensberg-Amathole Afromontane Fynbos and Lesotho Highlands Basalt Grassland sampling sites were excluded for analysis due to inconsistency caused by Chacma baboon (*Papio ursinus*) troops persistently tempering and removing field arthropod sampling equipment in all sampling stations. Below are images of selected sites and descriptions are given in their respective designated chapters.

The names given to the sampled sites are based on previous farm names that the park has adopted for the allocation and zoning sections. Selected sites for the

Selected sites for the Eastern Free State Sandy (Gm4) vegetation type included;

- Korfshoek (Korf_Gm4), Figure 1.6
- Honing Kloof (Honi_Gm4), Figure 1.7
- Twijhoek (Twij_Gm4), Figure 1.8
- Welverdiend (Welv_Gm4), Figure 1.9

Selected sites for the Basotho Montane Shrubland (Gm5) vegetation type included;

- QQ Mountain (QQMo_Gm5), Figure 1.10
- Avondrust (Avon_Gm5), Figure 1.11
- Diepkloof (Diep_Gm5), Figure 1.12
- Silasberg (Sila_Gm5), Figure 1.13

Northern Drakensberg Highlands Grassland vegetation type (Gd5) included;

- Blesbok Loop (Bles_Gd5), Figure 1.14
- Oribi Loop (Orib_Gd5), Figure 1.15
- Diepkloof (Diep_Gd5), Figure 1.16
- Tossline (Toss_Gd5), Figure 1.17

Mesic Highveld Grassland

Eastern Free State Sandy Grassland (Gm4)

Figure 1.6: Korfshoek site (Korf_Gm4) a biotope selected for the Eastern Free State Sandy Grassland in GGNHPark.

Figure 1.7: Honing Kloof site (Honi_Gm4) a biotope selected for the Eastern Free State Sandy Grassland in GGNHPark.

Figure 1.8: Twijhoek site (Twij_Gm4) a biotope selected for the Eastern Free State Sandy Grassland in GGNHPark.

Figure 1.9: Welverdiend site (Welv_Gm4) a biotope selected for the Eastern Free State Sandy Grassland in GGNHPark.

Basotho Montane Shrubland (Gm5)

Figure 1.10: QQ Mountain site (QQMo_Gm5) a biotope selected for the Basotho Montane Shrubland vegetation type in GGNHPark.

Figure 1.11: Avondrust site (Avon_Gm5) a biotope selected for the Basotho Montane Shrubland vegetation type in GGNHPark.

Figure 1.12: Diepkloof site (Diep_Gm5) a biotope selected for the Basotho Montane Shrubland vegetation type in GGNHPark.

Figure 1.13: Silasberg site (Sila_Gm5) a biotope selected for the Basotho Montane Shrubland vegetation type in GGNHPark.

Drakensberg Grassland

Northern Drakensberg Highlands Grassland (Gd5*)*

Figure 1.14: Blesbok Loop site (Bles_Gd5) a biotope selected for the Northern Drakensberg Highlands Grassland in GGNHPark.

Figure 1.15: Oribi Loop site (Orib_Gd5) a biotope selected for the Northern Drakensberg Highlands Grassland in GGNHPark.

Figure 1.16: Diepkloof site (Diep_Gd5) a biotope selected for the Northern Drakensberg Highlands Grassland in GGNHPark.

Figure 1.17: Tossline site (Toss_Gd5) a biotope selected for the Northern Drakensberg Highlands Grassland in GGNHPark.

General overview of the thesis

This study aimed to evaluate the state of environmental and ecological conservation of the sub-Alpine and Afromontane ecoregions found in GGHNPark, applying the Biotope Quality Index (*BQI*) (Bredenhand, 2014) as a monitoring tool that uses species assemblages and populations (as bioindicators) to help assess the effects of environmental disturbances in each grassland vegetation bioregion of the highlands. The study was designed to demonstrate different ecosystem, biotic community and GIS ecology analysis applications for which *BQI* can be used in conservation biology. The sites considered for sampling had a gradient of ecological (soil erosion) and human-influenced disturbances (historical agricultural activities, housing buildings, road constructions, pollution), plant encroachment and invasion in different vegetation types. The study also proposes a Biotope Conservation Status (*BCS*) measure as an additional step for *BQI* which assists to determine impact of the main disturbance variable on vegetation type natural environmental conservation status.

The thesis is presented in an investigative and experimental research base with chapters in the article manuscript format, including a biotic index and bioassessment review chapter and four biotic community research chapters that dealt with assessing impacts of environmental disturbances on community clusters of biotic assemblages and functional biotic groups by demonstrating the application of Biotope Quality Index

(*BQI*) and Biotope Conservation Status (*BCS*) measures as species surrogates for possible usage in conservation management strategies for each selected sampled vegetation type or biotope.

Chapter 2: Biomonitoring and biotic index review: continuous ecosystem evaluation and proposition to establish guidelines for biotic index nomenclature

This review article highlights the development and usage of biotic assessment in environment domains (air/atmospheric, water/aquatic and land/terrestrial) in quantifying impact of environmental disturbance onto biotic communities and ecosystem functioning and services. The chapter also indicates statistical or arithmetic syntax formulation on main ecological and environmental topics that different biotic indices and bioassessment focus on and in which country/state as well as world regions. We also propose standard guidelines to assist in description and assign name to biotic indices (or bioassessment) could assist to create a universal working database, avoid ambiguity and conflicts to establish newly developed indices, as well as assist to track continuity for amendments, adaptations and modifications to a developed index, in contributing to continuous biomonitoring and evaluation of ecosystems.

Chapter 3: Adopting Biotope Quality Index for Vegetation Ecology and its application as a Biomonitoring tool for the Mesic Highveld Grasslands in Golden Gate Highlands National Park, South Africa.

In this chapter, the Biotope Quality Index (*BQI*) was adopted using plants as sampling data taxon and vegetation floristic composition as a measure to indicate the impact of environmental disturbance using ambient standard scores per sampled site for the Mesic Highveld Grassland. We tested BQI_{veg} statistical compatibility against three commonly used plant abundance-cover scales i.e. Braun-Blanquet, Domin and percentage ratios, then also tested *BQI* in major vegetation representative taxa divisions, namely for 1. dicots (BQI_{dic}); 2. monocots (BQI_{mon}); 3. Graminoids (BQI_{gmd}), 4. forbs (BQI_{for}), 5. Poaceae grasses (BQI_{poa}) and with all species data (BQI_{veg}) for the Eastern Free State Sandy Grassland and the Basotho Montane Shrubland vegetation types. Environmental disturbances and their effects on plant abundance-cover and communities were identified and quantified using species richness and diversity, floristic community clustering responses against disturbance gradient. Then, the addition of the Biotope Conservation Status (*BCS*) to indicate the impact of disturbances that had a significant influence and the use of the Biotope Quality Index (BQI_{veg}) as a geospatial ArcGIS Feature analysis parameter to illustrate the impact on each Mesic Highveld Grassland bioregion.

Chapter 4: Evaluating effects of Plant Encroachment and Soil Erosion towards Arthropod Communities using Biotope Quality Index for Eastern Free State Sandy Grassland in Golden Gate Highlands National Park, South Africa.

Chapter 5: Biomonitoring alien plant invasion and soil erosion effects onto Basotho Montane Shrubland Grassland using Biotope Quality Index Arthropod Assemblages in Golden Gate Highlands National Park, South Africa

It was noticed from the data that each bioregion of Mesic Highveld Grassland in Chapters 4 and 5 has different environmental disturbances that affect them in various ways. Therefore, for the Eastern Free State Sandy vegetation type, arthropod abundances and assemblages were used to highlight the impact of plant encroachment and soil erosion on species richness, diversity and biotope quality (*BQI*), and for the Basotho Montane Shrubland bioregion we highlight the impact of plant invasion and soil erosion for similar species surrogate measures. Then, the Biotope Conservation Status (*BCS*) was included to indicate impact of significant disturbances, as well as to demonstrate the use of Biotope Quality Index (*BQI*) as a geospatial GIS vector for a visual map illustrating the impact on each Mesic Highveld Grassland bioregion.

Chapter 6: Ecological bioassessment using Biotope Quality Index to evaluate Northern Drakensberg Highlands grassland in Golden Gate Highlands National Park, South Africa

Lastly, the use of the Biotope Quality Index (BQI) was demonstrated as a potential species surrogate for rapid bioassessment of environmental conservation conditions in the Northern Drakensberg Highlands Grassland vegetation type. We used both arthropod assemblages and plant abundance-cover communities as bioindicators to quantify and indicate current natural ecological and environmental conditions of the Northern Drakensberg Highlands Grassland in Golden Gate Highlands National Park.

Chapter 7: General conclusion

In conclusion, the adoption of the Biotope Quality Index (*BQI*) was managed using vegetation ecology data to highlight the impact of environmental disturbance for Mesic Highveld and Drakensberg Grassland bioregions. The various applications of the Biotope Quality Index (*BQI*) were also demonstrated to indicate the ecological and environmental disturbance effects on biotic communities. Additionally, Biotope Conservation Status (*BCS*) was proposed as an additional step to indicate the impact of significant disturbances in different vegetation type. Therefore, from the results and application demonstrations, the *BQI* recommended to be adopted as a species surrogate and biodiversity conservation management tool for Afromontane grassland bioregions in Golden Gate Highlands National Park and related areas.

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CHAPTER 2 : Biotic Indices Review: Continuous Ecosystem Evaluation and Proposal to Establish Guidelines for Biotic Indices Nomenclature

Abstract

Bioassessments and biotic indices are used to evaluate the current state and changes in the ecosystem over time, through aggregated function and scoring systems for better conservation monitoring. The brief review focuses on 108 commonly used bioassessments and biotic indices retrieved from 4816 published articles and electronic report databases. The domains of the aquatic ecosystem had developed and established more than the terrestrial and atmospheric domains. Most developed countries, namely USA, Great Britain, Canada, France, Spain, Australia, and the Netherlands, have a diverse interest of stakeholders (research, conservation management and government departments) playing a role in data management systems for biomonitoring tools, while most developing countries are still in the developmental phase on the continent of South America and Africa. Propositions of standard guidelines to describe and assign names to biotic indices could assist in creating a universal working database, avoiding ambiguity and conflicts to established and newly developed indices, helping track continuity for amendments, adaptations and modifications to a developed index, contributing to continuous biomonitoring and evaluation of ecosystems.

Keywords

Bioassessment, Biomonitoring, Continuous ecosystem monitoring, Ecological disturbance, Ecosystem functioning, Environmental quality indices

Introduction

Anthropogenic activities and natural disturbances have influenced the deterioration of natural ecosystems worldwide that include overexploitation of resources (Seppelt et al., 2011), deforestation, agriculture and mining land transformation (Swift et al., 2004), industrial waste pollution (Hilsenhoff, 1987a), air pollution. In turn, these anthropogenic activities play a role in extreme weather patterns, climate change (Brotherton and Joyce, 2015; Murphy et al., 2020) and natural disasters (Naeem et al., 2009; Singh et al., 2018). Disturbance can be a one-time event or occur over a longer period depending on the type and magnitude, which then can cause different impacts at various ecosystem levels (Orzell et al., 2003). Growing concerns of natural ecosystem deterioration create conservation challenges for managers, scientists, and policy makers which drives recognition of the need to develop ecological assessment and monitoring tools for better restoration, conservation, and protection of natural resources (Wallace, 2007; Mace et al., 2012; Schneiders et al., 2012). Commonly used environmental monitoring efforts make use of biological components (indicators or surrogates) within an ecosystem to assist in developing assessment strategies (Gonçalves and Menezes, 2011). Monitoring approaches for environmental deterioration varies and these biological components are used at different levels (individual species, population, community and ecosystem) (Carignan and Villard, 2002; Tierney et al., 2009). Among pioneering the use of biological components for ecological assessments and disturbance classifications started during early developments of the Saprobien System by Kolkwitz and Marsson (1908) to evaluate sewage pollution effects in river systems using plankton and zooplankton and (Carignan and Villard, 2002; Tierney et al., 2009) and used macroinvertebrates in the Illinois River. This showed that some species or communities are sensitive to changes in environmental conditions that could have a direct or indirect influence on their survival (Niemi and McDonald, 2004; Dizdaroglu, 2015). The consideration of species or communities that can give some form of indication on conditions and status of the environment has contributed better to conservation practices.

Biological monitoring works on the basic assumption that an ecosystem in its natural or pristine state is with less or no disturbance (Cairns et al., 1993), therefore would provide enough resources such as food and habitat (Niemeijer and de Groot, 2008), perform ecological function efficiently (Bredenhand, 2014), resilience to disease (Tilman et al., 2014), to maintain population and community sizes (Loreau, 1998), for a better sustainable ecosystem (Arnoldi et al., 2016). The most common effective management approach is the establishment of quality indices to help with monitor and assess the state of environmental or ecological conservation. Most biotic indices data consist of species presence or abundance (incidence), or abundances values collected from either individual or sample unit (Chao, 2001; Colwell et al., 2012), measured physical-chemical parameters (Maxted et al., 2000) and descriptive weighted disturbance (Landwehr and Deininger, 1976; Lobdell et al., 2011) or a combination of these methods. Disturbance tolerance values are assigned to taxa depending on their sensitivity to exposure (Hilsenhoff, 1987b), or taxa are grouped (ranked) based on a distribution range of described sites (Stark, 1998). In addition, biotic index values are then graded depending on the degree of disturbance, referred to as tolerance value (Hilsenhoff, 1987b), taxon scores (Armitage et al., 1983), quality values (Chutter, 1972a), and sensitivity grade number (Chessman, 1995). The variation of biotic indices brings flexibility in practice and application, in that employed methods differ i.e., qualitative and quantitative protocols (Murphy, 1978), as well as processed parametric and non-parametric methods separately (Chiu et al., 2014). These results are later superimposed to reflect an intrinsic state of ecosystem conditions. Stark (1998) highlights that a taxon has a range of sensitivity to disturbances present in a specific geographical location sampled, and hence each biotic index should be specifically developed for a particular region. Biological indices or assessments have been developed globally considering heterogeneity within biological communities, complex ecological interactions and processes (van Oudenhoven et al., 2012), exposure to multiple parameters simultaneously capable of causing various stresses (Rosenberg et al., 2004). However, complexity in ecology has the benefits of helping to maintain the status and functioning of ecosystem conservation (Finazzi et al., 2013). On the other hand, complexity presents challenges to fully understand ecological structure, interactive processes, and function, hence the establishment of a holistic approach to better explain 'the complexity' using biotic assessments from species to ecosystem level, employing laboratory bioassays and field trail observations, single to multiple parameter influence analysis, specific to the approach of general methods, then later developing more efficient, cost effective, and better predictive (accurate) models.

Suggestions have been made in relation to reference databases and sites to compare both laboratory bioassays and field observations. Reynoldson et al. 1997) explained how reference sites can be used as control sites to create reference conditions to compare test sites, with physical, chemical and biological attributes, to set baseline conditions. Similarly, Maxted et al. (2000) and Gerritsen (1995) defined the biotic index (operation) as a function developed from reference databases that reflects sensitivity to biological degradation. Furthermore, two schools of thought recognise biotic indices in terms of datasets type based, namely as qualitative, quantitative and (or) mixed method type (Murphy, 1978), as well as multimetric and multivariate methods (Reynoldson et al., 1997) to make objective and informed decisions. Both analytical approaches differ in practice, but are developed using similar principles and using the same data. Vlek et al. (2004) highlights that the development of biotic indices began around the 1900's with the Saprobien System (Kolkwitz and Marsson, 1909) using the Illinois River bottom and shore fauna species data (Richardson, 1929) by selecting indicator organisms in Germany and Netherlands. Then followed by the 1940s statistician developing a species community diversity function (Simpson,1949; Weaver and Shannon, 1949). The ecosystem disturbance scoring system was introduced with the establishment of the Trent Biotic Index (Woodiwiss, 1964) in Britain, which was followed later by Indice Biotique (Tuffery and Verneaux, 1968) in Italy, as well as the Biotic Index for the South African rivers (Chutter, 1972) in South Africa. The trend grew toward the 1980s seeing establishment of more inclusive multimetric methods with development of Index of Biotic Integrity (Karr et al., 1986) and British Biological Monitoring Working Party (Armitage et al., 1983). Other inclusive assessment types were the multivariate methods, namely the River Invertebrate Prediction and Classification System (RIVPACS) (Wright et al., 1984) and Macroinvertebrate Community Index (Stark, 1993). Subsequently, there was a positive suggestion highlighted that the integration of multimetric and multivariate methods might be beneficial to the development of predictive modelling tools for better ecosystem structure and function monitoring strategies (Collier, 2009).

Currently, the regularly used developed biotic indices have shown a great variation range in focus topics that contribute to ecosystem condition assessments, including species richness (Chao and Chiu, 2016), diversity (Brown, 1997; Gessner et al., 2001), feeding groups (Ramírez et al., 2014), functional groups (Brussaard, 1998; Faber, 1991), disturbance response towards pollution (Hilsenhoff, 1987a; Kowalska et al., 2018; Yuan et al., 2020), habitat or biotope quality (Diaz et al., 2004; Johnson, 2007), habitat stability (Tilman, 1996; Allesina and Tang, 2011), ecosystem resilience (Willis et al., 2010; Loreau and Haegeman, 2015) and ecosystem functioning (Gessner et al., 2001; Lavorel and Garnier, 2002). Furthermore, three major ecological domains, namely atmospheric (air) (Kassomenos et al., 2012), aquatic (water) (Lenat, 1993; Metcalfe, 1989) and terrestrial (soil) (Ayuke et al., 2011) have been considered, as well as a fair representation of biological life-forms such as Bacteria (proteobacteria, acidobacteria, actinobacteria) (Schloter et al., 2018; Hermans et al., 2020), Fungi (lichen) (Perlmutter, 2010; Will-Wolf et al., 2015), Bryophyta (moss) (Author, 1990; Govindapyari et al., 2010), Plantae (flowering plants) (Puczko et al., 2018) and Animalia (nematods, annelids, birds, fish, insects) (Sutadian et al., 2016; Bünemann et al., 2018).

However, in addition to challenges posed by open and complex natural ecosystems to formulate a realistic biotic index, ecosystem conditions are subjected to change over time (Colwell et al., 2012), some are related to the establishment of biotic indices whereby variable effects were eclipsed (overshadowed) (Lumb et al., 2011; Mori and Christodoulou, 2012), and inconsistencies during follow-up supplementary long-term assessments. Many studies have proposed possible solutions to minimise and reduce above indices arithmetic formulation syntax challenges such as sampling rarefaction (Chao et al., 2014), variable bias (Landwehr and Deininger, 1976; Kachroud et al., 2019), incomplete data sets (Rana and Ganguly, 2020), and model assumption data corrections (Dejong, 1975; Sutherland et al., 2018). Another challenge is with the nomenclature of a developed bioassessment tool, whereby other names do not align with index definition and application, while others share more than one name, or even have more than one name for different versions. Although initial efforts have been made to assist in the process of developing biotic indices select, observe and measure biological (taxa) variables as well as ecological parameters (Ferreira, 2000; Boonsoong et al., 2009), this review proposes a standard framework to guide qualitative bioassessment or biotic index establishment, definition, nomenclature and application, which might be overseen by regulatory guidelines.

Methods and Materials

Data collection

This brief review includes common developed qualitative bioassessment and biotic indices to highlight their definition, nomenclature and application, as well as additional studies to highlight continuous usage of biotic indices in ecosystem monitoring and conservation studies or programs. The review considered published research articles that were accessible online, and the search engines used were PubMed, Web of Science, ScienceDirect and Google Scholar, the search filter criteria included the year 1960 to 2021, keywords and phrases ‘air or soil or water biotic index’, ‘environmental or ecological biological assessment’, search by country and region ‘Africa’, ’Europe’, ’Canada’, ’USA’, ’Asia’, ’Australia’, search by research focus ‘species stress tolerance’, ‘disturbance sensitivity’, ’biotope or habitat quality’, ‘ecosystem functioning or resilience or stability’.

Data cleaning and validation

The retrieved publications recovered from each search engine were screened for cleaning, validation, and relevance according to the above selection criteria, which resulted in 108 commonly used biotic index cited from 4816 articles (supplementary material: Table 0.1). The Biotic Index recorded information included the establishment state (develop, adapt or modify), authors and year, ecosystem domain (air, soil or water), research focus (functional group, stress tolerance, disturbance sensitivity, biotope quality, functionality, resilience, stability), and arithmetic syntax. Then, for continuous evaluation, each biotic index was used to find relevant studies that have applied it as an evaluation tool.

Data analysis

Comparative statistics using the frequency of citations for each biotic index to create geographical developmental references using the ArcMap (ArcGIS version 10.8) (Crawley, 2002; Crawley, 2012). Multiple correspondence analysis of biotic indices was used to detect similarity in assessment profiles, included domain, taxon, focus endpoints as variables, using the CCA package (R-statistical version 3.6.3) (Oksanen, 2010). A Biotic Indices usage period chart was created to highlight and indicate the continuation of the assessment using Adobe Illustrator (Adobe Creative Cloud) using four aggregation method categories namely additive, multiplicative, weighed means and hybrid arithmetic method (Shumway and Stoffer, 2006).

Results and Discussion

Biomonitoring assessments can be categorised according to Gonçalves and Menezes (2011) into the following forms, namely, protocols for rapid water quality assessments (54%), predictive bioassessment tools (19 %), biotic indices (11 %), ecological indices (10 %) and functional trait groups (6%) respectively. Majority of assessments were carried out in North America (51 %) was mentioned in most biomonitoring assessments, followed by Europe (35 %), Oceanica (9 %), Africa (6 %), Asia (4 %) and the least for South America (4 %) as shown in Figure 2.1. Most biomonitoring assessments were developed to monitor and evaluate aquatic ecosystems, especially rivers (48 %), estuaries (16 %), wetlands (6 %), lakes (6 %) and streams (5 %), while terrestrial systems (11 %) and atmospheric systems (8 %) were developed as well. Concerns about environmental conditions started in the early 1900s when water pollution started to become real by affecting available potable water for humans in most industrialised countries (Dallas, 2021; Eriksen et al., 2021), and so the eminent public health effects led to the establishment of ecosystem assessments. Demonstrations to indicate degradation of water quality were made possible by a shift to include biological organisms and communities by Kolkwitz and Marsson (1909) in Germany and the Netherlands, which initiated the practice of biotic indices in Europe. Additionally, the fishery industry and aquatic recreation activities that influenced surface water biological assessment diversified to include river, lakes, wetlands and estuary systems in most parts of Europe and North America (Swift et al., 2004; Lumb et al., 2011; Lee and Lautenbach, 2016). Then, the rest of the world began to adapt pioneer European indices models by modifying these indices to suit their own regional fauna and water conditions like South Africa (Chutter, 1972b), Australia (Chessman, 2003) and New Zealand (Stark, 1993).

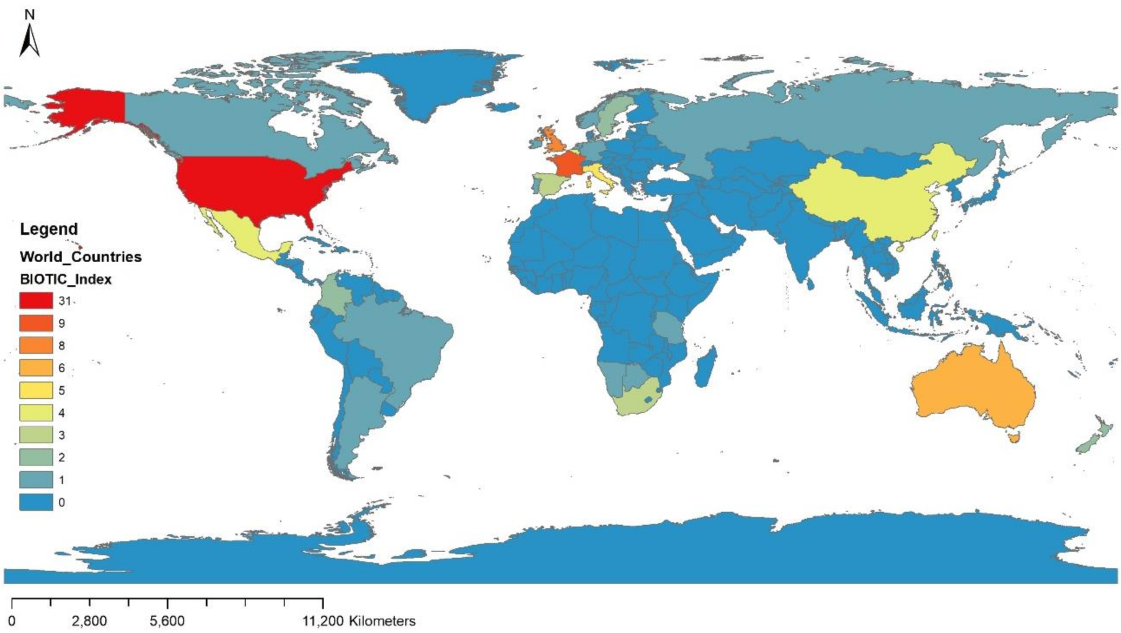

Figure 2.1: World representation for the development of bioassessments and biotic indices.

The inclusion of terrestrial and atmospheric ecosystem assessments also began in Europe, North America and Asia. The most developed biotic indices to assess and monitor terrestrial ecosystems (for land, soil and riparian zones) were derived from existing aquatic data protocols (Figure 2.2). Efforts to monitor ecosystem conditions in the terrestrial domain were also soil degradation-related disturbances (Liu et al., 2014; Vidal Legaz et al., 2017; Bünemann et al., 2018). In contrast, atmospheric biotic indices were developed to address concerns about human health challenges caused by air pollutants (Plaia and Ruggieri, 2011; Kassomenos et al., 2012). The atmosphere as an ecosystem niche for biological communities has not been extensively studied like other domains, however, arboreal lichen organisms (fungal spores) are used to indicate levels of pollution in air quality models (Gorai et al., 2015; Will-Wolf et al., 2015).

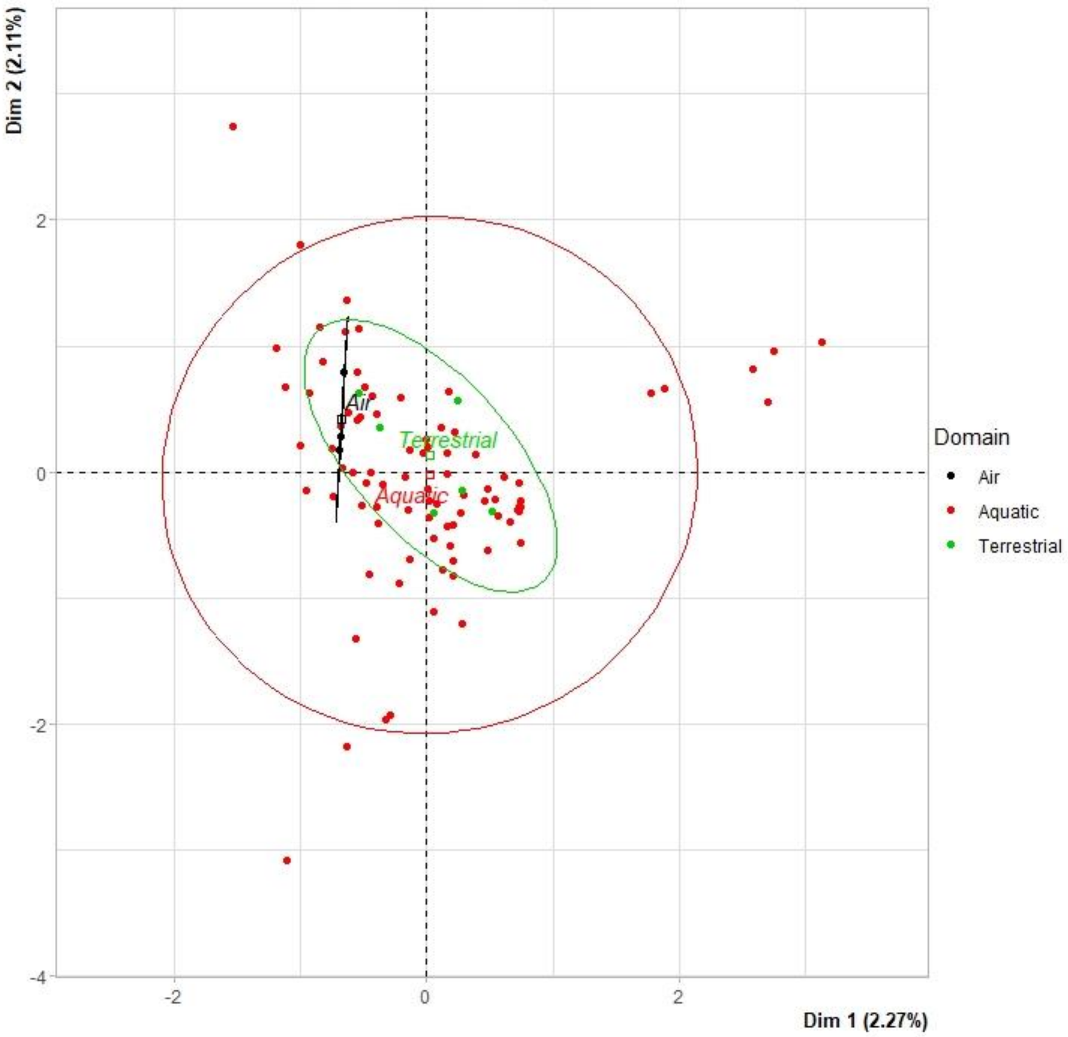

Figure 2.2: Multiple correspondence analysis indicating relatedness of biotic indices citation frequencies, arithmetic formulation and ecosystem domain.

The global acceleration in the use of biotic indices could be seen during the period from 1988 to 2012, as many countries started to consider conservation and commit to global issues around environment and natural ecosystems (Fisher and Kerry Turner, 2008; Kontogianni et al., 2010; Mace et al., 2012). But, the period between 1996 and 2004 showed an rapid increase in the development of biotic indices, since many countries and scientists established new indices, with about 45 % for ecosystem assessments established in that period (Figure 2.3). Again, the shift to include other ecosystem types, consideration of both environmental and public health concerns, as well as inclusive participation from relevant stakeholder, ecologists and environmental practitioners for better ecosystem resource usage and management. Since the suggestion to consider reference datasets and conditions (Reynoldson et al., 1997; Bailey et al., 2014a), many biotic indices have shown continued use by other scientists either for application or to adopt and develop other indices. Most countries from Europe, America, and Australia who considered a reference condition approach have among the most constituently used biotic indices that keep being modified for better outcomes to evaluating ecosystem conditions and climate change modelling (Cairns et al., 1993; Niemeijer and de Groot, 2008; la Notte et al., 2017).

Figure 2.3: Time series bar illustrates the development of biotic assessment and indices and period of use; (a) biotic indices development-usage distribution bar graph and, (b) biotic indices usage frequency indicating continuous use of a developed index protocol, with aggregation methods highlighted with colour variation.

Description and nomenclature of bioassessment and biotic indices

There has been an argument for developing standard guidelines on how to develop bioassessments (Barbour et al., 2006; Abbasi and Abbasi, 2011) and biotic indices (Murphy, 1978; Ponti et al., 2009). There have been constructive and controversial suggestions on how to refer and apply certain biotic indices. Specific features that are involved in building a biotic index should be included to describe terms and define the index. A description of biotic indices takes into consideration a general approach that includes specification of key common aspects, namely ecosystem domain, biological taxa used, environmental disturbance effects, arithmetic syntax, and status condition. Therefore, from the above, standard guidelines can be suggested on how biotic indices should be defined and named. Here are some examples for terms to describe biotic indices;

Single-metric biotic indices,

	Description
1.	could be described by specifying its arithmetic application, i.e., Shannon-Wiener Index is described as the degree of variation for species abundance proportion, by measuring uncertainty of predicting the species for a random sampled community (Barnes et al., 1998). or
2.	could be described by specifying its taxa and ecological focus i.e., Dragonfly Biotic Index (DBI) is described as a compound index based on geographical distribution, conservation status, and ecological sensitivity of dragonflies and damselflies species (Simaika and Samways, 2009), or

Additive-metric approach for multimetric and multivariate indices,

	Description
3.	could be described by specifying ecological domain, ecological focus and involved biotic indices i.e., Benthic Index of Biotic Integrity (B-IBI) is a multimetric tool that describes the biological condition of stream sites and their surrounding habitat based on the diversity and relative abundance of the benthic (bottom dwelling) macroinvertebrates found at the site (Kerans and Karr, 1994), or
4.	could be described by specifying ecological domain, ecological focus, application and involved biotic indices i.e., River Invertebrate Prediction and Classification System (RIVPACS) is a predictive model built from biological and environmental multivariate data used to assesses biological condition of rivers by comparing the taxa observed and reference sites (Wright et al., 2000).

The description of the biotic index should be clearly detailed to avoid confusion and contradiction to its name, description and application. Assigning names to biotic indices is another important step that assists to identify assessment tools that must be applied in specific regions and ecological domains. Chutter (1994) indicated that biological components are unique to a particular local of their distribution and occur in different physiochemical conditions, so adoption for the application of certain biotic indices should modify the test conditions to complement local taxa responses. Newly developed or adoption of an existing biotic index requires an index name to be identified with and what it relates to. Basic principles from already existing codes of international nomenclature for biological organisms, namely International Codes of Botanical Nomenclature (ICBN) and International Codes of Zoological Nomenclature (ICZN), could be adopted. For review purposes, we have put aside choice of language for naming preference, but we will use the standard English language terminology for our proposition examples. The following are some suggestions and recommendations as guidelines for nomenclature of biotic indices (Table 2.1);

Table 2.1: Suggested founding principles for biotic indices nomenclature guidelines:

	Description
1.	the guideline does not interfere with both biological and mathematical scientific procedures, which must not be subject to regulation or restraint, but should be reflected on as a tool for a standard process to assign names.
2.	the nomenclature does not determine the inclusiveness or exclusiveness of any biotic index, but rather provides a name that is to be used for a biotic index after the respective authors have detailed clear research focus and usage limits.
3.	to devise a name-bearing code that allows names to follow a universal 'Principle of Priority' process (ecosystem domain, research focus, taxa used, geographic region, arithmetic syntax, assessment type), and should be applied without infringing on biological and mathematical scientific procedures.
4.	to devise a system to assign names to indices using multiple or 'collective' metrics and variates assessments, in that they should attach a certain 'name-bearing code' following the universal 'Principle of Priority' process.
5.	to devise a system to assign names to indices that are adopted and modified from 'name-bearing' indices.
6.	the application for a name should go through a validation process to avoid ambiguity and conflicts, the use of the same name and acronym for different indices.
7.	the guide should also provide researchers aiming to establish new biotic indices and bioassessments with a clear set of rules on how to assign working names until they are approved. A set of rules should include a system to determine whether any name, previously proposed, is available and with what priority; whether the name requires amendment for its correct use, and to enable the name-bearing type of the index it denotes to be ascertained (and, when necessary, to be fixed).
8.	if needs be, there could be an organised entity structure assigned to oversee the nomenclature process to have efficiency to the process, accountability and sustainability. Its working name could be the 'International Guide for Biotic Indices and Assessment Nomenclature (IGBIAN)'.

Proposed nomenclature for bioassessments and biotic indices

These founding principles take into account the challenges that will result in biotic indices and assessments that are already published and assigned to a specific name. These could be designed as the ‘name-bearing’ part of the indices that will be adapted or modified from the previously developed index. There should also be limits to how much an index be allowed to be modified, in that it does not lose its founding purpose (guided by description, arithmetic syntax, ecosystem domain, and taxon), and if so, any amendments with substantial modification should be assigned into a new index. Again, we suggest that the author(s) should also be acknowledged with an attached citation (author, year of publication) next to the name of the biotic index (Table 2.2). The establishment of such a guide should ensure that information is not lost during the process and that information security and sustainability is maintained throughout.

Examples of how names could be assigned to biotic indices;

Definitions

<u>Adopt</u>: an approach to use an existing index protocol without changing its research focus, description, sampling and analysis process(es), but only curating it to fit taxa unique to a region.

<u>Amend</u>: an approach to make minor changes to improve an already existing index protocol in order to make it fairer or more accurate, or to reflect changing circumstances.

<u>Develop</u>: an approach to create a procedure towards establishing a new index protocol, i.e., with a specified detailed description.

<u>Modify</u>: an approach to adjust an existing index protocol to suit another ecosystem domain, but with significant changes to its description.

Principle of Priority

- **ecosystem domain**, i.e., aquatic, atmosphere, and terrestrial
- **research focus**, i.e., diversity, disturbance sensitivity-tolerance, functional traits groups, community structure or habitat integrity, resilience, sustainability, functioning and quality.
- **taxa**, i.e., bacterial, fungi (incl. lichens), moss, algae, protozoan (microbial), plantae, vertebrate (fish, birds, frogs), macroinvertebrates (annelids, arthropods, echinoderms)
- **geographic region**, i.e., country or state
- **arithmetic syntax**, i.e., additive geometric or arithmetic mean, weight averages, multimetric or multivariate hybrids.

- **assessment type** i.e., sampling scale, rapid assessment or biotic index

Table 2.2: Nomenclature of biotic assessment and indices suggestions

	Naming suggestions
1.	Example of a name-bearing index • **Index of Biotic Integrity** (Karr, 1981) assigned to a developed biotic index Examples of modification to Index of Biotic Integrity including sampling scale and ecosystem, • **The Benthic Index of Biotic Integrity** (Kerans and Karr, 1994) for rivers • **The Benthic Index of Biotic Integrity** (Weisberg et al., 1997) for estuaries Suggested name form with parenthesis • **River Benthic Index of Biotic Integrity** (Kerans and Karr, 1994), added a prefix 'River', sampling scale 'Benthic' to the name-bearing part 'Index of Biotic Integrity', or • **The Benthic Index of Biotic Integrity$_{\mathbf{(river)}}$** (Kerans and Karr 1994), added a prefix sampling scale 'Benthic' to the name-bearing part 'Index of Biotic Integrity', as well as the ecosystem denoted as a subscript in brackets,
2.	Example of a name-bearing index • **Hilsenhoff's Biotic Index** (Hilsenhoff, 1977) was assigned to a developed biotic index. Examples of significant major modifications to Hilsenhoff's Biotic Index for description, sampling scale but preservation of research focus and ecosystem domain, • **Family-Level Biotic Index** (Hilsenhoff, 1988), to form a rapid assessment. Suggested name form with parenthesis • **Family-Level Biotic Index$_{\mathbf{(Hilsenhoff's\ Biotic\ Index)}}$** (Hilsenhoff, 1988), acknowledging that the founding index could be done by denoting a subscript in brackets.

continues…..

	Naming suggestions
3.	Example of a name-bearing index • **Soil Biological Quality Index** (Parisi et al., 2005) was assigned to a developed biotic index. Examples of main founding taxa and amendments to Soil Biological Quality Index for description, different taxa, • **Soil Biological Quality Index$_{\mathbf{(arthropod)}}$** (Parisi et al., 2005), to denote founding taxa by a subscript in brackets, • **Soil Biological Quality Index$_{\mathbf{(collembola)}}$** (Parisi and Menta, 2008), to denote collembola amendment taxa by a subscript in brackets, • **Soil Biological Quality Index$_{\mathbf{(earthworms)}}$** (Fusaro et al., 2018), to denote earthworms amendment taxa by a subscript in brackets
4.	Example of a name-bearing index • **Biological Monitoring Working Party** (ISO-BMWP, 1980) assigned to a developed biotic index Examples of adoption of the main founding multimetric assessment to suit different regions, • **Iberian Biological Monitoring Working Party** (Bonada, 2003), added a prefix 'region name' to the name-bearing part, or • **Biological Monitoring Working Party$_{\mathbf{(Iberian)}}$** (Bonada, 2003), to the denote adopted region by a subscript in brackets Examples of adoption with major changes to develop new index from the main founding multimetric assessment to suit different regions, • **South African Scoring System$_{\mathbf{(BMWP)}}$** (Chutter, 1998) • **Stream Invertebrate Grade Number-Average Level$_{\mathbf{(BMWP)}}$** (Chessman, 1995) • **Macroinvertebrate Community Index$_{\mathbf{(BMWP)}}$** (Stark, 1985) to acknowledge the founding index we denote by a subscript in brackets.

continues…..

	Naming suggestions
5.	Example of the same index with multiple amendments, to suit ecosystem; • **South African Scoring System$_{\mathbf{(BMPW)}}$** (Chutter, 1998), founding author and year of publication • **South African Scoring System$_{\mathbf{(BMPW)}}$** Version 4 for lotic systems with low to moderate flow regimes (Dallas, 1995) • **South African Scoring System$_{\mathbf{(BMPW)}}$** Version 5 for lotic systems with low to moderate flow regimes (Dickens and Graham, 2002) An example of further modifications of an index with multiple amendments should acknowledge the most recent adoption regional version. • **Namibian Scoring System$_{\mathbf{(SASSA\text{-}V5)}}$** (Palmer and Taylor, 2004) • **Tanzania River Scoring System$_{\mathbf{(SASSA\text{-}V5)}}$** (Kaaya et al., 2015) • **Zambian Invertebrate Scoring System$_{\mathbf{(SASSA\text{-}V5)}}$** (Dallas et al., 2018) • **Okavango Assessment System$_{\mathbf{(SASSA\text{-}V5)}}$** (Dallas, 2009) for non-wadeable, deltaic aquatic biotopes.
6.	usage of acronym should also be used for both reference and syntax formulation purposes; **South African Scoring System$_{\mathbf{(BMPW)}}$** (Chutter, 1998), founding author and year of publication • **SASS** (Chutter, 1998), should be abbreviated without denoted subscripts, or • **SASS-ver.5** (Dickens and Graham, 2002), should be abbreviated with a suffix to indicate a modified version applicable. **Index of Biotic Integrity** (Karr, 1981) founding author and year of publication • **IBI** (Karr, 1981) • **Benthic Index of Biotic Integrity** (Kerans and Karr, 1994) for rivers • **B_IBI-riv.** (Kerans and Karr, 1994) • **Benthic Index of Biotic Integrity** (Weisberg et al., 1997) for estuaries • **B_IBI-est.** (Weisberg et al., 1997).

Concluding Remarks

The scientific tradition to assign names to bioassessments and biotic indices was dependent on the authors involved, and there has not been a standard protocol to help guide the process. Continuous biomonitoring and evaluation of ecosystems can be achieved by creating a working database and using reference sites (Reynoldson et al., 1997; Bailey et al., 2014b). In addition, a standard nomenclature system can assist in tracking the amendments, adaptations and modifications of a biotic index by acknowledging authors, promotion of name-bearing guide and usage of grammar parentheses and subscripts to index assigned names (Table 2.2). Furthermore, the nomenclature guide can indicate whether we are just recycling already existing indices with no innovation, highlight ecosystem domains that need to be included for biomonitoring, diversify research focus and environmental challenges, as well as for development of more accurate, region-specific predictive models.

The establishment of a standardised procedure is recommended, as some of these indices lack detail in their pre-size collection of data, and potentially your regulators can be involved in creating these protocols. Then, another aspect is to incorporate them into conservation policies and develop educational programmes to employ these tactics. The continuous collection and centralised data system for large-scale meta-analysis at higher levels of conservation and climate change can also be established. These organised data systems can develop into centres that focus on assisting in the identification (all using same identification systems or morphospecies coding), statistical analysis and data processing to create a consistency, which in turn will develop more accurate new area specific indices as we get a better understanding of indicator species. Furthermore, the authoritative entity will also supervise the developmental and establishment processes, as well as assist in tracking changes and modifications of biotic indices.

Based on the principles and guidelines suggested, the standard protocol might assist in avoiding challenges of ambiguity or conflict between index description and assignment of names, as some assigned names and acronyms have duplicates, while others have been adopted and modified multiple times, so that the founding authors are not acknowledge for their scientific contribution. Furthermore, some natural ecosystems in continuum systems like streams, rivers, wetlands and estuaries that are connected between countries or states to perform unified ecosystem functioning (Abbasi and Abbasi, 2011; Norris, 1995). The establishment of such an entity can also be used to promote international collaboration for scientific and practitioner personnel towards ecosystem integrity and functioning assessment protocols. The uniform nomenclature system can be linked to existing assessment reference databases to make work much more effective.

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CHAPTER 3 : Adopting Biotope Quality Index for Vegetation Ecology and its Application as a Biomonitoring Tool for Mesic Highveld Grasslands Bioregion in Golden Gate Highlands National Park, South Africa.

Abstract

Montane temperate grassland bioregions are prone to threats posed by anthropogenic disturbances such as land transformation for agricultural fields, overgrazing, habitat fragmentation, soil erosion, uncontrolled fires, and plant invasion. In this study, the Biotope Quality Index (*BQI*) was adopted using plants to evaluate grassland vegetation bioregions and the impact of disturbances. The adoption of *BQI* was evaluated using three criteria measures for abundance-cover and density (Braun-Blanquet scale, Domin scale and density cover). The *BQI* application was also tested separately for major common and dominant species taxa that included monocots, dicots, grasses, forbs and all recorded plant representatives. Qualitative and quantitative data were collected to measure environmental disturbances in Golden Gate Highlands National Park. The vegetation composition of both evaluated Mesic Highland grassland bioregions was dominated by the community of *Themeda triandra* - *Eragrostis chloromelas* species, as well as the Poaceae and Asteraceae families. *BQI* had a negative correlation to soil erosion for Eastern Free State Sandy grassland with the Honing Kloof site attributed with a higher degree of soil erosion, less ground vegetation cover, less diversity (H = 0.469) and a *BQI* value less than 0.001, the Korfshoek and Twijhoek sites were attributed with drier top soil with greater encroachment of *Seriphium plumosum* shrubs, lower to moderate vegetation cover, diversity (H = 1.025 and H = 1.096) with lower *BQI* values of 3.253 and 7.969 respectively, while the Welverdiend site had the highest vegetation ground cover, with higher diversity (H = 1.323) and *BQI* value of 28.180 respectively. Plant invasion and *BQI* had inconclusive correlation for Basotho Montane Shrubland grassland due to opposing effects by *Acacia mearnsii* tree at the Avondrust site was attributed to less ground vegetation cover, lower diversity (H = 0.893) and a *BQI* value of 0.419, however, the QQ Mountain site had a significant presence of *Cynodon dactylon* grass but with highest vegetation ground cover, with higher diversity (H = 1.471) and *BQI* value of 23.824 respectively. Lastly, Biotope Conservation Status (*BCS*) score classified sites that require conservation efforts as highly disturbed status (Honing Kloof, Avondrust, Korfshoek and QQ Mountain), poorly conserved status (Twijhoek and Diepkloof) and good conserved status (Welverdiend and Silasberg). *BQI* can be used as a surrogate to assess disturbance impacts on structural properties of vegetation communities, and *BCS* measure can be used to indicate conservation status and progress of ecological restoration efforts in grassland bioregions.

Keywords

Biodiversity, Biomonitoring, Biotope Conservation Status, Environmental Disturbance, Ground Vegetation Cover, *Themeda triandra, Eragrostis chloromelas,* Species community

Introduction

Grassland biomes cover more than 30 % of the surface area of terrestrial land and occur in a wide range of climatic regimes both temperate and tropical (Hu et al., 2016; New, 2019). The grassland biome hosts ecologically important ecosystems such as wetlands and other freshwater systems (Dixon et al., 2014), which are among terrestrial bioregions sensitive to disturbance (Tilman and Downing, 1994). Furthermore, ecosystems of the grassland biome play ecological roles offering services for climate and water regulation, soil and biodiversity conservation (Neke and Du Plessis, 2004), as well as providing human benefits like agricultural practices (crops and livestock production), natural medicine and tourism (Allen et al., 2011). Although grassland bioregions have a peculiar response to disturbance, conservation efforts are still being proposed due to continued environmental disturbance pressure (Han et al., 2020). Major threats to grassland conservation include ecological disturbances, namely drought, soil erosion and climate change challenges (Dixon et al., 2014), as well as anthropogenic activities that transform the land surface such as human settlement development, agricultural crop field practices, introduction of alien plants, uncontrolled wildfires (Neke and du Plessis, 2004; Prinsloo et al., 2021).

