Understanding the spatial relations of wetland plant functional traits and environmental gradients incorporating remote sensing

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Dedication

To my parents:
Goodman Lungana Nondlazi
(*Tolo Dlangamandla, Mchenge, Zulu, Mabha-nekhazi, Mlambo-aw'welwa, Ngeny-nkomo, Tshoba, Tshatwa, Jikazi, sizukulu sikaDisncane*)
Cecilia noKwakha Nikiwe : daughter of the Madikiza family
(*Mathiyane Zengele, Hlubi, Makhwange, sizukulu sikaMahlangabeza*)
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I, Basanda X. Nondlazi declare that:

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2021

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This thesis is compiled as stand-alone research papers, therefore repetition in sections such as methods are necessary. Following below are a list of the research papers published, submitted, or in preparation form. These papers were developed and written by the first author under the supervision of the second, third and fourth authors.

- 1. Basanda X. Nondlazi, Moses Azong Cho, Heidi van Deventer, Erwin Sieben Determining the wetland-dryland boundary of depressions using littoral gra- dient analysis of soil edaphic factors. Published in Wetlands Journal DOI:10.1007/s13157-021-01430-9
- 2. Basanda X. Nondlazi, Moses Azong Cho, Heidi van Deventer, Erwin Sieben Determining the wetland-dryland boundary of depressions using functional traits of littoral vegetation. Complete draft in internal review, ready for submission
- 3. Basanda X. Nondlazi, Moses Azong Cho, Heidi van Deventer, Erwin Sieben Determining the wetland boundary of depressions using Hyperspectral remote sensing analysis of littoral gradient soils. Complete draft in internal review, ready for submission
- 4. Basanda X. Nondlazi, Moses Azong Cho, Heidi van Deventer, Erwin Sieben Determining the wetland boundary of depressions using gradient analysis of littoral vegetation with Sentinel-2A indices. Complete draft in internal review, ready for submission
- 5. Basanda X. Nondlazi, Moses Azong Cho, Heidi van Deventer, Erwin Sieben Determining the boundary of depressional wetlands using littoral vegetation community structure. Complete draft in internal review, ready for submission

I, Professor Moses Azong Cho as supervisor of the PhD study hereby consent to the submission of this PhD Thesis.

On date 26 August 2021 ....

Signature..[redacted].............

Acronyms

AGB................Above Ground Biomass
BD...................Bulk Density
C.....................Carbon
$C0_2$..................Carbon dioxide
DEM...............Digital Elevation Model
DIWA.............Directory for Important Wetland of Australia
e.g. ....*Exempli gratia* "for example" or "such as."
EC...................Electric Conductivity
et al. ....*et alia* Latin for "and others."
etc...................*Et cetera*
GEE................Google Earth Engine
GIS..................Geographic Information System
GPS.................Global Positioning System
HGM................HydroGeoMorphic
ID.....................Identification
LAI...................Leaf Area Index
MLD.................Mpumalanga Lake District
MSBI................Misra Soil Brightness Index
N.......................Nitrogen
NDSI................Normalised Difference Salinity Index
NDWI...............Normalized Difference Water Index
PCA..................Principal Component Analysis
PPR..................Prairie Pothole Region
Ramsar.............A Convention on Wetlands of International Importance signed in Ramsar, 1971
REDNDVI ....Normalized Difference RedEdge using the first red edge position
S-2A..................Sentinal 2 A
SCI....................Soil Composition Index
SMC..................Soil Moisture Content
US ....United States
USGS................United States Geological Survey
VMC................Vegetation Moisture Conent

Key concepts

Edaphic factors: are factors related to the soil. The qualities or properties that may characterize the soil such as soil moisture content, bulk density, soil temperature or chemical pH and salinity.

Wetland ecotone: is an abrupt change in vegetation; resulting in a narrow ecological zone between two different, homogeneous and adjacent community types. In the littoral zone of a lake, an ecotone is the transition zone of distinct aquatic communities that vary throughout the year according to seasonality. For example between dryland and a water body (Burton and Tiner 2009), between rivers and their floodplains (Cummins and Wilzbach 2008), between marine and terrestrial.

Wetland boundary/threshold: The outer edge of the wetland ecotone. Wetland threshold: The exact point where one distinct equatorial region meets another; the point that can be considered as the true dividing line between the two ecosystems. Remote sensing measuring or studying an object from a distance.

Hyperspectral imaging: remote sensing that collects and processes information across the electromagnetic spectrum in the nanometre range.
Multispectral: sensors of airborne recording radiation from the visible parts of the electromagnetic spectrum.

Wetland delineation: determination of precise boundaries on the ground through field surveys.

Wetland classification: systematic arrangement of wetlands in groups or categories according to established specific criteria.

Wetland monitoring: observing and checking the changes in the quality and size or extents of wetlands over a period, thus keeping them under systematic review.

Wetland inventory: is a dataset containing information on the wetland of a country such as location, size and ecological data
Edaphic factors: soil properties that affect the diversity of organisms living in the soil environment.

Endorheic wetlands: are wetland with a drainage basin that normally retains water and allows no outflow to other external bodies of water, such as rivers or oceans, but drainage converges instead into lakes or swamps, permanent or seasonal.

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Abstract

To monitor and predict the response of wetland ecosystems to climate change over large scales, we must improve the delineation of the extents and classification of wetlands. We therefore need to gain a deeper understanding of the spatial relation between vegetation functional traits and edaphic factors. Remote sensing is a time and cost-efficient way to proceed with large-scale monitoring of wetland ecosystems and understaning these spatial relations. However, it has historically been difficult to use remote sensing data for large-scale environmental monitoring due to the cost. Landsat was the first sensor edition to have its products made freely available. However, for over thirty years, its broad spectral resolution (30 m) has limited the spectral understanding of many target objects, including the spatial relation between soil and vegetation along the littoral gradient of depressional wetlands. Recently, Sentinel-2A data has been made freely available, with upto 10 m spatial resolution and a greater spectral resolution. This free availability has sparked an increase in the number of studies that test Sentinel-2A's utility in improving applications that previously relied on Landsat. Hence in this thesis, the utility of Sentinel-2A indices in conjunction with field data in determining the wetland boundary's width, position, and pattern is tested. Some of the field data require expensive laboratory processing. Hence, a secondary aim was to test the utility of hyperspectral remote sensing indices in estimating and supplemnting or substituting these laboratory results. In this regard Hyperspectral indices were compared with laboratory data to detect the width, position, and pattern of wetland boundaries.

The advent of hyperspectral remote sensing provided the first near complete band composition in the Visible (VIS), near-infrared (NIR), and shortwave infrared (SWIR) regions of the electromagnetic spectrum (EM). Hyperspectral variables associated with soil physical, chemical and biological properties are detectable according to their reflectance in the wavelength range from 400 to 2500 nm. Vibrations produce reflectance signals in bonds between atoms in the soil, weak overtones, the stretching and bending of dominant compounds and electronic transitions of the EM. You can also use the average shift of the