Floristic composition and structure of terrestrial biomes are main components that are considered to describe ecosystems and bioregions (Archibold, 1995; van der Maarel, 2005). Flowering plants (angiosperms) are part of the ecological primary producer key to convert solar energy to bioenergy (Gibbons and Freudenberger, 2006), base source for the food chain (Fisher and Turner, 2008), as well as the provision of essential resources for soil moisture regulation, symbiotic relationships toward shelter, and breeding sites for various organisms (Kontogianni et al., 2010). In South Africa, the temperate grassland biome is one of the three largest bioregions covers 339 237 km^2 for 27.9 % of land surface area (Mucina and Rutherford, 2006; O'Connor et al., 2011), but only 2.04 % of the grassland biome is conserved (protected) and mainly in highland regions (Carbutt et al., 2011). Approximately 60 % (7750 km^2) of the grassland biome in South Africa has been irreversibly degraded due to intense land exploitation (Little et al., 2013), and it is now one of the most threatened biomes in South Africa (Neke and Du Plessis, 2004; Kaiser et al., 2008). There are international initiatives including the Aichi Biodiversity Target 11 of the Convention on Biological Diversity (CBD, 2012) and the Temperate Grasslands Conservation Initiative (TGCI) of a joint International Grasslands-Rangelands Congress (Mark, 2012), aimed at helping to understand the value of the remaining large contiguous and intact tracts of grasslands that support landscape-scale processes (Peart, 2008; TGCI, 2010; TGCI, 2012).

For the purpose of this study, the definition of terrestrial biotope by Connor et al. (2004) was inherited from an ecosystem approach through a structural and functional spatial quantifiable biogeographic unit perspective. To avoid ambiguity, we assign biotope to a community concept and habitat to species concept that are defined by their heterogeneity, complexity and spatial scale (McCoy and Bell, 1991).

Generally, a habitat represents physical conditions (with geographical location, physiographical features and physiochemical environment) that surround a species, species population and assemblage of species or community (Olenin and Ducrotoy, 2006). However, Dimitrakopoulos and Troumbis (2019) define a biotope as a topographical unit (landscape) characterised by similar environmental (physical) conditions, a specific assemblage and composition of plants and animals that are restricted by a clearly defined boundary. Furthermore, in this study, quality as a status (or condition) is described for a biotope that can sustain an optimal number of species and population sizes with optimal heterogeneous composition, and even when under perturbational stress, continue to supply sufficient resources with adequate resistance to sustain its ecological processes (Calow, 1992; Carignan and Villard, 2001; Bredenhand, 2014).

Evolutionary traits that angiosperms acquired and adapted over time for a certain region, namely reproductive strategies, seed dispersal pollination, structural form, community composition, and distribution patterns, could be used as characters during biomonitoring and assessment programs (Niemi and McDonald, 2004; Niemeijer and de Groot, 2008; Puczko et al., 2018). A bioindicator is a species, taxa, or community of organisms that is sensitive to change in environmental conditions of a biotope (Markert et al., 2003). Several studies have highlighted utilisation of angiosperms as biological indicators (bioindicators) in that some species or community response towards ecological disturbance and change (a particular or subset of ecological factors), that can also indicate resilience, stability and competence of an ecosystem to continue functioning (van der Maarel, 2005; Naeem et al. 2009, Tett et al. 2013). Biological attributes such as change in species relative abundance, biodiversity level assemblages, community or functional group compositions can assist in measuring impact of environmental disturbance, as well as monitoring conservation programs progress (Chen et al., 2015; Chin et al., 2015). Disturbance is regarded as a scaled process which can be categorised according to different traits including extent of impact, intensity, frequency, duration and spatial extant (Pickett and White, 1985; Willis et al., 2010), and could be measured coupled with species temporal and permanent responses to develop better long-term monitoring methods. However, ecosystem quality is seen as an aspect that has the potential to measure disturbance impact by indicating conservation status and response ability for individual species, biotope (ecological functioning unit) and ecosystem to perform their respective functions (Bouyer et al., 2007; Syrbe and Walz, 2012; Lee and Lautenbach, 2016). Development of rapid and long-term bioassessment conservation tools for grassland biotopes to respond towards impact disturbance, in order to highlight a cohesive biotic, chemical and physical response towards disturbance (Sharpley et al., 2012; Thach et al. 2018).

Consistent integrity patterns and diversity of plant communities can be more resistant to disturbances in that it helps maintain the biotic structure, functioning and species diversity of ecosystems such as grasslands (Tilman and Downing, 1994) and can be altered by environmental disturbances (Huntly, 1991). Although ecological events that occur in grasslands make it peculiar, as fire and grazing help to maintain the integrity and functioning of the biotope (O'Connor and Bredenkamp, 1997; O'Connor, 2005). Several studies have highlighted that adequate fire and moderate grazing enhance species diversity compared to unburned and ungrazed lands (Milchunas et al., 1988; O'Connor, 2011). The development of bioassessment and biotic indices depends on the rationale for agriculture or ecology, ecological aspects such as integrity, quality, health, function, or viability (Gibbons and Freudenberger, 2006), which then influence the choice of attributes to measure. A brief overview of proposed biotic indices and bioassessment techniques used in grasslands includes the Estimated Species Richness to evaluate the number of species observed in a sample (Chao, 1984), Shannon-Wiener Diversity Index for species variation proportion at a trophic level (Shannon and Weaver, 1949), Simpson-Yule Diversity Function for variation within communities (Simpson, 1964), relationships among vascular plants and their environment stressors, Ellenberg Indicator Values semi-quantitative relationships in agricultural environments (Ellenberg et al., 1991), Floristic Quality Assessment Indices evaluates naturalness state of habitat in response to disturbance (Sellers et al., 2001), some multivariate tools such as the Vegetation Condition Index to assess ecological and agronomic concerns of highland sourvelds in South Africa (Hardry and Hurt, 1989), Habitat Complexity Score to evaluate forests habitats in Australia (Catling and Burt, 1995), and multimetric tools that includes the Index of Biotic Integrity using plant communities in mid-Atlantic regions freshwater systems (Simon et al., 2001), Biodiversity Benefits Index for evaluating economic value of forests in the USA (Oliver and Parks, 2003), and Rapid Appraisal of Riparian Conditions to assess state of river riparian along Australian freshwater systems (Jansen et al., 2004).

To apply quantitative statistical ecology methods, an individual is defined both in a subjective and objective context, in that an individual plant originates from a single root system (Fenner, 1997), deemed to measure abundance using cover estimates, frequency and density (Kent and Coker, 1992). Species cover is considered as an estimate occupied by an individual plant species above the soil surface independent of any overhanging cover of other species in the sampled unit (Fehmi, 2009). The Biotope Quality Index was developed using arthropod assemblage (species occurrence counts) as a measure of mean proportion and standard deviation degree, to indicate the impact of the disturbance level and assign a conservation status to a biotope (Bredenhand, 2014). The Biotope Quality Index expression model is among statistical formulations that assess the relationship between variables or patterns within a variable of a sample unit of the observed population using standard scores, e.g., as in Pearson's Correlation Coefficient which is used to measure association between two variables, and for normalising data, which adjusts data points from the sample or population mean threshold. In this study, standard scores are referred to as the number of standard deviations by which a random data point value (raw score) is above or below the mean value of an observed sample as a measure of biotope quality (Bredenhand, 2014; Glantz et al., 2016). In that, Figure 3.1 indicates that data points that are above the sample mean would have a positive standard score and those below would be with negative scores (Aho, 2014).

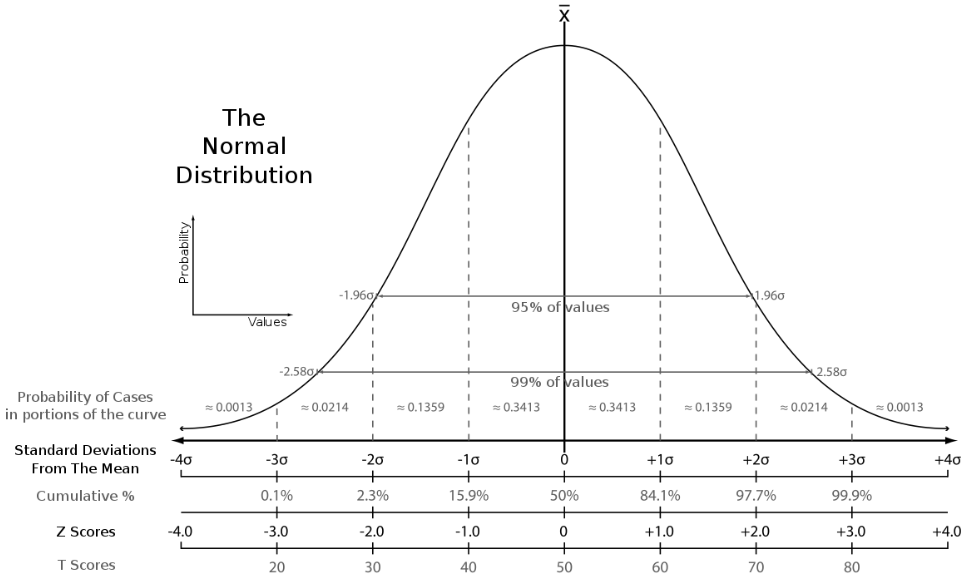

Figure 3.1: Basic illustration of sample standard deviation and mean from the general theory of probability (adapted from Aho, 2014)

The expression of the Biotope Quality Index (below) considers the evolutionary history of the integrity unit, and assumes that any biotope in a naturally good conserved state can act to sustain the arthropod population to optimum species numbers as well as their ecological processes.

$$\text{Biotope Quality Index } (BQI) = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right)$$

if $\log \sigma > 0$……………………………… (Bredenhand, 2014)

Expression elements represented include the abundance of *i*th morphospecies (x_i), *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ). Therefore, the aim is to adapt the Biotope Quality Index (BQI_{veg}) from a plant perspective to measure the environmental conditions of a biotope. Therefore, the expression elements above would be replaced as follows; abundance (species density cover estimate) of *i*th plant species (x_i), *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ). An ordinal scale is used to determine species abundance-cover estimates by assigning percentage values depending on how much ground cover a species occupy above soil surface within a sampling quadrant (Kent, 2012). The *BQI* biotic index considers vegetation data properties when sampling to quantify biotopes, in that redundancy and noise created by sites or quadrants with similar species and species composition (statistical duplicates), whereby data is reduced to individual species only with abundance-cover above mean across all sampled sites variance (Wildi, 2017).

It is challenging to quantify plant abundance using only counted data due to the difference in natural growth and structure form, in that there is no clear distinction between an individual with one to multiple stems, or those with a rhizome root system (below ground) and stolon root system (above ground) growth connection system (Austin, 2005). Generation overlaps for co-occurring monocarpic annual and biennial species, as well as polycarpic perennial herbaceous species, in one sampling quadrant (Austin, 1990; Bridge, 1993). Most regularly used species abundance-cover scales include Braun-Blanquet scale with five class partitions (Braun-Blanquet, 1952) and Domin scale with ten class partitions (Floyd and Anderson, 1987). Both species abundance-cover scales allow integration to use density counts in order to increase accuracy of cover estimate measure. The *BQI* calculation has four main steps which include;

- log transformation of relative species abundance (or abundance-cover)
- considering data entries equal to or above relative species abundance mean only (selecting biotope representative species)
- compute the difference between relative abundance and mean, then divide by standard deviation for each species (or morphospecies)
- aggregate the sum of all species per site into *BQI* value, then assign a grade biotope status category (poor, moderate, good and excellent conditions)

In this study, the aim was to develop a biotic index that helps to evaluate the effects of environmental disturbance using plants (grass, herbaceous, shrubs and trees) species abundance-cover instead of arthropod abundances, by adopting the Biotope Quality Index (Bredenhand, 2014), identifying possible disturbance bioindicators, as well as proposing a novel Biotope Conservation Status classification method for the Eastern Free State Sandy and Basotho Montane Shrubland Grassland bioregion in the Golden Gate Highlands National Park.

Materials and Methods

Study site

Golden Gate Highlands National Park (GGHNPark) covers a surface area of 342 km^2 (32758 ha), located between latitude of 28°27’ S to 28°37’ S and longitude of 28°33’ E to 28°42’ E, with elevation ranging from 1892 m to 2837 m in Figure 1 (Rademeyer and van Zyl, 2014). GGHNPark is located on the north-eastern Maloti-Drakensberg Mountain border between the Kingdom of Lesotho, the Eastern Free State and the Kwa-Zulu Natal Province. The Eastern Free State region is characterised by higher summer rainfall receiving between 800 mm and 1600 mm, with a warmer mean annual temperature ranging from 13°C and 30°C, and again lower winter rainfall between 500 mm and 760 mm and with a mean annual temperature ranging from 1°C to 26°C (Grab et al. 2011). Climate in the region experiences summer thunderstorms, winter frost, and snow falls, creating a moist cold highveld ecoregion that is mainly characterised by grasslands, herbs, along with occasional trees and shrubs (Mucina and Rutherford, 2006; Telfer et al., 2012; Rademeyer and van Zyl, 2014).

In this study, vegetation type categories as described by Mucina and Rutherfiord (2006) were considered as biotope units (Figure 3.2), to which a unit has a dominant and common floristic composition and structure, as well as clear boundaries. The GGHNPark hosts two subgroups of the Mesic Highveld Grassland vegetation types named Eastern Free State Sandy (Gm4) and Basotho Montane Shrubland (Gm5) within the Afromontane elevation level zone, which are considered sourveld (at some parts mixed) grasslands that are distinguished by their geomorphology, elevation, rainfall, and soil substrate properties. The Eastern Free State Sandy (Gm4) grassland is characterised by mudstone, sandstone and shale soil forms, on a flat to slightly uneven terrain with rangelands reaching 2025 m elevation. Vegetation is dominated by graminoids making up grass (Poaceae), herbs (mostly Asteraceae) and sparse short shrub communities (Eckhardt et al., 2006; Kay et al., 2006). The Basotho Montane Shrubland (Gm5) grasslands are situated on steep rocky Clarens sandstone with uneven slopes, characterised by mixture of taller shrubs, lower shrubs, grass communities and herbs (Mucina et al., 2006). Four sites were selected for each Mesic Highveld grassland subgroup taking into account ecological (soil erosion) and human-influenced disturbances (historical agricultural activities, housing buildings, road constructions, pollution level), as well as plant encroachment by invasive species. Selected sites for Eastern Free State Sandy were Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4), while those for Basotho Montane Shrubland were QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5), respectively (Figure 3.2).

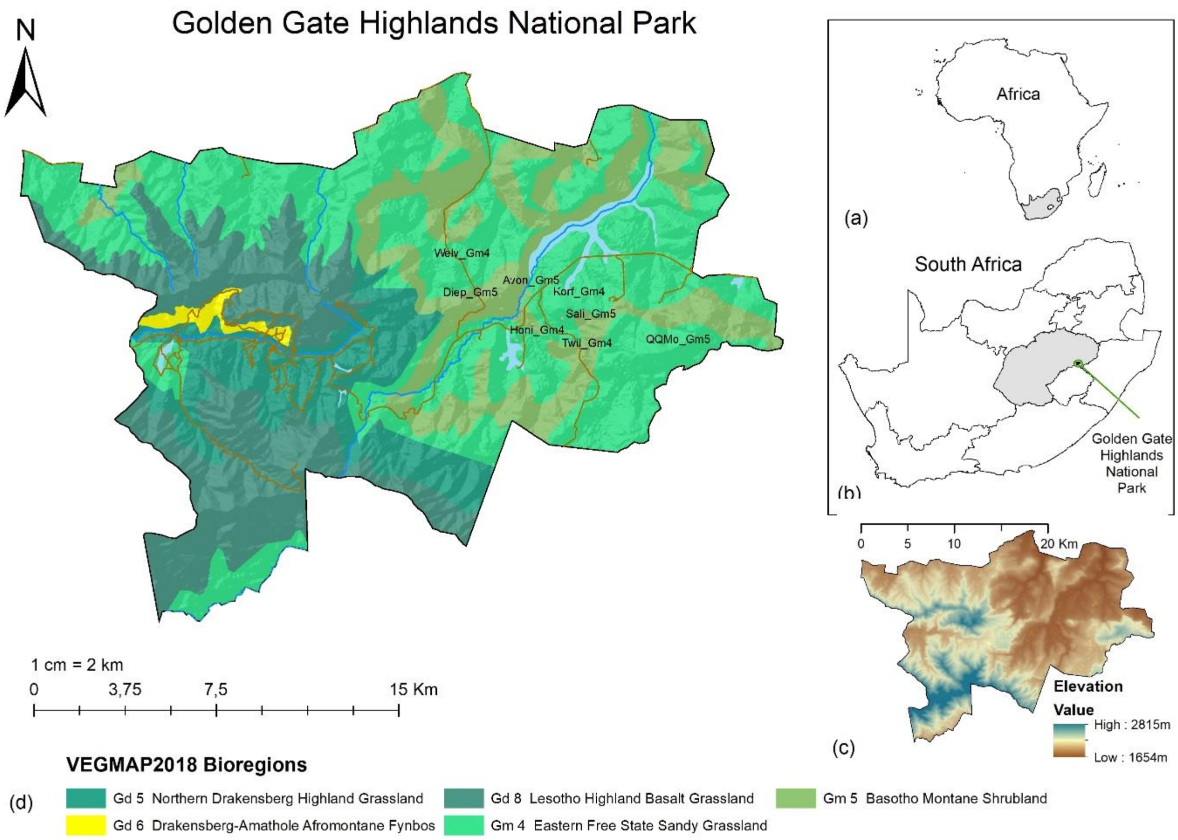

Figure 3.2: Location study area that illustrates (a) the African continent, (b) eastern section of the Free State Province highlighted in South Africa outline, (c) elevation and hillshade created of GGHNPark using the DEM image from USGS Explore Earth, and (d) GGHNPark that highlights sample sites within Mesic Highveld vegetation bioregions (Gm4 and Gm5) from SANBI GIS VEGMAP (2018) using ArcGIS.

Vegetation survey

Plant samples were recorded four times between seasons during September 2016 to September 2018. Random nested sample plots of 6 m x 5 m (area of 30 m^2) quadrats were made along the 200 m transect to ensure variation of plants within plots. Plant counts were recorded and species cover estimated using modified Domin cover-abundance scale (Eeckhardt et al., 1993; Brand et al., 2008). The identification and confirmation of plants was done at the Botany Herbarium University of Free State Qwaqwa campus at the Department of Plant Sciences herbarium, and the species names were confirmed using Germishuizen and Meyer (2003) as well as the names updates at www.pza.sanbi.org website.

Environmental data and disturbances

Environmental data was collected using field observation and DEM image in ArcMap (ArcGIS version 10.8) to measure the aspect, slope, elevation, latitude and longitude. Environmental disturbances were categorised according to ecological (soil erosion) and anthropogenic activities (plant invasion, agriculture, building infrastructure, pollution) (supplementary material: Table 0.3). Plant encroachment-invasion ratio was determined by recording the presence and percentage cover of plant species classified as encroachers and alien invaders (supplementary material: Table 0.2). Recorded anthropogenic activities that contribute to disturbance included historical agricultural activities (prior to sampled areas were declared as a national park), housing and road constructions, land pollution level, current wildlife and human activity zones as well as conservation rehabilitation programs (supplementary material: Table 0.4). Each sampled site was described based on current ecological conditions and historical activities (before and after the status of the national park).

Biotope Quality Index

The adopted Biotope Quality Index (Bredenhand, 2014) that uses terrestrial vascular plants in temperate highland grasslands using the modified Domin species abundance-cover scale was used to measure and indicate the environmental status of each sample site. All species abundance-cover data variations in plant species abundance-cover and their community composition were used to measure the effects of environmental change and disturbance. The Biotope Quality Index (*BQI*) considers only common representative species that occur above average samples per site and also includes the distance from average for best fit. Firstly, three commonly used subjective species abundance-cover and density scales were considered, namely, species density percentage cover (graded from 0 - 100%), Braun-Blanquet (seven abundance scaled points) and Domin species abundance-cover scales to increase accuracy of cover estimate measure (Table 3.1). The three abundance and density scales were computed separately to prepare the vegetation data in BQI_{veg} and to check their suitability.

Table 3.1: Abundance-cover scales used in collecting vegetation data for the Domin 10-point and Braun-Blanquet 7-point scale.

Abundance-cover scale	**Abundance score**
Domin (10 points)	
91-100%	10
76-90%	9
51-75%	8
34-50%	7
26-33%	6
11-25%	5
4-10%	4
<4% (with many individuals)	3
<4% (with several individuals)	2
<4% (with few individuals)	1
Braun-Blanquet (7 points)	
76-100%	6
51-75%	5
26-50%	4
6-25%	3
2-5%	2
<2% (with few individuals)	1
<2% (one individual)	+

Ordinal data for abundance-cover and density were transformed prior to analysis. Species abundance-cover data were transformed with logarithm ($\log_{10}$) to assume normality of individual variable distribution, homogeneity of variance, or linearity between data points (Kent, 2012). Data transformation took into consideration different growth strategies between species, plant strata differences, uneven distribution of species abundances and seasonal structural differences between species, in order to estimate abundance-cover data suitable for BQI_{veg} to determine state of biotope quality. The logarithm transformation of species abundance-cover data does not include the addition of 1 as, $\log X_i \neq \log (X_i+1)$, since the presence and absence of a species holds spatial and temporal ecological significance within a sampled biotope. Therefore, after selecting the preferred abundance-cover scale for plant species observations, the data values are transformed as follows;

Let X_i represent plant species abundance-cover (using any ordinal scale)

Then, $x_i = X_i$ ……………………………….(1)

$$(BQI)\,veg = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right)$$

……………….(2)

Conditions of equation can only be true if there were plant species present in sampling unit (biotope) and with existing standard deviation for $\log \sigma > 0$

Therefore, Bredenhand's (2014) Biotope Quality Index BQI_{veg} was adopted to be used from a plant perspective. In that, the expression elements of BQI_{veg} are represented as follows; species abundance-cover estimate of *i*th plant species (x_i), *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ). Meanwhile, likewise with animals, biotic indices are created based on separation of life-forms (fish, mammals, birds or arthropods only) in that they present different life strategies and do not fall on the same measure of scale. Again with plants, for adopting *BQI* we include separation of vegetation taxa representatives, namely, 1) all seed bearing and non-seed bearing, flowering and non-flowering, as well monocots and dicots, because they are all included when describing biotope or ecological bioregion, then expression into BQI_{veg}; furthermore separate measure of scales, 2) dicots only with expression into BQI_{dic}; 3) monocots only with expression into BQI_{mon}; then due to determinant dominant and descriptive terrestrial plant groups for temperate highlands grasslands, 4) graminoids (grasses, forbs, sedges, restios and juncas) only with expression into BQI_{gmd}; 5) Forbs only with expression into BQI_{for}; 6) as well as Poaceae (grasses) only with expression into BQI_{poa}, respectively.

Data analysis

For further application and evaluation of the environmental status we continued with species abundance-cover data using the 10-point Domin abundance scale for sampled biotopes. The Plantae BQI_{veg} version was then used for further ecological analysis. Descriptive statistics were used on plant abundance-cover data to measure mean, standard deviations and relative abundances-cover for each sampled site (Legendre and Legendre, 1998; Gardener, 2012). The Kruskal-Wallis H-test and Dunn post-hoc test were used to compare the mean variation of plant taxa between sites with uneven sample sizes using ranked means. Sample-based rarefaction (Gotelli and Colwell, 2001) was used to determine the probability of species rare occurrence (singleton and doubleton frequencies) for each site. The species surrogate included methods of biotic indices to calculate estimated species richness using Chao2 estimator (Chao, 1984), diversity indices using Simpson-Yule (Simpson, 1949) and Shannon–Wiener function (Weaver and Shannon, 1949). A modified two-way species cluster analysis (biclustering) using the Euclidean distance measure and the Ward hierarchical cluster method to simultaneously identify possible subgroups of the vegetation community with the Heatmaply package in R-Studio (Dominici et al., 2008; Lunghini et al., 2013). The association of environmental or ecological disturbance as a gradient to influence abundance-cover and density clusters was evaluated using Canonical Correspondence Analysis (CCA) using abundance-cover data in R-Studio with the vegan package (Gardener, 2014).

Soil erosion was determined using the RUSLE model (Renard et al., 1997) that considered the degree of plant cover, soil type, moisture and depth. Plants that were considered for encroachment and invasion were identified within a 6m x 5m plot along 200 m vegetation sampling transect, and encroachment-invasion percentage cover was determined per site. The effect of environmental or ecological disturbance on abundance-cover and density dynamics was determined using permutational multivariate analysis of variance (PERMANOVA) using in the R-Studio (ADONIS, simulations ran 23 permutations to congruency, in vegan package) and only disturbances with significant influence were considered for further analysis (Gardener, 2014). The sites were related based on plant and environmental data with both qualitative and weight qualitative data using Bray-Curtis dissimilarity (Bray and Curtis, 1957). Then, we also propose a novel category classification of the Biotope Conservation Status (*BCS*) that would be determined by the product of the two main disturbance variables and its influence on the *BQI* of each sampled site as follows (Table 3.2);

$$DBQI_i = \frac{BQI_i}{(d_1 \times d_2)_i}$$

……………………………………………………………………………………………(3)

$$BQI_{mean} = \frac{1}{N}\sum BQI_n$$

……………………………………………………………………………………………(4)

$$Biotope\ Conservation\ Status\ (BCS) = \frac{DBQI_i}{BQI_{mean}} \times 100$$

……………………………………………………………………………………………(5)

Table 3.2: Biotope Conservation Status category classification score illustrating conditions for Biotope Quality Index and degree of disturbance.

***BCS* value (%)**	**Biotope Conservation Status Rating**
91- 100	Pristine Conserved Status
71 – 90	Good Conserved Status
50 – 70	Fairly Conserved Status
26 – 49	Poorly Conserved Status
0 – 25	Highly Disturbed Status

Geospatial modelling for the Biotope Quality index to indicate current ecological conditions of the Eastern Free State Sandy (Gm4) and Basotho Montane Shrubland (Gm5) grassland vegetation types in the GGHNPark, a modified bootstrap jackknife 2 using 999 permutations (Manly, 1991) to predict asymptotic Biotope Quality Index values with varying population sizes from the calculated mean and standard deviation for neighbouring and adjacent related sites. The vegetation types were divided into units of sampling zones per locality, and then the predicted values of the asymptotic Biotope Quality Index estimates were assigned to each polygon of a sampling zone using ArcGIS Map (version 10.5) to highlight the area of environmental concern. Shapefiles for continent and country maps of the world were accessed from the Esri GIS database, while the provincial, RSA protected areas (SANParks), roads and surface water body shapefiles of South Africa were accessed from the Department of Forestry, Fisheries and the Environment (DFFE) EGIS database, and the vegetation bioregion VEGMAP 2018 (for South Africa, Lesotho and Swaziland) shapefiles were accessed from the SANBI Biodiversity GIS database.

Results

The mesic grassland vegetation was greatly represented by Tracheophytes (vascular plants) with a total of 94 plant species observed from 2 Clades (Polypodiophyta and Angiospermae), 3 Classes, 19 Orders, 33 Families were used for analysis. Distinctively, Basotho Montane Shrubland (Gm5) had the most plants observed with 91 species represented in 33 Families, 19 Orders, 3 Classes and 2 Clades, with most in the QQ Mountain, Silasberg and Diepkloof sites, as well as least in the Avondrust site (Table 3.3). Eastern Free State Sandy (Gm4) had at least 51 species observed from 12 Families, 8 Orders, 2 Classes and 1 Clades, with most in Welverdiend, Twijhoek and Korfshoek sites, as well as least in Honing Kloof site.

Table 3.3: Summary of plant taxa represented for the Mesic Grassland observed for the Eastern Free State Sandy and Basotho Montane Shrubland vegetation types in GGHNPark.

	Phylum	**Clade**	**Class**	**Order**	**Family**	**Species**
Korf_Gm4	1	1	2	6	8	24
Honi _Gm4	1	1	2	1	1	3
Twij_Gm4	1	1	2	7	9	32
Welv_Gm4	1	1	2	8	10	46
QQMo_Gm5	2	2	3	17	30	75
Avon_Gm5	1	1	2	8	10	24
Diep_Gm5	1	1	2	11	16	49
Sila_Gm5	2	2	3	14	19	50

* Abbreviations, Eastern Free State Sandy Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4), while those for Basotho Montane Shrubland were QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5).

Furthermore, the composition of plant life-forms counts in Basotho Montane Shrubland was composed of 38.41 % herbs and forbs, 34.36 % grass and sedge, 10.11 % shrubs, 10.11 % trees, and 1.11 % climbers and vines, with all represented in the QQ Mountain site, most in the Silasberg and Diepkloof sites, and the least in the Avondrust site (Figure 3.3). Eastern Free State Sandy has the least plant life-forms counts with 66.56 % grass-sedge, 25.38 % herbs and forbs, as well as 8.06 % shrub, with most represented in the Welverdiend site, most in Twijhoek and Korfshoek sites, with the least in Honing Kloof site. There was significant difference in abundance-cover between the Eastern Free State Sandy sites with a $F_{3,200}$ of 4.025 ($p < 0.01$) mainly between the Welverdiend and Honing Kloof sites, however, there was no significant difference between the abundance-cover of the Basotho Montane Shrubland sites.

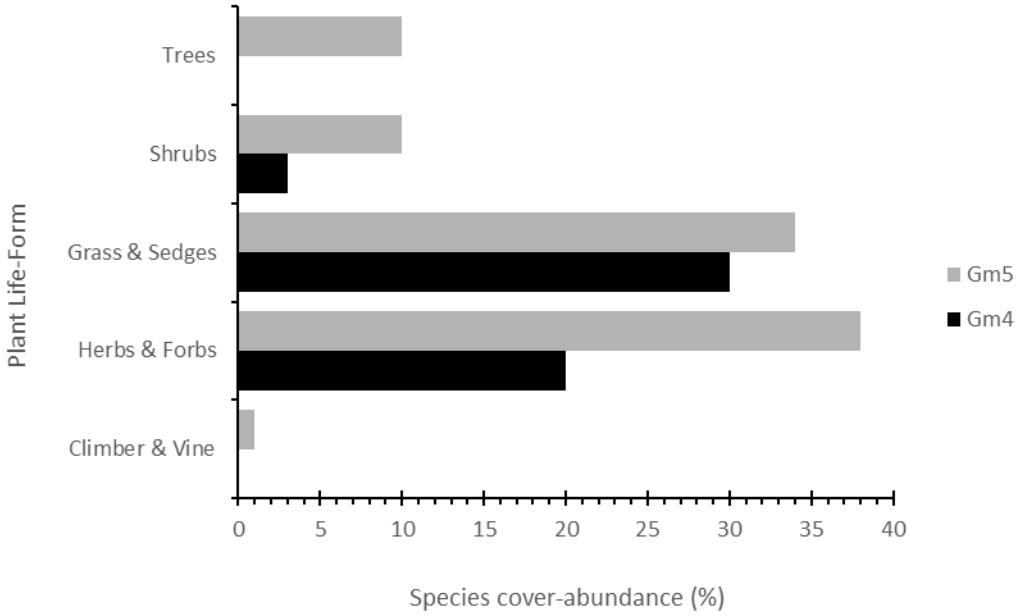

Figure 3.3: Plant life-forms observed in Mesic Grassland abundance-cover composition comparison for the Eastern Free State Sandy and Basotho Montane Shrubland vegetation types in GGHNPark.

Plant taxa assemblages and community composition

There was a mixture of short and tall grasses for Eastern Free State Sandy grassland with a plant family composition represented mostly by Poaceae (86 %), Asteraceae (6 %) and Cyperaceae (2 %) (Figure 3.4). The hierarchical species community cluster differentiated into two main communities, namely *Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn. - Erg.chl.*) and *Andropogon appendiculatus* - *Eragrostis capensis* (*And.app. – Erg.cpn.*) communities (Table 3.4), with four sub-communities that included (1) *Thm.trn. - Erg.chl. - Tristachya leucothrix,* (2) *Thm.trn. - Erg.chl. - Aristida diffusa,* (3) *Thm.trn. - Erg.chl. – Andropogon appendiculatus* and (4) *And.app. – Erg.cpn. - Eragrostis curvula,* respectively (Figure 3.5). The Basotho Montane Shrubland had dense and open shrubland with a composition of the plant family represented mainly by Poaceae (85 %), Asteraceae (5 %), Lobeliaceae (2 %), Oxalidaceae (2 %), Orchidaceae (2 %) and Fabaceae (2 %) (Figure 3.6). There were two main hierarchical species community clusters, namely *Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn. - Erg.chl.*) and *Aristida diffusa* - *Pteridium aquilinum* (*Ars.dff. – Ptr.aql.*) communities (Table 3.4), with four sub-communities that included (1) *Thm.trn. - Erg.chl. - Tristachya leucothrix,* (2) *Thm.trn. - Erg.chl. - Leucosidea sericea* (3) *Thm.trn. - Erg.chl. – Helichrysum pilosellum* and (4) *Ars.dff. – Ptr.aql. - Helichrysum callicomum,* respectively (Figure 3.7). There was a significant difference observed for the abundance-cover of the Eastern Free State Sandy (Gm4) grassland with $F_{3,200}$ of 4.024 (p < 0.01) mainly between the Welverdiend (Welv_Gm4) and Honing Kloof (Kloof_Gm4) sites, while there was no significant difference for the abundance-cover in the Basotho Montane Shrubland (Gm5) grassland sites.

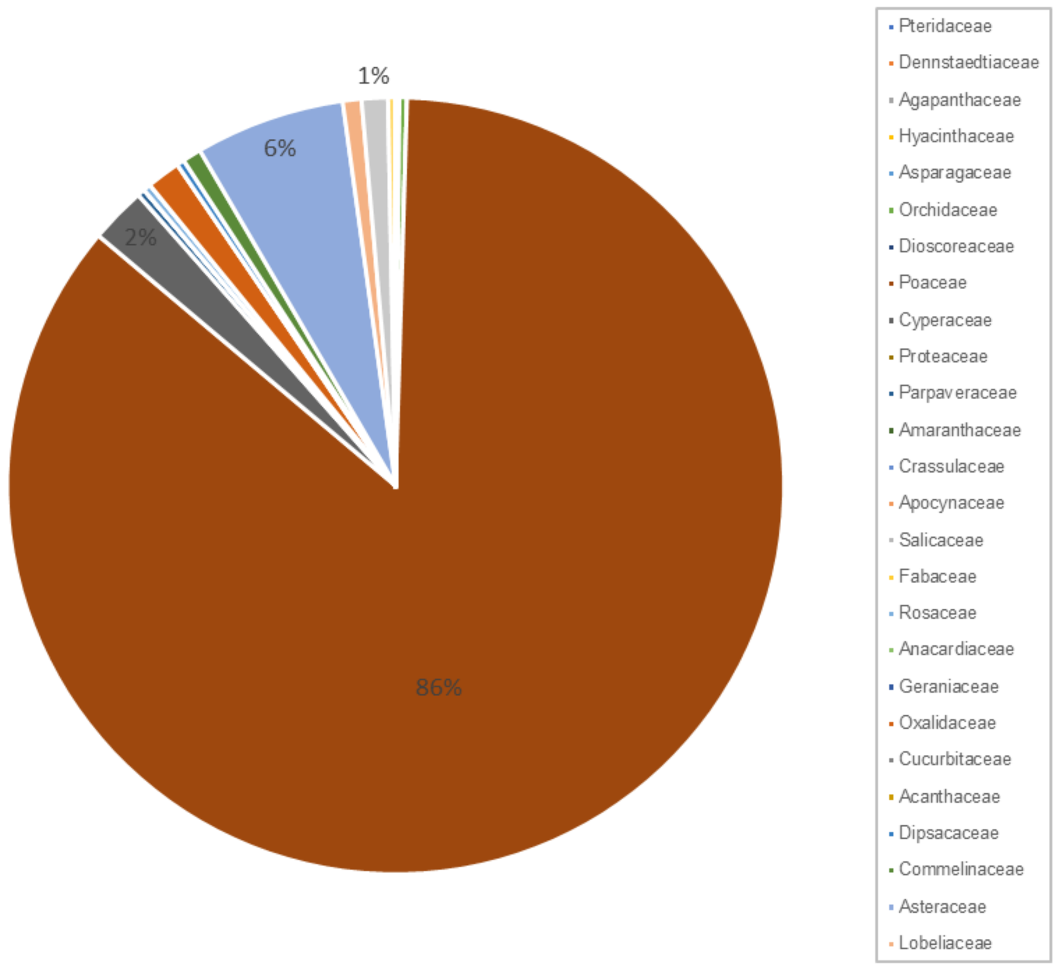

Figure 3.4: The composition of vegetation taxa for the Eastern Free State Sandy (Gm4) grassland vegetation type in GGHNPark.

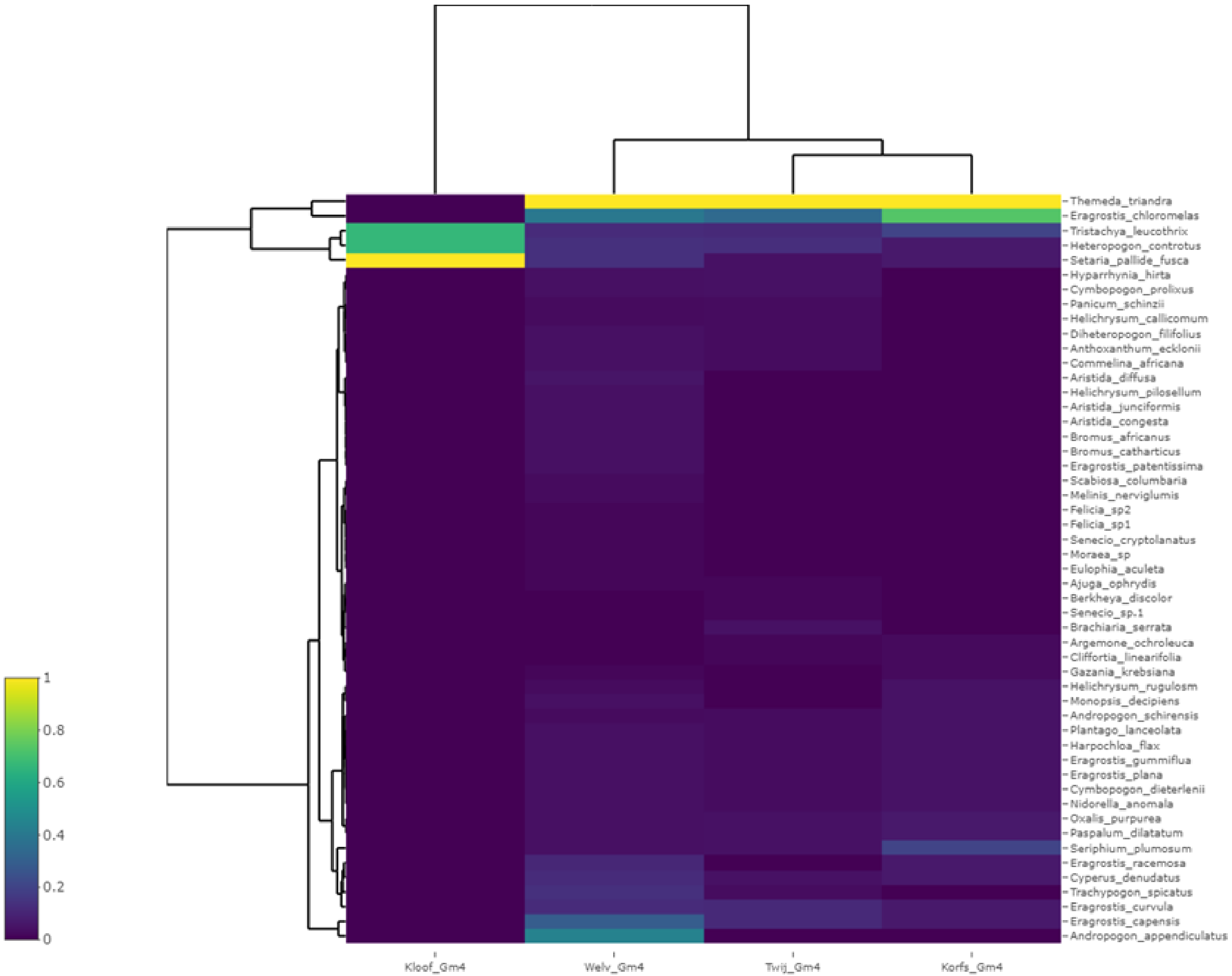

Figure 3.5: Illustration of plant community structures consolidated using a two-way species cluster (Euclidean distance measure and Ward hierarchical cluster) for the Eastern Free State Sandy (Gm4) grassland vegetation type in GGHNPark.

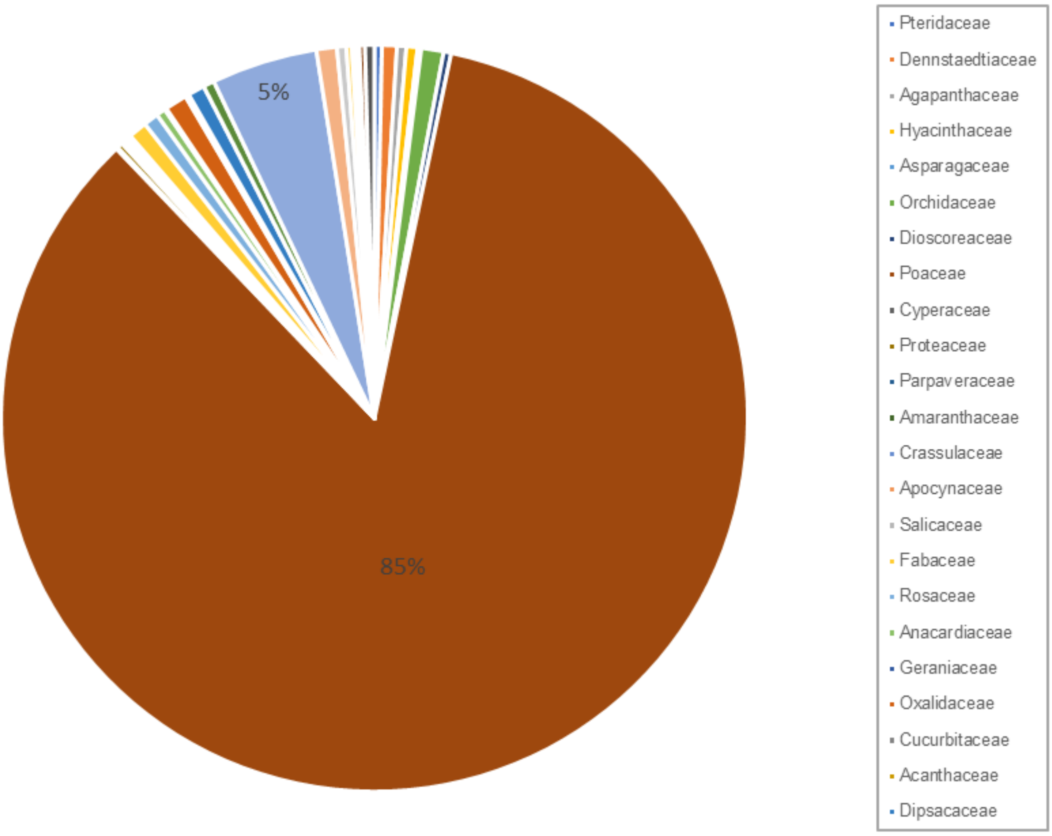

Figure 3.6: Illustration of the floristic composition structure of the Basotho Montane Shrubland (Gm5) grassland vegetation type in GGHNPark.

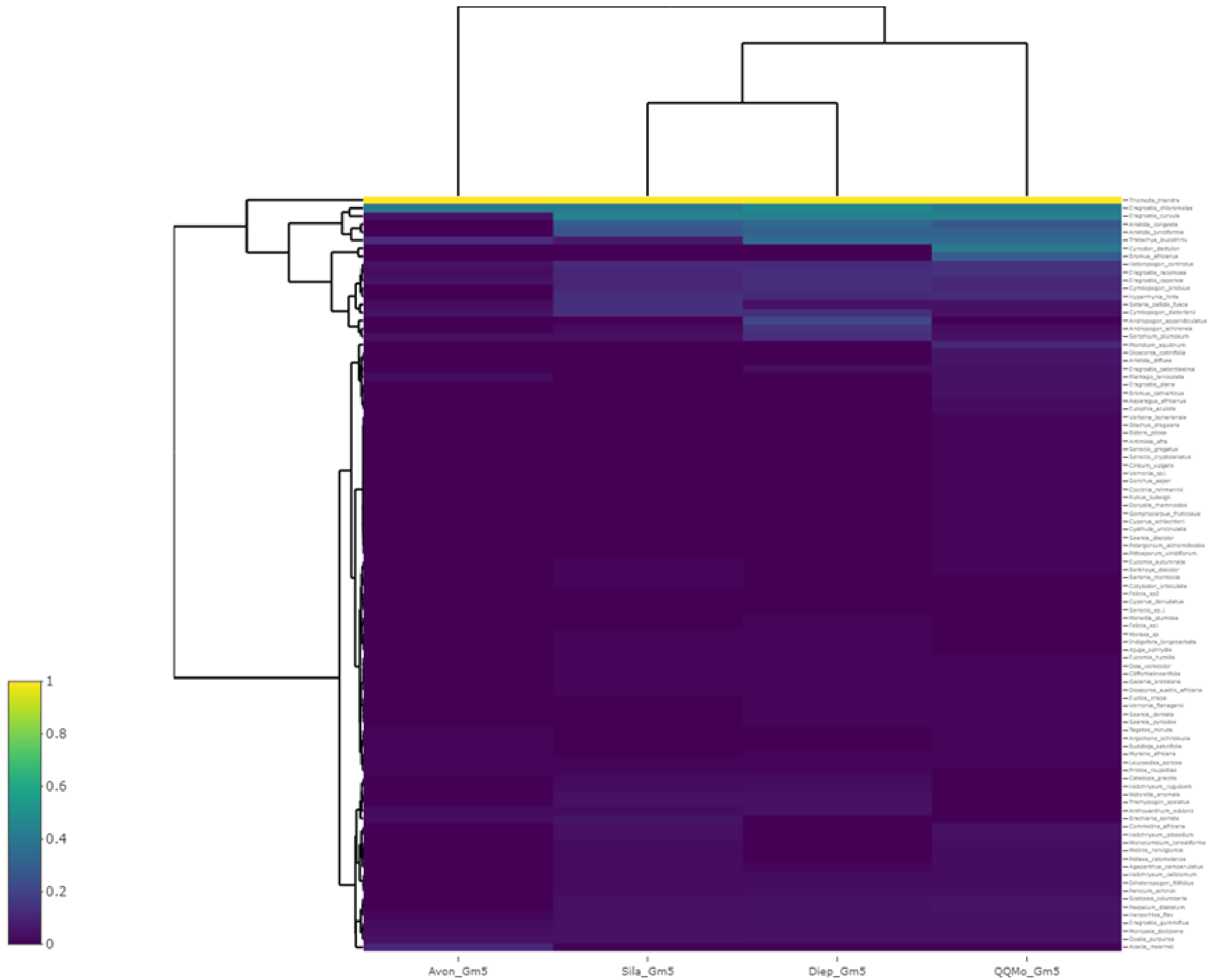

Figure 3.7: Illustration of the plant community structures consolidated using a two-way species cluster (Euclidean distance measure and Ward hierarchical cluster) of the Basotho Montane Shrubland (Gm5) grassland vegetation type in GGHNPark.

Table 3.4: Vegetation classification illustrating floristic composition and structure for Mesic Highveld Grassland vegetation unit for Eastern Free State Sandy (Gm4) and Basotho Montane Shrubland (Gm5) vegetation types in GGHNPark.

Vegetation classification (physiognomy)
Eastern Free State Sandy (Gm4)
1. *Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn. - Erg.chl.*) community
1.1. *Thm.trn. - Erg.chl. - Tristachya leucothrix* sub-community
1.2. *Thm.trn. - Erg.chl. - Aristida diffusa* sub-community
1.3. *Thm.trn. - Erg.chl. - Andropogon appendiculatus*
2. *Andropogon appendiculatus - Eragrostis capensis* (*And.app. – Erg.cpn.*) community
2.1. *And.app. – Erg.cpn. - Eragrostis curvula* sub-community
Basotho Montane Shrubland (Gm5)
1. *Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn. - Erg.chl.*) community
1.1. *Thm.trn. - Erg.chl. - Tristachya leucothrix* sub-community
1.2. *Thm.trn. - Erg.chl. - Leucosidea sericea* sub-community
1.3. *Thm.trn. - Erg.chl. -– Helichrysum pilosellum* sub-community
2. *Aristida diffusa - Pteridium aquilinum* (*Ars.dff. – Ptr.aql.*) community
2.1. *Ars.dff. – Ptr.aql. - Helichrysum callicomum* sub-community

Environmental disturbance impact on plant abundance-cover

The disturbance factors that were shown to have the greatest influence on plant abundance-cover and density were soil erosion and plant invasion, while other variables had less influence. In the Eastern Free State Sandy grassland, soil erosion was highest in Honing Kloof, Korfshoek and Twijhoek, while it was least in Welverdiend sites. The plant invasion was mostly caused by encroachment of *Seriphium plumosum* shrub which was higher at the Korfshoek and Twijhoek sites. In Basotho Montane Shrubland, there were two prominent invaders, namely *Cynodon dactylon* grass in the QQ Mountain site and *Acacia mearnsii* trees in the Avondrust site, while soil erosion was uniformly lower in all sites. Canonical eigenvalues indicated linear combinations with moderate gradients (between 0.090 and 0.262) for a percentage variance of 6.579 % for a relative inertia adjusted to abundance-cover, and while a percentage variance of 71.146 % was for relative weighted averages and site environmental factor classes for sites of the Eastern Free State Sandy grassland (Figure 3.8a). Sites of the Basotho Montane Shrubland had a percentage variance of 4.278 % for relative inertia adjusted to abundance-cover, as well as a percentage variance of 63.693 % for relative weighted averages and site environmental factor classes (Figure 3.8b). Other environmental variables that showed influence were agriculture farming history and wild population densities for both the Eastern Free State Sandy and the Basotho Montane Shrubland sites.

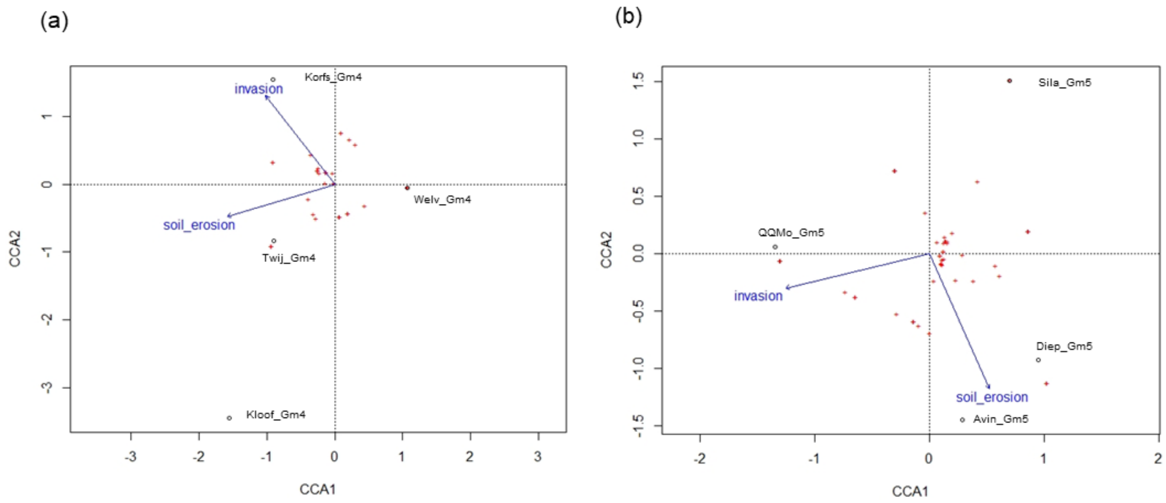

Figure 3.8: Canonical Correspondence Analysis (CCA) showing a constrained ordination for the degree of plant invasion and soil erosion disturbance influence on plant abundance-cover, (a) the Eastern Free State Sandy, and (b), the Basotho Montane Shrubland grassland vegetation types in GGHNPark.

Adopting Biotope Quality Index (BQI) for vegetation biomonitoring

Testing plant abundance-cover and density data scale

The Braun-Blanquet and Domin plant abundance-cover scale as well as percentage density data had a more or less similar variance range between sites. There was a strong negative correlation between soil erosion and the Biotope Quality Index (*BQI*) with the correlation coefficient value for the Braun-Blanquet scale (R^2 = 0.822), the Domin scale (R^2 = 0.822) and the density percentage cover scale (R^2 = 0.817) for the Eastern Free State Sandy sites. However, soil erosion had moderate relations with *BQI* in the Basotho Montane Shrubland sites with Braun-Blanquet scale correlation coefficient (R^2 = 0.196), Domin scale (R^2 = 0.196) and density percentage cover scale (R^2 = 0.175). Plant invasion indicated a weak association with *BQI* for both the Basotho Montane Shrubland and the Eastern Free State Sandy sites with the correlation coefficient for Braun-Blanquet scale (R^2 = 0.050), Domin scale (R^2 = 0.056) and density percentage scale (R^2 = 0.055), as well as with the correlation coefficient for Braun-Blanquet scale (R^2 = 0.013), Domin scale (R^2 = 0.014) and density percentage scale (R^2 = 0.018) respectively. However, the comparative response between *BQI* and the degree of disturbance had an inverse proportional influence for both soil erosion (Figure 3.9) and plant invasion (Figure 3.10) for the Eastern Free State Sandy and Basotho Montane Shrubland sites using all abundance-cover scales.

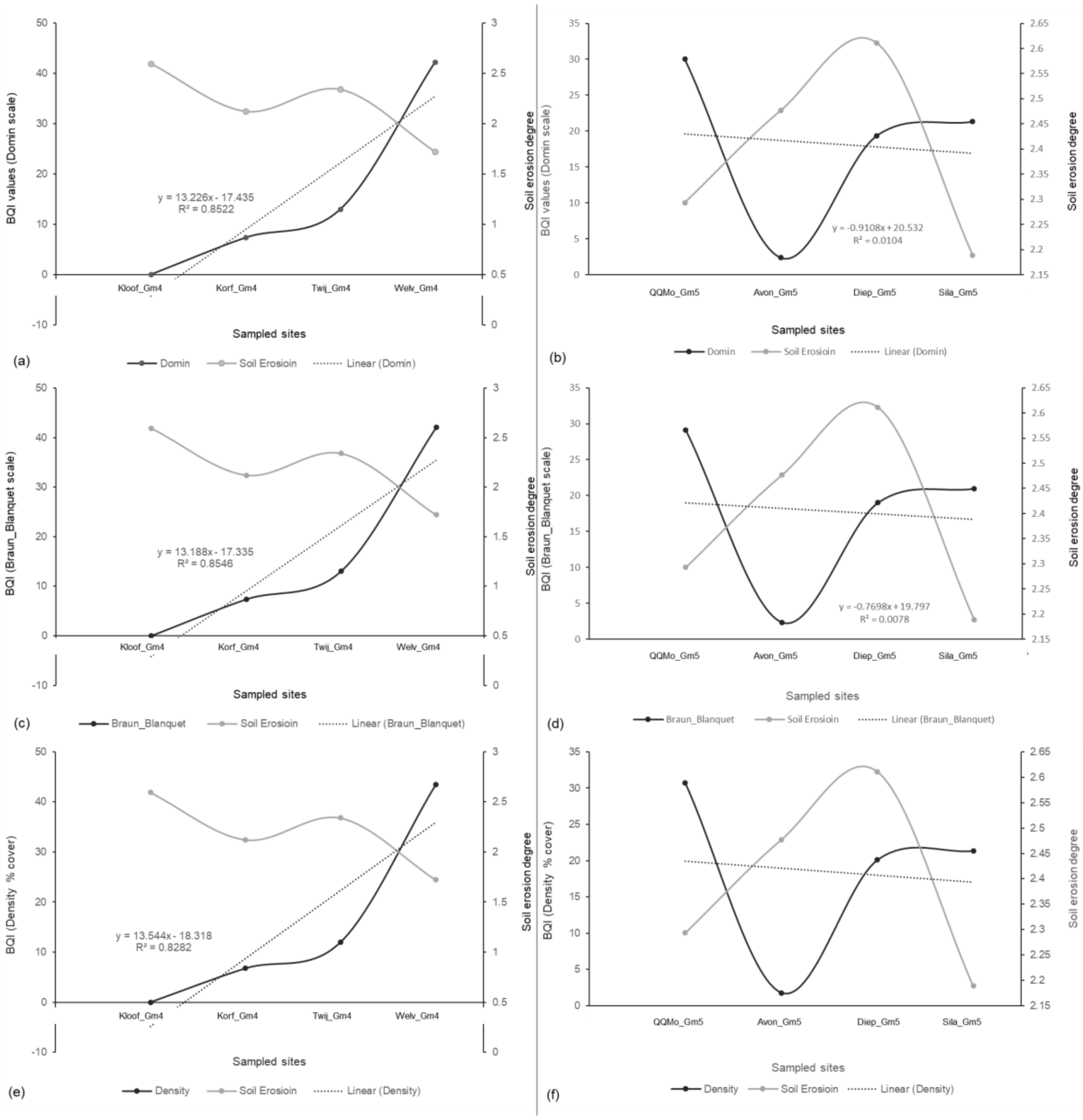

Figure 3.9: Comparison of plant abundance-cover scale (Braun-Blanquet, Domin and Density) formats to assess relationship between Biotope Quality Index (*BQI*) and soil erosion disturbance degrees for the Eastern Free State Sandy (Gm4) and Basotho Montane Shrubland (Gm5) grassland.

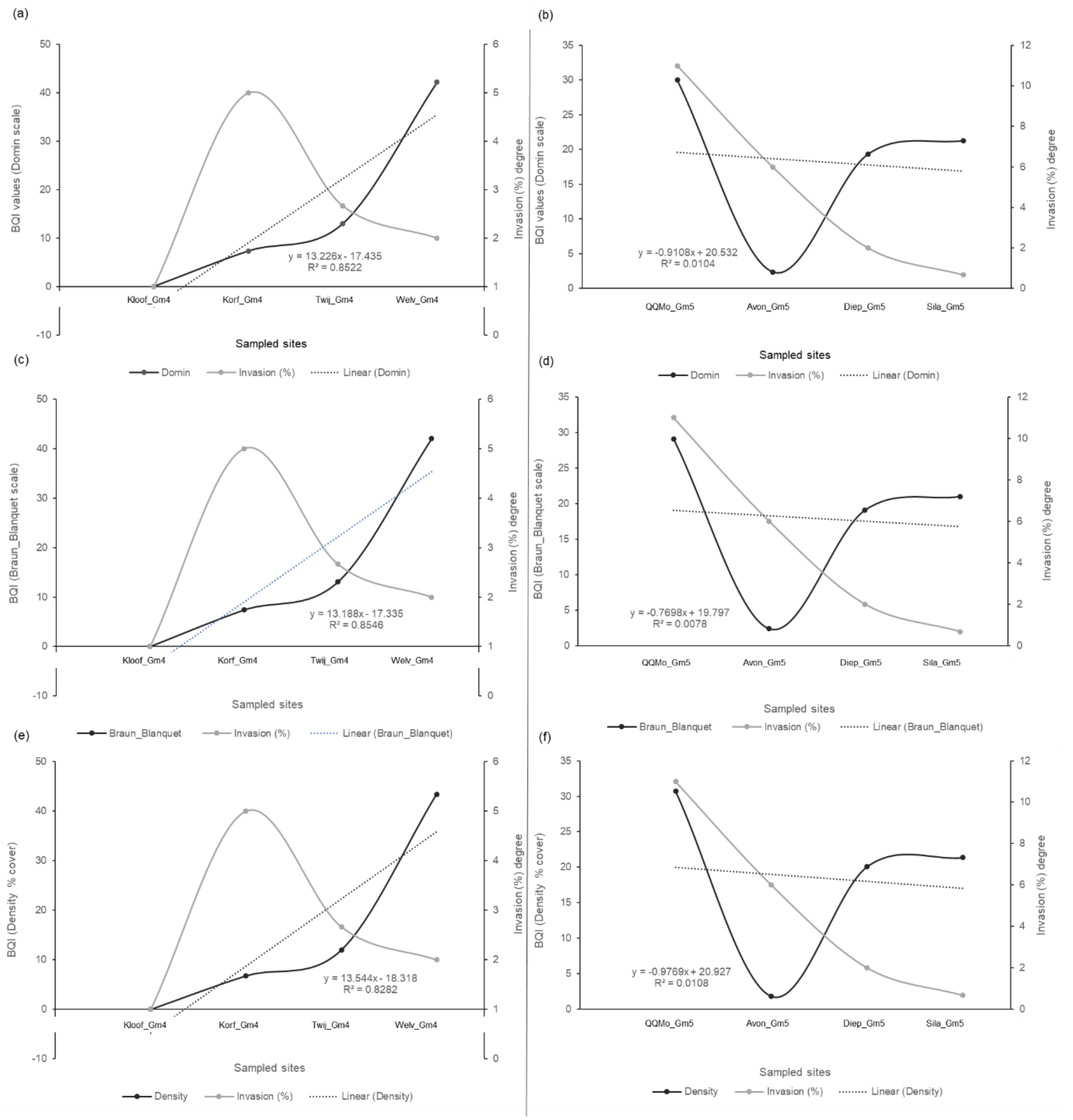

Figure 3.10: Comparison of plant abundance-cover scale (Braun-Blanquet, Domin and Density) formats to assess relationship between Biotope Quality Index (*BQI*) and plant invasion disturbance degrees for the Eastern Free State Sandy (Gm4) and Basotho Montane Shrubland (Gm5) grassland vegetation types in GGHNPark.

Testing BQI plant taxa representative groups

Most representative taxa from the vegetation bioregion tested indicated a positive correlation for the *BQI* values along the degree of disturbance gradient. There was a higher correlation coefficient value in BQI_{gra} (grasses) with R^2 of 0.852, BQI_{mon} (monocots) with R^2 of 0.845, BQI_{veg} (all plants) with R^2 of 0.845, BQI_{for} (forbs) with R^2 of 0.778, BQI_{dic} (dicots) with R^2 of 0.744, and least for BQI_{gmd} (graminoids) with R^2 of 0.723 (Figure 3.11), for the Eastern Free State Sandy grassland vegetation type in GGHNPark. However, the QQ Mountain site had the highest BQI value and degree of disturbance (Figure 3.9 and Figure 3.10) but all other sites conformed to the disturbance gradient. Furthermore, the Basotho Montane Shrubland grassland vegetation type had a higher correlation coefficient value for BQI_{dic} (dicots) with R^2 of 0.164, BQI_{for} (forbs) with R^2 of 0.099, BQI_{veg} (all plants) with R^2 of 0.056, BQI_{gmd} (graminoids) with R^2 of 0.035, BQI_{mon} (monocots) with R^2 of 0.018 and least for BQI_{gra} (grasses) with R^2 of 0.001 (Figure 3.11 and Figure 3.12).

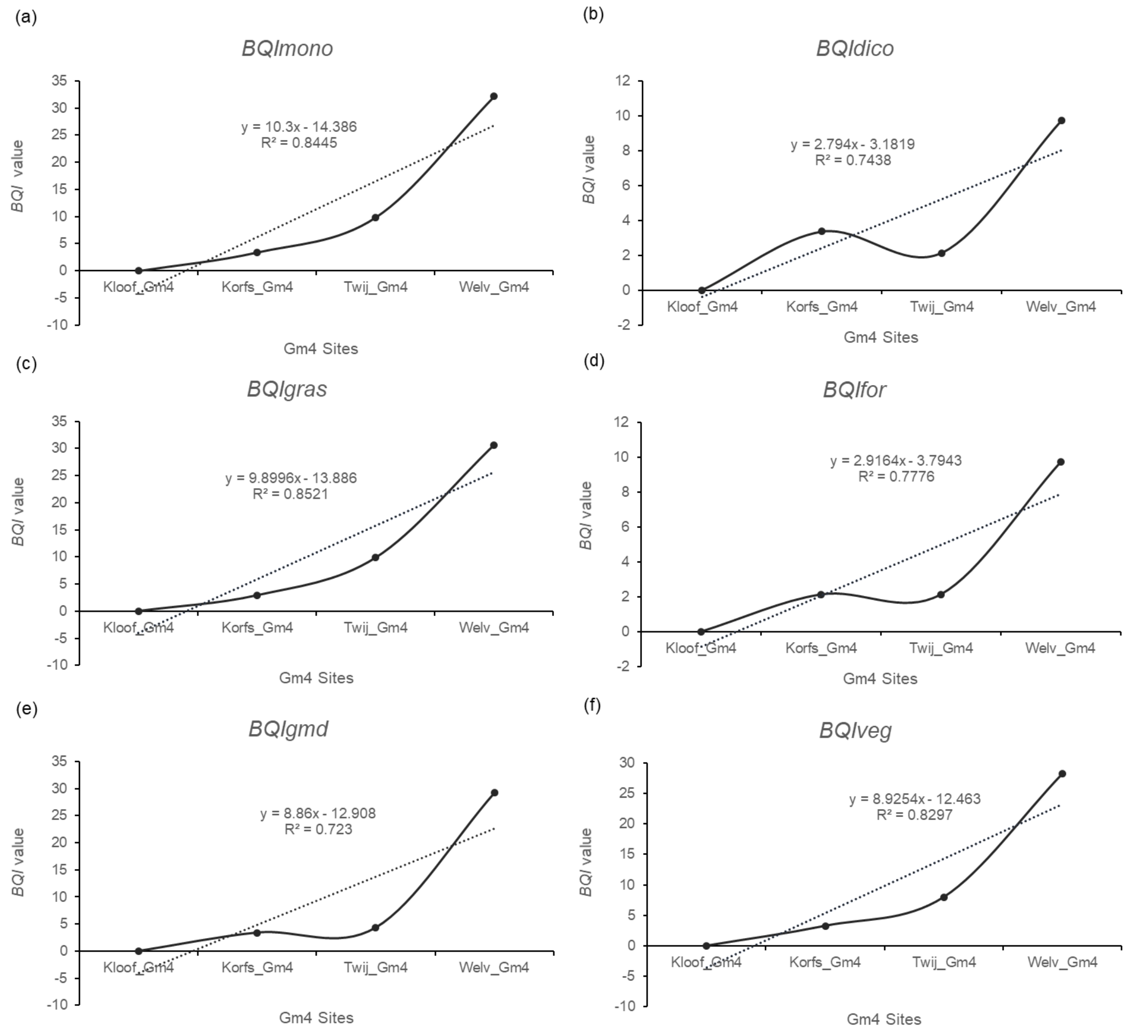

Figure 3.11: The representative categories of vegetation taxa for the Biotope Quality Index (*BQI*) were (a) monocots BQI_{mon}, (b) dicots BQI_{dic}, (c) grasses BQI_{gra}, (d) forbs BQI_{for}, (e) graminoids BQ_{gmd}, and (f) all plants BQI_{veg}, for the Eastern Free State Sandy (Gm4) grassland vegetation type in GGHNPark.

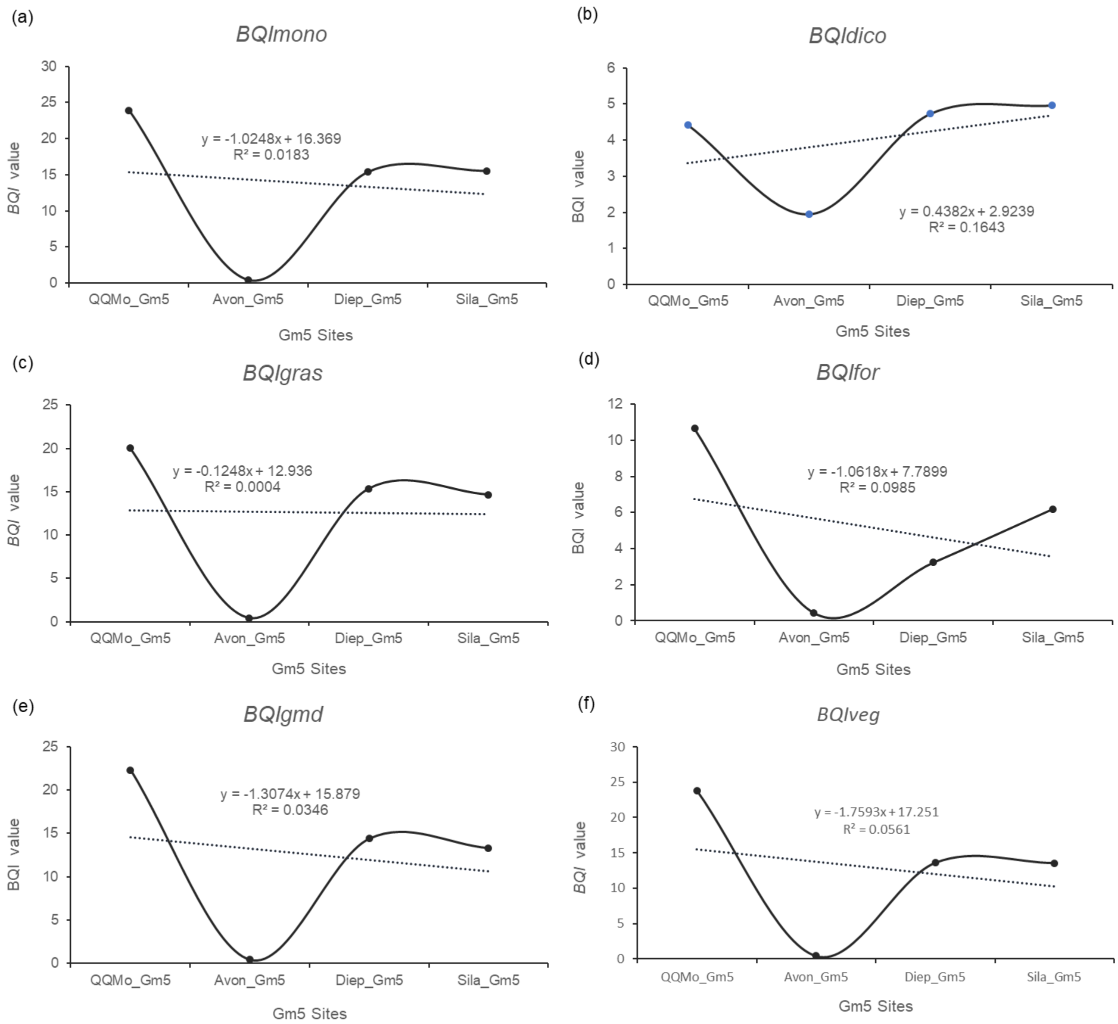

Figure 3.12: Categories of representative vegetation taxa for the Biotope Quality Index (*BQI*) were (a) monocots BQI_{mon}, (b) dicots BQI_{dic}, (c) grasses BQI_{gra}, (d) forbs BQI_{for}, (e) gramoniods BQ_{gmd}, and (f) all plants BQI_{veg}, for the Basotho Montane Shrubland (Gm5) grassland vegetation type in GGHNPark.

Biotope Conservation Status (BCS) measure

The extent of the main disturbance variables to the conservation status of the biotope indicated that the sites with a higher degree of disturbance had a lower *BCS* value. In that, the Eastern Free State Sandy grassland Welverdiend site had a higher *BCS* score of 83.086 % attributed to a relative higher *BQI* value, a lower degree of disturbance and good conservation status, while other sites, namely Honing Kloof with a lower *BCS* value of 0.001 %, Korfshoek *BCS* score of 3.114 % and Twijhoek *BCS* score of 12.114 %, were attributed to a relative lower *BQI* value, a higher degree of disturbance and highly disturbed status. The Basotho Montane Shrubland had the highest *BCS* score of 72.330 % at Silasberg site attributed by a relative moderate *BQI* value, a lower degree of disturbance and good conservation status, Diepkloof site had a *BCS* score of 20.270 % attributed by relative moderate *BQI* value, lower degree of disturbance and poorly conserved condition, while the lowest was for the QQ Mountain site with a *BCS* score of 7.74 % but attributed with a relative higher *BQI* value, higher degree of disturbance and poorly disturbed status, and lastly, the Avondrust site with *BCS* score of 0.219 % attributed by a relative low *BQI* value, a high degree of disturbance and highly disturbed status (Table 3.5). The Biotope Quality Index as geospatial modelling Feature analysis parameter was able to visually highlight areas within that park with a varying degree of disturbance for the Eastern Free State Sandy (Figure 3.13a) and the Basotho Montane Shrubland sites (Figure 3.13b), while the *BQI* estimates were simulated for the representation of other sites with similar vegetation and ecological status.

Table 3.5: Biotope Quality Index and Biotope Conservation Status scores for the Eastern Free State Sandy and Basotho Montane Shrubland sites in GGHNPark.

	Eastern Free State Sandy				Basotho Montane Shrubland			
	Kloof	Korfs	Twij	Welv	QQMo	Avon	Diep	Sila
BQI	0.001	3.253	7.968	28.180	23.824	0.419	13.605	13.565
Erosion (°)	2.595	2.121	2.339	1.722	2.293	2.477	2.611	2.293
Invasion (°)	0.001	5.00	2.667	2.001	11.000	6.000	2.000	0.667
BCS (score %)	0.001	3.114	12.967	83.086	7.348	0.219	20.270	72.330
BCS (condition)	HDC	HDC	HDC	GCC	HDC	HDC	PDC	GCC

*abbreviations: Biotope Quality Index (*BQI*); for site names Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4), Welverdiend (Welv_Gm4), QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5); for Biotope Condition Status (*BCS*) Highly Disturbed Status (HDC), Good Conserved Status (PDC) and Good Conserved Status (GCC) respectively.

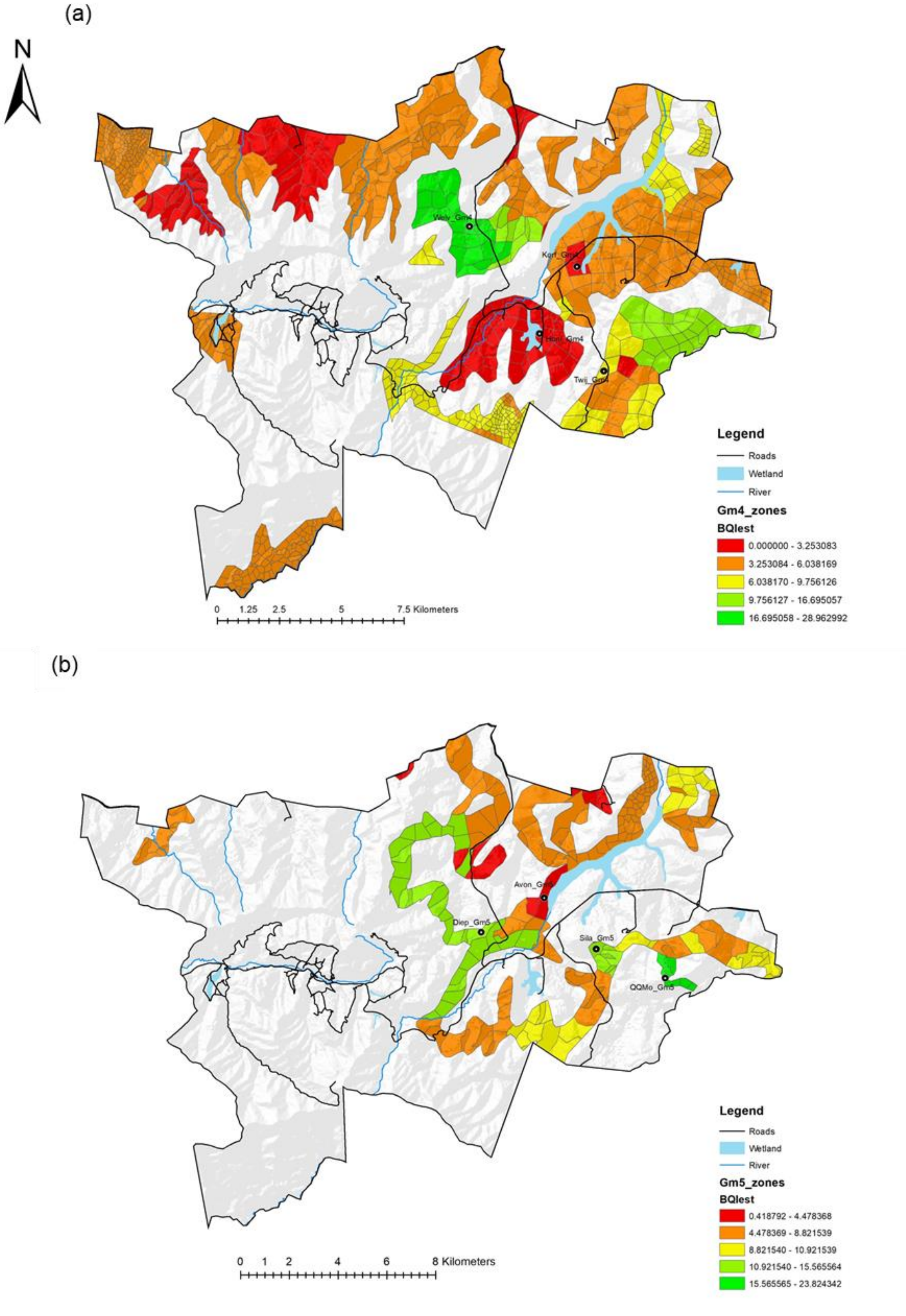

Figure 3.13: Ecological monitoring with the Biotope Quality Index (*BQI*) as a geospatial Feature analysis parameter for (a) Eastern Free State Sandy (Gm4), (b) and Basotho Montane Shrubland (Gm5) vegetation type sites in GGHNPark.

Supplementary species and diversity surrogates

The Eastern Free State Sandy grassland had mixed species surrogate population composition patterns, in that the Welverdiend site had higher estimated species richness (S_{max} = 50.083), Shannon-Wiener diversity (H = 1.323) and a rarefaction of 1.167 as attributes, while the Twijhoek site had moderate estimated species richness (S_{max} = 32.962), Shannon-Wiener diversity (H = 1.096) and a rarefaction of 0.385 as attributes, the Korfshoek site had an estimated species richness (S_{max} = 24.500), Shannon-Wiener diversity (H = 1.024) and a rarefaction of 0.333 as attributes, and lastly, the Honing Kloof site with lower estimated species richness (S_{max} = 3.000), Shannon-Wiener diversity (H = 0.469) and a rarefaction of 0.001 as attributes (Figure 3.14a,c). However, the Basotho Montane Shrubland grassland had its unique mixed species surrogate population composition patterns, in that QQ Mountain had higher estimated species richness (S_{max} = 132.800), Shannon-Wiener diversity (H = 1.471) and a rarefaction of 3.400 as attributes, while the Diepkloof site had moderate estimated species richness (S_{max} = 76.000), Shannon-Wiener diversity (H = 1.333) and a rarefaction of 3.000 as attributes, the Silasberg site had estimated species richness (S_{max} = 58.909), Shannon-Wiener diversity (H = 1.326) and a rarefaction of 1.273 as attributes, and lastly, the Avondrust site had lower estimated species richness (S_{max} = 27.500), Shannon-Wiener diversity (H = 0.893) and a rarefaction of 1.000 as attributes (Figure 3.14b,d).

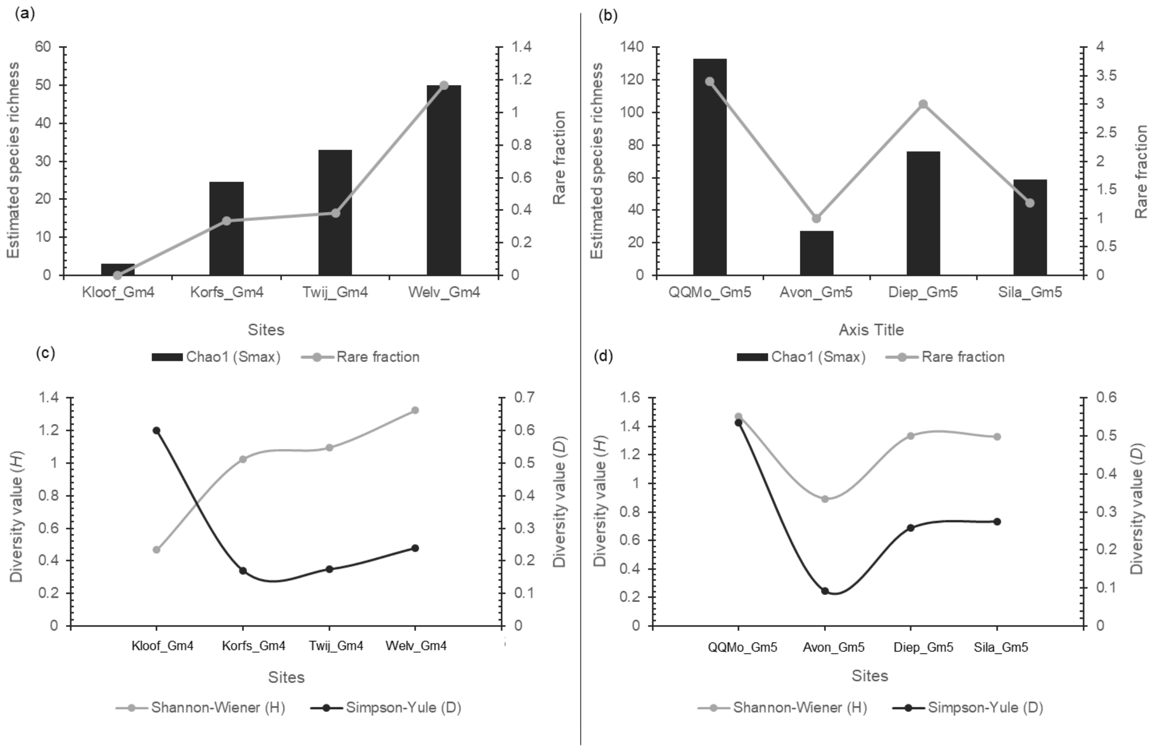

Figure 3.14: Comparison of species surrogates for (a, b) Chao1 Index and Rarefaction (c, d) Shannon-Wiener diversity and Simpson-Yule diversity index between Eastern Free State Sandy and Basotho Montane Shrubland sites in GGHNPark.

Discussion

The composition of floristic life-form was different between the Mesic Highveld grassland bioregion with five vegetation layers namely climber and vine, herb and forb, grass and sedge, as well as shrub and tree represented for the Basotho Montane Shrubland, while only three vegetation layers that included herb-forb, grass-sedge and shrub were observed the in Eastern Free State Sandy grassland. The variation in floristic life-form composition is attributed by geomorphology, elevation, soil substrate properties and micro-climatic conditions (Bredenkamp and Brown, 2003). In that, Basotho Montane Shrubland rocky slopes with shallow well-drained soils favour establishment of more woody plants (shrubs and trees) with a mosaic open and dense shrubland strip patches, than adjacent relative flat drier soil Eastern Free State Sandy grassland surface with prominent mixture of short and tall grasses (Mucina and Rutherford, 2006; Brand et al., 2009). The neighbouring bioregions were both dominate by the *Themeda triandra* - *Eragrostis chloromelas* grass community, and were further distinctly differentiated by the *Andropogon appendiculatus - Eragrostis capensis* grass community for the Eastern Free State Sandy as well as *Aristida diffusa* - *Pteridium aquilinum* grass-fern community for the Basotho Montane Shrubland. Additionally, Poaceae and Asteraceae species contributed to most abundance-cover and diversity at most sites. However, ecological disturbances are also known to influence grassland communities, including low temperatures, frost and snow (Mucina and Rutherford, 2006), soil degradation and erosion (Pietola et al., 2005), fire and grazing pressure (O'Connor and Bredenkamp, 1997; Augustine and McNaughton, 1998; Short et al., 2003). Most studies suggest that there are complex synergistic processes (biotic and abiotic interactions) that work together to manage and maintain the floristic composition and structure of highland grassland vegetation types prone to disturbance.

Honing Kloof, Korfshoek and Twijhoek sites had the least species abundance-cover and diversity observed attributed with higher soil erosion degree, compared to the Welverdiend site with relatively higher species abundance-cover and diversity with lower soil erosion degree with modified two-way ordinal hierarchical clustering. Again, Canonical Correspondence Analysis assisted in finding environmental disturbance variables that significantly influenced plant abundance-cover between evaluated sites. The historical data records of Golden Gate Highlands National Park (GGHNPark) indicated that the Mesic Highland bioregion was previously used as agricultural crop fields, especially along the river valley where the soil was rich in nutrients in the region (Rademeyer and van Zyl, 2014). Most of the Eastern Free State Sandy vegetation types, which included Honing Kloof, Korfshoek and Twijhoek were agricultural field farms except the Welverdiend site, before the farms were integrated into GGHNPark. Many studies have indicated that land cultivation and transformation can amplify soil erosion and topsoil loss (Wang et al., 2004; Huan et al., 2019), reduce the native plant abundance-cover ratio (Gossner et al., 2016), richness (Smart et al., 2006), and diversity (Dornelas et al., 2014). The extent of land degradation state and the type of restoration process can affect the duration to restore larger fields in grassland areas (Andrade et al., 2015), therefore, the recovery process in Honing Kloof and Korfshoek, which were used intensively as agricultural fields, are still with low plant species abundance-cover compared to other sites even after that section was incorporated into the GGHNPArk protected area. The results coincide with previous studies indicating plant sub-communities that prominently had *Tristachya leucothrix, Cymbopogon dieterlenii and Hyparrhynia hirta* that are regarded as drier grass communities in this lower soil moisture affinity in the Eastern Free State Sandy sites (Mucina and Rutherford, 2006). Dense vegetation cover can assist in controlling soil erosion in that it reduces surface water runoff (Puigdefàbregas, 2005) and alleviates runoff denudation of the soil (Li et al., 2005). Therefore, sites with a lower ground surface vegetation coverage were more prone to higher soil erosion.

The *Seriphium plumosum* was the common invasive shrub which mainly encroached on drier grass communities, which is reported to be an aggressive *Cymbopogon-Themeda* grass community (Snyman, 2011a). *Seriphium plumosum* has a faster growth rate to outcompete for space and resources with native plants (Snyman, 2009), capability to modify soil characteristics and reduction of grass density-cover (Snyman, 2011b), and as a result, reducing grazing capacity of grassland (Wepener et al., 2008). However, it is unclear how *Seriphium plumosum* shrub changed or adapted to become an invasive shrub, especially in and around fields that were previously used for intensive crop farming practices (Wepener, 2007; Snyman, 2011b). On the other hand, two invasive plants have been reported for the Basotho Montane Shrubland vegetation type, namely *Cynodon dactylon* grass at the QQ Mountain site and *Acacia mearnsii* trees at the Avondrust site. The two invasive plants had different impacts on the affected sites, in that *Cynodon dactylon* growth was restricted to patches within native plant shrubland communities. Regardless of restrictions, it is reported by Fonseca et al. (2013) that *Cynodon dactylon* was still able to outcompete immediate neighbouring plant cover on site and change native plant community structure. In contrast, invasion of *Cynodon dactylon* on a site with higher species abundance-cover, diversity and *BQI*, in that, the native plant diversity might have assisted in limiting expansion to restrict its invasion. Many studies have reported that local biotic integrity and diversity play a role in resilience and stability as an ecosystem service (Fischman, 2004). Furthermore, the Avondrust site was reported to have a high invasion of *Acacia mearnsii* trees and was attributed to very low plant abundance-cover and diversity. The *Acacia mearnsii* trees have been reported to release allelochemicals that alter soil structure properties and nutrient flow (Moyo and Fatunbi, 2010), as well as outcompete native flora for space and resources, such as groundwater (de Neergaard et al., 2005), thus making conditions unfavourable for native plants to establish well. Additionally, montane shrubland slopes have a thin topsoil layer and bedrock closer to the ground surface, which could affect weathering and leaching for nutrient-retention capacity of the soil (Grab and Knight, 2015). Overall, the estimated abundance-cover and species representation between the Eastern Free State Sandy sites were different due to the intense effects of ecological disturbances that showed significant differences, while there was not much difference between the Basotho Montane Shrubland sites.

The Biotope Quality Index was able to indicate and demonstrate areas of conservation concern in the GGHNPark for vegetation bioregion monitoring. Three commonly used vegetation abundance-cover scales and density measure that include Domin, Braun-Blanquet scale and percentage density were proved to be all compatible as data collection and recording for Biotope Quality Index, but taking into consideration normalised log transformed data. However, modified Domin and Braun-Blanquet scales with more than 6 ordinal abundance-cover estimate classes work better to increase variance among collected plant species per site to include rare uncommon individuals. Although the Biotope Quality Index only considers species with abundances that are above the site average for evaluation, the overall *BQI* value also depends on standard deviation at the time of collection. Furthermore, vegetation types are named after common and dominant plant species communities, results indicated that although Poaceae and Asteraceae were the most common families, and a positive correlation was observed for specific plant taxa (monocots, dicots, forbs, and grass) for the *BQI* measure, we advise that all vegetation data list which include larger species variation forms must be considered when calculating *BQI* value to avoid bias and eclipsing of variable effects (Abbasi and Abbasi, 2011; Green and Chapman, 2011; Bredenhand, 2014). Again, ecological integrity dataset which considers dominant, common and rare occurring species reflects more realistic and accurate estimated abundance, ecosystem interactions and responses (Colwell et al., 2012; Chao et al., 2014).

The Biotope Quality Index also indicated its ability to be used as a disturbance monitoring surrogate to detect the impact of plant invasion and soil erosion on floristic composition and structure of highland grassland vegetation types. Therefore, sites with a higher *BQI* value were attributed to a lower degree of disturbance, especially the Welverdiend (represented by Gm4) and Silasberg (represented by Gm5) sites. Sites with a higher degree of disturbance had a lower *BQI* value, namely in Honing Kloof prone to soil erosion, Korfshoek prone to encroachment by *Seriphium plumosum* shrub and invasion by *Acacia mearnsii* tree in Avondrust site. The Biotope Quality Index was able to highlight the resilience properties of the moist Basotho Montane Shrubland at the QQ Mountain site with higher diversity and *BQI* value. This area was able to limit and restrict the impact of *Cynodon dactylon* grass invasion. A biotic index must be able to indicate impact of tested environmental disturbance variables (Diaz et al., 2004), also reflect possible status of environmental conditions (Brussaard, 1998), and resilience-stability capacity of ecosystem under perturbing disturbance (Lee and Lautenbach, 2016). Therefore, the Biotope Quality Index has properties that should be exempted for ecological or environmental biomonitoring using vegetation units as surrogates to measure the impact of disturbance and to evaluate the conservation status of bioregions. Furthermore, baseline data for reference sites can be used to estimate the *BQI* value through site conditions and description relatedness to other similar sites, for its application as a geospatial data Feature analysis parameter to highlight the environmental condition of highlands grassland bioregions on a graphic GIS map illustration.

The Biotope Conservation Status (*BCS*) score measure is suggested as an additional step for the Biotope Quality Index (*BQI*) in evaluating vegetation types. The proposed *BCS* scoring (percentage) system indicated that sites that are prone to a higher degree of disturbance, that is, Avondrust (Gm5) and Honing Kloof (Gm4) had a lower *BCS* score and classified as a highly disturbed status, the Diepkloof (Gm5) had a moderate *BCS* score and classified poorly conserved status, and while sites with a higher *BCS* score were Silasberg (Gm5) and Welverdiend (Gm4) with a lower degree of disturbance as well as good conserved status category. Plants make more reliable indicators of long-term environmental change and disturbance monitoring units (Gibbons and Freudenberger, 2006; LaPaix et al., 2009). Therefore, Biotope Conservation Status scoring is recommended as an additional step for Biotope Quality Index evaluation methods, to assist in categorising environmental conservation status of impacted sites.

Conclusion

The plant species community *Themeda triandra - Eragrostis chloromelas* was common and dominated the two adjacent Mesic Highland grassland vegetation types. The relatively flat valley surface of the Eastern Free State Sandy grassland was distinct with the *Andropogon appendiculatus - Eragrostis capensis* grass community and was attributed to higher soil erosion effects as well as the invasion of *Seriphium plumosum* encroaching shrub, while rocky slopes of the Basotho Montane Shrubland grassland had *Aristida diffusa* - *Pteridium aquilinum* grass-fern community and was attributed to lower uniform soil erosion effects as well as invasion by *Cynodon dactylon* grass and *Acacia mearnsii* trees. The sites that were mostly impacted by soil erosion were historically used as crop farming and cultivation, to whom they were likely introduced plant species (grass, shrub and tree respectively) in the region.

The Biotope Quality Index provides information on both biotic integrity and the quality conditions of grassland vegetation types. The Biotope Quality Index should be used for evaluating vegetation bioregions for ecology research, environmental management and GIS as a species surrogate to indicate effects of soil erosion, encroachment by local species and invasion by alien species. Depending on the type of scale or method of species abundance-cover and density, data log transformation is required to execute the *BQI* formulation to normalise the dataset. The *BQI* will only be applicable if there is variance observed between species abundance-cover or density per site and among species. It is recommended that after defining the criteria to describe and differentiate bioregions into biotope units, the species data sets should be analysed separately per biotope units, even for bioregion comparison studies or projects. Additionally, the Biotope Conservation Status (*BCS*) score can be used as a supporting step to measure the progress of environmental conservation efforts within protected bioregions and areas.

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CHAPTER 4 : Evaluating Effects of Plant Encroachment and Soil Erosion towards Arthropod Communities using Biotope Quality Index for Eastern Free State Sandy Grassland in Golden Gate Highlands National Park, South Africa.

Abstract

Temperate grassland bioregions are prone to environmental disturbance such as agricultural practices, herbivore overgrazing, habitat fragmentation, uncontrolled fires and alien plant invasion. The Biotope Quality Index could be used to identify and evaluate the magnitude and extent of the influence of environmental disturbance on grassland vegetation type using arthropod assemblages. Sampling in the form of active net sweeping and pitfall trapping were used to collect arthropods in the Eastern Free State Sandy grassland vegetation type in the Golden Gate Highlands National Park. Composition of arthropod taxon included 68 % Insecta, 13 % Entognatha, 6 % Arachnida, 5 % Diplopoda, 5 % Chilopoda, 2 % Acari and 1 % Malacostraca. The Biotope Quality Index had greater variation between sites being highest in Welverdiend (*BQI* = 137.533), Korfshoek (*BQI* = 111.183), Twijhoek (*BQI* = 109.812) and least for Honing Kloof (*BQI* = 11.058). There was a negative correlation between soil erosion and Biotope Quality Index (R^2 = 0.602), while plant encroachment and the Biotope Quality Index had a positive correlation (R^2 = 0.374). The woody shrubs showed a significant impact were *Seriphium plumosum* and *Tagetes minuta*. The Biotope Quality Index indicated that the Honing Kloof and Korfshoek sites were previously used as farmlands had poor ecosystem status, compared to Welverdiend and Twijhoek sites with higher *BQI* values and good ecosystem statuses. The Biotope Quality Index was successful to detect sites that were negatively affected by soil erosion and invasive woody plants, as well as to highlight the magnitude of each environmental disturbance. A holistic grassland recovery initiative is recommended and the Biotope Quality Index can be used as a biomonitoring tool to assess progress towards the restoration of heavily impacted sites of the Eastern Free State Sandy grassland.

Keywords

Arthropod assemblages, Afromontane, Bioassessment, Biodiversity, Disturbance, Insect Community, Invasive plants, *Seriphium plumosum*

Introduction

The grassland biome is a biogeographical region characterised by a floristic composition that is dominated by Poaceae grasses and with a frequent inclusion of other graminoid (Cyperaceae, Juncaceae and Restionaceae), herbaceous plants and low shrubs (Dixon et al., 2014). The grassland biome covers more than 30 % of global terrestrial land surface area (Mucina and Rutherford, 2006), occurring in a wide range of both temperate and tropical climate regimes (New, 2019). In South Africa, temperate grassland covers 339 237 km^2 (27.9 %) of land surface area, from elevation ranging from 300 m to 3482 m (a.s.l.). South African grasslands are among the three largest biomes along with the Savanna and Nama-Karoo biomes making up a total of 80 % in surface area (Carbutt et al., 2011). South Africa grassland biome has four major regional bioregions that include the Mesic Highveld, Dry Highveld, Sub-Escarpment and Drakensberg Grassland units located mainly in interior and north-eastern regions (Mucina and Rutherford, 2006). The grassland biome hosts important ecosystems such as highlands terrain, grassveld, wetlands and other freshwater systems (Dixon et al., 2014), that play a role for services such as climate and water regulation, soil and biodiversity conservation (Neke and du Plessis, 2004), as well as providing human benefit resources for agricultural practices (crop and livestock farming), mining, natural medicine and tourism (Allen et al., 2011).

However, most grassland bioregions worldwide are under pressure due to environmental disturbances that is caused by overgrazing and uncontrolled wildfires (New, 2019), anthropogenic activities human settlement development, agricultural crop field modifications and introduction of alien plants (Prinsloo et al., 2021), and challenges of climate change i.e., prolonged shift in annual temperature and precipitation patterns (Dixon et al., 2014). Threats to grassland biomes include habitat fragmentation, degradation, alien species invasion, altered ecosystem functioning and biodiversity loss (O'Connor, 2005; Wang et al., 2020). Only 2.04 % of the grassland biome is conserved in South Africa, mainly highland regions, and, while about 60 % (7750 km^2) is in an irreversible degraded state (Little et al., 2013), and it is most likely the most threatened biome in the country (Kaiser et al., 2008).

In terrestrial ecosystems, grasslands have a single layer in terms of plant height difference, which play a role in providing a vast range of non-woody graminoid, herbs and forbs variation (Carbutt et al., 2011). Understanding the evolutionary relationship between arthropods and ecological processes within grassland bioregions still poses an information gap in that, formation of important historical and biogeographic processes leading to the outline local species composition and diversity is still understudied (Joern and Laws, 2013; New, 2019). The local arthropod diversity found in grassland vegetation types is mainly influenced by ecological disturbances related to grazing, fire, and climate change, which are the primary ecological and evolutionary drivers that shape grassland systems (Knapp et al., 1998). Additionally, there have been growing conservation concerns about negative effects caused by intense land use on the loss of arthropod diversity in grassland regions (Gossner et al., 2016). The local life history and population dynamics of many epigeal and soil dwelling arthropods are driven by environmental disturbance regimes experienced from the surrounding grassland bioregion mosaic.

Arthropod species play a vital role in the restoration and conservation of grasslands worldwide. Most restoration programs that involved arthropods were focused on ecosystem processes and functioning (Mills et al., 1993), in that arthropods can be ecosystem engineers and modifiers that influence community structure and ecological integrity (Weisser and Siemann, 2003, Samways, 2005). Some are key role players as pollinators, decomposers, predators and parasitoids for biocontrol (Ollerton et al., 2003; Hudewenz et al., 2012; Maas et al., 2021), vectors of diseases causing agents, as well as being herbivores of invasive plants (Steffan-Dewenter and Tscharntke, 2002). Furthermore, arthropods are also used for environmental conservation practices as tools for monitoring ecological integrity (Krause and Culmsee, 2013), functioning (Ebeling et al., 2018), stability (Morris, 2000), resilience (Joern and Laws, 2013) and quality (Bredenhand, 2014). Insects and other arthropods have physiological and behavioural characteristics that make them better candidates as indicators of ecological change for better environmental monitoring and conservation practices (Samways, 2005).

Regardless of the important ecosystem services that grasslands provide, dynamics and impacts of anthropogenic and ecological disturbances are poorly understood in South African grasslands bioregions (O'Connor and Kuyler, 2009). Bioassessments and biotic indices utilise arthropod taxa groups to measure and monitor changes in ecosystem integrity, functioning and quality for environmental and biodiversity conservation practices (Weisser and Siemann, 2003, Samways, 2005). The quality of the grassland ecosystem reflects the functional state of ecological integrity within a bioregion, and can be monitored using biotic indices to inform conservation programmes, general public and decision-makers about the state of the ecosystem. The Biotope Quality Index is a numeric expression which uses the mean proportion of common occurring arthropods assemblage over standard deviation (log transformed), to highlight the level of disturbance to indicate the conservation status of a biotope (Bredenhand, 2014). The Biotope Quality Index assumes that any biotope in a good (pristine) conserved state, can sustain the arthropod population to optimum species numbers and their ecological processes.

$$\text{Biotope Quality Index } (BQI) = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right) \text{ if } \log \sigma > 0$$

The represented symbols include, abundance of *i*th morphospecies (x_i), found in the *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ). Bredenhand (2014) developed and demonstrated that the Biotope Quality Index has a strong correlation to disturbance levels in the Shale Fynbos bioregion which can be used as a biomonitoring tool at Jonkershoek Nature Reserve in South Africa. The Biotope Quality Index was able to highlight effects of *Pinnus radiata* plantation disturbance and seasonal change towards arthropod assemblages. Therefore, the current study adopted the Biotope Quality Index to evaluate the effects of environmental disturbance on arthropod species assemblages that occur in the Eastern Free State Sandy grassland vegetation types at the Golden Gate Highlands National Park.

Materials and Methods

Study site

Golden Gate Highlands National Park (GGHNPark) is situated along the north-eastern Maloti-Drakensberg Mountain border linking the Kingdom of Lesotho, eastern Free State and Kwa-Zulu Natal Province, cover surface area of 342 km^2 (32758 ha), located between geospatial quadrant with latitude of 28°27' S to 28°37' S and longitude of 28°33' E to 28°42' E, with altitudinal range from 1892 m to 2837 m in Figure 4.1 (Rademeyer and van Zyl, 2014). The eastern Free State region is characterised by summer rainfall which ranges between 800 mm to 1600 mm, with a mean annual temperature ranging from 13°C to 30°C, and a winter rainfall range of 500 mm to 760 mm as well as a mean annual temperature ranging from 1°C to 26°C (Grab et al., 2011). The GGHNPark is within a moist and cold highveld ecoregion that is mainly characterised by grasses and herbs with occasional shrubs and trees (Mucina and Rutherford, 2006; Telfer et al., 2012, Rademeyer and van Zyl, 2014).

The vegetation type categories described by Mucina and Rutherfiord (2006) were considered units of biotopes as shown in Figure 4.1 with a dominant common flora and clear boundaries. The Mesic Highveld Grassland Bioregion is distinctively characterised by grass (Poaceae) communities of *Eragrostis curvula*, *Tristachya leucothrix* and *Themeda triandra*, and Asteraceae herbs namely *Helichrysum* spps., *Vernonia* spps., and *Berkheya* spps. (Eckhardt et al., 2006; Kay et al., 2006). Four sites were selected for sampling in the Eastern Free State Sandy (Gm4), elevation range of 1654 to 1800 m for vegetation type. Site selection took into account ecological disturbance (soil erosion), human influenced disturbances (historical agricultural activities, housing buildings, road constructions, pollution), plant encroachment and invasion. Selected sites were named Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4) respectively.

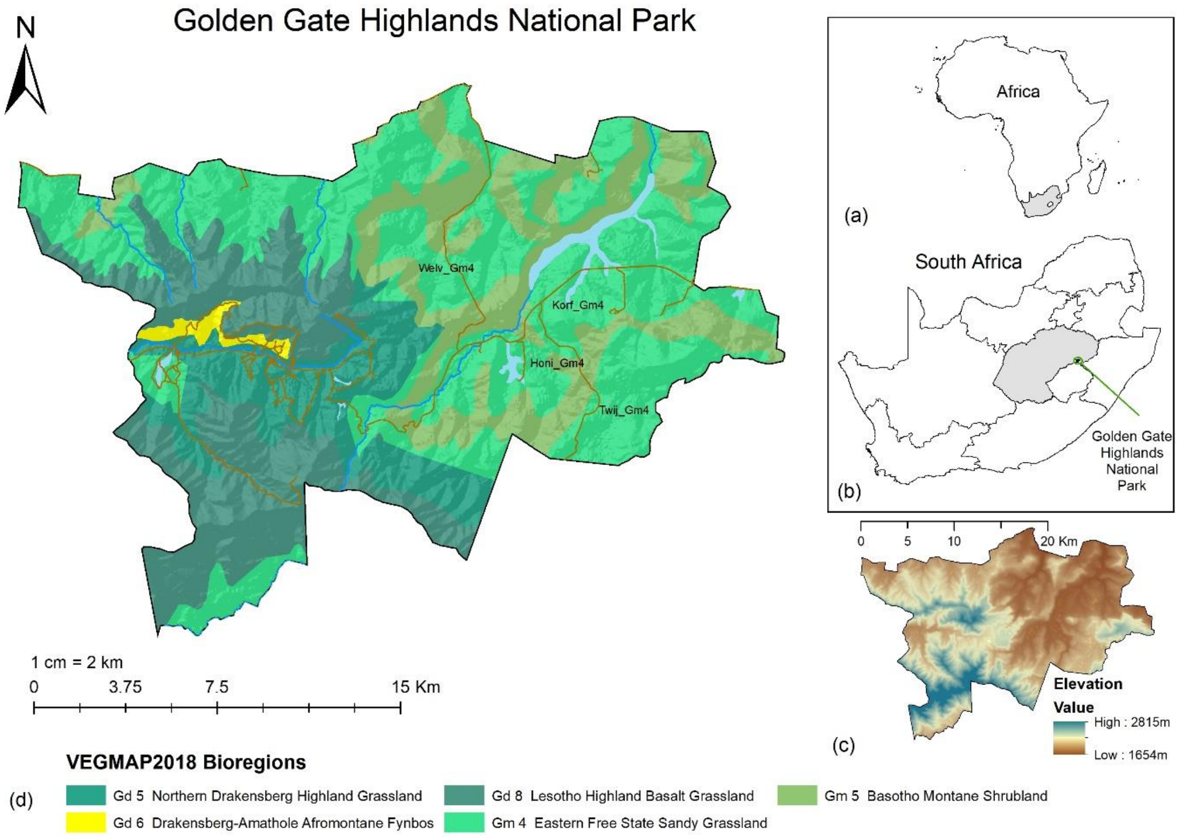


Figure 4.1: Location of study area illustrating (a) the continent of African, (b) eastern section of the Free State Province highlighted in South Africa outline, (c) GGHNPark elevation and hillshade created using DEM image from USGS Explore Earth, and (d) GGHNPark highlighting sampled sites within Gm4 vegetation type from SANBI GIS VEGMAP (2018) using ArcGIS.

Arthropod collection

Arthropod samples were collected using passive pitfall trapping, active net sweeping and field observations every two weeks of the month from September 2016 to September 2018. Pitfall sampling was carried out by inserting 10 randomly arranged 250 ml plastic jars (60 mm diameter X 100 mm height) into the ground surface, in a 200 m transect and with 20 m spacings between (Yekwayo et al., 2018). A pitfall solution of 50 ml containing ethanol and propylene glycol (3 v:1 v ratio) was used as a killing and preservation agent. The lid of the plastic jar was modified to elevate 20 mm from the ground surface to restrict rainwater runoff, prevent evaporation of the pitfall preservation solution, large objects and predators (vertebrates and invertebrates) from clogging the trap. An insect net with a 400 mm diameter steel hoop and mesh fabric sheet, was used to sweep the vegetation along the 200 m transect adjacent to the pitfall traps during the day. Field observations were also included to record aerial insects visible on shrubs and trees within a 200 m sampling transect. After the reference collection was compiled, the catch and release method was used for net sweeping samples. The collected arthropods samples were sorted and identified to family and genus taxa level, and later morphospecies. A reference (pseudonym) was assigned to each morphologically distinct specimen (Gerlach et al., 2013). The reference collection was created and housed at the University of the Free State Qwaqwa campus entomology collection. Only adult life stages were considered and data from the sampling methods was pooled for analysis.

Ecological disturbances

Ecological data was collected to measure disturbance at each sampled site. Data included topography and elevation, plant cover, soil type, as well as soil moisture were used to determine soil erosion. Plant encroachment-invasion was determined by recording the presence and percentage cover of plants classified as encroachment and alien invasives. Anthropogenic activities that contribute to disturbance were also recorded along historical agricultural activities (prior to sampled areas were declared as a national park), housing and road constructions, land pollution, current wildlife and human activity zones as well as conservation rehabilitation programs. Each sampled site was described based on current ecological conditions and historical activities (before and after the status of national park).

Data analysis

Descriptive statistics was used from arthropod counted data to measure mean, standard deviations and relative abundances for each sampled site (Legendre and Legendre, 1998; Gardener, 2012). Kruskal-Wallis H-test and the Dunn post-hoc test were used to compare the mean variation of arthropod taxa between sites with uneven sample sizes using ranked means. Sample-based rarefaction (Gotelli and Colwell, 2001) was used to determine species rare (singleton and doubleton frequencies) occurrence probability for each site. The species surrogate included biotic indices methods to calculate estimated species richness using Chao2 (Chao, 1984), diversity indices using Simpson-Yule Index (Simpson, 1949) and Shannon–Wiener function (Weaver and Shannon, 1949) as well as Biotope Quality Index (Bredenhand, 2014) to indicate the environmental status of each sample sites. The comparative association of species surrogates as a gradient to influence arthropod distributions was evaluated whose co-variation is plotted using the Canonical Correspondence Analysis (CCA) R-Studio's (log transformed abundance data) with the vegan package (Gardener, 2014).

Soil erosion was determined using the Revised Universal Soil Loss Equation (RUSLE) and then quantitative erosion estimates were grouped into erosion risk classes (supplementary material: Table 0.3) (Bartsch et al., 2002). Plants that were considered for encroachment and invasion were identified within 200 m x 100 m arthropod sampling transect, and percentage cover of invasion was determined per site (supplementary material: Table 0.2). Quantitative methods were used to determine and classify the degree of disturbance for the pollution ratio, aggregation of buildings, the density of wildlife and human activity per site (supplementary material: Table 0.4). The effect of environmental or ecological disturbance on the dynamics of arthropod community was determined using a permutational multivariate analysis of variance (PERMANOVA) using the R-Studio (ADONIS, 23 permutation simulations ran to congruency, in the vegan package) and only disturbances with significant influence were considered for further analysis. Moreover, significant quantitative factors were placed for nonmetric multidimensional scaling ordination (NMDS), transformed arthropod abundance using hellinger, with a Bray-Curtis as an ordination distance method (the vegan R-studio) to distinguish clustering influence between sampled sites. The sites were related based on arthropod and environmental data with both qualitative and weight qualitative data using Bray-Curtis dissimilarity (Bray and Curtis, 1957).

The Biotope Quality index was used for geospatial modelling to illustrate current ecological conditions of the Eastern Free State Sandy (Gm4) grassland bioregion in the GGHNPark, a modified bootstrap jackknife 2 using 999 permutations (Manly, 1991) to predict the values of the asymptotic Biotope Quality Index with varying population sizes from the calculated mean and standard deviation for neighbouring and adjacent related sites. The Gm4 bioregion was divided into units of sampling zones per locality, and then, predicted asymptotic Biotope Quality Index values were assigned to each polygon (sampling zone unit) using ArcGIS Map (version 10.5) to highlight the area of environmental concern. Shapefiles for world continental and country maps were accessed from Esri GIS database, while South Africa's provincial, SA protected areas (national parks), roads and surface water body shapefiles were accessed from Department of Forestry, Fisheries and the Environment (DFFE) EGIS database, and vegetation bioregion VEGMAP 2018 (for South Africa, Lesotho and Swaziland) shapefiles were accessed from SANBI Biodiversity GIS database.

Results

There were a total of 6079 adult arthropod specimens represented in seven Classes, 22 Orders, 105 Families and 227 Morphospecies used for analysis. The composition of arthropod Class taxon included 68 % Insecta, 13 % Entognatha, 6 % Arachnida, 5 % Diplopoda, 5 % Chilopoda, 2 % Acari and 1 % Malacostraca for overall the Eastern Free State Sandy grassland bioregion. The most dominant arthropod families in all sampled sites were Entomobryidae, Formicidae, Pholcidae, Thripidae, Cicadellidae, Aphididae, Tetranychidae, Tenebrionidae, Chrysomelidae and Anthomyiidae. The sites with the most abundant arthropods were Welverdiend and Twijhoek representing highest taxa groups (Table 4.1). The Korfshoek and Honing Kloof sites had the least abundance of arthropods with few taxa represented and more morphospecies occurring in very low frequencies (singleton and doubleton) with a high rarefaction value per site. There was a significant difference between ranked arthropod abundance means for the four uneven sample site sizes with Kruskal-Wallis chi-squared (94.736, df = 3, $p < 0,05$). Furthermore, the Dunn post-hoc test indicated that the arthropod abundances at the Welverdiend and Twijhoek sites were highly significantly different from those at the Korfshoek and Honing Kloof sites.

Table 4.1: Summary of arthropod taxa Class, Order, Family, morphospecies counts and rarefaction for Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4) Eastern Free State Sandy grassland vegetation type in GGHNPark.

		Taxon counts			Morphospecies		
	Class	Order	Family	T.count	singleton	doubleton	rarefaction
Korf_Gm4	4	16	61	116	42	12	0.466
Honi_Gm4	7	18	63	94	53	14	0.713
Twij_Gm4	7	21	94	175	33	14	0.269
Welv_Gm4	6	20	83	177	26	10	0.203
Kruskal-Wallis chi-squared = 94.736, df = 3, p-value <0,05							
Comparison of Gm4 sites mesns by group (Benjamini-Hochberg)							
	Honi_Gm4		Korf_Gm4		Twij_Gm4		
Korf_Gm4	3.291	0.0006**					
Twij_Gm4	7.065	0.0000***	3.731	0.0001**			
Welv_Gm4	8.985	0.0000***	5.773	0.0000***	2.275	0.0115*	

*ref. evaluated at $p < 0.05$ significant difference is indicated by an asterisk *, highly significant ** and ***. Abbreviations, total counts (T.count), Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4)

Arthropod assemblages and composition

The majority of arthropods in sampled sites were composed of class Insecta, Entognatha and Arachnida with few Chilopoda, Diplopoda, Acari and Malacostraca represented (Figure 4.2). The arthropod community in the Welverdiend site was composed of 73 % Insecta, 15 % Entognatha, 7 % Arachnida, 3 %, Acari, 1 % Diplopoda and 1 % Malacostraca. The arthropod community at the Twijhoek site were composed of 77% Insecta, 13 % Entognatha, 5 % Arachnida, 2 % Acari, 1 % Chilopoda, 1 % Diplopoda and 1 % Malacostraca. The arthropod community at the Korfshoek site was composed of 77 % Insecta, 16 % Entognatha, 6 % Arachnida and 1 % Malacostraca. While, the Honing Kloof site arthropod community was composed of 68 % Insecta, 22 % Entognatha, 4 % Arachnida, 2 % Acari, 2 % Malacostraca, 1 % Chilopoda and 1 % Diplopoda respectively.

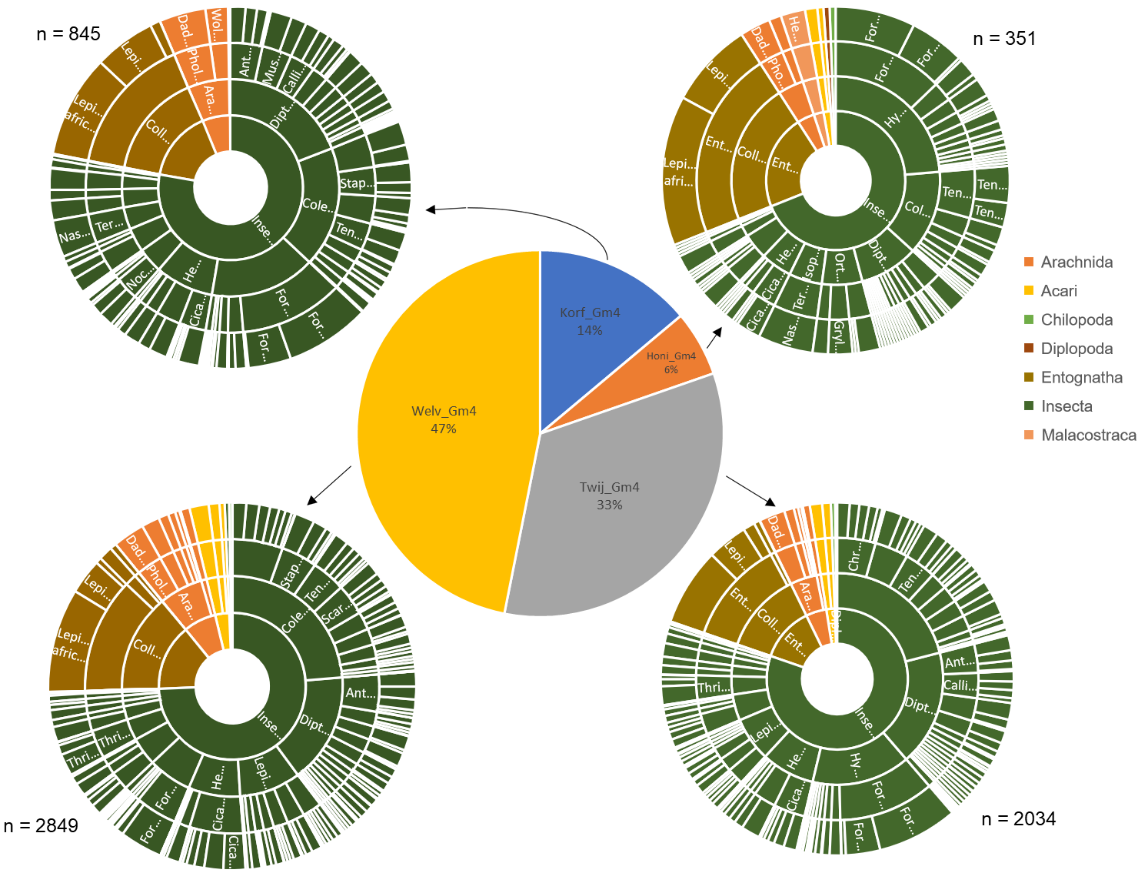


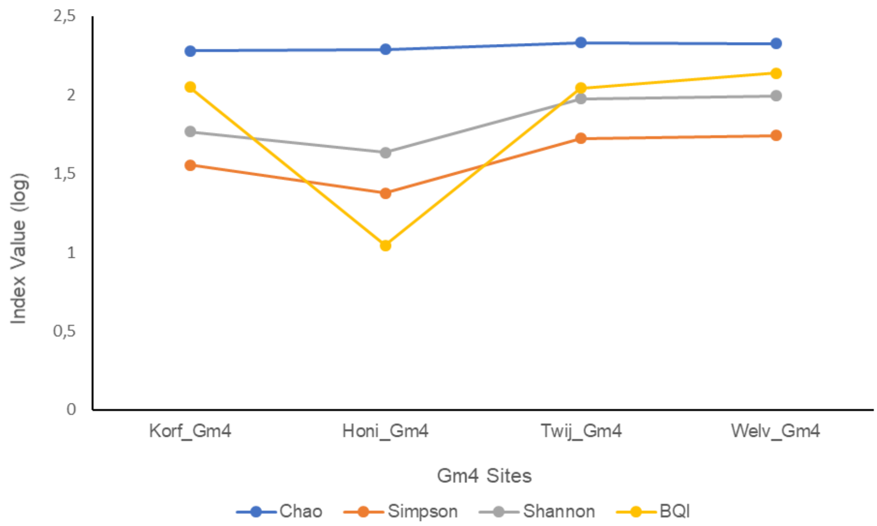
Figure 4.2: Arthropod composition illustrating Class, Order, Family and morphospecies taxon for Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4) sites in Eastern Free State Sandy grassland vegetation type in GGHNPark.

Biotope Quality Index (BQI) and Species surrogates

The Biotope Quality Index showed great variation between sites with the highest *BQI* value of 137.533 at Welverdiend site, a *BQI* value of 111.183 at Korfshoek site, a *BQI* value of 109.812 at Twijhoek site, and the least *BQI* value of 11.058 for Honing Kloof site (Figure 4.3a). The estimated species richness of the arthropod was relatively higher at Twijhoek site (S_{max} = 213.893), and Welverdiend site (S_{max} = 210.800) sites, and lower at Korfshoek site (S_{max} = 194.321), and the least at Honing Kloof site (S_{max} = 189.500). There was great variation with arthropod diversity between the evaluated sites, with Simpson-Yule and Shannon-Wiener diversity index higher for the Welverdiend site (D = 55.302 and H = 1.995) and Twijhoek site (D = 52.922 and H = 1.976), while lower at Korfshoek site (D = 35.921 and H = 1.766) and Honing Kloof site (D = 23.790 and H = 1.635) respectively.

The Biotope Quality Index and species surrogates indicated significant association with arthropod abundance patterns with a clear distinction between strongly positively correlated at the Welverdiend site, while strongly negatively correlated at Honing Kloof site (Figure 4.3b). The sum of all canonical eigenvalues was 0.522 (of total Inertia) with a mean eigenvalue of 0.173 for CCA1 and CCA2 computation simulation used to explain variation of 33.984% by the provided variables.

(a)



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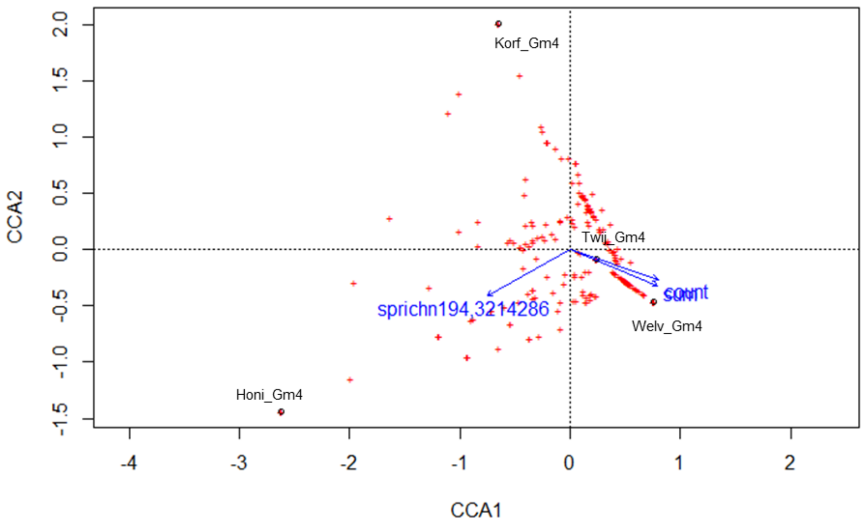

Figure 4.3: Evaluated biotic indices namely estimated species richness, diversity and quality indices, a) determined Chao Index, Shannon-Wiener Function, Simpson-Yule Index and BIotope Quality Index response, b) relation between arthropod abundances and biotic indices (canonical correspondence analysis) for selected sites in Eastern Free State Sandy grassland vegetation type in GGHNPark.

Biotope Quality Index and ecological disturbances

Soil erosion and Biotope Quality Index were negatively correlated with a correlation coefficient value (R^2 = 0.602) (Figure 4.4a), while association between plant encroachment and Biotope Quality Index had a positive correlation with a correlation coefficient value (R^2 = 0.374) (Figure 4.4b), for the Eastern Free State Sandy grassland bioregion. The observed plant encroachers were woody shrubs of the Asteraceae family, namely *Seriphium plumosum* with higher percentage of plant coverage and present in all sampled sites than *Tagetes minuta* with scattered distribution.

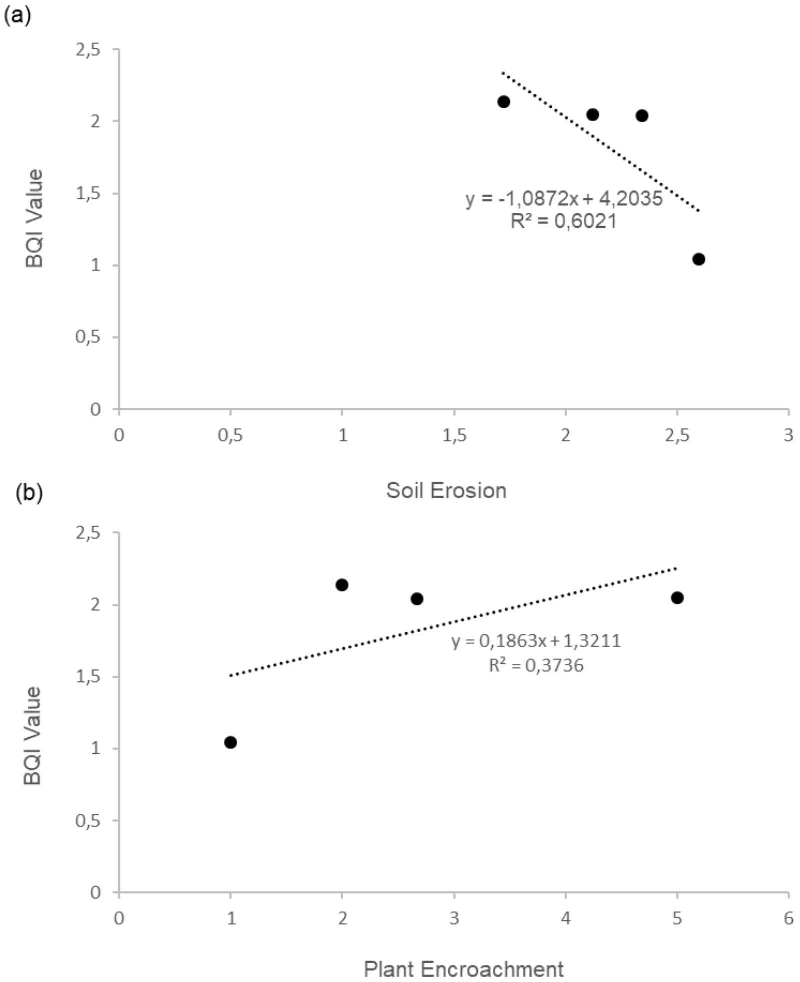

Figure 4.4: Relationship between Biotope Quality Index to environmental disturbances, a) with top soil erosion, and, b) plant encroachment for selected sites in Eastern Free State Sandy grassland vegetation type in GGHNPark.

Environmental disturbance impact on arthropods

The environmental disturbance factors that had the greatest influence on the composition of arthropod community using PERMANOVA included historical agricultural fields (farm) with correlation coefficient (R^2 = 0.581, $p < 0.05$), development of buildings (R^2 = 0.557, $p < 0.05$), and presence of pollution (R^2 = 0.581, $p < 0.05$).

The clustering of arthropod communities using NMDS (with stress value of 0.160) indicated that there was a range of common shared morphospecies between historical agricultural fields (farm and non-farm) sites (Figure 4.5a). The Venn diagram illustrates that there were 150 arthropod families shared between all the sampled sites, with 66 families unique to the sites without an agricultural practices record, and only 11 families unique to the sites that were used as farm fields (Figure 4.5b). The non-farm sites (Welverdiend and Twijhoek) had the most arthropod families of 216 and previous farm sites had least with 161 (Korfshoek and Honing Kloof). Furthermore, Bray-Curtis dissimilarity dendrogram indicated that arthropod community composition and environmental factors for site Welverdiend and Twijhoek were more closely related, than those for Korfshoek and Honing Kloof respectively (Figure 4.5c).

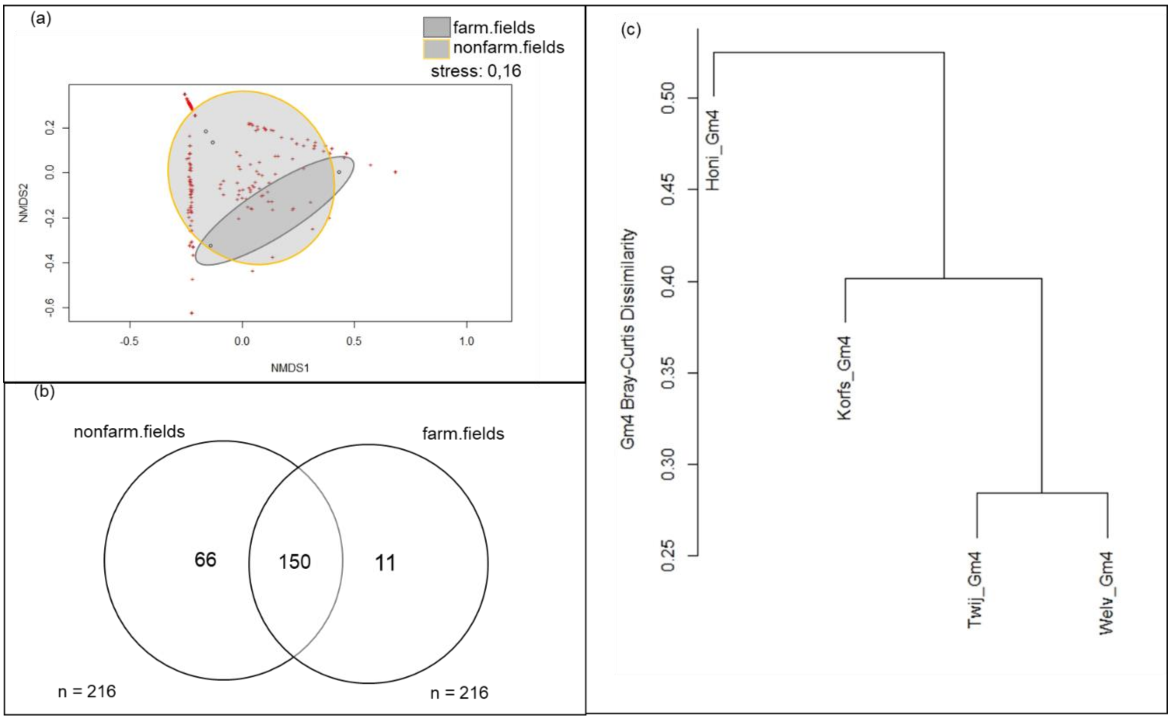

Figure 4.5: Site relation with common shared arthropod families and morphospecies, a) between previously farm and non-farm fields (non-metric multidimensional scaling), b) shared and unique arthropod families for previously farm and non-farm fields (venn diagram), and, c) cladogram indicating association using dissimilarity chart (Bray-Curtis dissimilarity).for Korfshoek (Korf_Gm4), Honing Kloof (Honi_Gm4), Twijhoek (Twij_Gm4) and Welverdiend (Welv_Gm4) in Eastern Free State Sandy grassland vegetation type in GGHNPark.

Biotope Quality Index as geospatial modelling vector

The quantitative review of Rademeyer and van Zyl (2014) indicated that Golden Gate Highlands National Park (GGHNPark) was declared a national park in 1963 covering only 1792 ha of north-western region (Figure 4.6a), over time incorporated other neighbouring farms to 11630 ha, and the rest surrounding areas used privately as farm fields. Additionally, most of the adjacent farms to GGHNPark were active until Qwaqwa National Park was established in 1991 covering the north-eastern regions which covered 21128 ha, and all agricultural activities ceased. However, GGHNPark and Qwaqwa National Park merged later in 2008 to cover a total surface area of 32758.32 ha for the current area demarcation (Figure 4.6b). the Biotope Quality Index estimates (*BQIest*) for sites that resemble vegetation and landscape similarities were divided into five categories of environmental status range (Figure 4.6b), to whom sites with lower *BQIest* value ranging between 24.234 – 32.000 characterised by a higher degree of disturbance and poor biotope quality conditions included the Honing Kloof site and related areas, a *BQIest* value ranging between 32.100 – 100.465 for the Korfshoek site and related areas attributed to higher degree of disturbance and poor biotope quality conditions, while *BQIest* value were higher between 109.902 – 124.178 attributed to less degree of disturbance and moderate biotope quality conditions at the Twijhoek site and related areas, as well as for the Welverdiend site and related areas with 124.178 – 160.224 *BQIest* value range attributed to less degree of disturbance and good biotope quality conditions respectively.

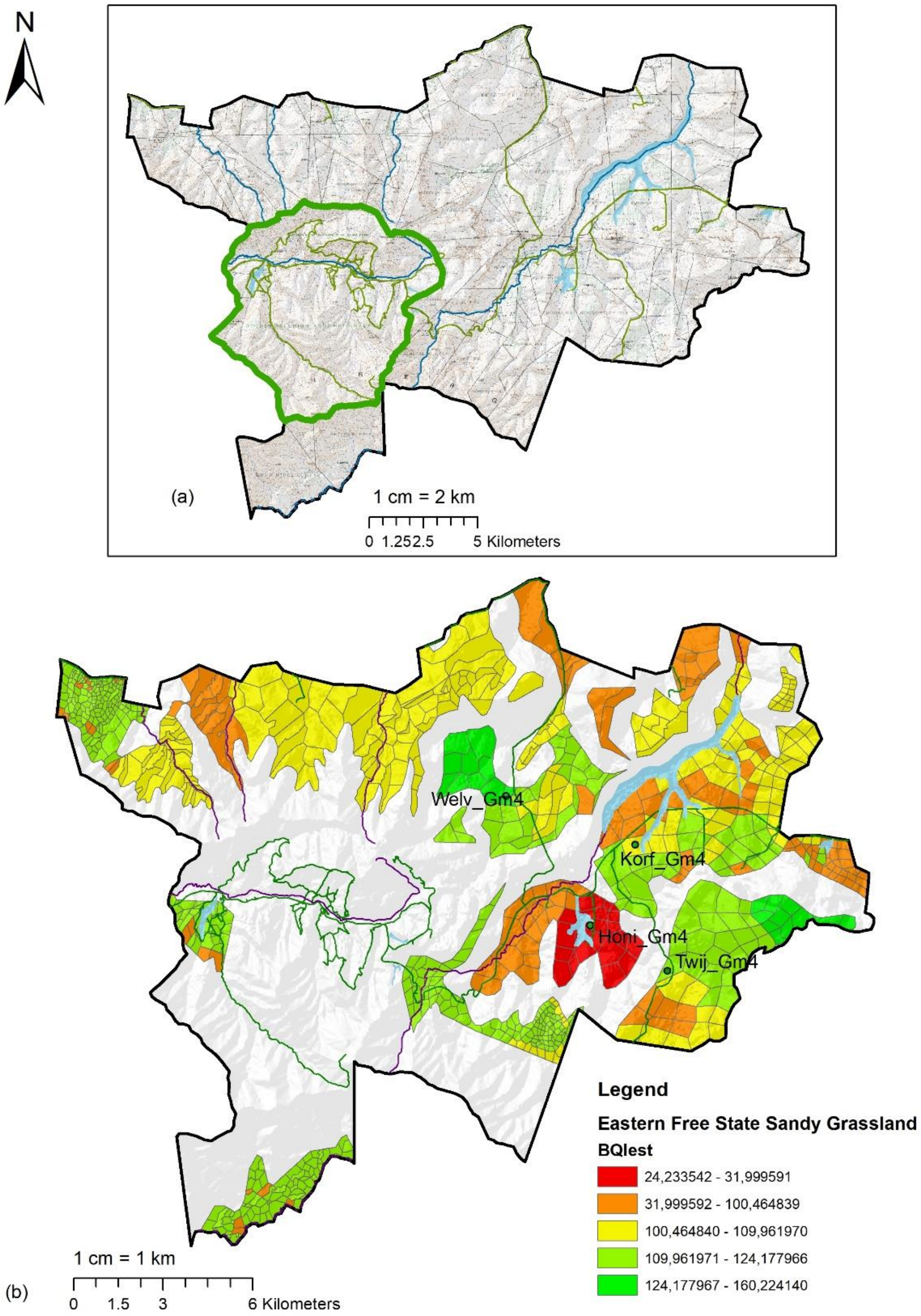

Figure 4.6: Comparison using Geospatial Biotope Quality Index, a) previous GGHNPark 1940 demarcation, and b) current GGHNPark demarcation Biotope Quality Index evaluation highlighted for sites in Eastern Free State Sandy (Gm4) grassland vegetation type in GGHNPark.

Discussion

Mesic Highveld Grassland bioregion sites that had the most arthropod morphospecies, species richness and diversity were those with better Biotope Quality Index value and less disturbances in this study. The decline in arthropod abundance in grassland along a distance gradient was hypothesised to be tied to positive relationship between graminoid and arthropod herbivory diversity (Haddad et al., 2019). In that, most arthropod herbivores display some degree of specialisation and a more plant diverse community with less environmental disturbances should provide a wide range of resources in turn influencing greater abundance of herbivores (Novotny et al. 2002). Hence, the Welverdiend and Twijhoek sites exhibited less degree of environmental disturbance and had relatively more arthropod abundance than those of the Korfshoek and Honing Kloof sites with a higher degree of disturbance.

The arthropod assemblages had three common dominating taxa, namely Entognatha, Insecta and Arachnida, although abundance numbers varied greatly between sampled sites. Another study in montane grassland from central Argentina concurred with a similar pattern arthropod composition (Cagnolo et al., 2002), that collembola and insects constitute more representation within arthropod communities. The common and dominant arthropod families represented were Entomobryidae and Formicidae at all sampled sites of the Eastern Free State Sandy grassland bioregion. Arthropod families of Entomobryidae and Formicidae have been reported with larger abundance number in grasslands (New, 2019), while another studies used them as bioindicators of ecological changes, considering Entomobryidae for intense land use (Chauvat et al., 2007), and Formicidae for agricultural practices (Simons and Weisser, 2017). The representation of family composition was similar across all sites but to varying degrees. Sites that had higher levels of disturbance had a large representation of morphospecies that occur in very low frequencies as singleton and doubleton abundance counts. Similarly, arthropod communities that were collected from non-native vs native grasslands subjected to higher degree of herbivory disturbance indicated a larger pool of morphospecies as singleton and doubleton in other studies (Farrell et al., 2015).

Background information of GGHNPark on the historical activities indicated that most of the sites including Korfshoek and Honing Kloof in the Eastern Free State Sandy grassland vegetation type were used for agricultural practices (Rademeyer and van Zyl, 2014). Conventional farming involves clearing and transformation of native grasslands into monoculture of selected cultivated crops, application of chemicals for pest control and frequent top soil tillage (McIntyre and Martin, 2002). Consequently, conventional farming leads to reduced topsoil depth, degraded soil structure and soil compaction, losses of soil organic matter (SOM), and nutrient depletion, that ultimately contribute to soil erosion (Morgan, 2005; Seitz et al., 2018). The combination effects of intense land use through agricultural practices and consequential soil erosion was reported to reduce arthropod abundance and diversity in natural temperate grasslands (Pfestorf et al., 2013; Wang et al., 2020). The Korfshoek and Honing Kloof sites had relatively lower abundance and diversity, while the Welverdiend and Twijhoek sites with no history of farming had comparatively higher abundance and were more diverse. The presence of graminoid diversity in semi-natural grasslands created an opportunity to promote establishment of ground and soil dwelling arthropods (Sabais et al., 2011; Zhao et al., 2020).

Woody shrubs of the Asteraceae family namely *Seriphium plumosum* and *Tagetes minuta* were recorded to be encroaching the Mesic Highveld Grassland sites at a significant rate. There are a few theories into causes of woody plant encroachment in grasslands, that include overgrazing, extended drought period, low soil nutrient and exclusion of frequent fire (Ward, 2005). In addition to this uncertainty, the common factor related to invasion of woody shrub encroachers is on abandoned cultivated farm lands that are easier to be encroached by shrubs (Avenant, 2015). Woody shrubs tend to take advantage of abundant disturbed land by spreading into it, and outcompeting local grass and forbs for resources and space as land tries to restore. However, the Biotope Quality Index was able to detect effects of both soil erosion and woody shrub encroachment in sampled sites, in that the Honing Kloof site had very low arthropod frequencies and degree of morphospecies abundances variance was little in conjunction with higher percentage of soil erosion. Additionally, the Korfshoek site also had significantly higher disturbance levels of soil erosion and percentage degree woody shrub encroachment, that correlated with lower arthropod frequencies and degree of variance in morphospecies abundances. Therefore, the ecological status of the Honing Kloof and Korfshoek sites were classified to be poor biotope quality conditions due to very low Biotope Quality Index values. The Biotope or ecosystem quality represents the capacity of Mesic Highveld Grassland to perform its ecological functions and be able to maintain and sustain the local population even at their optimal density levels (Bredenhand, 2014).

Other factors that showed the potential to influence arthropod abundances from PERMANOVA included housing development and signs of pollution in the park. There have been various reports that anthropogenic activities such as human settlement development (e.g. farm houses and tourism attractions) bring along aspects of secondary influence to introduce pollution and land modifications (Birkhofer et al., 2017). Furthermore, there were remnants of the old farm houses near the Honing Kloof and Korfshoek sites, that indicate that there was a negative human added involvement in the current poor ecological conditions. Moreover, the Welverdiend site with no disturbances had relatively higher Biotope Quality Index value and exhibited good biotope conditions. This includes a diverse number of arthropods that were found in greater abundances indicating that the biotope could maintain and sustain optimum population densities that play important roles in fulfilling ecosystem functions.

Conclusion

The Biotope Quality Index was able to help in detecting areas that were affected by soil erosion and woody plant encroachment, as well as highlighting the magnitude of each environmental disturbance in the Eastern Free State Sandy grassland vegetation type of the GGHNPark. Even after 56 years since the park was declared a protected area, land that was used intensively for cultivation seems to recover naturally slowly from negative effects. Hence, for sites with poorly represented Mesic Highveld Grassland biotope integrity and quality, a holistic land restoration programme is recommended to recognise sites that were previously used as cultivation for crops before they were included to be part of GGHNPark. The Biotope Quality Index can be used as a biomonitoring tool to assess progress towards the restoration of heavily impacted Eastern Free State Sandy grassland sites.

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CHAPTER 5 : Plant Invasion and Soil Erosion Impact towards Arthropod Assemblages using Biotope Quality Index for Basotho Montane Shrubland in Golden Gate Highlands National Park, South Africa.

Abstract

The Basotho Montane Shrubland grassland is a biotope situated on steep rocky slopes prone to environmental disturbances such as soil erosion, plant invasion, grazing and fire. Biomonitoring using the Biotope Quality Index is used to assess the magnitude and extent of environmental disturbances in montane shrubland grassland bioregion using arthropod assemblages. The ground dwelling arthropods were collected using active net sweeping and pitfall trapping in the Golden Gate Highlands National Park. The results highlighted the composition of arthropod assemblage represented by taxon which included 77 % Insecta, 15 % Entognatha, 6 % Arachnida, 1 % Diplopoda, 1 % Chilopoda and 1 % Malacostraca. There was great variation in the Biotope Quality Index between sites being high at QQ Mountain (*BQI* = 174.740), Diepkloof (*BQI* = 98.606), while the least at Silasberg (*BQI* = 77.634) and Avondrust (*BQI* = 76.572). The Biotope Quality Index had a negative correlation with soil erosion (R^2 = 0.018), and a positive correlation with plant invasion and (R^2 =0.616). The QQ Mountain site had *Cynodon dactylon* grass invasion but also with a higher arthropod diversity and good biotope conditions. Contrary, the Avondrust site had *Acacia mearnsii* tree invasion but seen to influence a lower arthropod diversity and poor biotope conditions. The Biotope Quality Index could be used as a species surrogate for biomonitoring Basotho Montane Shrubland grassland vegetation type in response to ecological impacts of plant invasion and soil erosion.

Keywords

Acacia mearnsii, Arthropod Community, *Cynodon dactylon*, Bioassessment, Biodiversity, Ecological Disturbance, Montane Shrubland, Species Surrogates

Introduction

Southern Africa has a variety of vegetation units that constitute the Mesic Highveld Grassland bioregion distinguished on the basis of their topography, geological substrate features, elevation and climate conditions (Mucina and Rutherford, 2006). Montane shrubland grasslands are situated along steep rocky slopes that are lined with woody shrubs, herbaceous plants and sporadic trees (Bellingham, 1998). Montane shrubland grasslands are ecotones on mountain or hill slopes along the transition zone between two vegetation units and which acts as a buffer region driven by hostile ecological conditions (Mucina and Rutherford, 2006). Ecotones ecosystems are sensitive and susceptible to negative effects of ecological disturbance (Serrano-Ortiz et al., 2007). The montane shrubland grasslands create a unique, versatile geomorphological and vegetation mosaic that provides resources that are utilised as alternative food source and refugia for local fauna (Koyama et al., 2015). Most of these ecosystems are ecologically distinct and have interesting characteristics, but are poorly studied due to the fact that they are less economically valuable, and hence there is little consideration of conservation in temperate regions.

Montane shrubland strips have the potential to act as ecological buffers against wildfires and soil erosion for highland grassland resilience (Lomolino et al., 1989; Torresani et al., 2019). Fire is an ecological tool that is used to maintain integrity and functioning in grassland biomes (Scheiter et al., 2012; Morris et al., 2020), to limit woody and alien plants from encroaching into montane shrubland areas (de Villiers and O'Connor, 2011; Murphy and Bowman, 2012), as well as to assist in vegetative regeneration of dormant bud resprouting and seed germination (Foulkes et al., 2021). The patchy and rocky structure of the montane shrubland ecozone play an important role in influencing the effects on reduction of intensity and severity of fire by narrowing the barrier of the wildfire corridor between adjacent continuing grassland fields (Conver et al., 2018). Additionally, montane shrubland ecotones have the ability to channel excessive rainwater runoff away from adjacent grass fields (Xu et al., 2019), collect eroded topsoil (Morris and Moses, 1987), limit movement of topsoil (Ochoa-Cueva et al., 2015), in that it reduces intensity and severity impact of soil erosion. Montane shrubland grassland ecotones are understudies, and many ecological aspects are not fully understood that include holistic functioning and the extent of ecological disturbance on its conditions towards local biodiversity.

In South Africa, most montane shrublands are considered to be vulnerable and threatened conservation biotopes. Some studies have indicated that montane grasslands hold some level of localised vegetation endemism (O'Connor and Kuyler, 2009), with herbs *Lessertia tenuifolia* and *Leucaena latisiliqua* at sandstone Basotho Montane Shrubland, the regional sub-Escapement limited distribution range for small tree *Encephalartos frederici-guilielmi*, low shrubs *Eriocephalus africanus* and *Senecio acutifolis* at Tarkastad Montane Shrubland, and with Drakensberg mountain extending to Griqualand east range to accommodate drier low shrubs *Euryops tysonii*, *Relhania acerosa*, *R*. *dieterlenii*, herbs *Diascia integerrima* and *Helichrysum elegantissimum* at

Senqu Montane Shrubland (Mucina and Rutherford, 2006). The evolutionary hostility to ecological perturbation of montane shrubland biotope associations of the endemic biodiversity of plants and arthropods is still to be explored.

Bioassessment and biotic indices could be used to evaluate and monitor state of montane shrublands and impacts posed by the nature of hostility to ecological disturbances towards local biodiversity. The ability of insects and other arthropods to be sensitive to change in environmental conditions has contributed to their potential as biomonitoring tools (Benefer et al., 2016). Arthropods' physiological and behavioural adaptations, local life history and population dynamics enables them to assist in indicating ecological changes for better environmental monitoring and conservation practices (Samways, 2005). Many studies utilise arthropods to monitor ecological integrity (Krause and Culmsee, 2013), functioning (Ebeling et al., 2018), stability (Morris, 2000), resilience (Joern and Laws, 2013) and quality (Bredenhand, 2014). Conservation practices in montane shrubland ecotones can consider evaluating ecosystem quality that reflects ecological integrity and functional status within a bioregion. Biotic indices are used to inform conservation programs, the general public and decision-makers about the state of the ecosystem. Bredenhand (2014) developed a species surrogate (numeric expression) called Biotope Quality Index, which make use of common occurring arthropod assemblage mean proportion and standard deviation degree in a population, to indicate disturbance level impact, and assign a conservation status for a biotope.

The expression of the Biotope Quality Index (below) takes into consideration evolutionary history of the integrity unit and assumes any biotope that is in a naturally good conserved condition, can act to sustain the arthropod population to optimum species numbers as well as their ecological processes.

$$\text{Biotope Quality Index } (BQI) = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right) \text{ if } \log \sigma > 0$$

The *BQI* expression arithmetic elements represented, include abundance of *i*th morphospecies (x_i), *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ). The range of variation of the Biotope Quality Index indicates a strong correlation with degree of disturbance, and temporal effects of seasonal change in the Shale Fynbos bioregion at Jonkershoek Nature Reserve in South Africa (Bredenhand, 2014). Therefore, the Biotope Quality Index was adopted to evaluate effects of environmental disturbance using assemblages of arthropod species that occur in the Basotho Montane Shrubland bioregion in the Golden Gate Highlands National Park.

Materials and Methods

Study site

The study was carried out at the Golden Gate Highlands National Park (GGHNPark) (Figure 5.1), which is located in the north-eastern Maloti-Drakensberg Mountains along the border that connects the Kingdom of Lesotho, eastern Free State and western Kwa-Zulu Natal Province. GGHNPark is located between the geospatial quadrant within latitude of 28°27' S to 28°37' S and longitude of 28°33' E to 28°42' E, covers a surface area of 342 km^2 (32758 ha), with an altitudinal range of 1892 m to 2837 m shown in Figure 5.1(c). Climate conditions are characterised by summer rainfall receiving approximately 800 mm of mean seasonal precipitation, with a mean annual temperature ranging from 13°C to 30°C, and while winter rainfall of approximately 300 mm mean seasonal precipitation and with mean annual temperature ranging from 1°C to 26°C (Grab et al., 2011). The highland ecoregion is within a moist cold that is mainly characterised by graminoids, forbs, herbs, woody trees and shrubs (Mucina and Rutherford, 2006; Telfer et al., 2012).

The biotope units are recognised as vegetation bioregion categories described by Mucina and Rutherfiord (2006) as in Figure 5.1(d) and with a clear ecological boundary in this study. The Basotho Montane Shrubland Grassland bioregion is on a steep rocky Clarens sandstone formation slope, characterised by a mixture of taller shrubs *Buddleja salviifolia*, *Euclea crispa* subsp. *ovata*, *Olea europaea* subsp. *africana*, *eucosidea sericea and Searsia dentata, lower shrubs Gnidia capitata, Euphorbia stricta var. cuspidata, Felicia filifolia subsp. filifolia and Myrsine africana, grass communities of Aristida congesta, Eragrostis racemosa, Hyparrhenia hirta, Cymbopogon pospischil, Tristachya leucothrix and Themeda triandra, and herbs namely Helichrysum nudifolium var. nudifolium, Hermannia depressa, Vernonia oligocephala and Dicoma anomala*. (Eckhardt et al., 2006; Kay et al., 2006). Sampling was carried out at four sites on the Basotho Montane Shrubland Gm5 (elevation range 1480 m to 1940 m). The sites were selected based on consideration of ecological disturbances such as soil erosion, human influenced disturbances (pollution), plant encroachment and invasion. The selected sites were named QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5) respectively.

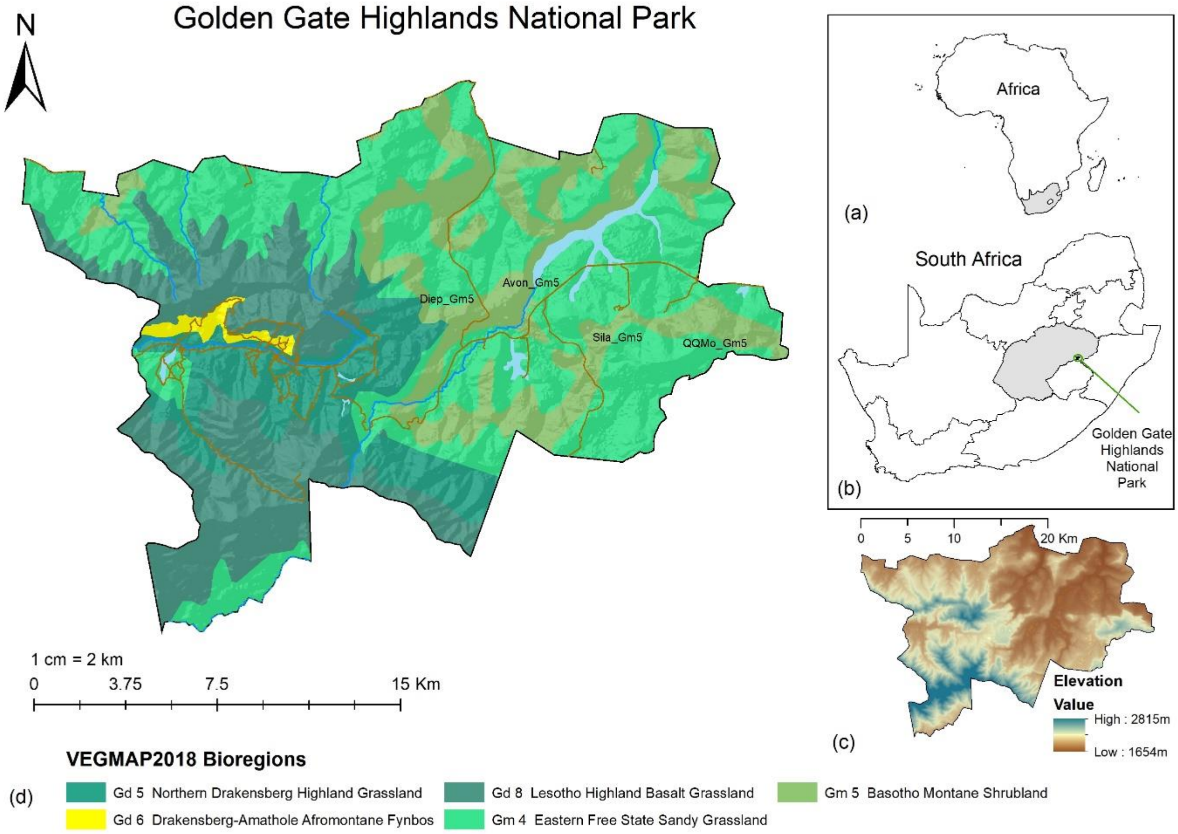

Figure 5.1: The location of study area illustrating (a) the continent of Africa, (b) the eastern section of the Free State Province highlighted in the outline of South Africa, (c) elevation and hillshade of GGHNPark created using DEM image from USGS Explore Earth, and (d) GGHNPark highlighting sampled sites within Gm5 vegetation bioregions from SANBI GIS VEGMAP (2018) using ArcGIS.

Arthropod sampling

Ground-dwelling arthropods were collected using a combination of sampling techniques namely unbaited pitfall trapping, active net sweeping and field observations after every two weeks of month from September 2016 to September 2018. First, pitfall trap sampling was conducted by inserting 10 randomly arranged 250 ml of plastic jars (60 mm diameter X 100 mm height) into the ground surface, in a 200 m transect and 20 m spacing in between. A 50 ml pitfall solution containing ethanol and propylene glycol (3 v:1 v ratio) was used as a killing and preservation agent (Yekwayo et al., 2018). The plastic jar lid was modified to elevate the lid up by 20 mm from the ground surface to restrict rainwater runoff, prevent evaporation of the pitfall preservation solution, as well as restrict large objects and predators (vertebrates and invertebrates) from clogging the trap. Second, net sweeping was performed using a steel hoop of 400 mm diameter and mesh fabric sheet. The arthropods were swept on vegetation along a 200 m transect adjacent to pitfall traps during the day. Lastly, field observations were also included to record aerial insects visible on the vegetation and surroundings in sampling 200 m transect. The catch and release method was used for net sweeping samples after the reference collection was compiled. The collected arthropod samples were sorted and identified to family and genus taxa level using Scholtz and Holm (2008) for insects, Horak et al (2015) for ticks and Dippenaar-Schoeman (2014) for spiders, and then, later assigned morphospecies with reference (pseudonym) to each morphologically distinct specimen (Gerlach et al., 2013). The arthropod reference collection was created and housed at the Qwaqwa campus entomology collection of the University of the Free State. Only the adult life stages of arthropods were considered and data from all sampling methods was pooled for analysis.

Ecological disturbances

Quantitative and qualitative data were collected to measure disturbance at each sampled site to evaluate ecological and environmental disturbance variables. Site topography and elevation, plant cover, soil type, depth and moisture were used to determine soil erosion. Plant encroachment and invasion were measured by recorded presence and percentage cover of plants that are classified as encroachment and alien invasive (supplementary material: Table 0.2). Anthropogenic activities that were considered to contribute to disturbance were also recorded, and included road site constructions, footpaths, land pollution level, current wildlife and human activity zones (supplementary material: Table 0.4), within the 200 m x 100 m arthropod sampling transect range.

Data analysis

Presence data was processed using descriptive statistical analysis to measure mean, standard deviations and relative abundances for each sampled site (Legendre and Legendre, 1998). The arthropod taxa mean were compared between sites with uneven sample sizes using the Kruskal-Wallis H-test ranked means and Dunn post-hoc test for specific site comparison. Sample-based rarefaction was used to determine the rare occurrence probability of species (singleton and doubleton frequencies) for each site (Gotelli and Colwell, 2001). Species surrogate biotic indice methods were included to calculate estimated species richness using Chao2 (Chao, 1984), diversity indices using the Simpson-Yule diversity (Simpson, 1949) and Shannon–Wiener function (Weaver and Shannon, 1949) as well as the Biotope Quality Index (Bredenhand, 2014) to indicate environmental status of each sample site. The association by effect for species surrogates (biotic indices) as a gradient to influence arthropod distributions was evaluated using the Canonical Correspondence Analysis (CCA) R-Studio (using log transformed abundance data) with the vegan package (Gardener, 2012) .

Degree of soil erosion was determined using the modified Revised Universal Soil Loss Equation (RUSLE) and quantitative erosion estimates were grouped into erosion risk classes (Bartsch et al., 2002). The quantitative methods were used to determine and grade the degree of disturbance for pollution ratio, building aggregation, wildlife density and human activity (supplementary material: Table 0.3) per site. The effect of environmental and ecological disturbance onto arthropod community dynamics was determined using permutational multivariate analysis of variance (PERMANOVA) in R-Studio (ADONIS, simulations ran 23 permutations to congruency, in vegan package) and only disturbances with significant influence were considered for further analysis (Gardener, 2012). Additionally, significant quantitative disturbance factors were placed for Nonmetric multidimensional scaling ordination (NMDS), transformed arthropod abundance using hellinger, with a Bray-Curtis as an ordination distance method (vegan R-studio) to distinguish clustering influence between sampled sites (Gardener, 2012). Then, sites were related based on arthropod and environmental data using both quantitative and weight qualitative data using Bray-Curtis dissimilarity (Bray and Curtis, 1957).

The potential of the Biotope Quality Index as a geospatial modelling vector to indicate current ecological conditions of the Basotho Montane Shrubland (Gm5) grassland bioregion in the GGHNPark, was measured by determining estimated Biotope Quality Index values with a modified bootstrap jackknife 2 using 999 permutations (Manly, 1991), to predict the asymptotic Biotope Quality Index values with varying population sizes from known calculated mean and standard deviation for neighbouring and adjacent related sites. The Gm5 bioregion sites were divided into sampling zone polygon units per locality, and then, the predicted asymptotic Biotope Quality Index values were assigned to each sampling unit polygon using ArcGIS Map (version 10.5) to highlight areas of environmental concern. The shapefiles for continental and country maps were accessed from the Esri GIS database, while the provincial, South African protected areas (national parks), roads and surface water body shapefiles were accessed from EGIS database of the Department of Forestry, Fisheries and the Environment (DFFE), and the vegetation bioregion VEGMAP 2018 (for South Africa, Lesotho and Swaziland) shapefiles were accessed from SANBI Biodiversity GIS database.

Results

A total of 10 520 adult arthropod individuals were represented in six classes, 24 orders, 107 families and 220 morphospecies for analysis. The overall arthropod composition included main class taxa with 70 % Insecta, 17% Entognatha, 6% Arachnida, 3 % Diplopoda, 3 % Chilopoda and 1 % Malacostraca for the Basotho Montane Shrubland grassland vegetation type. The most dominant arthropod families in all sampled sites were Entomobryidae, Formicidae, Phalangiidae, Thripidae, Cicadellidae, Aphididae, Tetranychidae, Tenebrionidae, Chrysomelidae and Anthomyiidae. The QQ Mountain and Diepkloof sites had most abundant arthropods with the highest taxa groups represented, while the Silasberg and Avondrust sites had the least arthropod abundance with few taxa represented and more morphospecies occurring in very low frequencies (singleton and doubleton) with a high rarefaction value per site (Table 5.1). A significant difference was observed between the ranked abundance for sites for uneven sample size with a Kruskal-Wallis chi-squared of 45.602 (df = 3, $p < 0,05$). The Dunn post-hoc test further indicated that there was significant difference mainly between QQ Mountain and Avondrust sites, Diepkloof and Avondrust sites, as well as Silasberg and Avondrust sites, while there was no significant difference between Silasberg and QQ Mountain.

Table 5.1: Summary of arthropod taxa class, order, family, morphospecies counts and rarefaction at the QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5) sites for the Basotho Montane Shrubland grassland vegetation type in GGHNPark.

		Taxon counts			Morphospecies		
	Class	Order	Family	T.count	singleton	doubleton	rarefaction
QQMo_Gm5	6	22	102	182	25	12	0.204
Avon_Gm5	6	7	23	95	41	15	0.596
Diep_Gm5	6	6	21	78	30	17	0.610
Sila_Gm5	6	7	19	82	23	13	0.444
Kruskal-Wallis chi-squared = 45.602, df = 3, p-value <0,05							
Comparison of the means of Gm5 sites by group (Benjamini-Hochberg)							
	Avon_Gm5		Diep_Gm5		QQMo_Gm5		
Diep_Gm5	2.473	0.0080*					
QQMo_Gm5	6.665	0.0000***	4.139	0.0001**			
Sila_Gm5	2.809	0.0037*	0.389	0.349	3.649	0.0003*	

*ref. evaluated at $p < 0.05$ significant difference is indicated by an asterisk *, highly significant ** and ***. Abbreviations, total counts (T.count), QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5).

Arthropod assemblages and composition

All sampled sites had a common shared arthropod class dominated by Insecta, Entognatha and Arachnida with few Chilopoda, Diplopoda and Malacostraca represented (Figure 5.2). The arthropod community at the QQ Mountain site was composed of 77% Insecta, 15% Entognatha, 6% Arachnida, 1% Diplopoda, 1% Chilopoda and 1% Malacostraca. The community of arthropods at the Diepkloof site was composed of 71% Insecta, 21% Entognatha, 7% Arachnida, 1% Chilopoda, 1% Diplopoda and 1% Malacostraca. The arthropod community at the Silasberg site was composed of 75% Insecta, 18% Entognatha, 6% Arachnida, 1% Chilopoda, 1% Diplopoda and 1% Malacostraca. However, Avondrust site arthropod community was composed of 70% Insecta, 21% Entognatha, 8% Arachnida, 2% Malacostraca, 1% Chilopoda and 1% Diplopoda respectively.

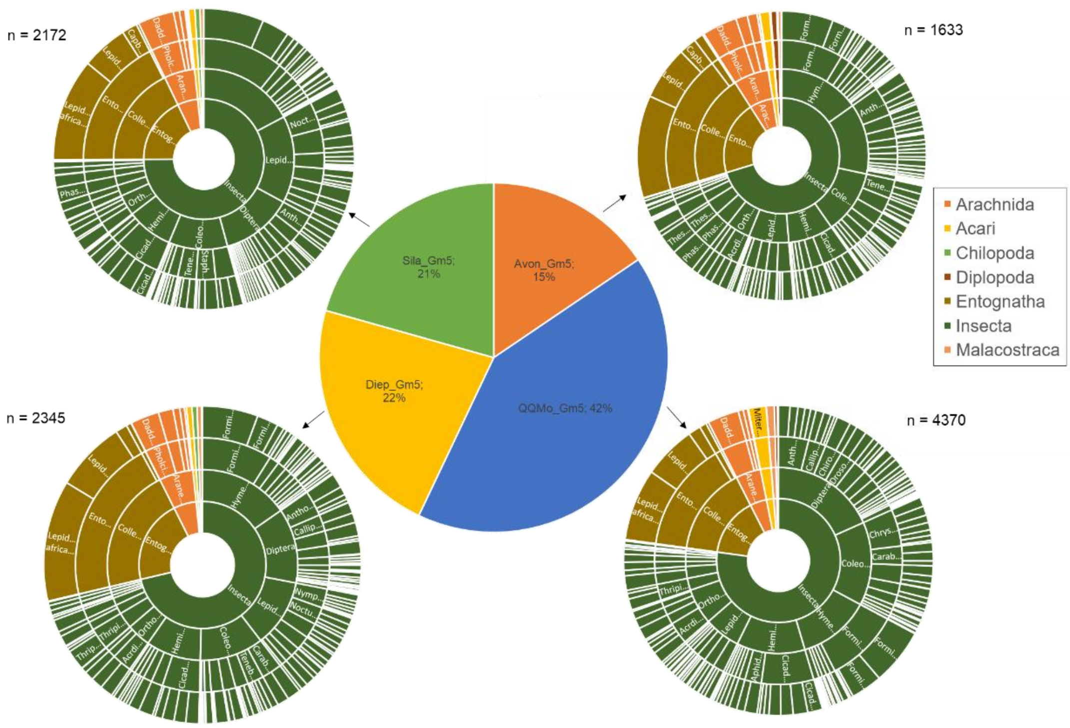

Figure 5.2: Histograms illustrate and compare arthropod community composition with class, order, family and morphospecies taxon between the QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5) for the Basotho Montane Shrubland grassland vegetation type in GGHNPark.

Biotope Quality Index (BQI) and species surrogates

The biotic species surrogates had different response patterns for sampled sites for the Biotope Quality Index, Simpson-Yule diversity, Shannon-Wiener function and estimated species richness for the Basotho Montane Shrubland vegetation type. The Biotope Quality Index between sites indicated variation with the highest *BQI* value of 174.739 at the QQ Mountain site, the Diepkloof site with a *BQI* value of 98.606, and while the least for the Silasberg site with a *BQI* value of 77.634 as well as the Avondrust site with a *BQI* value of 76.572 (Figure 5.3a). There was also variation in arthropod diversity between evaluated sites, with Simpson-Yule diversity and Shannon-Wiener function higher at the QQ Mountain site (D = 50.133 and H = 1.958) and Avondrust (D = 36.351 and H = 1.889), but lower at the Silasberg site (D = 37.149 and H = 1.868) and the Diepkloof site (D = 33.458 and H = 1.878) respectively. On the contrary, estimated species richness between sites was relatively higher at the Avondrust site (S_{max} = 224.033), and the QQ Mountain site (S_{max} = 207.042), and while lower at the Diepkloof site (S_{max} = 195.471) and least at the Silasberg site (S_{max} = 173.346).

There was a mixed relationship between species surrogates with arthropod abundances patterns with a clear distinction between strongly positively correlation for the Diepkloof site and strongly negatively correlation for the QQ Mountain site (Figure 5.3b). The sum of all canonical eigenvalues was 0.359 (total Inertia) with eigenvalue between of 0.065 (CCA3) and 0.162 (CCA1) which can account to explaining 22.33% of influence by species surrogates to the observed arthropod abundances variation among sampled sites.

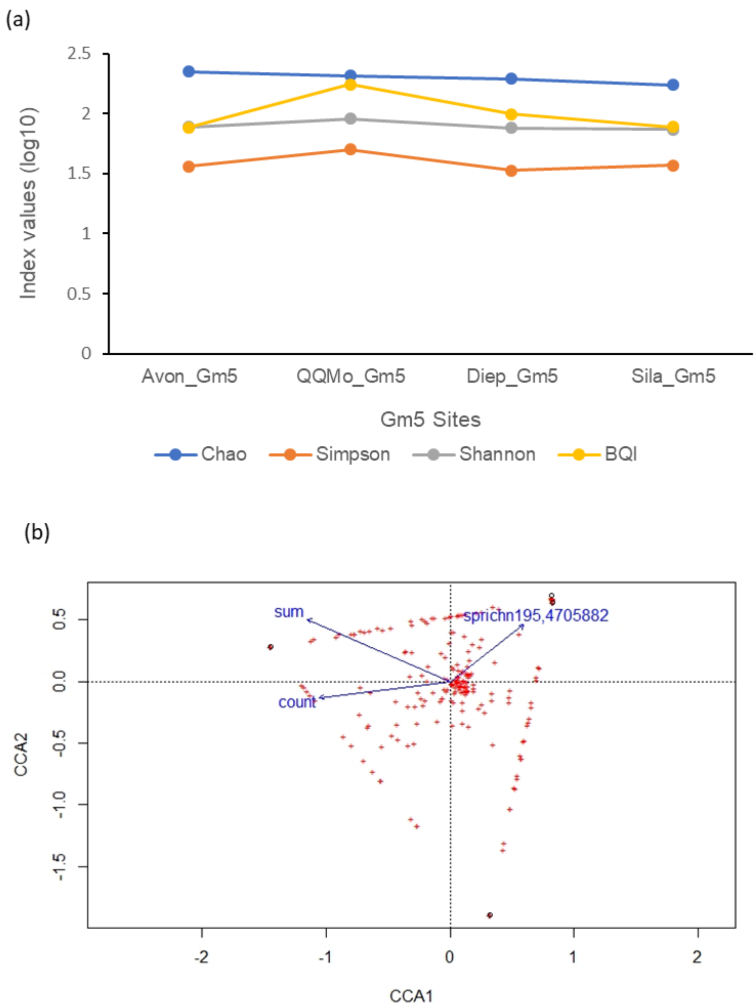

Figure 5.3: Evaluated biotic indices namely estimated species richness, diversity and quality indices, a) determined Chao Index, Shannon-Wiener function, Simpson-Yule diversity and Biotope Quality Index influence, b) relation between arthropod abundances response and biotic indices effect (canonical correspondence analysis) for selected sites for the Basotho Montane Shrubland grassland vegetation type in GGHNPark.

Biotope Quality Index and ecological disturbances

The association between alien plant invasion and the Biotope Quality Index was positively correlated with a correlation coefficient value (R^2 = 0.616) (Figure 5.4b), and again positive weak correlation between soil erosion and the Biotope Quality Index had a correlation coefficient value (R^2 = 0.018) (Figure 5.4a). Site Diepkloof and Silasberg had active and shallow footpaths with moderate rill and sheet erosion type, while Avondrust had a combination of water and rill-sheet erosion types. categorised Significant invasive alien plant recorded for selected sites of the Basotho Montane Shrubland grassland vegetation type included grass species of *Cynodon dactylon* (Family, Poaceae) at the QQ Mountain site, and tree species of *Acacia mearnsii* (Family, Fabaceae) at the Avondrust site.

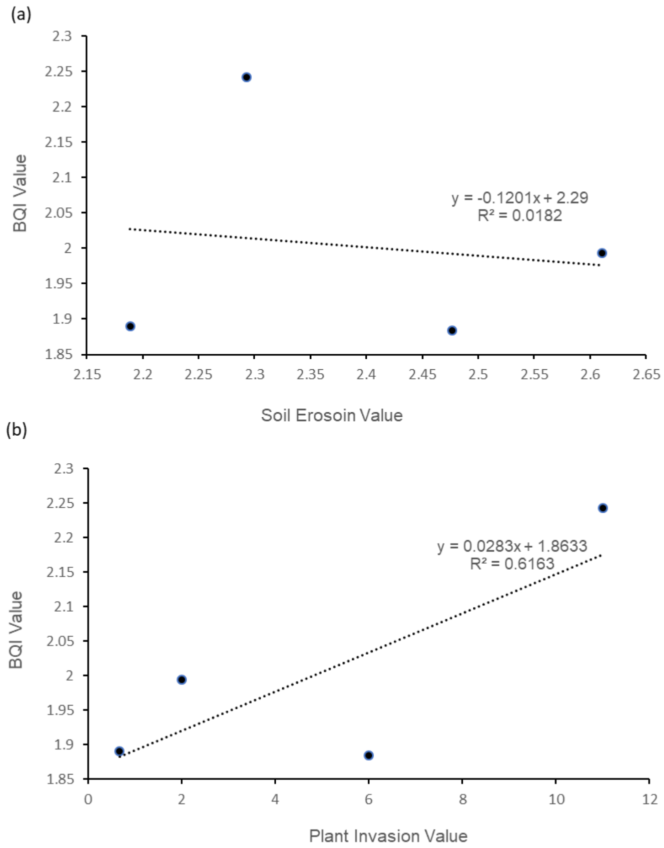

Figure 5.4: Relationship between Biotope Quality Index to environmental disturbances, a) with top soil erosion, and, b) alien plant invasion for selected sites in Basotho Montane Shrubland grassland vegetation type in GGHNPark.

Environmental disturbance impact on arthropods

Wildlife activity had the greatest influence on the composition of the arthropod community using PERMANOVA of all environmental disturbance factors tested with higher correlation coefficient (R^2 = 0.738) but without significant difference. The Venn diagram illustrates that there were 132 families of arthropod shared between sites with and without alien plant invasion, 58 families were unique to sites without invasion and only 83 families were unique to sites with invasion (Figure 5.5a). Alien plant invasion had a low correlation coefficient (R^2 = 0.279) with no significant difference as well. The clustering of the arthropod communities using NMDS had a stress value of 0.19, indicating that there was a larger range of commonly shared morphospecies between sites with invasion by alien plants (Figure 5.5b). The Diepkloof and Silasberg sites without plant invasion had most arthropod families of 215 observed, and while sites with invasion of alien plants had the least families of 190 at the QQ Mountain and Avondrust sites. The composition of the arthropod community and association of the environmental factors are illustrated by the Bray-Curtis dissimilarity dendrogram, which indicated that the Diepkloof and Silasberg sites were more closely related, than those for the QQ Mountain and Avondrust sites (Figure 5.5c).

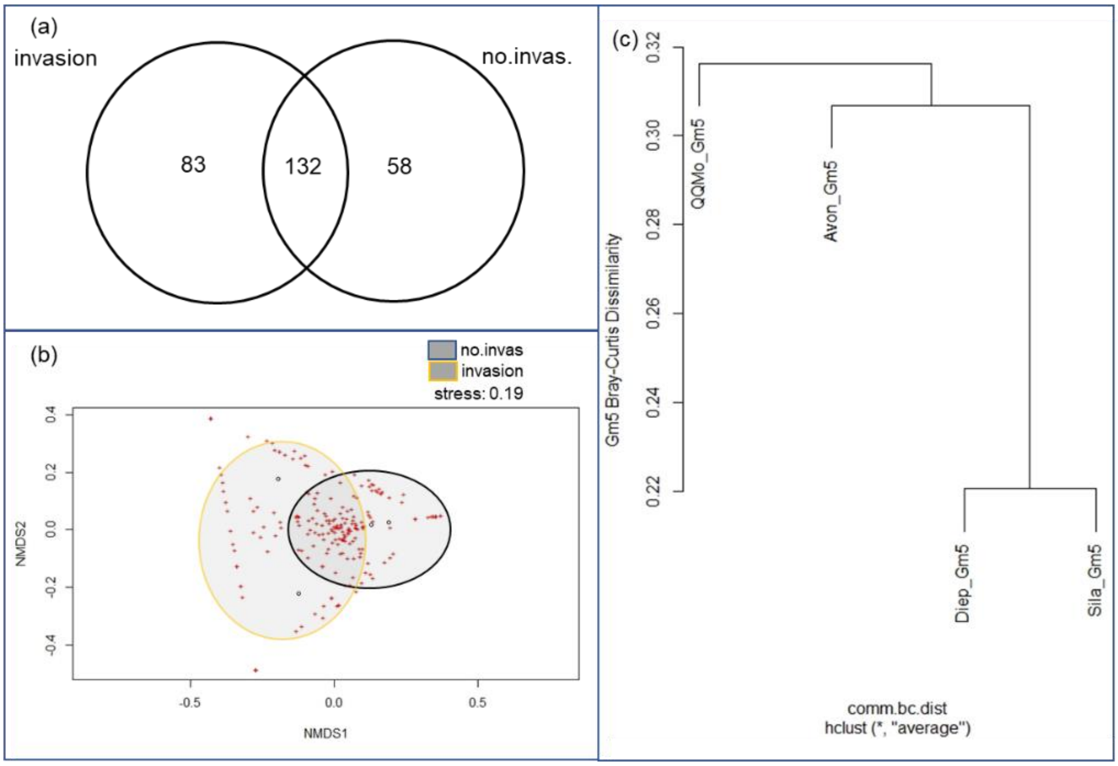

Figure 5.5:Site relation with common shared arthropod families and morphospecies, a) between previously farm and non-farm fields (non-metric multidimensional scaling), b) shared and unique arthropod families for previously farm and non-farm fields (Venn diagram), and, c) cladogram indicating association using dissimilarity chart (Bray-Curtis dissimilarity) at the QQ Mountain (QQMo_Gm5), Avondrust (Avon_Gm5), Diepkloof (Diep_Gm5) and Silasberg (Sila_Gm5) for the Basotho Montane Shrubland grassland vegetation type in GGHNPark.

Biotope Quality Index as geospatial modelling vector

The Biotope Quality Index estimates (*BQI*est.) were based on shared attribute similarities between sites that included the vegetation, landscape similarities and environmental factors, which were then divided into five range environmental condition categories (Figure 5.6b). The QQ Mountain site had the highest *BQI* value of 174.739 (moderate disturbance, but with conserved conditions), the Diepkloof site had a *BQI* value of 98.606 (less disturbance, moderate conditions), while the Silasberg site had the lowest *BQI* value of 77.634 (less disturbance, poor conditions) and for the Avondrust site with a *BQI* value of 76.572 (higher disturbance, poor conditions). The observations of the additional sites to the GIS model indicated a great variation of the Basotho Montane Shrubland relief profiles with steeper to moderate slopes, asymmetrical valleys, with a combination of convex and concave landforms.

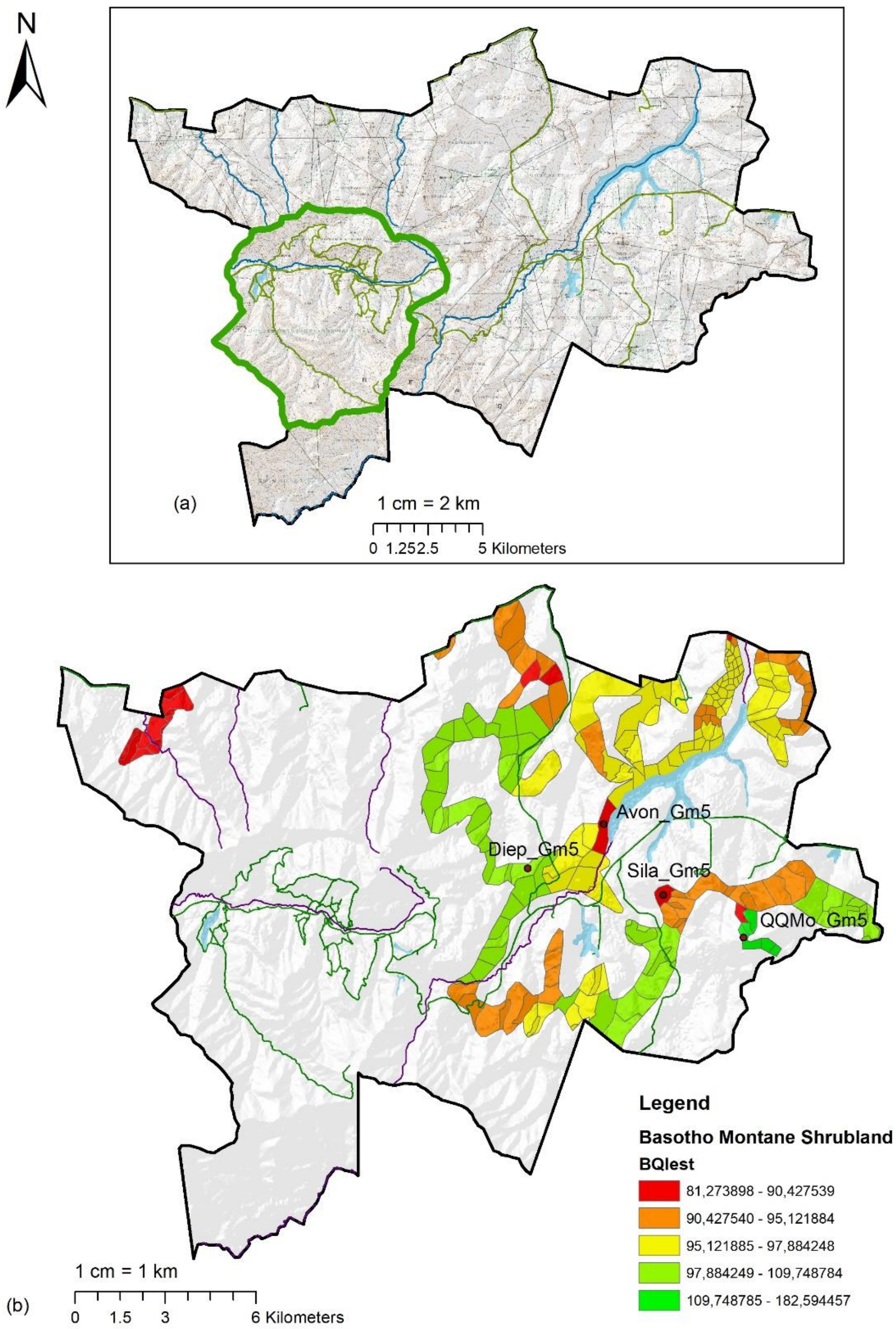

Figure 5.6: Comparison of ecological conditions using Geospatial Biotope Quality Index (*BQI*est.), for a) previous GGHNPark 1940 area border demarcation, and b) the current GGHNPark area border demarcation using Biotope Quality Index to highlight sites for the Basotho Montane Shrubland (Gm5) grassland in GGHNPark.

Discussion

Arthropod morphospecies abundance, species richness and diversity representation had variation in response to type of disturbance between sites. Other studies indicated that composition and diversity of ground dwelling macro-arthropod taxa are mainly represented by Insecta, Collembola and Araneae (Fanga and Winchester, 1999) in both temperate woody shrubland (Sanchez and Parmenter, 2002) and open grassland biotopes (van Schalkwyk et al., 2021). Majority of insect taxa represented were herbivores that included order Homoptera (families Cicadellidae, Aphididae and Lygaeidae), Orthoptera (family Acrididae), Thysanoptera (family Thripidae) and Coleoptera (family Chrysomelidae). The shrubland patches in grassland biomes (or woody grasslands) could provided a variety of food sources for plant phloem and xylem sucking, leaf chewers and growing bud feeders (Hudewenz et al., 2012; van Schalkwyk et al., 2021). Other prominent insect groups in this study included pollinators such as order Lepidoptera (Nymphalidae and Pieridae), Coleoptera (Scarabaeidae and Chrysomelidae), and Diptera (Bombyliidae). Plant variability at the Basotho Montane Shrubland vegetation type included herbaceous and shrubby Asteraceae family species of *Helichrysum* spp., *Felicia* spp., and *Senecio* spp., which offer bright yellow, red, pink and purple compound flowering that attracts localised pollinating insects (Mucina et al., 2006; New, 2009). The Basotho Montane Shrubland showed great differences between biotic species surrogates whereby the species richness was higher at the Avondrust site with high degree of disturbance. The Chao’s Index takes into account species that occur in very low abundances such as singleton and doubleton, as well as the even distribution of species cross sites (Leitao at al., 2016). The montane temperate shrubland ecotones might have local factors contributing to arthropods adapting to occurring in very low abundances in disturbed biotopes (Harris et al., 2004), in that most have resorted to lower abundances due to the nature of hostile rocky steep slope relief environments.

Changes in ecosystems have different effects towards arthropod structure and population dynamics, species richness and diversity, in response to both spartial and temporal ecological variables (Heino et al., 2008). The Biotope Quality Index was able to show better arthropod response from both plant invasion and soil erosion at the sampled sites. In this study, plant invasion had opposing effects on arthropod abundance, species richness, diversity and environmental status, in that the type of invasive plant affected the ecological response differently at the Basotho Montane Shrubland vegetation type. The QQ Mountain site was recorded with a significant invasion of *Cynodon dactylon* (family Poaceae) grass and lower erosion levels but with a higher arthropod abundance frequency, diversity and a good environmental (biotope) condition. The *C. dactylon* grass is listed as an invasion grass that could have originate in Africa (Wu, 2011), which is reported to compete with native wild grass and other graminoids for space thus alter composition and structure of vegetation community in most grassland biomes (Fonseca et al., 2013; Guido et al., 2016). In the QQ Mountain site, although *C. dactylon* grass covered a significant ground surface area, hostile shrubland ecotone ecological factors seemed to limit its ability to outcompete and change floristic structure. Other studies have shown positive correlation responses between certain types of plant invasion and arthropod species surrogates (Sax, 2002; Pearson, 2009), as well as between plant and arthropod diversity (Hertzog et al., 2016). Floristic composition and diversity play an important role as primary producers within a biotope, and both aspects influence the ecosystem functioning, resilience and stability in montane shrubland ecotones (Haddad et al., 2001; Perner et al., 2005). The synergistic influences from ecological and biotic factors could have contributed to the QQ Mountain site with higher resilience, variation of vegetation layers and hence promoted arthropod diversity.

However, certain non-native invasive plants can negatively affect floristic composition, structure and functioning of the local ecosystem (biotope) (Ehrenfeld, 2003; Mack and D'Antonio, 2003). The *Acacia mearnsii* (family Fabaceae) tree invasion at the Avondrust site was also associated with a higher degree of soil erosion and resulted in the frequency and diversity of the low arthropod abundance as well as poor biotope conditions. The *A. mearnsii* trees were introduced for ornamental and commercial purposes from Australia that then became an aggressive invasive alien plant in South Africa (van Wilgen et al., 2011; Chan et al., 2015). Majority of *Acacia* species are known to use more water than native grass, shrubs and trees flora (de Neergaard et al., 2005), and altering soil structure and nutrient flow (Moyo and Fatunbi, 2010), as well as the potential to reduce the ability of a biotope to provide necessary resources to local fauna. The invasive ability of *A. mearnsii* trees also includes the release of allelochemicals to alter soil chemistry and suppress growth of native plant species (van Kleunen et al., 2010), thus often creating an open ground soil surface, increasing the risk of soil erosion (D'Antonio, 1992), reducing plant cover and arthropod diversity.

Although each of the sampled sites presented a unique response of arthropod composition to environmental disturbance, those that were more prone to soil erosion and plant invasion shared common nested community assemblages. The PERMANOVA indicated that plant invasion could better explain arthropod abundances and composition pattern variations in relation to other types of environmental disturbances tested. Plant invasion has more pronounced negative effects on grassland biomes, in that it could alter localised fauna and flora compositions (Sharafatmandrad and Mashizi, 2021). The dissimilarity dendrogram illustrated that there were shared environmental conditions and arthropod taxa between the Diepkloof and Silasberg sites as well as the Avondrust and QQ Mountain sites. Naturally, montane shrubland ecotones are prone to some level of soil erosion due to relief rocky steep slopes (Kwok and Eldridge, 2016), that then allow both plants and arthropods to develop life and behaviour adaptation to reproduce in lower abundance averages, due to the limiting hostile conditions as seen in the Diepkloof and Silasberg sites. The evaluations using the Biotope Quality Index indicated its potential as an ecological geospatial Feature analysis parameter with sites similar to the description of QQ Mountain (vegetation and landform) displayed good environmental (biotope) conditions, sites similar to the Diepkloof and Silasberg had moderate environmental conditions, and while sites related to the Avondrust has poor environmental conditions for the Basotho Montane Shrubland vegetation type in the GGHNPark.

Conclusion

The representatives from Insecta and Entognatha families seemed to be well adapted to montane shrubland vegetation type with great community composition for arthropods between different slope degrees and landforms. The impacts of environmental disturbance on the abundances and composition of arthropods were different between shrubland sites. The Biotope Quality Index and other species surrogates responded differently to plant invasion and soil erosion between sampled sites. Plant invasion was shown to act contrary to influence both the increase and decrease in arthropod diversity as well as biotope quality. The ecology of the QQ Mountain site played a role in the resilience of invasion of *Cynodon dactylon* grass associated with better arthropod diversity. While, the invasion *Acacia mearnsii* trees at the Avondrust site influenced lower arthropod diversity. The Biotope Quality Index showed a better response of arthropods to environmental disturbances both for soil erosion and plant invasion than other species surrogates. Lastly, the Biotope Quality Index indicated potential to be used in conservation programs for biomonitoring and geospatial studies for the Basotho Montane Shrubland vegetation type.

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CHAPTER 6 : Ecological Bioassessment using the Biotope Quality Index to Evaluate the Northern Drakensberg Highlands Grassland

Abstract

Montane highland grasslands are among bioregions susceptible to ecological alteration and change due to threats posed by climate change, anthropogenic and environmental disturbances. The study was aimed to demonstrate the use of Biotope Quality Index and Biotope Conservation Status as species surrogates to evaluate the conservation state condition of the Northern Drakensberg Highlands (Gd5) grassland vegetation type, using plant abundance-cover and communities as well as arthropod morphospecies assemblages. Vegetation surveys were carried out using 6 m x 5 m nested sampling plots, quantified and identified to species, while arthropods were collected using pitfall trapping and sweep netting, quantified and identified to morphospecies, in four Gd5 sites (Blesbok Loop, Diepkloof, Oribi Loop and Tossline) at the Golden Gate Highlands National Park (GGHNPark). The results indicated that the Gd5 vegetation type is characterised by Poaceae, Asteraceae, Thymelaeaceae, Oxalidaceae, Lobeliaceae, Rosaceae, Proteaceae, and Orchidaceae families of plants with a dominance of grass-herb floristic composition with sporadic shrubs and trees. There two main dominating plant communities were *Themeda triandra – Eragrostis chloromelas* and *Themeda triandra – Aristida diffusa* which further divided into four sub-communities. Analysis Arthropod assemblages were characterised by families Entomobryidae, Cicadellidae, Thripidae, Anthomyiidae, Lycaenidae, Xylorctidae, Formicidae and Lycosidae with mostly herbivorous, omnivorous and predatory individuals. A two-way analysis indicated five arthropod morphospecies communities namely (1) *Lepidocyrtinus* sp.1 (Entomobryidae) – Agalliinae sp.3 (Cicadellidae), (2) Thripidae sp.2 (Thripidae) – *Capbrya* sp.1 (Entomobryidae), (3) Anthomyiidae sp.2 (Anthomyiidae) – Formicinae sp.3 (Formicidae), (4) Encyrtidae sp.2 (Encyrtidae) – Rutelinae sp.1 (Scarabaeidae), and (5) Thripidae sp.3 (Thripidae) – Chrysomelidae sp.6 (Chrysomelidae) respectively. The Biotope Quality Index (*BQI*) was higher for plants and arthropods at Tossline (BQI_{veg} = 28.323: BQI_{art} = 92.937), Diepkloof (BQI_{veg} = 27.353: BQI_{art} = 79.826), Oribi Loop (BQI_{veg} = 13.753: BQI_{art} = 49.157) and Blesbok Loop (BQI_{veg} = 16.701: BQI_{art} = 43.512) sites respectively. All sampled sites were above good conservation status due to no significant influence on the measured environmental (anthropogenic and ecological) disturbances towards the abundances of biotic taxa for both plants and arthropods. The Biotope Quality Index and Biotope Conservation Status using both plant communities and arthropod assemblages showed the potential to be used as a species surrogates for a rapid bioassessment and biomonitoring for the Northern Drakensberg Highlands grassland vegetation type at the GGHNPark.

Keywords

Afromontane bioregions, Arthropod assemblages, Biomonitoring, Biotic Communities, Biotope Conservation Status, Disturbance Level, Vegetation Type

Introduction

There is a global increase in the rate of degradation of natural ecosystems due to escalated large-scale anthropogenic disturbances and ecological stochastic events leading to pollution, land transformation, habitat fragmentation, alien invasions and climate change (IPBES, 2019; Eriksen et al., 2021). Ecosystem and biodiversity conservation programs use biological assessments (bioassessments) as tools to monitor changes posed by negative impacts of ecological disturbances on the biotic community structure and diversity (Metcalfe, 1989; Rosenberg, et al., 2004), disturbance tolerance response (Hilsenhoff, 1987), habitat status condition (Gazendam et al., 2011), as well as ecosystem functioning (Cummins et al, 1989; Wang et al., 2016), quality (Kitching et al., 2000), resilience (Gallopìn, 2006) and stability (Simboura and Zenetos, 2002). Bioassessment protocols involve biotic indices as a practical-technical approach to track and quantify impact of environmental disturbances (Barbour et al., 1999), with assumption that biological population fluctuations and community structure composition can be attributed to ambient environmental conditions (Hobbs and Suding, 2009), in that evolutionary relationships and links created through ability of biotic units to adapt (Callicott, 2000), and for capacity of the environmental to support and maintain optimal heterogeneous conditions and populations, even when under perturbing exposure to disturbance (Pickett et al., 2007; Hinojosa et al., 2010). Most of the ecological restoration efforts in protected areas are aimed at habitat rehabilitation, increase and maintain biodiversity, as well as intent to restore ecosystem functioning in respect to holistic biotope challenges of interest (Gaston, et al., 2002; Allan and Castillo, 2007). There have been international (Temperate Grasslands Conservation Initiative, TGCI) and local conservation (South African Biodiversity Institute, SANBI and Maloti Drakensberg Transfrontier Programme, MDTP) initiatives aimed at increasing grassland area protection, establish rehabilitation and restoration programs (TGCI, 2011; Carbutt and Martindale, 2014).

A protected area is a recognised and dedicated geographic space that aims to maintain functioning ecosystems, biomes and play a role for species refugia as well as long-term nature conservation (Boitani, et al., 2008; Dudley, 2008; Muhumuza and Balkwill, 2013). Temperate grasslands are among the most threatened and least protected biomes in terms of global surface area cover if compared to other highly practised biomes such as savannas, mangroves, subtropical and tropical forests (Neke and du Plessis, 2002; Bertzky, et al., 2012; Driver et al., 2012). However, the grasslands biome is the second largest covering a surface area of 360 589 km^2 in South Africa, to which approximately 98 % is unprotected (prone to varying degrees of disturbance), and only 2 % is formally conserved as protected areas especially in the montane and highland along the Maloti-Drakensberg mountain escarpment (Carbutt et al., 2011). The local temperate grassland biome is represented by four bioregions that include Drakensberg Grassland, sub-Escarpment Grassland, Dry Highveld Grassland and Mesic Highveld Grassland, that further form 72 vegetation sub-units clustered according to their similar vegetation structure, macroclimate (average annual rainfall, temperatures and frost), and a similar disturbance events and regime namely frequent fire, grazing and soil erosion (Mucina and Rutherford, 2006; Carbutt et al., 2011). However, southern African bioregion is mostly represented by indigenous temperate grasslands with a high endemic ratio per surface area cover especially in the Maloti-Drakensberg mountain escarpment (O'Connor, 2005; Clark et al., 2011). Additionally, the eastern Free State Maloti-Drakensberg, western KwaZulu-Natal uKhahlamba-Drakensberg, north Eastern Cape Witteberg-Drakensberg and Lesotho Maloti Mountain make up the Drakensberg Alpine Centre of plant endemism and their conservation status is not fully known (Carbutt and Edwards, 2006).

Although most of bioassessment and biomonitoring arthropod protocols have been developed for terrestrial and aquatic ecosystems (Morris, 2000), application of each needs to considering variation in specific regional and environmental parameters (Simboura and Zenetos, 2002; Pryke and Samways, 2012). In grassland biomes, arthropod protocols make use of species surrogates to evaluate and quantify the impact of ecological disturbances such as fire (Little at al., 2013; Pressler et al., 2018), soil erosion (Menta, 2012; van der Merwe et al., 2020) and grazing (Jankielsohn et al., 2001; O'Connor, 2005; Klink et al., 2015; Shezi et al., 2021). The bioassessment can adopt a univariate, multivariate and (or) multimetric approach (Brehm et al., 2003; Beier and de Albuquerque, 2015), to monitor the spatial and temporal changes over time. The sampling scale is an important aspect when evaluating and monitoring disturbance effects using species surrogate such as estimated species richness and evenness, diversity, community structures and functional feeding groups, also to assess impact on quality and functioning of the ecosystem (Bredenhand, 2014), as well as to measure stability and resilience capacity during perturbation exposure to disturbance dynamics (Gàmez-Viruês et al., 2015; Eckert et al., 2019). The biotope is a geographical unit of the vegetation type or a habitat that can be bordered by appropriate boundaries and which is characterised by its biota (Lincoln et al., 1998; Allaby, 2010), whom can be used as working ecological or ecosystem scale for bioassessment and biomonitoring. The Biotope Quality Index expressed below takes into consideration the evolutionary history of biotic integrity units, and assumes that the biotope in its naturally good conserved and heterogeneous state, can act to sustain biotic units to optimum species populations to perform ecological roles and processes. Bredenhand (2014) developed a biotic surrogate, the Biotope Quality Index (*BQI*), using the responses of the arthropod population and community, as a tool to assess the impact of environmental disturbance and state of ecological condition. The Biotope Quality Index (*BQI*) like other indices (e.g., bivariate correlation, regression, or multiple regression analysis and clustering) which use standardised values formulation, of non-parametric biotic assemblages using morphological species abundances counts (or plant cover/density).

Let x species abundance (cover/density);

$$\text{Biotope Quality Index } (BQI) = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right)$$

if $\log \sigma > 0$……………………………… (Bredenhand, 2014)

The expression elements represented, include the abundance of *i*th species/morphospecies (x_i), *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ), and this is only true if there was variance (σ) in the mean ($\overline{x}_i$) of observed population, in that $\log \sigma > 0$ for natural and heterogeneous biotopes.

The Golden Gate Highlands National Park (GGHNPark) falls along the north-eastern regions of Maloti-Drakensberg mountain with mainly Clarens and Elliot Formation (sandstone, siltstones and mudstone) and occasional basalt soil deposits as well as cold moist highlands sourveld (Grab et al., 2011). Plant diversity is usually greater in nutrient-poor sandstone soil than on nutrient-rich dolerite soil substrates (Tilman, 1988; O'Connor, 2005), creating a mosaic of locally restricted endemic species ranges. Among the main objectives of GGHNPark is the ongoing conservation and protection of highlands biodiversity with its natural environment as well as the sustainable functioning of freshwater and other grassland ecosystems (SANParks, 2014).

GGHNPark is among Drakensberg endemic vegetation dominated by forbs, short *Leucosidea* spp. shrubs and scattered *Protea* spp. trees (Mucina and Rutherford, 2006) offering optional refugia and food source for arthropods. The park hosts important taxa of four vulnerable southern slopes endemic butterfly species namely *Pseudonympha paragaiko* (Nymphalidae: Golden Gate Brown), *Torynesis orangica* (Nymphalidae: Golden Gate Widow), *Orachrysops montanus* (Lycaenidae: Golden Gate Blue) and *Durbania amakosa sagittata* (Lycaenidae: Qwaqwa Rocksitter) (Mecenero et al., 2013) with a unique stratified arachnid-insect composition along shrub patches (Botha et al., 2020). Additionally, Drakensberg endemic plants include *Alepidea pilifera* herb (Apiaceae), *Chironia peglerae* herb (Gentianaceae), *Elaphoglossum drakensbergense* fern (Dryopteridaceae), *Merwilla dracomontana* herb (Hyacinthaceae) which are most likely sensitive and vulnerable to grassland environmental changes (Mucina and Rutherford, 2006; Raimondo et al., 2009; Crouch et al., 2011). Most research relating to environmental studies in the GGHNPark has been on the geomorphology landscape (Groenewald, 1986; Grab et al., 2011; Telfer et al., 2012), bird species inventory list (Hutsebaut et al., 1992; de Swardt and van Niekerk, 1996; Barnes et al., 1998), alien plant invasion (Mukwada et al, 2015), plant inventory lists (Roberts, 1966; Daeman et al., 2010), a few in plant community structure and vegetation type classification (Mucina and Rutherford, 2006), fire and soil erosion effects on soil macrofauna (van der Merwe et al., 2020), influence of patchy shrublands-forests on arthropod communities (Botham et al., 2020), and not much has been done on arthropod and plant communities relationships. The current study aims to evaluate the response of community structures of arthropods and plants to environmental disturbances, as well as ecological quality and conservation status using the Biotope Quality Index for the Northern Drakensberg Highlands grassland vegetation type.

Materials and Methods

Study site

Golden Gate Highlands National Park (GGHNPark) cover surface area of 342 km^2 (32758 ha), is located on the north-eastern Maloti-Drakensberg Mountain border between the Kingdom of Lesotho, eastern Free State and Kwa-Zulu Natal Province, located between latitude of 28°27' S to 28°37' S and longitude of 28°33' E to 28°42' E, with elevation ranging from 1892 m to 2837 m and can be seen in Figure 6.1 (Rademeyer and van Zyl, 2014). The eastern Free State province region is characterised by higher summer rainfall of approximately 850 mm mean annual precipitation, with a warmer mean annual temperature ranging from 13 °C to 30 °C, and lower winter rainfall of approximately 500 mm mean annual precipitation and with mean annual temperature ranging from 1 °C to 26 °C (Mucina and Rutherford, 2006; Grab et al., 2011).

The study adopts bioregion and vegetation type categories as described by Mucina and Rutherfiord (2006) and regards them as biotope units (Figure 6.1), to which the Northern Drakensberg Highlands Grassland (Gd5) is a vegetation type with common dominant flora and clear ecological boundaries. The Northern Drakensberg Highlands Grassland (Gd5) unit is characterised by steep to moderate slopes of broad valleys, and its geomorphology, elevation, rainfall and soil substrate properties influence the indigenous vegetation composition of the sour highland grasslands. Four sites were selected considering ecological disturbances e.g. soil erosion, plant encroachment and invasion. The selected sites were named Blesbok Loop (Bles_Gd5), Oribi Loop (Orib_Gd5), Diepkloof (Diep_Gd5) and Tossline (Toss_Gd5).

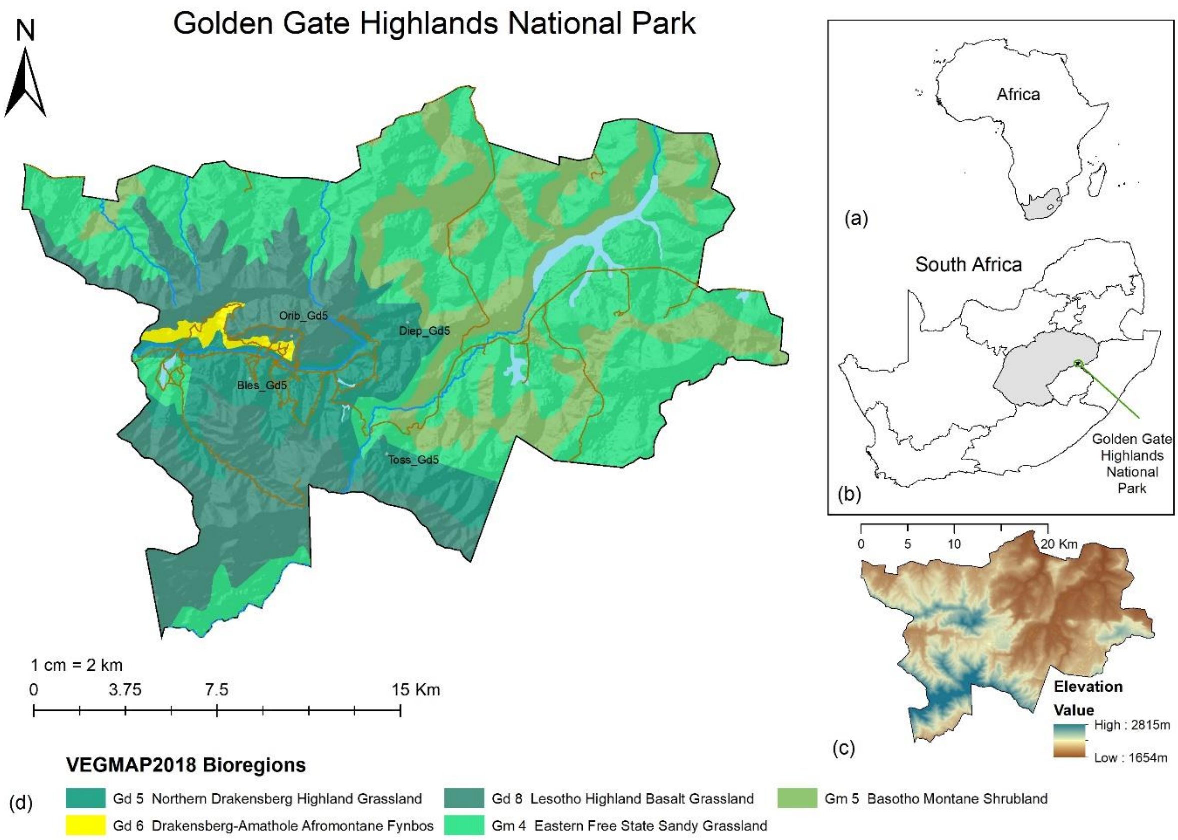

Figure 6.1: Location of the study area illustrating (a) the continent of Africa, (b) eastern section of the Free State Province highlighted in outline of South Africa, (c) elevation and hillshade of GGHNPark created using DEM image from USGS Explore Earth, and (d) the GGHNPark highlighting sampled sites of the Northern Drakensberg Highlands Grassland (Gd5), adopted from SANBI GIS VEGMAP (2018) using ArcGIS.

Vegetation survey

Vegetation sampling was conducted during the September 2016 to September 2018 period along with the arthropod survey. The nested sampling plots of 6 m x 5 m quadrants (area of 30 m^2) were recorded along a 200m transect to ensure variation of plants within sites. Plant counts were recorded and species cover estimated using the modified Braun-Blanquet abundance-cover scale (Brand et al., 2008). Identification and confirmation of plant species was done at Botany Herbarium of the Department of Plant Sciences at the University of Free State Qwaqwa campus, and species names were confirmed using literature by Moffett (1997) as well as Germishuizen and Meyer, (2003).

Arthropod sampling

A combination of sampling techniques was used to collect ground-dwelling arthropods namely unbaited pitfall trapping, active net sweeping and field observations. Pitfall trap sampling was conducted by inserting 10 randomly arranged

250 ml plastic jars (60 mm diameter X 100 mm height) into the ground surface, on a 200 m transect with 20 m spacing in between. A 50 ml pitfall solution of ethanol and propylene glycol (3 v:1 v ratio) was used as a killing and preservation agent (Yekwayo et al., 2018). The plastic jar lid was modified to elevate the lid 200 mm above ground surface to restrict excess rainwater runoff, prevent evaporation of pitfall preservation solution, large objects and predators (vertebrates and invertebrates) from clogging the trap. Furthermore, net sweeping was done using a 400 mm diameter steel hoop with a mesh fabric sheet, which arthropods were swept along the 200 m vegetation transect adjacent to pitfall traps during the day. In addition, field observations were also included to record aerial insects visible on vegetation in the 200 m sampling unit and surroundings. The catch and release method was used for net sweep samples after reference collection was compiled. Collected arthropod samples were sorted and identified to family and genus taxa level using Scholtz and Holm (2008) for insects, Horak et al (2015) for ticks and Dippenaar-Schoeman (2014) for spiders, and then later assigned morphospecies reference (pseudonym) to each morphologically distinct specimen (Gerlach et al, 2013). Only adult arthropod life stages were considered and data from the sampling method was pooled for analysis per site.

Environmental data and disturbances

Environmental data was collected to measure the aspect, slope, elevation, latitude and longitude. Environmental disturbances were categorised as ecological (natural) and anthropogenic activities to help describe sampled sites based on current ecological conditions, that included plant invasion, soil erosion, road site constructions, footpaths, land pollution level, current wildlife and human activity zones. Soil erosion was determined using the modified RUSLE model (Renard et al., 1997) considering degree of plant cover, soil type, moisture and depth (supplementary material: Table 0.6). Plant encroachment-invasion was determined by recording the presence and percentage cover of plant species classified as encroachment and alien invasive (supplementary material: Table 0.5). Anthropogenic activities that contribute to disturbance were also recorded historical agricultural activities (prior to sampled areas being declared as a national park), housing and road constructions, land pollution, current wildlife and human activity zones as well as conservation rehabilitation programs (supplementary material: Table 0.7).

Data analysis

Descriptive statistical analysis was used to process recorded species presence data to measure mean, standard deviations and relative abundances (cover/density) for each sampled site. The population mean of arthropod and plant species taxa was compared separately between sites with uneven sample sizes using ranked means for Kruskal-Wallis H-test and Dunn post-hoc test. The sample-based rarefaction was used to determine species rare occurrence probability (singleton and doubleton frequencies) for each site (Gotelli and Colwell, 2001). Species surrogate biotic indices methods were included to measure estimated species richness using the Chao2 estimator (Chao, 1984), diversity indices using Simpson-Yule diversity (Simpson, 1949) and Shannon–Wiener function (Weaver and Shannon, 1949) as well as Biotope Quality Index (Bredenhand, 2014) to indicate ecological conditions of each samples site. We also propose a category classification of Biotope Conservation Status (*BCS*) (Table 6.1) which would be determined by product of the main two disturbance variables and its influence on the *BQI* of each sampled site as follows;

$$DBQI_i = \frac{BQI_i}{(d_1 \times d_2)_i}$$

……………………………………………………………………………………………(2)

$$BQI_{mean} = \frac{1}{N}\sum BQI_n$$

……………………………………………………………………………………………(3)

$$Biotope\ Conservation\ Status\ (BCS) = \frac{DBQI_i}{BQI_{mean}} \times 100$$

……………………………………………………………………………………………(4)

Table 6.1: Biotope Conservation Status category classification score illustrating conditions for Biotope Quality Index and degree of Disturbance

BCS value (%)	Biotope Conservation Status Rating
91- 100	Pristine Conserved Status
71 – 90	Good Conserved Status
50 – 70	Fairly Conserved Status
26 – 49	Poorly Conserved Status
0 – 25	Highly Disturbed Status

A modified two-way species cluster analysis (biclustering) using the 'Euclidean' distance measure and the 'Ward' hierarchical cluster method were simultaneously used to identify possible arthropod and vegetation community subgroups with Heatmaply package in the R-Studio (Dominici et al., 2008, Lunghini et al., 2013). The association of environmental or ecological disturbance as a gradient that influences arthropod assemblage and plant abundance-cover clusters was evaluated using the Canonical Correspondence Analysis (CCA) in the R-Studio with vegan package (Oksanen, 2010).

Qualitative and quantitative methods used to determine the effects of environmental and ecological disturbance degree on the dynamics of the arthropod and plant community using the permutational multivariate analysis of variance (PERMANOVA) in the R-Studio (ADONIS, simulations ran 23 permutations to congruency levels, in vegan package) and only disturbances with significant influence were considered for further analysis (Gardner, 2014). Furthermore, significant quantitative disturbance factors were placed for nonmetric multidimensional scaling ordination (NMDS), transformed arthropod abundance using the Hellinger method, with a Bray-Curtis measure as an ordination distance method (vegan R-studio) to distinguish clustering influence between sampled sites (Gardner, 2014). Then, sites were related based on arthropod, plant and environmental data using both quantitative and weight qualitative data using Bray-Curtis dissimilarity (Bray and Curtis, 1957).

Then, the Biotope Quality Index was used for geospatial modelling as a vector to indicate current ecological conditions of the Northern Drakensberg Highlands Grassland (Gd5) grassland vegetation type in the GGHNPark. The Biotope Quality Index estimates (*BQI*est.) were predicted based on site relatedness using bootstrap jackknife 2 using 999 permutations (Manly, 1991), with varying population sizes from the calculated mean and standard deviation of the neighbouring and adjacent related sites. Ecological and topographic barriers were used to divided Northern Drakensberg Highlands Grassland into sampling polygon zones units per locality and then, predicted asymptotic Biotope Quality Index values were assigned to each polygon zone using ArcGIS Map (version 10.5) to highlight the area of environmental concern. Shapefiles for continental and country maps were accessed from the Esri GIS database, while provincial, SA protected areas (national parks), roads and surface waterbody shapefiles of South Africa were accessed from the EGIS database of the Department of Forestry, Fisheries and the Environment (DFFE), and vegetation bioregion VEGMAP 2018 (for South Africa, Lesotho and Swaziland) shapefiles were accessed from the SANBI Biodiversity GIS database.

Results

Vegetation composition

The vegetation composition of the Northern Drakensberg Highlands grassland recorded a total of 62 plant species from two clades (Polypodiophyta and Angiospermae), three classes (Polypodiopsida, Monocotyledonae and Eudicotyledonnae), nine orders and 14 families respectively. The plant life-form composition was represented by 37.590 % grass and sedges, 23.360 % herbs and forbs, 2.300 % shrubs as well as 1.200 % trees in general (Figure 6.2). The plant families of Poaceae and Asteraceae were the most common and abundant in all sampled sites, with more species (n = 52) represented in the Tossline site which was mainly characterised by 31 % Poaceae, 7 % Asteraceae, 4 % Orchidaceae, 2 % Cyperaceae, 1 % Rosaceae and 1 % Thymelaeaceae assemblage. Furthermore, the Diepkloof site (n = 46) was mainly characterised by 27 % Poaceae, 6 % Asteraceae, 2 % Cyperaceae, 1 % Orchidaceae, 1% Proteaceae and 1 % Thymelaeaceae assemblage, the Blesbok Loop site (n = 41) was characterised by 24% Poaceae, 7 % Asteraceae, 2 % Orchidaceae, 2 % Cyperaceae, 2 % Lamiaceae and 1 % Oxalidaceae assemblage, and lastly, the Oribi Loop site (n = 38) was mainly characterised by 21 % Poaceae, 8 % Asteraceae, 2 % Orchidaceae, 2 % Cyperaceae, 1 % for Lamiaceae and Oxalidaceae assemblage respectively (Figure 6.3). The Blesbok Loop and Oribi Loop sites had a similar plant family taxa composition than to Tossline and Diepkloof sites (Figure 6.3e). However, there was significant difference of $F_{3,244}$ = 13.594 (p < 0.01) observed in abundance-cover especially between sites, namely the Diepkloof and Oribi Loop, Tossline and Oribi Loop, as well as Blesbok Loop and Diepkloof sites.

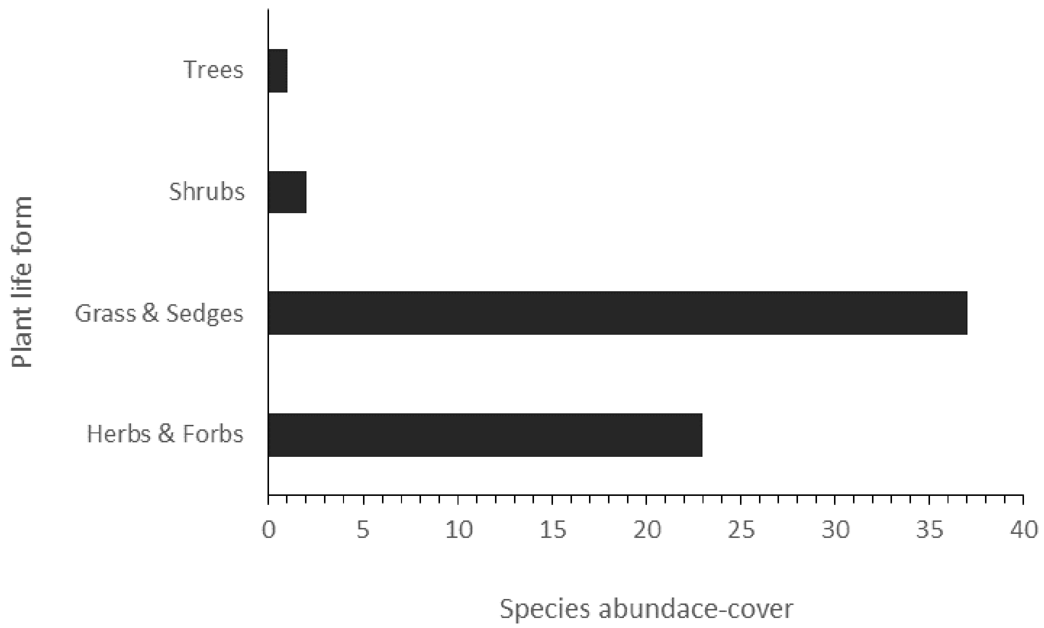

Figure 6.2: Plant life-forms observed for the Northern Drakensberg Highlands grassland included grasses and sedges, herbs and forbs, shrubs as well as trees abundance-cover composition in the GGHNPark.

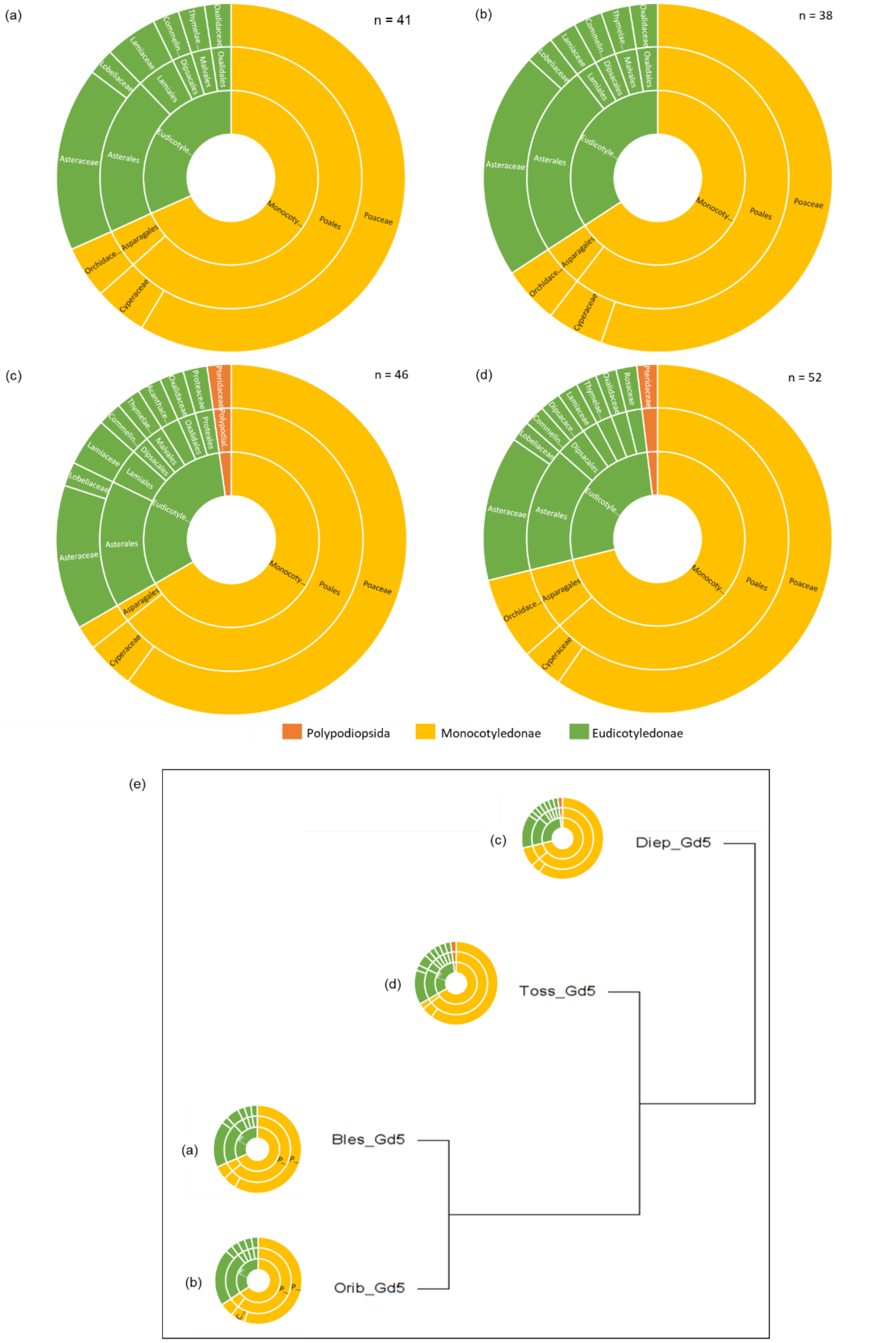

Figure 6.3: Comparative plant taxa composition (a) Blesbok Loop (Bles_Gd5), (b) Oribi Loop (Orib_Gd5), (c) Diepkloof (Diep_Gd5), (d) Tossline (Toss_Gd5) and (e) illustrating family taxa composition similarity between sites for Northern Drakensberg Highlands grassland vegetation type in the GGHNPark.

The two-way hierarchical species community cluster for vegetation composition of the Northern Drakensberg Highlands vegetation type was dominated by two main communities (Table 6.2), namely the *Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn*. - *Erg.chl*.) and *Themeda triandra – Aristida diffusa* (*Thm.trn*. – *Art.dff*.) communities, with four additional sub-communities that included (1), *Thm.trn*. - *Erg.chl* – *Eragrostis racemosa*, (2) *Thm.trn*. - *Erg.chl - Lasiosiphon burchellii,* (3) *Thm.trn*. – *Erg.chl – Aristida junciformis* and (4) *Thm.trn*. – *Art.dff*. - *Eragrostis curvula* respectively (Figure 6.4a). Furthermore, there were 28 species shared between all sampled sites, with 9 species unique to the Diepkloof site (e.g. *Aristida congesta*, *A. diffusa*, *Barleria monticola, Protea roupelliae* and *Vernonia flanaganii*), and the Tossline site with 3 unique species (e.g. *Leucosidea sericea*, *Eulophia aculeate* and *Moraea sp.*) assemblage observed (Figure 6.4b).

Table 6.2: Vegetation classification illustrating floristic composition and structure for Mesic Highveld Grassland vegetation unit for Northern Drakensberg Highlands Grassland (Gd5) vegetation type in GGHNPark.

Vegetation classification (physiognomy)		
Northern Drakensberg Highlands Grassland (Gd5)		
3.	*Themeda triandra* - *Eragrostis chloromelas* (*Thm.trn. - Erg.chl.*) community	
	3.1.	*Thm.trn. - Erg.chl. - Eragrostis racemosa* sub-community
	3.2.	*Thm.trn. - Erg.chl. - Lasiosiphon burchellii* sub-community
	3.3.	*Thm.trn. - Erg.chl. - Aristida junciformis* sub-community
4.	*Themeda triandra – Aristida diffusa* (*Thm.trn*. – *Art.dff*.) community	
	4.1.	*And.app. – Erg.cpn. - Eragrostis curvula* sub-community

*abbriviations: *Themeda triandra - Eragrostis chloromelas (Thm.trn. - Erg.chl.), Themeda triandra – Aristida diffusa* (*Thm.trn*. – *Art.dff*.).

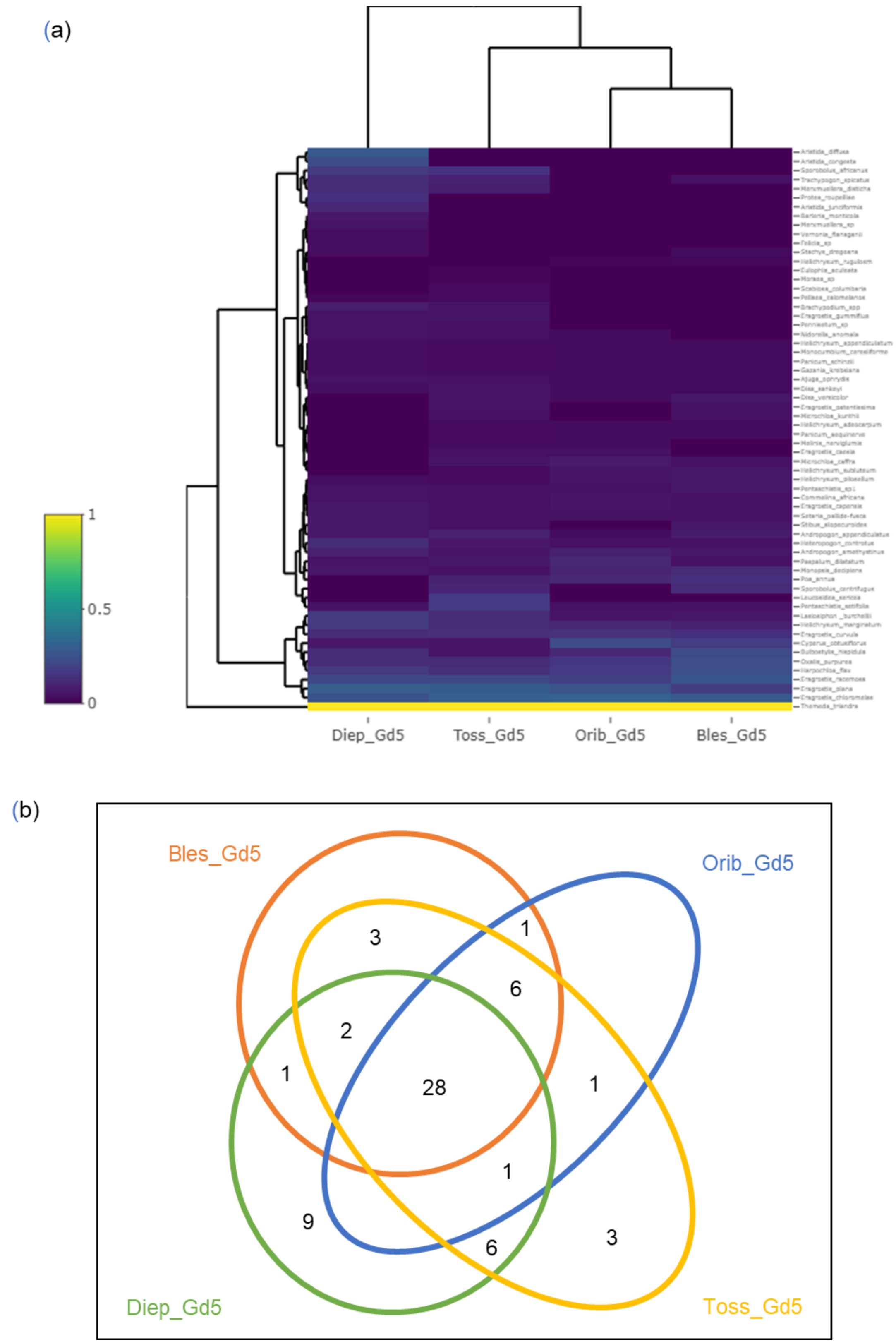

Figure 6.4: Species assemblages (a) a two-way hierarchical species community cluster using the Euclidean distance measure and Ward clustering method, (b) shared and unique species Venn diagram for the Northern Drakensberg Highlands vegetation type in GGHNPark.

Arthropod assemblages

A total of 3 246 adult arthropod individuals were collected from seven classes, 21 orders, 96 families and 202 morphospecies. The composition of arthropod taxa consisted of 76 % Insecta, 16 % Entognatha, 4 % Arachnida, 1 % Diplopoda, 1 % Chilopoda, 1 % Acari and 1 % Malacostraca generally for the Northern Drakensberg Highlands grassland vegetation type. Again, the feeding guilds that were observed included 45 % herbivores, 25 % predators, 4 % necrophagous, 6 % omnivores, 2 % haematophagous and 5 % scavengers arthropods (Figure 6.5), that were within taxa dominated by solitary, sub-social and eusocial behaviour arthropod life strategies.

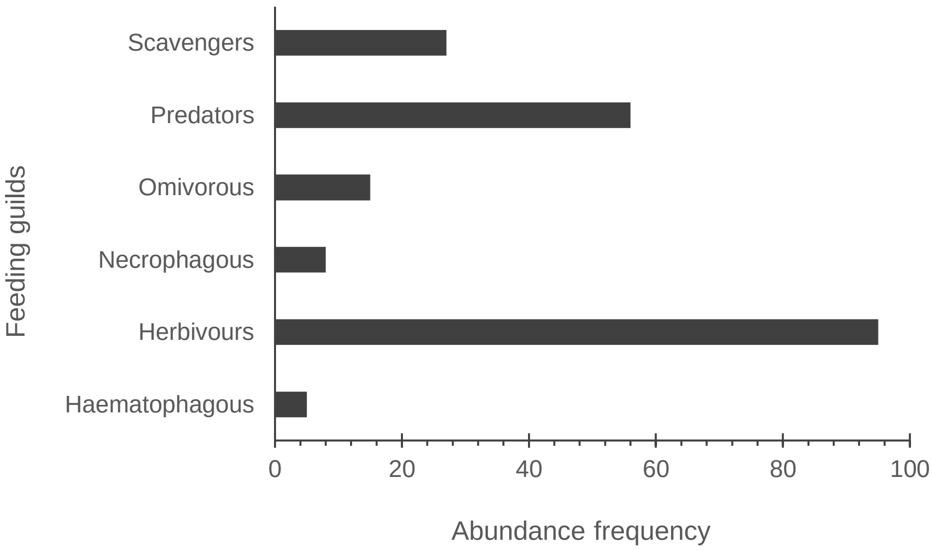

Figure 6.5: Composition of observed arthropod feeding guilds taxa highlighting herbivores, predators, necrophagous, omnivorous, haematophagous and scavengers.

Arthropod orders were dominated by 23 % Coleoptera, 20 % Diptera, 18 % Hymenoptera, 13 % Hemiptera, 9 % Hemiptera, 45 % Orthoptera, 2 % Araneae and 2 % Collembola, with most present in Diepkloof, Tossline, Oribi Loop and least in Blesbok Loop Site (Table 6.3). All sites had a morphospecies rarefaction value with about 32% to 47% composing of species occurring in singleton and doubleton counts of rare occurring species.

Table 6.3: Arthropod taxa counts for Diepkloof, Tossline, Oribi Loop and Blesbok Loop site observed in Northern Drakensberg Highlands Grassland in GGHNPark.

	Taxon counts			Morphospecies			
Sites	Class	Order	Family	T.count	singleton	doubleton	rarefaction
Bles_Gd5	5	15	51	98	23	22	0.459
Orib_Gd5	6	17	76	132	27	35	0.470
Diep_Gd5	7	20	75	139	25	35	0.432
Toss_Gd5	6	16	67	128	23	17	0.313

*abbreviations: Morphological species (morphospecies), morphospecies counts (T.count), Blesbok Loop (Bles_Gd5), Oribi Loop (Orib_Gd5), Diepkloof (Diep_Gd5) and Tossline (Toss_Gd5) sites, Golden Gate Highlands National Park (GGHNPark).

Common and dominant arthropod families included 15 % Entomobryidae, 6 % Noctuidae 6 % Cicadellidae, 5 % Formicidae, 5 % Anthomyiidae, 3 % Thripidae, 3 % Tenebrionidae, 3 % Staphylinidae, 3 % Peiridae, 3 % Scarabaeidae, 2 % Acrididae, Lycosidae, Pholcidae and Lycaenidae respectively. The Tossline and Diepkloof sites (n = 1117; n = 851) had the highest morphospecies abundance while the Oribi Loop and Blesbok Loop sites (n = 656; n = 622) were the lowest respectively (Figure 6.6). The two-way clustering revealed five morphospecies communities that includes (Table 6.4) (1) *Lepidocyrtinus* sp.1 (Entomobryidae) – Agalliinae sp.3 (Cicadellidae), (2) Thripidae sp.2 (Thripidae) – *Capbrya* sp.1 (Entomobryidae), (3) Anthomyiidae sp.2 (Anthomyiidae) – Formicinae sp.3 (Formicidae), (4) Encyrtidae sp.2 (Encyrtidae) – Rutelinae sp.1 (Scarabaeidae), and (5) Thripidae sp.3 (Thripidae) – Chrysomelidae sp.6 (Chrysomelidae) respectively (Figure 6.7). The sampled sites shared 45 families, with a family representative composition most closely similar for Tossline, Oribi Loop and Diepkloof sites sharing about 24 families, compared to Oribi Loop and Diepkloof sotes sharing 18 families as well as for Tossline and Oribi Loop sites sharing only 11 families. Furthermore, the Diepkloof site had the most unique family representatives observed with 16 morphospecies. There was a significant difference between ranked means of arthropod abundance with uneven site sample size using Kruskal-Wallis chi-squared of 20.783 (df = 3, p < 0,01), and Dunn post-hoc test specifically indicated that the arthropod morphospecies abundance mean at Tossline, Oribi Loop and Diepkloof sites was significant different from the Blesbok Loop site.

Figure 6.6: Arthropod assemblages observed at (a) Blesbok Loop (Bles_Gd5), (b) Oribi Loop (Orib_Gd5), (c) Diepkloof (Diep_Gd5) and (d) Tossline (Toss_Gd5) for the Northern Drakensberg Highlands vegetation type in GGHNPark.

Table 6.4: Arthropod community composition and structure for Mesic Highveld Grassland vegetation unit for Northern Drakensberg Highlands Grassland (Gd5) vegetation type in GGHNPark.

Arthropod community composition
Northern Drakensberg Highlands Grassland (Gd5)
1. *Lepidocyrtinus sp.1 (Entomobryidae)* – *Agalliinae sp.3 (Cicadellidae)* community
2. Thripidae *sp.2 (Thripidae)* – *Capbrya sp.1 (Entomobryidae)*community
3. Anthomyiidae sp.2 (Anthomyiidae) – Formicinae sp.3
4. Encyrtidae sp.2 (Encyrtidae) – Rutelinae sp.1 (Scarabaeidae)
5. Thripidae sp.3 (Thripidae) – Chrysomelidae sp.6 (Chrysomelidae)

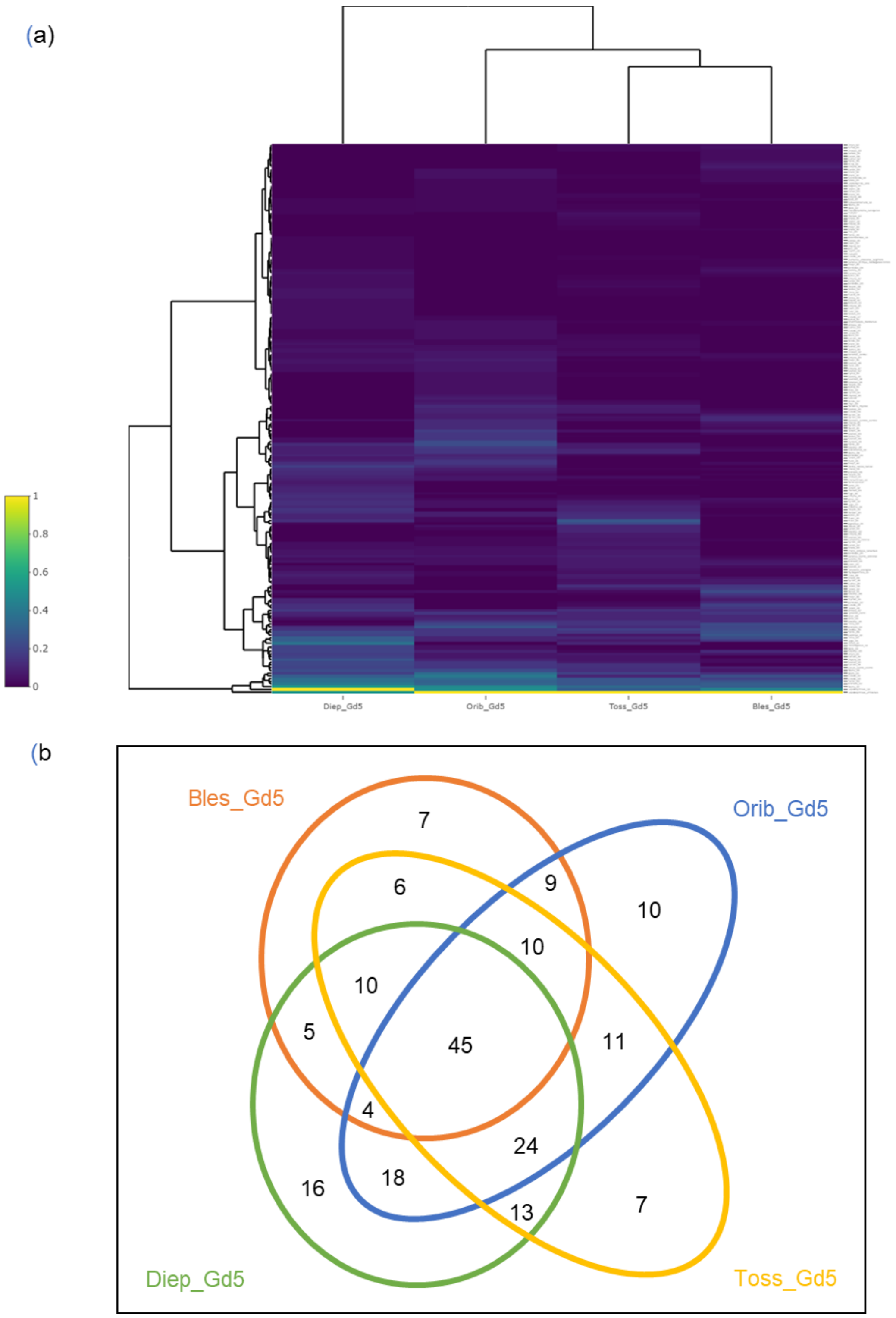

Figure 6.7: Arthropod morphospecies assemblage relatedness (a) a two-way hierarchical species community cluster using Euclidean distance measure and Ward clustering method, (b) shared and unique species Venn diagram for Northern Drakensberg Highlands vegetation type in GGHNPark.

Vegetation type disturbances

All qualitative and quantitative environmental disturbance parameters were restricted and limited with low-to-no human or tourism activity, as well as no infrastructure development for all sampled sites within restricted access park zone sections. The effect of soil erosion and alien invasion on the abundance of arthropods and plants did not have a significant influence using ADONIS.

Biotope Quality Index and other species surrogates

Bioassessment species surrogates responded differently compared to soil erosion and rarefaction effects for both evaluated plant and arthropod abundances. The Biotope Quality Index (*BQI*) was higher for plants and arthropods at Tossline (BQI_{veg} = 28.323: BQI_{art} = 92.937), Diepkloof (BQI_{veg} = 27.353: BQI_{art} = 79.826), Oribi Loop (BQI_{veg} = 13.753: BQI_{art} = 49.157) and Blesbok Loop (BQI_{veg} = 16.701: BQI_{art} = 43.512) sites respectively. Plant estimated species richness and Simpson-Yule diversity function were higher at Tossline (S_{max} = 52.182: D = 15.723), Diepkloof (S_{max} = 46.056: D = 17.635), Blesbok Loop (S_{max} = 41.056: D = 13.357) and Oribi Loop (S_{max} = 38.042: D = 11.729) sites respectively. While for arthropods, estimated species richness and Simpson-Yule diversity function were higher at Diepkloof (S_{max} = 148.929: D = 58.194), Tossline (S_{max} = 143.559: D = 44.151), Oribi Loop (S_{max} = 142.414: D = 55.314) and Blesbok Loop (S_{max} = 109.022: D = 34.886) sites respectively. The evaluation of floristic composition indicated that the estimated species richness and Biotope Quality Index had a higher positive correlation coefficient (R^2 = 0.746 and R^2 = 0.717) for both soil erosion and rarefaction, but lower for the Shannon-Wiener function (R^2 = 0.630) and least for Simpson-Yule diversity (R^2 = 0.417) respectively. The vegetation type had common plant species and taxa communities that were above mean abundance-cover observed per site namely from families of Poaceae, Asteraceae, Thymelaeaceae, Oxalidaceae, Lobeliaceae, Rosaceae, Proteaceae, and Orchidaceae. Furthermore, the estimated species richness, Biotope Quality index and Simpson-Yule diversity were high at the Tossline and Diepkloof sites, while low at the Blesbok Loop site and least at the Oribi Loop site. The Shannon-Wiener function indicated high diversity at Oribi Loop site with relatively high levels of soil erosion, while the Tossline, Diepkloof and Blesbok Loop sites had low diversity and low levels of soil erosion (Figure 6.8a-d).

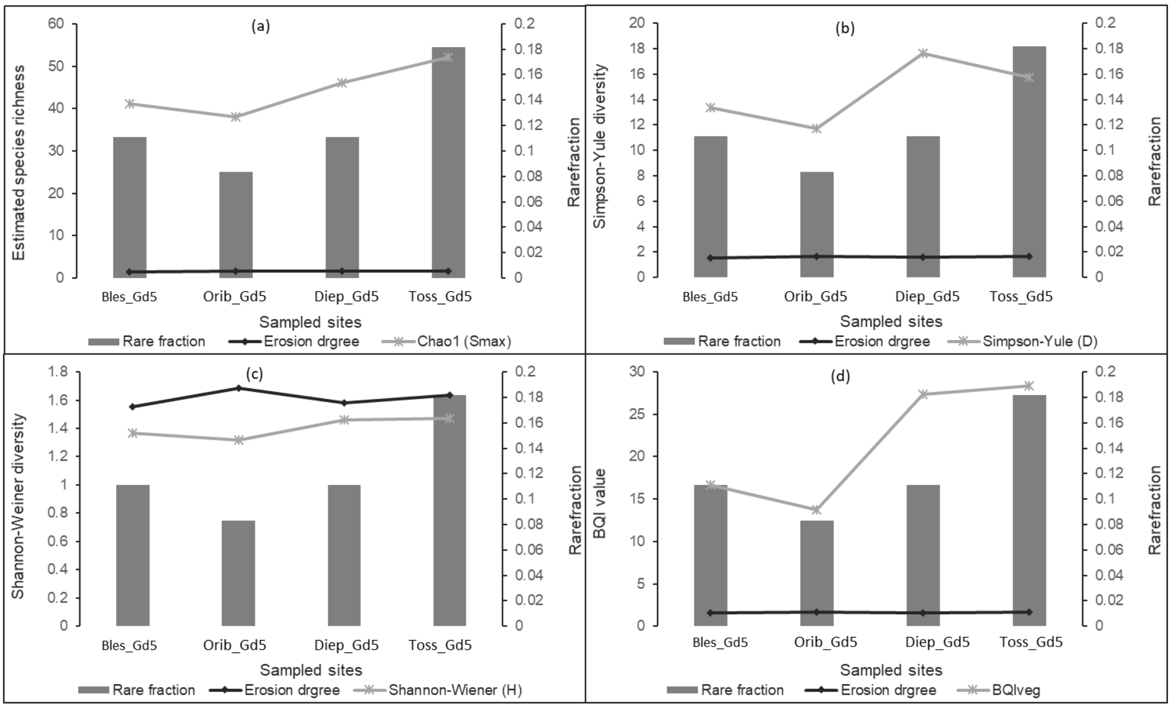

Figure 6.8: Bioassessment using response of vegetation species surrogates to soil erosion and rarefaction with (a) the estimated species richness, (b) Simpson-Yule diversity, (c) Shannon-Wiener function, and (d) Biotope Quality Index for the Northern Drakensberg Highlands vegetation type in GGHNPark.

Furthermore, response of species surrogates using arthropod assemblages indicated that the Biotope Quality index had a higher positive correlation coefficient (R^2 = 0.993) for both soil erosion and rarefaction, when compared to the estimated species richness (R^2 = 0.611), Shannon-Wiener function (R^2 = 0. 376) and least for Simpson-Yule diversity (R^2 = 0.1367) respectively. However, the estimated species richness, Simpson-Yule diversity and Shannon-Wiener diversity were higher at sites with low rarefaction namely the Oribi Loop and Diepkloof sites, while species surrogates were lower at the Blesbok Loop and Tossline sites with high rarefaction (Figure 6.9a-d). The common arthropod species and taxa assemblages in the vegetation type that were above mean abundance observed per site included families of Entomobryidae, Cicadellidae, Thripidae, Anthomyiidae, Lycaenidae, Xylorctidae, Formicidae and Lycosidae.

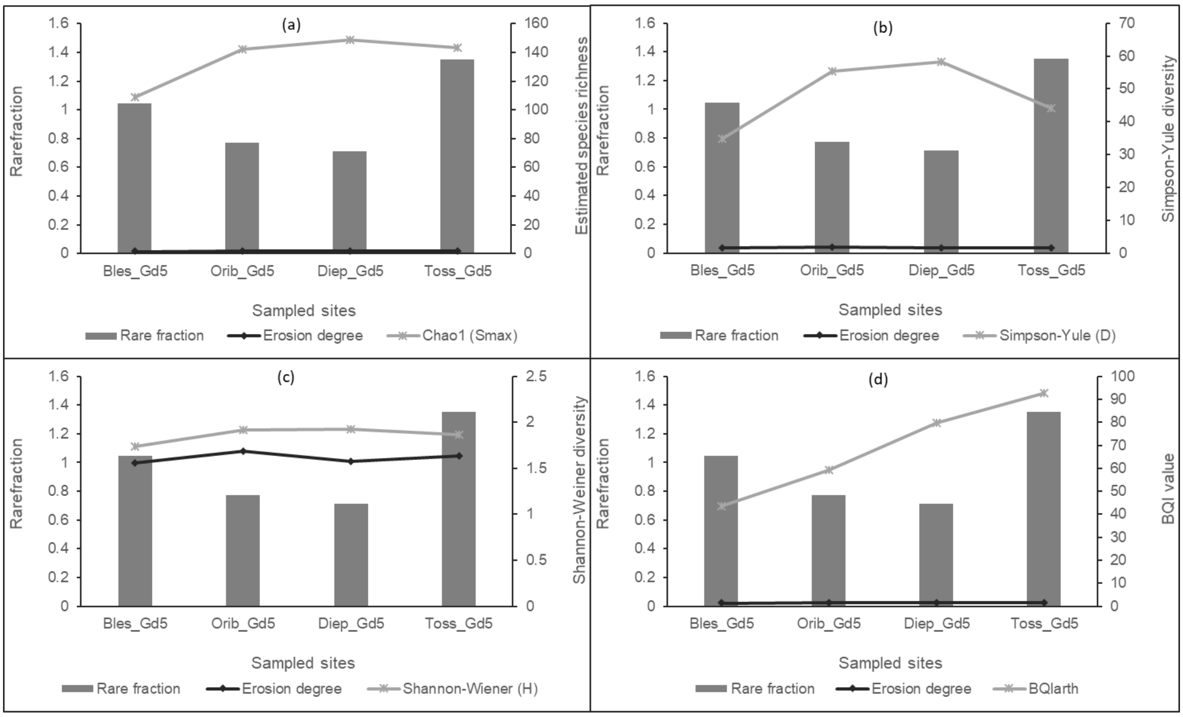

Figure 6.9: Bioassessment using response of arthropod species surrogates to soil erosion and rarefaction with (a) the estimated species richness, (b) Simpson-Yule diversity, (c) Shannon-Wiener function, and (d) Biotope Quality Index for the Northern Drakensberg Highlands vegetation type in GGHNPark.

Biotope Conservation Status

The Biotope Conservation Status using plant as measuring units indicated that sites attributed to relatively lower degree of disturbance, higher abundance-cover and rarefaction of species such as the Diepkloof site with a *BCS* score of 80.420 % and the Tossline site with a *BCS* score of 80.480 %, and while the Tossline site with a *BCS* score of 82.420 % using arthropods as a unit of measure, were classified to be pristine conserved status. Furthermore, sites attributed to uniform and average disturbance degree, higher abundance-cover and rarefaction of species included the Diepkloof site with a *BCS* score of 73.470 % which was classified to be in a good conserved status using arthropods as a unit of measure. However, sites attributed with lower degree of disturbance, moderate abundance-cover and rarefaction of species had mixed status with the Oribi Loop site (*BCS* score of 50.990 %) classified as fairly conserved status using arthropods as a unit measure, while the Blesbok Loop site (*BCS* score of 49.900 %) and the Oribi Loop site (*BCS* score of 37.900 %) was classified as poorly conserved status when using plants as a unit measure.

Discussion

The findings presented above indicate that both plants and arthropods can be used for rapid bioassessment with the inclusion of Biotope Quality Index as a species surrogate even in stable biotopes. The Northern Drakensberg Highlands grassland vegetation type revealed two main strata clusters of plant community that are dominated by Poaceae and Asteraceae, with its differential subcommunity having sporadic representatives of Rosaceae, Orchidaceae, Thymelaeaceae, Oxalidaceae and Proteaceae depending on topography and aspect of the vegetation unit. The dominant grass genera included species of *Themeda*, *Eragrostis*, *Aristida* and *Heteropogon*, which are regarded as main species for community composition of C_4 grass and sourveld for the Northern Drakensberg Highlands grassland (Ehleringer et al., 1997; O'Connor, 2005). The grass genera are adapted to annual field fire and regular herbivory to maintain its compositional stability (Tainton et al., 1980; Morris et al., 2020), which are located mainly in central summer-rainfall, relatively lower north-facing slope degrees with shallow nutritive soil in the Maloti–Drakensberg Mountains (Cowling, 1983; Scott, 2002). The dominance of *Themeda triandra* in the vegetation type can be attributed to its ability to exploit both the winter and summer growing seasons equally, thus gaining greater proportional cover in this study (Cowling, 1983), and it also used as an indicator in central regions for the untransformed or absence of intense cultivation of natural fields (Loftus et al., 2015; Muller et al., 2021).

Arthropod feeding guilds were comprising of mainly herbivorous, predatory, omnivorous, scavenger and necrophagous individuals, that are also reported as common feeding guilds in grassland regions (Morris, 2000; Joern and Laws, 2013; Botha et al., 2016). In this study, the above-ground arthropod community groups highlighted were dominated by species and taxa of *Lepidocyrtinus* spp. (Entomobryidae) springtails, Agalliinae spp. (Cicadellidae) leafhoppers, Lycosinae spp. (Lycosidae) spiders, Formicinae spp. (Formicidae) ants, Myrmicinae spp. (Formicidae) ants, Anthomyiidae flies, Xylorctidae moths, Thripidae thrips, Lygaeidae ground bugs, Staphylinidae rove beetles and Acrididae grasshoppers, who were common representatives of observed feeding guilds. Joshi et al (2004) suggested that guilds composition should not be homogenised for fair assessment, because arthropods differ in food source or host preference, plant part selection and mode of feeding. The vegetation type provided a variety of food sources for feeding niches of phloem sap and xylem solution feeding herbivores of Hemiptera (Cicadellidae and Lygaeidae), growth buds and leaves (foliage) cutter-chewing Orthoptera (Acrididae, Gryllidae, Chrysomelidae), as well as tissue scrapers, pollen and petal feeders (Thripidae and Scarabaeidae) could be driven by available plant life-forms, and while predatory insect were active hunters Staphylinidae of (Coleoptera) and passive hunters of Lycosidae (Araneae) (Tscharntke and Greiler, 1995; Haddad et al., 2015; Picker et al., 2019). Furthermore, dominance by Entomobryidae was also observed in montane grassland from Central Argentina (Cagnolo et al., 2002) which could have been due to its relatively higher reproductive rates and clustering sub-social behaviour (Cipola et al., 2020), and also reported to be used as a bioindicator for natural grass fields because of its sensitivity to disturbance and change in environmental change (Sautter et al., 1999; Sabais et al., 2011; Janion-Scheepers et al., 2015).

The majority of all sampled sites shared common dominant arthropod and plant families. Therefore, we recommend that coherent dominance by floristic community structures of *Themeda triandra* (Poaceae) grass and arthropod community assemblages of *Seira* spp. (*Lepidocyrtinus* spp.: Entomobryidae) springtails, as well as their change or fluctuation in composition should be used as a bioindicator taxa for bioassessment and biomonitoring programmes for the Northern Drakensberg Highlands vegetation type.

All evaluated environmental disturbances that included both qualitative and quantitative variables (pollution, tourism activity, foot paths, building constructions, alien invasion, plant encroachment and soil erosion) had no significant influence on arthropod and plant community structures as well as measured species surrogates. The Northern Drakensberg Highlands vegetation type section has been protected since GGHNPark was declared in 1963 to conserve highland grasslands, and it is classified within the limited access and human activity zones in the park (SANParks, 2013), therefore less impact by environmental disturbance has been observed. However, soil erosion indicated some level of variance between sampled sites, and was used as a compare measure against species surrogates, in that montane highland grasslands are naturally prone to some degree of soil erosion and plant communities are adapted to in the region (Mucina and Rutherford, 2006; Carbutt et al., 2011).

Using both arthropods and plants as units of measure for the Biotope Quality Index indicated higher correlation response with soil erosion at the sites than the estimated species richness, Simpson-Yule diversity and Shannon-Wiener function for grassland bioregion of the highlands. The Tossline site had the highest *BQI*, estimated species richness, Simpson-Yule diversity and Shannon-Wiener function values for both arthropods and plants. Potential vegetation type common species and taxa community and assemblage that were above mean abundance observed per site included Poaceae, Asteraceae, Thymelaeaceae, Oxalidaceae, Lobeliaceae, Rosaceae, Proteaceae, and Orchidaceae for plants as well as Entomobryidae, Cicadellidae, Thripidae, Anthomyiidae, Lycaenidae, Xylorctidae, Formicidae and Lycosidae for arthropods respectively, could be used as main representative taxa for biotope quality to monitor changes at the Northern Drakensberg Highlands grassland over time, as their evolutionary relationships are regarded as indigenous fauna and flora that is sensitive to indicate environmental disturbance in the area. Plant diversity increase creates more options for some special and general herbivorous arthropods (Tscharntk and Greiler, 1995), and opportunity for refugia in herb-grass, shrub and tree layers (Botham et al., 2020), influencing increase in arthropod species successional establishments, species richness and diversity (Sabais et al., 2011; Botha et al., 2016). The greater variation in plant and arthropod species representation could be attributed to limited environmental disturbance that ensured good biotope quality conditions and conservation status for better functioning of the highland grassland vegetation type.

The majority of sampled sites had a higher composition of arthropod rarefaction, with most morphospecies occurring in single and double counts, which could be an adaptation to a homogeneous highlands grassland strategy for arthropods to sustain a population in a unique hostile vegetation unit. The homogeneity of Poaceae-Asteraceae floristic composition strata and ecological conditions of the highland vegetation type could influence the dominance of grass-herb layer to create conditions that pose limitations of resources (McIntyre and Lavorel, 1994; Borer, et al., 2012; Prather et al., 2020) and general restrictions for other arthropods with variation of abundance count strategies to establish or change to suit the montane environment (Cagnolo et al., 2002). Naturally complimenting taxa units indicated that the estimated species richness of arthropods was higher in sites that were having a higher plant diversity and variation in life-form layers, for example, at the Diepkloof and Tossline sites, and relatively lower estimated species richness of arthropods at the Oribi Loop and Blesbok Loop sites. Some spatial differences for species surrogates observed between sites might have been as a result of microclimate variation brought by daily fluctuations (Tscharntk and Greiler, 1995), seasonal variation (Barnett and Facey, 2016), large herbivore impact (van Klink et al., 2014; Ferreira et al., 2020) and grassland fire intervals (Little et al., 2013) in the highlands grassland.

In general, the Biotope Conservation Status of the vegetation type indicated good standings, in that the BCS score ranged from fairly to good conserved biotope status, suggesting that ecological integrity cues (includes community assemblages, species richness and diversity) is still well conserved for arthropods and plants at the Northern Drakensberg Highlands grassland in the GGHNPark.

Conclusion

The Northern Drakensberg Highlands vegetation type was characterised by *Themeda triandra – Eragrostis – Aristida - Lasiosiphon burchellii* plant communities, and arthropod assemblages of representatives from Entomobryidae, Cicadellidae, Thripidae, Anthomyiidae, Lycaenidae, Xylorctidae, Formicidae and Lycosidae families. The majority of observed arthropods have adopted abundance levels below the mean population which could be due to the homogeneity of vegetation in the region. This highlands grassland vegetation type was attributed to low-to-limited environmental disturbances. The Biotope Quality Index measure was able to indicate its potential as a species surrogate for bioassessments and biomonitoring, in that higher *BQI* values were observed in sites with greater variation in plants life-forms and arthropods feeding group, also in agreement with the estimated species richness and diversity measures.

We recommend the use of the Biotope Quality Index and Biotope Conservation Status for rapid bioassessments and biomonitoring in protected areas, rangeland and natural highland grasslands.

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CHAPTER 7 : General Conclusion

The development and use of biotic indices has resulted in the establishment of better environmental conservation, ecosystem rehabilitation and restoration programs. The complexity of biological and ecological systems as well as the influence of disturbance (natural or anthropogenic) require a holistic approach which integrates the use of biological components (species, population, community and biotope level) that could reflect historic, current and long-term change in environmental conditions of terrestrial biomes and bioregions (Kerans and Karr, 1994; Fierro et al., 2018). The application of the Biotope Quality Index (*BQI*) developed by Bredenhand (2014) and the novel Biotope Conservation Status (*BCS*) (in this study) proved their potential as a species surrogate and status classification measure that can be utilised during bioassessments and biomonitoring for environmental conservation, ecosystem rehabilitation and restoration programs in protected areas as well as extended temperate highland grassland bioregions.

The review of the biotic indices in Chapter 2 highlighted that the aquatic system domain still receives more research attention in developing indices than others, even though there has been great progress for terrestrial and atmospheric systems. Public health concerns during the farming and industrialisation era in the 1900s raised concerns about environmental conditions due to natural water system pollution and contamination in most industrialised countries (Hilsenhoff, 1987). The regions showing the most activity in the development and usage of biotic indices were in Europe, North America and Australasia, with Asia, South America and Africa still progressing in establishing and using bioassessment and biomonitoring protocols. The challenges in most developing and under developed countries are access to scientific resources and professional ecologist capacity to establish working programs to develop bioassessment and biomonitoring protocols. However, already existing biotic indices also had technical challenges in manner they were developed i.e., index description, naming, arithmetic formulation and amendments, that then created confusion in usage, bias in measured variables, biotic response specificity to region, ecosystem or variable, as well as some ambiguity in what indices are developed for and its application (Green and Chapman, 2011). However, with such challenges, the establishment of centralised regional reference data systems that will allow collaborative working groups and multi-dimensional approach to biotic indices application was proposed. Furthermore, it was suggested that an establishment of standard guidelines with clearly defined protocols will assist processes to develop, establish and usage of biotic indices, with some suggested examples and recommendations. Again, the establishment of an International Code of Biotic Indices Nomenclature (ICBIN) was proposed as a professional entity to oversee development and establishment processes, as well as assist in tracking changes and modifications of biotic indices.

The thesis was designed to adopt Biotope Quality Index (*BQI*) for vegetation ecology analysis, and to demonstrate different analysis as well as application methods by using the use of arthropods and plants. In Chapter 3, the study aimed to adopt *BQI* as a vegetation ecology species surrogate. This study tested three commonly used plant abundance-cover scales namely Braun-Blanquet, Domin and percentage cover, to show arithmetic compatibility for *BQI* application as a species surrogate tool. According to Archibold (1995) the majority of vegetation biomes and bioregions are named and described according to a dominant constant floristic compositional feature. So, it further tested and demonstrated by correlating significant disturbances with specific or representative *BQI* dominant plant taxa, which included 1. dicots (BQI_{dic}); 2. monocots (BQI_{mon}); 3. graminoids (BQI_{gmd}), 4. forbs (BQI_{for}), 5. grasses Poaceae (BQI_{poa}) and with all species data (BQI_{veg}). Our results indicated that BQI_{veg} would be a consensus tool, since *BQI* relies on comparative variance between observed plant species and sampled sites. The community ecology analysis indicated that the floristic composition of the Mesic Highveld Grassland was mainly represented by Poaceae, Asteraceae and Cyperaceae, with the Eastern Free State Sandy Grassland vegetation type characterised by communities of *Themeda triandra* - *Eragrostis chloromelas* and *Andropogon appendiculatus* – *Eragrostis capensis* species, with soil erosion highest at the Honing Kloof and Korfshoek sites as well as the encroachment by *Seriphium plumosum* shrub which was higher at the Korfshoek and Twijhoek sites. Then, the Basotho Montane Shrubland vegetation type was characterised by communities of *Themeda triandra* - *Eragrostis chloromelas* and *Aristida diffusa* - *Pteridium aquilinum* species with two prominent invaders namely *Cynodon dactylon* grass at the QQ Mountain site and *Acacia mearnsii* trees at the Avondrust site, while soil erosion was uniformly lower in all the Gm5 sampled sites.

According to Mucina and Rutherford (2006), the main floristic composition feature of the Mesic Highveld Grassland is Poaceae grasses and Asteraceae herbs-woody species that change in assemblages along altitudinal gradient and micro-habitat. *Cynodon dactylon, Seriphium plumosum* and *Acacia mearnsii* are among common invasive plants in the Free State Province grassland biome, which have a negative impact on native species richness, community composition, biodiversity and ecosystem functioning (Snyman, 2009; Moyo and Fatunbi, 2010; Fonseca et al., 2013). The sites that most greatly affected by soil erosion and plant invasion were among previously agricultural farm fields and close to human settlement in the Golden Gate Highlands National Park (GGHNPark) (SANParks, 2013; Rademeyer and van Zyl, 2014). The intense agricultural practices are known to cause great loss in plant biodiversity and cover (Birkhofer et al., 2017), as well as it would take a longer period to restore the land to its native grassland floristic structure (O'Connor and Bredenkamp, 1997). The Biotope Conservation Status (*BCS*) measure was able to indicate sites that are in a good conservation state the Silasberg (Basotho Montane Shrubland) and the Welverdiend (Eastern Free State Sandy Grassland) that could be used as reference sites, as well as those with a disturbed conservation state, i.e., Avondrust (Basotho Montane Shrubland) and Honing Kloof (Eastern Free State Sandy Grassland) that require GGHNPark to mitigate with strategies to address plant invasion and soil erosion challenges.

Furthermore, Chapter 4 was dedicated to demonstrate how the *BQI* can be used for biotic assemblages and community ecology analysis for the Eastern Free State Sandy Grassland vegetation type. The observed common arthropod taxa included Insecta, Entognatha, Arachnida, Diplopoda, Chilopoda, Acari and Malacostraca in all sampled sites. The most dominant arthropod families were Entomobryidae and Formicidae, which have been reported by previous studies to be common in grasslands (New, 2019), and with a potential to be used as bioindicators of ecological change (Chauvat et al., 2007; Simons and Weisser, 2017). The Honing Kloof site had a high degree of soil erosion with low species abundance, richness and diversity. The Biotope Quality Index was able to indicate sites where significant shrub encroachment and soil erosion impact was observed. The Korfshoek site also had a high degree of soil erosion while experiencing plant encroachment of *Seriphium plumosum* and *Tagetes minuta*, which was attributed to low species abundance, richness, diversity and biotope quality. The results from historical records indicated that some sections of GGHNPark were used for agriculture farming (Rademeyer and van Zyl, 2014), and plant encroachers of *S. plumosum* and *T. minuta* are common invaders of abandoned crop fields with drier soil (Avenant, 2015). The Honing Kloof and Korfshoek sites were considered to be in a poor biotope quality condition with low *BQI*, while the Welverdiend site with low disturbance had a higher *BQI* and exhibited good ecological conditions. The Biotope Quality Index demonstrated its potential as a species surrogate and ArcGIS geospatial feature analysis parameter for the bioassessment at the Eastern Free State Sandy Grassland.

Similarly, the *BQI* was tested to demonstrate that it should be used for biotic assemblages and community ecology analysis for the Basotho Montane Shrubland vegetation type in Chapter 5. The results indicated that Insecta, Entognatha and Arachnida were the common arthropod groups represented by families that included Entomobryidae, Formicidae, Phalangiidae, Thripidae, Cicadellidae, Aphididae, Tetranychidae, Tenebrionidae, Chrysomelidae and Anthomyiidae. As indicated above, Entomobryidae and Formicidae, are most common in the temperate grassland biomes (New, 2019). The Avondrust site had a higher plant invasion of *Acacia mearnsii* trees and low arthropod species abundances, richness and diversity. Most *Acacia* species in South Africa are known to be aggressive invaders that compete for resources (water and space) with native flora, and also alter soil structure and nutrient flow (Moyo and Fatunbi, 2010). On the contrary, the QQ Mountain site had *Cynodon dactylon* grass invasion but with a higher arthropod species abundance, richness and diversity. Many studies have reported opposing results on invasive species paradox, in that some are recorded in sites with low or reduced species richness and diversity (Simberloff et al., 2003), and while others are found in sites with high or increased species richness and diversity (Hejda et al., 2009). The Biotope Quality Index was able to show the significant impact of invasion while other species surrogates were with conflicting responses at the QQ Mountain site. Sites with low degree of disturbance namely the Diepkloof and Silasberg sites had a high *BQI* with good biotope conditions.

Chapter 6 demonstrates the use of the *BQI* and *BCS* for rapid bioassessment using arthropod assemblages and floristic composition clusters to evaluate the Northern Drakensberg Highlands Grassland vegetation type. The floristic structure was composed of Poaceae, Asteraceae, Thymelaeaceae, Oxalidaceae, Lobeliaceae, Rosaceae, Proteaceae, and Orchidaceae. The floristic clustering indicated two main communities that were dominated by *Themeda triandra* grass, i.e., *Themeda triandra* – *Eragrostis chloromelas* and *Themeda triandra* – *Aristida diffusa*. Most highland grasslands are characterised by dominance of *Themeda triandra* (Poaceae) grass and lower herbaceous plants of Asteraceae (O'Connor and Bredenkamp, 1997; Mucina and Rutherford, 2006). Arthropod assemblages were represented mostly by herbivorous, omnivorous and predatory taxa from Entomobryidae, Cicadellidae, Thripidae, Anthomyiidae, Lycaenidae, Xylorctidae, Formicidae and Lycosidae families. Arthropods were represented by five morphospecies communities that included *Lepidocyrtinus* sp.1 – Agalliinae sp.3, Thripidae sp.2 – *Capbrya* sp.1, Anthomyiidae sp.2 – Formicinae sp.3, Encyrtidae sp.2 – Rutelinae sp.1, and Thripidae sp.3 – Chrysomelidae sp.6 community clusters. The Biotope Quality Index values ranged fairly higher for the vegetation type, with no clear significant influence of measured environmental (anthropogenic and ecological) disturbances. The Biotope Conservation Status scores classified the Diepkloof and Tossline sites as good conservation status, while the Blesbok Loop and Oribi Loop sites had a fair conservation status for the Northern Drakensberg Highlands Grassland using both arthropod and plant communities.

In summary, the state of the Afromontane ecoregion in the Golden Gate Highlands National Park (GGHNPark) is in two situations, the old lower Qwaqwa National Park (Mesic Highveld Grassland) section has most of the sites with high disturbance due to soil erosion, alien plant invasion and drier woody shrub encroachment. The results were correlated with intense historic agricultural farming that was active until the 1990s before the section was protected. It is recommended that the park should start incorporating other ecosystem rehabilitation and restoration strategies rather than relying only on allowing the disturbed sections to restore naturally, which might take longer. The vegetation types that are within the higher original GGHNPark section (Drakensberg Grassland) are still in good conserved status (and status) with no significant disturbance effects, since the section was protected from 1963 limiting human access to reduce negative impacts.

It was also recommended that the park should include the Biotope Quality Index and Biotope Conservation Status measure for conservation management programmes to assist in monitoring progress of rehabilitation and restoration of ecosystems in the park, and to establish long-term data that can be used to evaluate the effects of climate change.

There are several points of interest that should be noted when using the Biotope Quality Index as a species surrogate and GIS geospatial feature analysis parameter;

- Select biotic communities that are evolutionary related and who are measured with the same scale, i.e., same animal class with similar life behavioural history and reproductive strategies.
- Use only one data abundance (or abundance-cover) scale in the same *BQI* equation.
- Transform data using log X_i and not (1 + log X_i), so as to highlight relevance of presence and absence taxa as an indication or response to disturbance.
- Make a reference baseline data inventory when adopting the *BQI* for other regions, and compare trends for that specific biotope or ecoregion.
- Try to include sampling sites that also represent variation of micro-habitat or biotope conditions (disturbed to pristine/natural), so as to have understanding on dynamics of ecoregion.
- Do not aggregate *BQI* of the vegetation type or biotope units for a representative condition state, to avoid bias and eclipsing of variables impacts and species representation among sites.

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SUPPLEMENTARY MATERIAL

Chapter 2

Table 0.1: Commonly used biotic assessments and indices chart for environmental and ecological biomonitoring

Indices	Year	Country_State	Continent	Domain	Ecosystem	Equation
TBI_WAr_ENG	1964	England	Europe	Water	River	Arithmatic_additive
BI_War_FRA	1968	France	Europe	Water	River	Arithmatic_additive
CBI_WAr_ENG	1968	England	Europe	Water	River	Arithmatic_additive
BISAR_WAr_RSA	1972	South_Africa	Africa	Water	River	Arithmatic_additive
HIS_WAr_USA	1975	Montana	USA	Water	River	Arithmatic_additive
ITI_WAr_USA	1978	Carlifornia	USA	Water	Estuary	Arithmatic_additive
TBI_WAr_FRA	1978	France	Europe	Water	River	Arithmatic_additive
EBI_WAr_BLG	1979	Belgium	Europe	Water	River	Arithmatic_additive
IBI_WAr_USA	1981	Great_Lakes	USA	Water	Stream	Arithmatic_additive
BBI_WAr_BLG	1983	Belgium	Europe	Water	River	Arithmatic_additive
BrBMWP_WAr_ENG	1983	England	Europe	Water	River	Arithmatic_additive
BQI_WAr_IRL	1985	Ireland	Europe	Water	River	Arithmatic_additive
ISO_WArM_USA	1985		USA	Water	Estuary	Arithmatic_multiplicative
WBI_WAr_USA	1988	Wisconsins	USA	Water	River	Arithmatic_multiplicative
EPT_WAv_USA	1988	North_Carolina	USA	Water	Stream	Arithmatic_average_method
ICI__WAr_USA	1989	Ohio	USA	Water	River	Arithmatic_hybrid_method
MI_TAr_FRA	1990	France	Europe	Terrestrial	Soil	Arithmatic_additive
SPSI__WAr_FRA	1991	France	Europe	Water	River	Arithmatic_hybrid_method
SQI_TAr_USA	1991	Michigan	USA	Terrestrial	Soil	Arithmatic_multiplicative
Barbour et al (1992)	1992			Water		Arithmatic_additive
ICS_WArM_USA	1992	Virginia	USA	Water	River	Arithmatic_multiplicative
NCBI_WAr_USA	1993	North_Carolina	USA	Water	Stream	Arithmatic_additive
RIVPACS_WHb_ENG	1993	England	Europe	Water	River	Arithmatic_hybrid_method

Biotic Indices table cont…

Indices	Year	Country_State	Continent	Domain	Ecosystem	Equation
SpBMWP_WAr_ESN	1993	Spain	Europe	Water	River	Arithmatic_additive
SQMCI_WArM_NZL	1993	New_Zealand	Australasia	Water	Stream	Arithmatic_additive
MRBI_WHb_AUS	1993	Australia	Australasia	Water		Arithmatic_hybrid_method
EQI_WAr_ENG	1994	England	Europe	Water	River	Arithmatic_additive
IEC_WHb_MEX	1994	Mexico	USA	Water	Estuary	Arithmatic_hybrid_method
PI_TAr_POR						
TARISS_WAr_TAN	2015	Tanzania	Africa	Water	River	Arithmatic_additive
DBI_WAv_RSA	2009	South_Africa	Africa	Water	River	Weighted_avarages
Thompson et al (2017)						
OKASS_WAr_BOT	2009	Botswana	Africa	Water	Wetland	Arithmatic_additive
MBM_WAr_CHI	2016	China	Asia	Water	River	Arithmatic_additive
Shulse et al (2009)						
Perlmutter 2010	2010	Carlifornia	USA	Air	Ambient Air	Arithmatic_additive
SIF_TAr_POT						
A__WHb_VEN						
EQI_WAr_USA	2014		USA	Water	River	Arithmatic_additive
B_IBI_WArM_USA	1994	Tennessee	USA	Water	Stream	Arithmatic_multiplicative
SQI_TAv_NET	1994	Nethalands	Europe	Water	Soil	Weighted_avarages
IBI_WAr_MEX	1995	Mexico	USA	Water	Stream	Arithmatic_additive
SIGNAL_WHb_AUS	1995	Australia	Australasia	Water	River	Arithmatic_hybrid_method
Fore et al (1996)				Water		
BrBMWQ_WHb_ESP	1996	Spain	Europe	Water	River	Arithmatic_hybrid_method
ASPT_WAv_ENG	1983	England	Europe	Water	River	Weighted_avarages
BTSI_WHb_USA	1996	Mid_Atlantic_Region	USA	Water	Estuary	Arithmatic_hybrid_method
BDI_WAr_FRA	1996	France	Europe	Water	Lake	Arithmatic_additive
BHQ_WAr_SWE	1997	Sweeden	Europe	Water	Estuary	Arithmatic_additive

Biotic Indices table cont…

Indices	Year	Country_State	Continent	Domain	Ecosystem	Equation
EBI_WAr_USA	1997	Massachusetts	USA	Water	Estuary	Arithmatic_additive
Breber (1997)						
Eu_IBI_WAr_USA	1997	Mid_Atlantic_Region	USA	Water	Estuary	Arithmatic_additive
SASS_WAr_RSA	1994	South_Africa	Africa	Water	River	Arithmatic_additive
CM_WHb_ENG	1997	England	Europe	Water	Stream	Arithmatic_additive
BICE_SQT_WAr_USA	1998	Missouri	USA	Water	River	Arithmatic_additive
MMI_WAv_NZL	1998	New_Zealand	Australasia	Water	Estuary	Weighted_avarages
PAR_WHb_USAL	1999		USA	Water	River	Arithmatic_hybrid_method
VPBI_WArM_USA	1999	Virginia	USA	Water	Estuary	Arithmatic_multiplicative
BCI_WHb_MEX	1999	Mexico	USA	Water		Arithmatic_hybrid_method
IEI_WHb_USA	2000	Michigan	USA	Water	Wetland	Arithmatic_hybrid_method
AZTI_MBI_WAr_FRA	2000	France	Europe	Water	Estuary	Arithmatic_additive
CPMI_WHb_USA	2000	Mid_Atlantic_Region	USA	Water	Stream	Arithmatic_hybrid_method
D_MI_WArM_GER	2000	Germany	Europe	Water	Lake	Arithmatic_multiplicative
EQUATION_WHb_POR	2000	Portugal	Europe	Water	Estuary	Arithmatic_hybrid_method
LMII_WHb_USA	2000	New_Jersey	Europe	Water	Lake	Arithmatic_hybrid_method
IBGN_WAr_BLG	2000	Belgium	Europe	Water		Arithmatic_additive
RIVPACS_WHb_ENG	2000	England	Europe	Water	River	Arithmatic_hybrid_method
AusRivAS_WHb_AUS	2000	Australia	Australasia	Water	River	Arithmatic_hybrid_method
MuLFA_WHb_VIE	2000	Vienna	USA	Water	River	Arithmatic_hybrid_method
ITC_WHb_RUS	2000	Russia	Europe	Water	River	
Wefering et al (2000)	2000				River	
BEAST_WHb_USA	2000		USA	Water		
MAPI_WArM_CHI	2001	China	Asia	Water	Estuary	Arithmatic_additive
BRI_WAv_USA	2001	Carlifornia	USA	Water	Estuary	Weighted_avarages
OEI_WHb_CAN	2001	Canada	USA	Water	Estuary	Arithmatic_hybrid_method

Indices	Year	Country_State	Continent	Domain	Ecosystem	Equation
LBII_WAr_USA	2001	Great_Lakes	USA	Water	Lake	Arithmatic_additive
LEHA_WHb_CHI	2001	China	Asia	Water	Lake	Arithmatic_hybrid_method
IBIR_WHb_USA	2001	Illinois	USA	Water	River	Arithmatic_hybrid_method
ICBEMP_WHb_USA	2001	Colombia	Sourth_America	Water	All	Arithmatic_hybrid_method
VPBI_WArM_USA	2001	Virginia	USA	Water	Estuary	Arithmatic_multiplicative
QBSc_TAr_ITA	2001	Italy	Europe	Terrestrial	Soil	Arithmatic_additive
EQI_WAr_USA	2001		USA	Terrestrial	Soil	Arithmatic_additive
SWAMPS_WAv_AUS	2002	Australia	Australasia	Water	Wetland	Weighted_avarages
BII_WAr_USA	2002	Oregon	USA	Water	River	Arithmatic_additive
EBI_DIPPER_WHb_ITA	2002	Italy	Europe	Water	Stream	Arithmatic_hybrid_method
MAIA_WHb_USA	2002	Mid_Atlantic_Region	USA	Water	Estuary	Arithmatic_hybrid_method
FBI_WAv_FRA	2002	France	Europe	Water	River	Weighted_avarages
ISI_WHb_NOR	2002	Norway	Europe	Water	River	Arithmatic_hybrid_method
EcoQI_WAv_ITA	2003	Italy	Europe	Water	Estuary	Weighted_avarages
Col_BMWP_WHb_USA	2003	Colombia	Sourth_America	Water	River	Arithmatic_hybrid_method
IEI_WArM_USA	2003	Mid_Atlantic_Region	USA	Water	Estuary	Arithmatic_multiplicative
NASS_WAr_NAM	2004	Namibia	Africa	Water	River	Arithmatic_additive
BQI_WArM_SWE	2004	Sweeden	Europe	Water	River	Arithmatic_multiplicative
SMAF_TAr_USA	2004	Washington	USA	Terrestrial	Soil	Arithmatic_additive
QBSar_TAr_ITA	2005	Italy	Europe	Terrestrial	Soil	Arithmatic_additive
QUALES_WAr_ESP	2005	Spain	Europe	Water	Wetland	Arithmatic_additive
ANNA_WHb_AUS	2004	Australia	Australasia	Water	River	Arithmatic_hybrid_method
FFG_WHb_BRA	1996	Brazil	Sourth_America	Water	River	Arithmatic_hybrid_method
NBI_WAv_USA	2007	New_York	USA	Water	River	Weighted_avarages
ElmPT_WAv_ARG	2007	Argentina	Sourth_America	Water	Stream	Weighted_avarages
BOPA_WAv_FRA	2007	France	Europe	Water	Lake	Weighted_avarages
Hasselbach et al 2005	2005	Alaska	USA	Air	Ambient Air	Arithmatic_hybrid_method
US EPA 200	2000	Oregon	USA	Air	Ambient Air	Arithmatic_hybrid_method

Table 0.2: Plant invasion measure chart for Mesic Highveld Grassland in GGHNPark

	Eastern Free State Sandy Grassland				Basotho Montane Shrubland			
	Korf _Gm4	Honi _Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila _Gm5
plant_encroach. (%)	15	3	8	6	11	6	6	2
alien_plant (%)	0	0	0	0	22	12	0	0
arthropod_invas. (%)	0	0	0	0	0	0	0	0
mean (%)	**5**	**1**	**2.6666667**	**2**	**11**	**6**	**2**	**0.666667**

Table 0.3: Soil erosion variables used for measure RUSLE for Mesic Highveld Grassland in GGHPark

	Eastern Free State Sandy Grassland				Basotho Montane Shrubland			
	Korf _Gm4	Honi _Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila _Gm5
type	rill_sheet	scalding	rill_sheet	water	rill_sheet	rill_sheet	rill_sheet	rill_sheet
top soil_silt	6	6	4	3	4	8	8	5
open ground	40	85	35	12	10	35	25	20
moisture_level	6	3	16	20	35	15	20	20
termite_mounts	3	2	3	0	0	0	0	0
boulders	0	0	0	0	8	2	12	6
footpath	few	few	few	none	few	few	few	few
LogValue								
top soil_silt	0.77815125	0.77815125	0.602059991	0.477121255	0.60205999	0.903089987	0.903089987	0.698970004
open ground	1.602059991	1.92941893	1.544068044	1.079181246	1	1.544068044	1.397940009	1.301029996
moisture_level	0.77815125	0.47712125	1.204119983	1.301029996	1.54406804	1.176091259	1.301029996	1.301029996
termite_mounts	0.477121255	0.30103	0.477121255	0	0	0	0	0
boulders	0	0	0	0	0.90308999	0.301029996	1.079181246	0.77815125
SUM	**3.635483747**	**3.48572143**	**3.827369273**	**2.857332496**	**4.04921802**	**3.924279286**	**4.681241237**	**4.079181246**
Sdev	**0.583509047**	**0.74437721**	**0.61123254**	**0.602512677**	**0.56633925**	**0.631176238**	**0.557744363**	**0.536538118**
erosion value	**2.121337655**	**2.5946916**	**2.339412643**	**1.721579052**	**2.29323108**	**2.476911838**	**2.610935914**	**2.188636228**

Table 0.4: Qualitative and quantitative data evaluating anthropogenic disturbance for Mesic Grassland in GGHNPark

	Eastern Free State Sandy Grassland				Basotho Montana Shrubland			
	Korf _Gm4	Honi _Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila _Gm5
farming_history	yes	yes	yes	no	no	no	no	no
housing_structures	no	yes	no	no	no	yes	no	no
conservation	no	no	no	no	no	no	no	no
polution	little	little	none	none	none	none	none	none
wildlife_activity	more	more	more	little	more	moderate	moderate	few

Table 0.5: Plant invasion measure chart for Drakensberg Grassland in GGHNPark

	Northern Drakensberg Highlands			
	Bles_Gd5	Orib _Gd5	Diep_Gd5	Toss_Gd5
encroachment (%)	0	0	0	0
alien_plant (%)	0	0	0	0
alien_arthrop. (%)	0	0	0	0
mean (%)	**0**	**0**	**0**	**0**

Table 0.6: Soil erosion variables used for measure RUSLE for Drakensberg Grassland in GGHPark

	Northern Drakensberg Highlands			
	Bles_Gd5	Orib _Gd5	Diep_Gd5	Toss_Gd5
type	water	water	water	water
top soil_silt	2	3	2	3
open ground	6	6	6	6
moisture_level	28	30	29	28
termite_mounts	0	0	0	0
boulders	0	0	0	0
footpath	none	none	none	none
LogValue				
top soil_silt	0.301029996	0.477121255	0.301029996	0.477121255

Table 0.7: Qualitative and quantitative data evaluating anthropogenic disturbance for Drakensberg Grassland in GGHNPark

	Northern Drakensberg Highlands			
	Bles_Gd5	Orib _Gd5	Diep_Gd5	Toss_Gd5
farming_history	no	no	no	no
housing_structures	no	no	no	no
conservation	yes	yes	yes	yes
polution	none	none	none	none
wildlife_activity	few	few	few	few

R-Studio step wise code for Biotope Quality Index

The Biotope Quality Index was developed using arthropod assemblage (species occurrence counts) as a measure of mean proportion and standard deviation degree, to indicate the impact of the disturbance level and assign a conservation status to a biotope (Bredenhand, 2014). The Biotope Quality Index expression model is among statistical formulations that assess the relationship between variables or patterns within a variable of a sample unit of the observed population using standard scores, e.g., as in Pearson's Correlation Coefficient which is used to measure association between two variables, and for normalising data, which adjusts data points from the sample or population mean threshold. In this study, standard scores are referred to as the number of standard deviations by which a random data point value (raw score) is above or below the mean value of an observed sample as a measure of biotope quality (Bredenhand, 2014; Glantz et al., 2016).

The expression of the Biotope Quality Index (below) considers the evolutionary history of the integrity unit, and assumes that any biotope in a naturally good conserved state can act to sustain the arthropod population to optimum species numbers as well as their ecological processes.

$$\text{Biotope Quality Index } (BQI) = \sum_{i=1}^{n} \left(\frac{\log x_{ij} - \log \overline{x}_i}{\log \sigma} \right)$$

$$\text{if } \log \sigma > 0 \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(Bredenhand, 2014)}$$

The represented symbols include, abundance of *i*th morphospecies (x_i), found in the *j*th sample (x_j), estimated mean ($\overline{x}_i$), and population standard deviation (σ).

The BQI calculation has four main steps which include;

- log transformation of relative species abundance (or abundance-cover)
- considering data entries equal to or above relative species abundance mean only (selecting biotope representative species)
- compute the difference between relative abundance and mean, then divide by standard deviation for each species (or morphospecies)
- aggregate the sum of all species per site into BQI value, then assign a grade biotope status category (poor, moderate, good and excellent conditions)

Code Chunk for Biotope Quality Index (*BQI*)

1: Prepare data in a format that R studio will recognize e.g. as .txt or .csv files

- Species abundance (abundance-cover/density cover) matrix with sampled sites arranged in columns and species in rows
- Import the species abundance file as a data frame into R-studio

```
data <- read.csv(file.choose() , sep = ';', header = TRUE, row.names = 1)
str(data)

## 'data.frame':    5 obs. of  5 variables:
##  $ site_A: int  0 4 2 1 1753
##  $ site_B: int  23 3 0 1 0
##  $ site_C: int  56 2 8 1 2367
##  $ site_D: int  5 8 0 2 1876
##  $ site_E: int  109 5 0 0 2354
```

2: log transform morphospecies abundances

```
data_log <- log10(data)
```

3: replace negative infinite values with 0

```
data_log1 <- do.call(data.frame, lapply(data_log, function(x) replace(x, i
s.infinite(x), 0)))
data_log1

##     site_A    site_B   site_C   site_D   site_E
## 1 0.000000 1.3617278 1.748188 0.698970 2.037426
## 2 0.602060 0.4771213 0.301030 0.903090 0.698970
## 3 0.301030 0.0000000 0.903090 0.000000 0.000000
## 4 0.000000 0.0000000 0.000000 0.301030 0.000000
## 5 3.243782 0.0000000 3.374198 3.273233 3.371806
```

4: calculate and add row mean and standard deviation onto data frame

```
data_log1$means <- apply(data_log1, 1, mean) #add mean column to data fram
e
data_log1$stdevs <- apply(data_log1 [1:5], 1, sd) #add standard deviation
column to data frame, modify depending on number of data rows and column
```

5: equate abundance from mean divided by standard deviation of each point by row

```
data_bqi <- do.call(data.frame, lapply(data_log1, function(x) x - data_log
1$means))
data_bqi

##          site_A     site_B     site_C     site_D     site_E means      s
tdevs
## 1 -1.169262473  0.1924654  0.5789256 -0.4702925  0.8681640     0 -0.345
51248
## 2  0.005605745 -0.1193330 -0.2954243  0.3066357  0.1025158     0 -0.369
47284
## 3  0.060205999 -0.2408240  0.6622660 -0.2408240 -0.2408240     0  0.151
67110
## 4 -0.060205999 -0.0602060 -0.0602060  0.2408240 -0.0602060     0  0.074
41871
## 5  0.591178023 -2.6526039  0.7215944  0.6206289  0.7192026     0 -1.168
61182

data_bqi1 <- do.call(data.frame, lapply(data_bqi, function(x) x / data_log
1$stdevs))
data_bqi1

##         site_A     site_B     site_C     site_D     site_E means      std
evs
## 1 -1.41943852  0.2336454  0.7027928 -0.5709165  1.0539169     0 -0.4194
385
## 2  0.02469693 -0.5257391 -1.3015350  1.3509289  0.4516483     0 -1.6277
670
## 3  0.15339300 -0.6135720  1.6873230 -0.6135720 -0.6135720     0  0.3864
280
## 4 -0.44721360 -0.4472136 -0.4472136  1.7888544 -0.4472136     0  0.5527
864
## 5  0.39837007 -1.7874785  0.4862522  0.4182158  0.4846404     0 -0.7874
785
```

6:convert negative into zero(0)

```
bqiP <- data_bqi1
bqiP[bqiP < 0] <- 0
bqiP

##        site_A    site_B    site_C    site_D    site_E means    stdevs
## 1 0.00000000 0.2336454 0.7027928 0.0000000 1.0539169     0 0.0000000
## 2 0.02469693 0.0000000 0.0000000 1.3509289 0.4516483     0 0.0000000
## 3 0.15339300 0.0000000 1.6873230 0.0000000 0.0000000     0 0.3864280
## 4 0.00000000 0.0000000 0.0000000 1.7888544 0.0000000     0 0.5527864
## 5 0.39837007 0.0000000 0.4862522 0.4182158 0.4846404     0 0.0000000
```

7: create a new data frame excluding

```
bqiMorph <- bqiP[,c(1:5)] #modify depending on number of data rows and col
umn
bqiMorph
```

```
##       site_A    site_B    site_C    site_D    site_E
## 1 0.00000000 0.2336454 0.7027928 0.0000000 1.0539169
## 2 0.02469693 0.0000000 0.0000000 1.3509289 0.4516483
## 3 0.15339300 0.0000000 1.6873230 0.0000000 0.0000000
## 4 0.00000000 0.0000000 0.0000000 1.7888544 0.0000000
## 5 0.39837007 0.0000000 0.4862522 0.4182158 0.4846404
```

8: calculate row/species sums and add BQI column

```
bqiMorph$BQI <- apply(bqiMorph, 1, sum) #add mean column to data frame
bqiMorph
```

```
##       site_A    site_B    site_C    site_D    site_E      BQI
## 1 0.00000000 0.2336454 0.7027928 0.0000000 1.0539169 1.990355
## 2 0.02469693 0.0000000 0.0000000 1.3509289 0.4516483 1.827274
## 3 0.15339300 0.0000000 1.6873230 0.0000000 0.0000000 1.840716
## 4 0.00000000 0.0000000 0.0000000 1.7888544 0.0000000 1.788854
## 5 0.39837007 0.0000000 0.4862522 0.4182158 0.4846404 1.787478
```

```
str(bqiMorph)
```

```
## 'data.frame':    5 obs. of  6 variables:
##  $ site_A: num  0 0.0247 0.1534 0 0.3984
##  $ site_B: num  0.234 0 0 0 0
##  $ site_C: num  0.703 0 1.687 0 0.486
##  $ site_D: num  0 1.351 0 1.789 0.418
##  $ site_E: num  1.054 0.452 0 0 0.485
##  $ BQI   : num  1.99 1.83 1.84 1.79 1.79
```

9: calculate column/site sums and add BQI column

```
bqiMorph[nrow(bqiMorph) + 1, (1:5)] <- apply(bqiMorph, 2, sum) #add mean c
olumn to data frame
```

```
## Warning in matrix(value, n, p): data length [6] is not a sub-multiple o
r
## multiple of the number of columns [5]
```

```
row.names(bqiMorph) [6] <- c('BQI')
str(bqiMorph)
```

```
## 'data.frame':    6 obs. of  6 variables:
##  $ site_A: num  0 0.0247 0.1534 0 0.3984 ...
##  $ site_B: num  0.234 0 0 0 0 ...
##  $ site_C: num  0.703 0 1.687 0 0.486 ...
##  $ site_D: num  0 1.351 0 1.789 0.418 ...
```

```
##  $ site_E: num  1.054 0.452 0 0 0.485 ...
##  $ BQI   : num  1.99 1.83 1.84 1.79 1.79 ...
```

10: export *BQI* table results, can modify table further using other packages

write.table(bqiMorph, file = “BQI_table.txt”, sep = “, row.names = TRUE, col.names = TRUE)

write.csv(bqiMorph, file = “BQI_table.csv”)

write.csv2(bqiMorph, file = “BQI_table.csv”)

11: exporting many tables as data frame list

```
bqilist <- list(data_log1, bqiMorph); names(bqilist) <- c("Abundance_Mean_
Stdev", "BQI_Summary")
bqilist

## $Abundance_Mean_Stdev
##     site_A    site_B   site_C   site_D   site_E     means    stdevs
## 1 0.000000 1.3617278 1.748188 0.698970 2.037426 1.1692625 0.8237500
## 2 0.602060 0.4771213 0.301030 0.903090 0.698970 0.5964542 0.2269814
## 3 0.301030 0.0000000 0.903090 0.000000 0.000000 0.2408240 0.3924951
## 4 0.000000 0.0000000 0.000000 0.301030 0.000000 0.0602060 0.1346247
## 5 3.243782 0.0000000 3.374198 3.273233 3.371806 2.6526039 1.4839921
##
## $BQI_Summary
##         site_A    site_B    site_C    site_D    site_E      BQI
## 1   0.00000000 0.2336454 0.7027928 0.0000000 1.0539169 1.990355
## 2   0.02469693 0.0000000 0.0000000 1.3509289 0.4516483 1.827274
## 3   0.15339300 0.0000000 1.6873230 0.0000000 0.0000000 1.840716
## 4   0.00000000 0.0000000 0.0000000 1.7888544 0.0000000 1.788854
## 5   0.39837007 0.0000000 0.4862522 0.4182158 0.4846404 1.787478
## BQI 0.57646000 0.2336454 2.8763679 3.5579991 1.9902056       NA

sink("BQIsummary_table.csv")
print(bqilist)

## $Abundance_Mean_Stdev
##     site_A    site_B   site_C   site_D   site_E     means    stdevs
## 1 0.000000 1.3617278 1.748188 0.698970 2.037426 1.1692625 0.8237500
## 2 0.602060 0.4771213 0.301030 0.903090 0.698970 0.5964542 0.2269814
## 3 0.301030 0.0000000 0.903090 0.000000 0.000000 0.2408240 0.3924951
## 4 0.000000 0.0000000 0.000000 0.301030 0.000000 0.0602060 0.1346247
## 5 3.243782 0.0000000 3.374198 3.273233 3.371806 2.6526039 1.4839921
##
## $BQI_Summary
##         site_A    site_B    site_C    site_D    site_E      BQI
## 1   0.00000000 0.2336454 0.7027928 0.0000000 1.0539169 1.990355
## 2   0.02469693 0.0000000 0.0000000 1.3509289 0.4516483 1.827274
## 3   0.15339300 0.0000000 1.6873230 0.0000000 0.0000000 1.840716
## 4   0.00000000 0.0000000 0.0000000 1.7888544 0.0000000 1.788854
## 5   0.39837007 0.0000000 0.4862522 0.4182158 0.4846404 1.787478
## BQI 0.57646000 0.2336454 2.8763679 3.5579991 1.9902056       NA
```

```
sink()
```

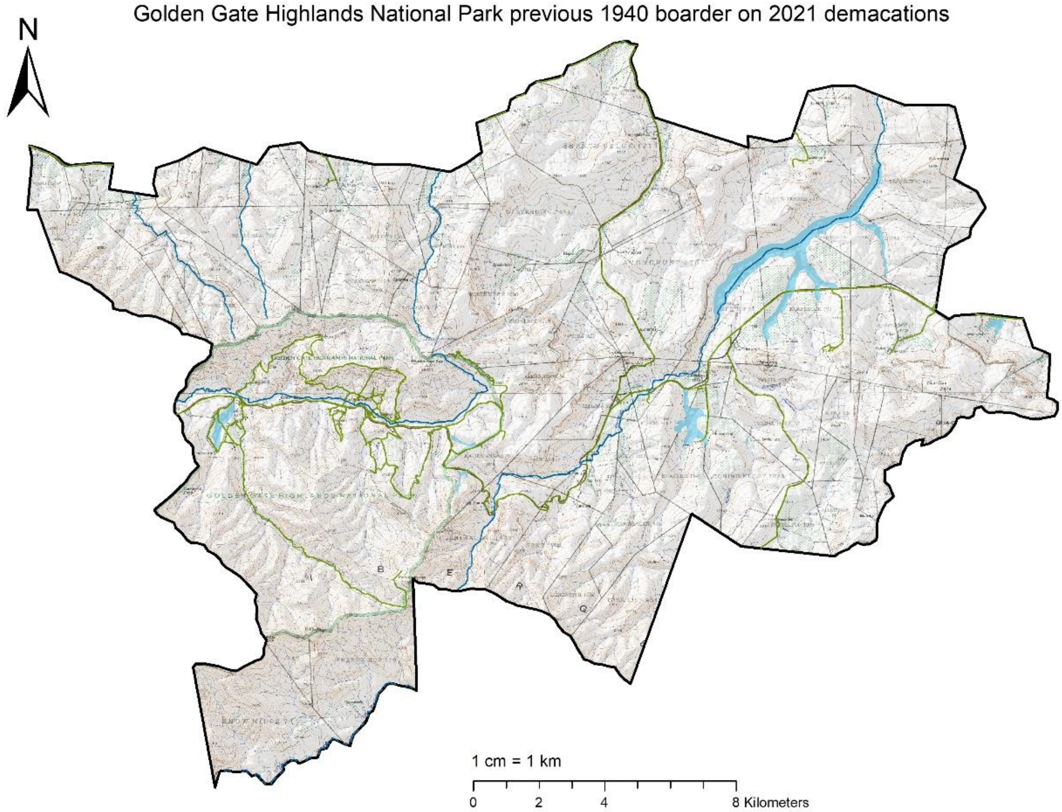
12: alternative exporting many tables as dataframe list option

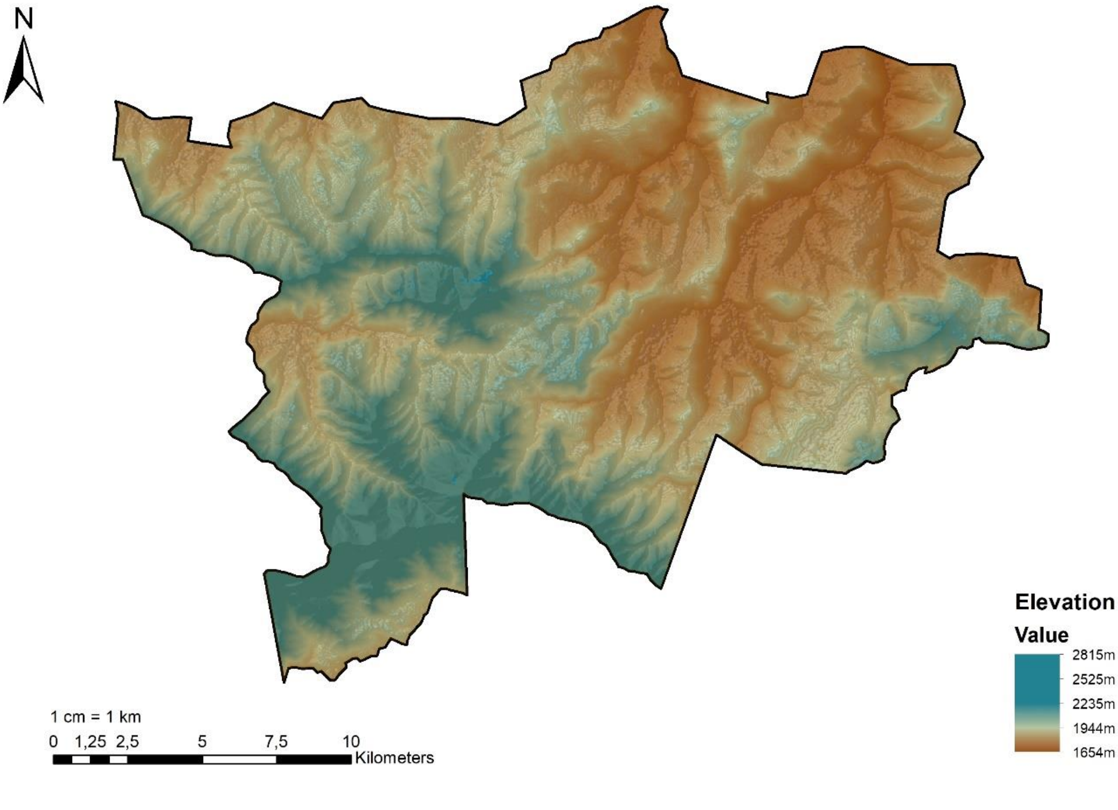
```
sink("BQIsummary_table.txt")
print(bqilist)

## $Abundance_Mean_Stdev
##     site_A    site_B   site_C   site_D   site_E     means    stdevs
## 1 0.000000 1.3617278 1.748188 0.698970 2.037426 1.1692625 0.8237500
## 2 0.602060 0.4771213 0.301030 0.903090 0.698970 0.5964542 0.2269814
## 3 0.301030 0.0000000 0.903090 0.000000 0.000000 0.2408240 0.3924951
## 4 0.000000 0.0000000 0.000000 0.301030 0.000000 0.0602060 0.1346247
## 5 3.243782 0.0000000 3.374198 3.273233 3.371806 2.6526039 1.4839921
##
## $BQI_Summary
##         site_A    site_B    site_C    site_D    site_E      BQI
## 1   0.00000000 0.2336454 0.7027928 0.0000000 1.0539169 1.990355
## 2   0.02469693 0.0000000 0.0000000 1.3509289 0.4516483 1.827274
## 3   0.15339300 0.0000000 1.6873230 0.0000000 0.0000000 1.840716
## 4   0.00000000 0.0000000 0.0000000 1.7888544 0.0000000 1.788854
## 5   0.39837007 0.0000000 0.4862522 0.4182158 0.4846404 1.787478
## BQI 0.57646000 0.2336454 2.8763679 3.5579991 1.9902056       NA

sink()
```

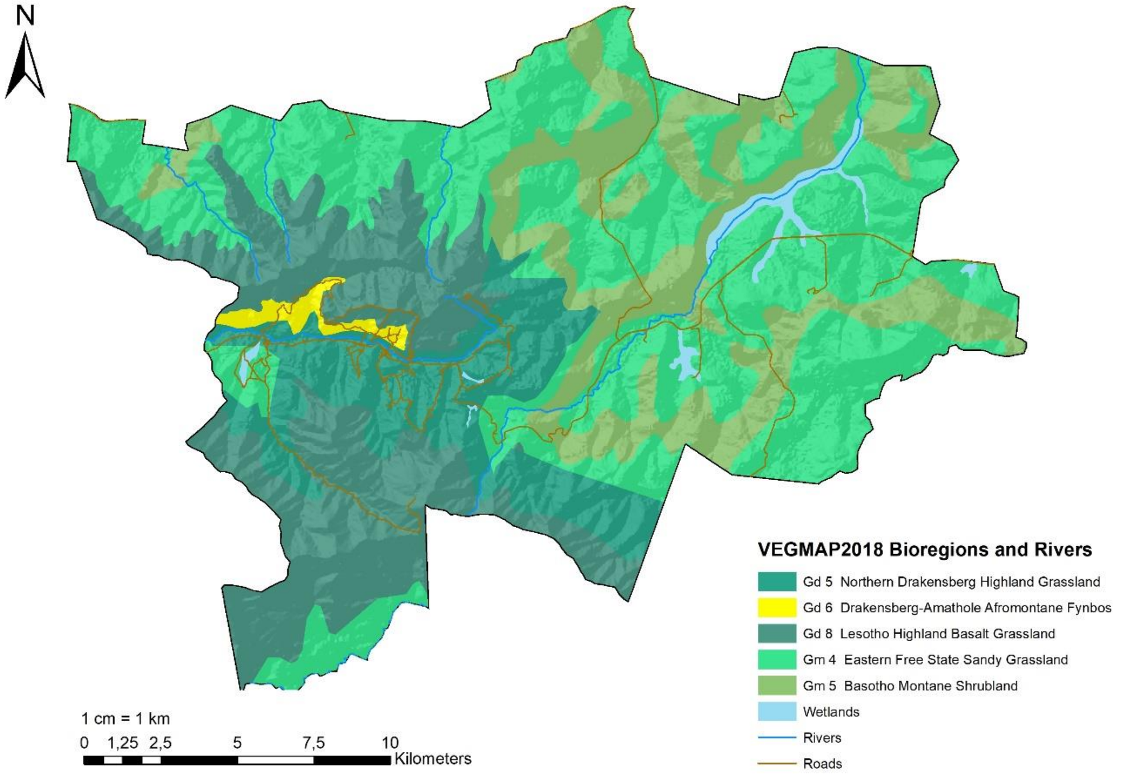
Biotope Quality Index (*BQI*) can be used for both animal and plant taxa data files.

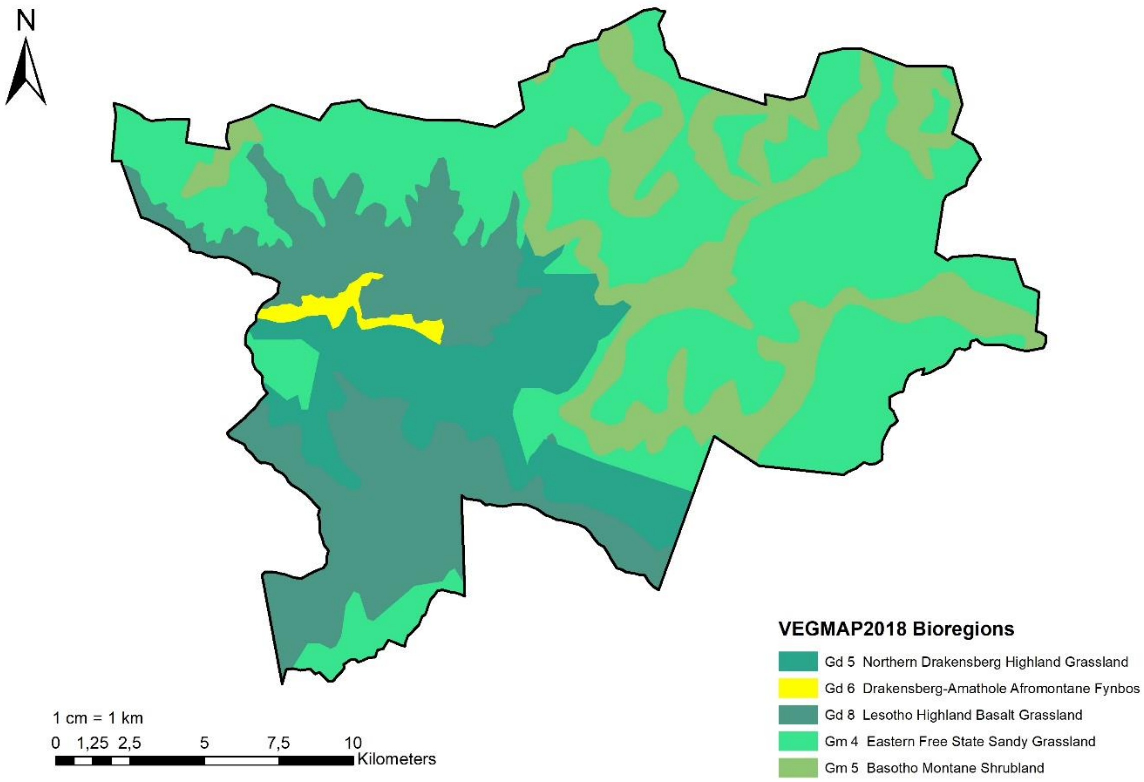
GIS material for Golden Gate Highlands National Park

Golden Gate Highlands National Park previous 1940 boarder on 2021 demacations






N
VEGMAP2018 Bioregions and Rivers
Gd 5 Northern Drakensberg Highland Grassland
Gd 6 Drakensberg-Amathole Afromontane Fynbos
Gd 8 Lesotho Highland Basalt Grassland
Gm 4 Eastern Free State Sandy Grassland
Gm 5 Basotho Montane Shrubland
Wetlands
Rivers
Roads
1 cm = 1 km
0 1,25 2,5 5 7,5 10
Kilometers


N
VEGMAP2018 Bioregions
Gd 5 Northern Drakensberg Highland Grassland
Gd 6 Drakensberg-Amathole Afromontane Fynbos
Gd 8 Lesotho Highland Basalt Grassland
Gm 4 Eastern Free State Sandy Grassland
Gm 5 Basotho Montane Shrubland
1 cm = 1 km
0 1,25 2,5 5 7,5 10
Kilometers

Plant list (Mesic Highveld Grassland)

Table 0.8: Plant list (density cover scale) recorded for Mesic Highveld Grassland in GGHNPark

Family	Genus	Species	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Pteridaceae	*Pellaea*	*calomelanos*	0	0	0	0	2	0	0	2
Dennstaedtiaceae	*Pteridium*	*aquilinum*	0	0	0	0	8	0	0	0
Agapanthaceae	*Agapanthus*	*campanulatus*	0	0	0	0	2	0	1	2
Hyacinthaceae	*Eucomis*	*autumnalis*	0	0	0	0	1	0	0	1
		humilis	0	0	0	0	1	0	1	1
	Merwilla	*plumbea*	0	0	0	0	0	0	1	0
Asparagaceae	*Asparagus*	*africanus*	0	0	0	0	2	0	0	0
Orchidaceae	*Disa*	*versicolor*	0	0	0	0	1	0	1	1
	Moraea	*sp*	0	0	0	1	0	0	1	1
	Eulophia	*aculeta*	0	0	0	1	2	0	0	0
Dioscoreaceae	*Dioscorea*	*cotinifolia*	0	0	0	0	4	0	0	0
Poaceae	*Andropogon*	*appendiculatus*	0	0	0	33	0	0	15	0
		schirensis	2	0	2	2	2	0	10	2
	Paspalum	*dilatatum*	3	0	3	3	4	0	3	3
	Panicum	*schinzii*	0	0	2	2	2	0	2	2
	Setaria	*pallide-fusca*	3	3	3	10	3	2	3	10
	Melinis	*nerviglumis*	0	0	0	2	2	0	0	2
	Monocumbium	*ceresiiforme*	0	0	0	0	2	0	0	2
	Trachypogon	*spicatus*	0	0	2	10	0	0	3	3
	Brachiaria	*serrata*	0	0	3	0	0	2	0	4

Family	Genus	Species	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Poaceae	*Anthoxanthum*	*ecklonii*	0	0	2	3	0	2	3	2
	Tristachya	*leucothrix*	9	2	8	9	25	9	24	5
	Eragrostis	*capensis*	3	0	8	22	8	3	9	8
		chloromelas	33	0	24	30	29	28	33	30
		curvula	3	0	8	9	32	3	30	33
		gummiflua	2	0	2	3	3	2	3	3
		racemosa	3	0	0	8	10	2	9	8
		patentissima	0	0	0	3	3	0	2	0
		plana	2	0	2	3	3	0	0	0
	Catalepis	*gracilis*	0	0	0	0	0	0	2	2
	Bromus	*catharticus*	0	0	0	3	3	0	0	0
		africanus	0	0	0	3	21	0	0	0
	Aristida	*congesta*	0	0	0	3	20	0	23	22
		diffusa	0	0	0	4	4	0	0	0
		junciformis	0	0	0	3	23	0	22	19
	Themeda	*triandra*	45	0	70	75	73	72	70	75
	Heteropogon	*controtus*	3	2	9	10	10	3	8	9
	Harpochloa	*flax*	2	0	2	3	3	1	3	3
	Diheteropogon	*filifolius*	0	0	2	3	2	0	2	2
	Cymbopogon	*dieterlenii*	2	0	2	3	3	3	9	10
		prolixus	0	0	3	3	8	0	9	8

Family	Genus	Species	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Poaceae	*Hyparrhynia*	*hirta*	0	0	3	3	9	0	8	10
	Cynodon	*dactylon*	0	0	0	0	30	0	0	0
Cyperaceae	*Cyperus*	*schlechteri*	0	0	0	0	1	0	0	0
		denudatus	3	0	3	9	0	0	0	0
Protea	*Protea*	*roupelliae*	0	0	0	0	0	1	1	1
Parpaveraceae	*Argemone*	*ochroleuca*	1	0	1	0	1	1	0	0
Amaranthaceae	*Cyathula*	*unicinulata*	0	0	0	0	1	0	0	0
Crassulaceae	*Cotyledon*	*orbiculata*	0	0	0	0	0	0	0	1
Apocynaceae	*Gomphocarpus*	*fruticosus*	0	0	0	0	1	0	0	0
Salicaceae	*Doryalis*	*rhamnoides*	0	0	0	0	1	0	0	0
Fabaceae	*Acacia*	*mearnsii*	0	0	0	0	0	8	0	0
	Indigofera	*longebarbata*	0	0	0	0	0	0	1	1
Rosaceae	*Leucosidea*	*sericea*	0	0	0	0	1	1	1	1
	Rubus	*ludwigii*	0	0	0	0	1	0	0	0
	Cliffortia	*linearifolia*	1	0	1	0	1	0	1	1
Anacardiaceae	*Searsia*	*dentata*	0	0	0	0	1	0	1	0
		pyriodes	0	0	0	0	1	0	1	0
		discolor	0	0	0	0	1	0	0	0
Geraniaceae	*Pelargonium*	*alchemilloides*	0	0	0	0	1	0	0	0
Oxalidaceae	*Oxalis*	*purpurea*	3	0	3	3	3	3	3	3
Cucurbitaceae	*Coccinia*	*rehmannii*	0	0	0	0	1	0	0	0

Family	Genus	Species	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Acanthaceae	*Barleria*	*monticola*	0	0	0	0	0	0	0	1
Dipsacaceae	*Scabiosa*	*columbaria*	0	0	0	2	3	0	3	3
Commelinaceae	*Commelina*	*africana*	0	0	2	3	3	0	0	3
Asteraceae	*Helichrysum*	*pilosellum*	0	0	0	3	3	0	0	3
		rugulosm	2	0	0	2	0	0	2	2
		callicomum	0	0	2	2	2	0	2	2
	Nidorella	*anomala*	2	0	2	3	0	0	3	3
	Sonchus	*asper*	0	0	0	0	1	0	0	0
	Vernonia	*sp1*	0	0	0	0	1	0	0	0
		flanaganii	0	0	0	0	1	0	1	0
	Cirsium	*vulgare*	0	0	0	0	1	0	0	0
	Seriphium	*plumosum*	9	0	3	3	3	3	9	3
	Senecio	*cryptolanatus*	0	0	0	1	1	0	0	0
		gregatus	0	0	0	0	1	0	0	0
		sp.1	0	0	1	0	0	0	0	0
	Berkheya	*discolor*	0	0	1	0	1	0	0	1
	Felicia	*spp1*	0	0	0	1	0	0	1	0
		sp2	0	0	0	1	0	0	0	0
	Gazania	*krebsiana*	1	0	0	1	1	0	1	1
	Artimisia	*afra*	0	0	0	0	1	0	0	0
	Bidens	*pilosa*	0	0	0	0	1	0	0	0

Family	Genus	Species	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
	Tagetes	*minuta*	0	0	0	0	1	1	0	0
Lobeliaceae	*Monopsis*	*decipiens*	2	0	0	3	3	2	3	3
Plantaginaceae	*Plantago*	*lanceolata*	2	0	2	3	3	2	0	0
Lamiaceae	*Ajuga*	*ophrydis*	0	0	1	1	0	0	1	1
	Stachys	*dregeana*	0	0	0	0	1	0	0	0
Verbenaceae	*Verbena*	*bonariensis*	0	0	0	0	1	0	0	0
Loganiaceae	*Buddleja*	*salviifolia*	0	0	0	0	1	1	0	0
Pittosporaceae	*Pittosporum*	*viridiflorum*	0	0	0	0	1	0	0	0
Myrsinaceae	*Myrsine*	*africana*	0	0	0	0	1	1	1	0
Ebenaceae	*Diospyros*	*austro-africana*	0	0	0	0	1	0	1	1
	Euclea	*crispa*	0	0	0	0	1	0	1	0

Arthropod list (Mesic Highveld Grassland)

Table 0.9: Arthropod morphospecies list recorded for Mesic Highveld Grassland in GGHNPark

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Pisauridae	Nurssp_01	0	0	3	25	1	4	11	3
Pholcidae	Daddyl_01	35	10	48	80	143	64	68	84
Thomisidae	Craps_01	0	0	6	18	23	0	0	3
Selenopidae	Walls_01	0	0	0	3	0	0	5	2
Salticidae	Jumos_01	0	0	8	11	3	18	16	13
Lycosidae	Wolfs_01	17	4	18	45	20	21	35	14
Solifugae	Sunsp_01	1	0	13	23	0	1	0	0
Tetranychidae	Miter_01	0	2	23	27	81	18	13	15
Ixodidae	Tickr_01	0	4	16	48	1	2	0	0
Scolopendridae	Scolo_01	0	2	8	0	2	1	0	0
Geophilidae	Geop_01	0	0	0	13	0	0	12	12
Spirostreptidae	Doratogonus	0	0	0	8	14	11	2	0
	Spirobolida	0	2	0	0	2	2	0	1
	Sphaerotherium	0	0	1	0	1	0	0	0
Entomobryidae	Capbrya	8	0	22	14	57	32	28	36
	Lepidocyrtinus africanus	78	49	145	260	328	187	288	240
	Lepidocyrtinus sp1	46	28	75	89	242	98	168	108
Hypogastruridae	Hypogastrura	0	0	11	35	12	17	12	6
Sminthuridae	Sminthurus	0	0	1	16	32	3	1	0
Lepismatidae	Lepisma_01	0	0	2	1	17	15	12	16
Termitidae	Nasut_01	28	18	25	16	23	8	12	15
Blattidae	Blatt_01	0	1	12	13	37	7	9	1
	Blatt_02	1	1	0	8	0	2	1	2
Gryllidae	Gryll_01	4	8	13	21	57	15	22	16

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Gryllotalpidae	Mole_01	0	0	2	0	8	0	1	12
Tettigoniidae	Tetti_01	0	0	6	5	36	0	0	2
Pyrogomorphidae	Pyrcom_01	5	5	12	32	45	12	13	18
Acrididae	Acrididae_01	1	7	14	26	53	16	28	5
	Acrididae_02	0	1	9	18	26	16	31	28
	Acrididae_03	8	0	9	11	8	0	0	0
	Acrididae_04	0	1	16	5	23	14	0	0
	Acrididae_05	12	0	0	0	16	0	0	0
Mantidae	Mant_01	8	0	17	18	25	8	12	21
Thespidae	Thesp_01	15	6	16	38	66	37	25	22
Phasmatidae	Phasm_01	3	0	18	24	36	45	29	36
Aphididae	Aphid_01	14	3	35	6	58	11	34	0
	Aphid_02	9	1	17	0	36	16	8	0
Psyllidae	Psyll_01	0	0	12	0	23	0	0	1
Cicadellidae	Cicade_01	1	1	1	18	68	18	18	28
	Cicade_02	2	0	24	25	54	6	31	38
	Cicade_03	16	8	12	56	56	13	28	34
	Cicade_04	1	0	19	48	66	16	28	28
	Cicade_05	2	1	8	2	19	2	0	8
	Cicade_06	2	0	3	11	23	7	6	1
	Cicade_07	1	0	6	6	12	1	0	0
Miridae	Mirid_01	7	0	0	7	35	5	0	1
Pyrrchoeoridae	Pyrrch_01	0	0	5	0	22	0	6	0
Lygaeidae	Lyga_01	0	0	12	2	6	8	7	13
	Lyga_02	0	0	0	0	8	1	8	3
Argacidae	Arga_01	3	0	2	1	0	11	12	1
Pentatomidae	Penta_01	2	1	1	34	13	15	26	23

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Pentatomidae	Penta_02	1	0	1	0	3	3	0	0
Reduviidae	Reduv_01	0	1	8	0	17	1	6	1
	Reduv_02	12	1	0	0	0	2	0	0
	Reduv_03	0	0	0	0	1	1	8	0
Thripidae	Thrip_01	16	0	35	56	51	23	48	28
	Thrip_02	1	1	13	23	25	1	6	1
	Thrip_03	0	0	1	1	0	3	2	0
	Thrip_04	0	0	8	9	15	3	8	0
Myrmeliontidae	Myrme_01	6	0	6	6	11	0	5	0
Chrysopidae	Chryso_01	0	1	3	2	9	5	7	1
Carabidae	Carab_01	12	0	18	7	23	0	32	23
	Carab_02	18	4	21	7	42	1	28	24
	Carab_03	1	1	0	13	42	9	0	11
Coccinellidae	Cocci_01	8	2	0	18	42	8	17	9
	Cocci_02	0	0	18	21	25	4	1	0
	Cocci_03	4	1	11	0	0	4	0	0
	Cocci_04	6	0	3	1	0	0	1	0
	Cocci_05	1	0	8	16	0	7	0	0
Tenebrionidae	Teneb_01	11	0	28	35	46	15	16	18
	Teneb_02	0	0	0	18	34	14	9	5
	Teneb_03	0	0	9	22	24	6	5	3
	Teneb_04	14	8	13	8	2	0	12	8
	Teneb_05	1	12	22	16	0	0	6	18
	Teneb_06	0	0	1	8	0	8	0	2
Staphylinidae	Staphyl_01	15	4	32	49	28	17	29	31
	Staphyl_02	4	0	18	35	15	4	0	4

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Staphylinidae	Staphyl_03	8	0	11	8	7	0	4	6
	Staphyl_04	0	0	9	16	9	0	3	15
	Staphyl_05	0	0	8	4	0	2	0	0
Chrysomelidae	Chryso_01	13	1	23	35	25	0	0	0
	Chryso_02	0	0	18	16	42	13	0	2
	Chryso_03	6	0	16	22	34	11	0	0
	Chryso_04	0	0	18	32	43	2	22	6
	Chryso_05	4	0	9	17	23	0	2	1
	Chryso_06	1	0	4	8	9	8	12	4
	Chryso_07	1	0	0	0	0	0	1	1
	Chryso_08	1	0	0	28	34	1	0	0
	Chryso_09	0	1	1	1	0	1	0	0
Melyridae	Astylus	0	0	8	0	1	0	0	1
Polyphagoid	Polyph_01	0	0	0	0	0	3	5	1
	Polyph_02	0	0	8	12	0	0	2	0
	Polyph_03	0	0	0	6	2	5	3	3
	Polyph_04	0	2	28	31	15	5	0	2
Scarabaeidae	Scarab_01	0	0	9	19	0	0	5	1
	Scarab_02	2	0	12	34	0	15	0	7
	Scarab_03	0	2	0	2	4	2	2	11
	Scarab_04	0	6	0	16	0	0	13	0
	Scarab_05	5	0	0	18	0	7	3	0
	Scarab_06	0	0	13	8	19	8	14	9
	Scarab_07	0	0	1	0	2	0	1	0
	Scarab_08	6	0	4	0	0	0	0	0

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Nitidulidae	Nitidu_01	0	1	4	12	23	0	0	0
	Nitidu_02	1	0	0	0	0	0	0	0
Sirtidae	Sirtid_01	0	0	0	1	0	1	0	0
Leiodidae	Leiod_01	0	0	0	0	1	1	0	0
Brentidae	Brent_01	0	0	0	2	0	4	0	2
	Brent_02	0	0	0	0	0	6	0	0
	Brent_03	0	1	3	9	0	1	0	1
Curculionidae	Curcul_01	1	0	12	25	5	0	1	0
	Curcul_02	1	0	1	8	13	0	2	0
	Curcul_03	6	0	1	0	0	2	2	0
	Curcul_04	1	0	4	11	1	0	0	0
Tipulidae	Tipul_01	11	0	8	18	0	7	1	6
Chironomidae	Chiron_01	0	0	14	36	64	3	7	7
	Chiron_02	8	0	2	8	34	0	1	0
Ceratopogonidae	Cerato_01	0	0	8	18	1	0	0	0
Psychodidae	Psycho_01	0	0	1	0	0	1	0	0
Scatopsidae	Scatop_01	8	3	11	14	7	0	0	0
Simuliidae	Simuli_01	0	0	1	0	1	0	0	7
Phoridae	Phorid_01	8	0	9	23	24	1	0	0
	Phorid_02	0	0	1	18	37	0	0	7
Syrphidae	Syrphi_01	4	1	0	16	17	1	0	0
	Syrphi_02	2	0	6	8	27	12	2	0
	Syrphi_03	0	0	5	13	3	2	6	2
	Syrphi_04	8	0	1	8	24	1	1	6
	Syrphi_05	0	1	0	1	5	9	7	2
	Syrphi_06	1	0	1	0	0	0	1	1

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Dolichopodidae	Dolich_01	2	0	0	8	1	5	5	6
Empididae	Empid_01	0	0	9	3	0	7	1	0
	Empid_02	0	1	1	0	1	8	1	4
Asilidae	Asili_01	0	1	1	1	0	1	1	0
Bombyliidae	Bomby_01	1	1	0	0	2	8	2	8
	Bomby_02	0	1	6	8	3	11	4	2
Tabanidae	Tabani_01	2	1	4	21	17	8	6	3
	Tabani_02	0	0	0	4	2	6	5	9
Stratiomyidae	Stration_01	0	0	1	1	9	7	8	2
	Stration_02	0	1	2	0	4	1	1	0
Conopidae	Conop_01	0	0	1	0	9	0	0	0
Psilidae	Psili_01	1	0	1	0	2	1	1	1
Ragionidae	Ragion_01	0	1	1	0	1	0	0	0
Tachinidae	Tachin_01	2	0	2	0	2	1	0	1
Anthomyiidae	Anthom_01	9	0	16	27	36	12	8	12
	Anthom_02	11	1	28	31	32	18	12	16
	Anthom_03	8	0	19	38	52	21	35	20
	Anthom_04	1	0	3	8	26	4	20	8
	Anthom_05	1	1	1	11	25	9	1	0
	Anthom_06	0	0	0	1	0	1	0	0
Cryptochetidae	Crypto_01	0	0	0	1	0	0	12	7
Agromyzidae	Agromyz_01	0	1	8	0	12	0	1	18
Muscidae	Musci_01	16	0	23	4	34	0	7	6
	Musci_02	12	2	17	12	22	12	32	28
	Musci_03	0	1	0	1	0	0	5	5

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Drosophilidae	Drosop_01	12	1	18	21	48	1	31	23
	Drosop_02	0	2	4	0	0	1	2	0
	Drosop_03	0	1	31	1	44	0	0	0
Tephritidae	Tephri_01	2	0	12	1	16	0	2	0
	Tephri_02	1	0	4	1	13	0	2	0
	Tephri_03	0	0	0	11	7	0	12	8
	Tephri_04	0	0	0	9	0	1	1	2
Calliphoridae	Calliph_01	0	0	15	2	18	0	0	0
	Calliph_02	8	1	16	35	33	13	26	18
	Calliph_03	5	1	9	6	13	11	23	9
	Calliph_04	11	6	19	0	36	6	0	0
Sarcophagidae	Sarcop_01	0	1	1	0	0	0	0	0
Scathophagidae	Scathop_01	6	0	11	10	19	4	4	26
Sepsidae	Sepsi_01	0	1	8	1	17	0	0	0
Pulicidae	Pulici_01	0	0	0	0	6	1	1	0
Nymphalidae	Asterocampa	0	0	0	4	0	0	8	4
	Danaus	4	0	8	8	14	2	2	3
	Telchinia	1	0	6	1	0	0	1	3
	Catacroptera cloanthe cloanthe	0	0	0	1	6	9	8	11
	Precis octavia sesamus	0	2	0	11	0	3	9	0
	Junonia orithya madagascariensis	0	0	5	9	0	0	9	9
	Junonia hierta cebrene	0	0	12	12	2	5	4	3
	Vanessa cardui	0	0	8	14	14	5	16	12
Nymphalidae	Asterocampa	0	0	0	4	0	0	8	4
	Danaus	4	0	8	8	14	2	2	3
	Telchinia	1	0	6	1	0	0	1	3

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Nymphalidae	Catacroptera cloanthe cloanthe	0	0	0	1	6	9	8	11
	Precis octavia sesamus	0	2	0	11	0	3	9	0
	Junonia orithya madagascariensis	0	0	5	9	0	0	9	9
	Junonia hierta cebrene	0	0	12	12	2	5	4	3
	Vanessa cardui	0	0	8	14	14	5	16	12
Lycaenidae	Leptomyrina lara	1	0	2	4	6	20	2	8
	Lycaena clarki	0	0	2	15	11	2	14	18
	Zizeeria_01	0	0	0	1	21	5	18	12
	Zizeeria_02	0	0	0	5	11	5	8	6
Pieridae	Belenois aurota aurota	1	6	12	26	22	16	13	32
	Pontia helice helice	0	0	1	4	6	0	11	0
	Colias electo electo	1	0	4	8	20	2	5	12
	Catopsilia florella	1	1	12	17	12	7	16	26
Hesperiidae	Hesper_01	0	1	8	6	11	4	7	15
Sphingidae	Hemaris thysbe	0	1	9	11	17	4	22	34
Noctuidae	Noctu_01	0	1	3	8	1	0	1	1
	Noctu_02	9	0	11	8	16	8	11	18
	Noctu_03	11	0	17	21	28	8	16	32
Noctuidae	Noctu_04	2	1	0	8	17	7	16	23
	Ctenuchinae	0	0	1	1	3	1	8	6
Notondontidae	Noto_01	0	0	6	12	16	9	12	18
	Noto_02	1	0	2	19	5	0	16	8
Geometridae	Geome_01	0	1	8	23	16	0	16	25

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Ichneumonidae	Ichneu_01	0	2	0	2	8	0	8	3
	Ichneu_02	8	6	11	11	16	12	19	13
Braconidae	Braco01	0	0	2	6	2	1	4	3
	Braco02	1	2	0	1	1	7	5	8
	Braco03	0	0	2	0	1	1	11	8
	Braco04	0	0	3	0	5	0	8	3
	Braco05	0	1	1	0	0	1	1	3
Figitidae	Figit01	0	0	0	0	0	1	7	0
Dryinidae	Dryin01	2	0	14	8	12	6	11	17
	Dryin02	1	0	2	1	8	8	2	8
Aphelinidae	Aphel01	1	0	1	0	1	0	2	1
	Aphel02	0	3	8	0	0	4	9	12
Ceraphronidae	Cerap01	0	0	1	0	0	1	1	0
	Cerap02	1	0	0	1	1	1	0	0
	Cerap03	0	0	0	1	1	1	1	0
Elasmidae	Elasm01	0	0	0	3	1	1	1	1
Ceraphronidae	Ceraph_01	0	1	6	0	1	2	0	0
Cynipidae	Cynip_01	0	1	2	0	0	2	0	1
Encyrtidae	Encyr01	1	2	0	0	1	0	1	0
	Encyr02	5	1	0	0	1	0	0	0
	Encyr03	8	0	1	4	13	15	12	18
	Encyr04	0	1	0	4	0	0	8	8
	Encyr05	0	1	0	2	1	0	0	0
	Encyr06	0	0	0	0	8	1	0	0

Family	Morphosp	Korfs_Gm4	Kloof_Gm4	Twij_Gm4	Welv_Gm4	QQMo_Gm5	Avon_Gm5	Diep_Gm5	Sila_Gm5
Platygasteridae	Platy01	0	0	0	0	0	5	1	0
	Platy02	0	1	0	0	0	0	5	3
	Platy03	0	1	0	2	0	8	0	0
	Platy04	3	0	2	0	4	7	8	0
	Platy05	1	0	0	1	0	1	0	2
	Platy06	0	0	4	0	1	1	2	0
Pteromalidae	Pterom_01	0	0	0	0	8	0	0	13
	Pterom_02	0	4	0	0	0	0	0	0
Scelionidae	Sceli01	0	6	1	0	0	0	0	0
Halictidae	Halict_01	0	2	0	0	2	6	0	0
Vespidae	Vesp_01	1	1	0	1	0	0	0	0
	Vesp_02	1	0	1	0	3	1	7	3
Apidae	Apid_01	1	2	21	4	28	13	15	12
Formicidae	Formi01	1	1	1	0	1	1	1	1
	Formi02	0	0	8	2	12	7	17	18
	Formi03	64	26	150	108	257	94	132	140
	Formi04	32	18	64	18	143	38	51	64
	Formi05	1	0	0	3	8	1	8	12
Agnaridae	Hemilepistus	1	8	3	11	36	6	11	8

APPENDICES

Official legal sampling permits and agreement

UFS Ethical clearance approval permit

UNIVERSITY OF THE FREE STATE
UNIVERSITEIT VAN DIE VRYSTAAT
YUNIVESITHI YA FREISTATA

UFS·UV
NATURAL AND AGRICULTURAL SCIENCES
NATUUR- EN LANDBOUWETENSKAPPE

Faculty of Natural and Agricultural Sciences

06-Nov-2015

Dear **Serero Modise**

Ethics Clearance: **Ecological analysis of Afromontane grasslands in the eastern Free State using the novel Biotope Quality Index**

Principal Investigator: **Serero Modise**

CONDITIONALLY APPROVED

This letter confirms that a research proposal with tracking number: **UFS-HSD2015/0482** and title: **Ecological analysis of Afromontane grasslands in the eastern Free State using the novel Biotope Quality Index** was given ethics clearance by the Ethical Committee pending clarification of the following:

Permits required for collection of samples

Please ensure that the ethical standards committee is notified should any substantive change(s) be made, for whatever reason, during the research process. This includes changes in investigators. Please also ensure that a brief report is submitted to the ethics committee on completion of the research. The purpose of this report is to indicate whether or not the research was conducted successfully, if any aspects could not be completed, or if any problems arose that the ethical standards committee should be aware of.

Note:

1. This clearance is valid from the date on this letter to the time of completion of data collection.
2. Progress reports should be submitted annually unless otherwise specified.

Yours Sincerely

Prof. Neil Heideman
Chairperson: Ethics Committee
Faculty of Natural and Agricultural Sciences

Natural and Agricultural Sciences Research Ethics Committee
Office of the Dean: Natural and Agricultural Sciences
T: +27 (0)51 401 2322 | F: +27 (0)51 401 3728 | E: Heidemannj@ufs.ac.za
Biology Building, Ground Floor, Room 9 | P.O. Box/Posbus 339 (Internal Post Box G44) | Bloemfontein 9300 | South Africa
www.ufs.ac.za

SANParks sampling permit approval

APPL ICATION FOR A PERMIT TO COLLECT NATURAL RESOURCES IN SOUTH AFRICAN NATIONAL PARKS

Scientific Services
Savanna and Arid Node

The information requested below must be completed in full before the issuing of a permit can be considered.

A. RENEWAL OF EXISTING PERMIT

- Provide the current permit number here:
- If the information on the original permit has not changed, it is only necessary to give the reason for renewal below; no additional information is required:-

- If the information has changed, proceed with section B below.

APPLICATION FOR A NEW PERMIT

1. What is the reason for collecting the material? (Mark with X)				Research		X	Other (give details):		
Do you currently have a registered research project with SANParks?			If yes, provide project number:		Expiry date:			No	X
2. Personal details of collector	Title	Mr	Initials	SA	Surname	Modise			
3. ID or Passport number	8906225945088				Nationality	South African			

4. Residential address (if applying in personal capacity)
347N Bluegumbosch, Witsieshoek, 9870

5. Institution (if not applying in a private capacity):
University of the Free State

6. Description of material to be collected, and amount required:
Arthropods: Arachnida, Myriapods, Crustaceans, Hexapoda (Entognatha & Insecta) (2 haplotype specimens for each individual species representative for each invertebrate class reference)
Angiosperms (1 specimens for each individual representative for each plant species reference)

7. Period for which permit is required: 2016 - 2017

8. Will live material be collected? If yes, give details if different from above details: No

9. NB: If the material is to be removed from the KNP, a State Veterinary Movement permit must be obtained.

10. Locality where the material is to be collected : Golden Gate Highlands National Park

11. Other relevant information

13. Additional conditions: Capture and release will be used for any previously collected specimens

South African
NATIONAL PARKS

The signatures below indicate that the collection permit has been approved for research number:

____

Manager Scientific Services: Date:

.... 1/11/2016..........

APPROVED ☒ **NOT APPROVED** ☐

Reason for decision: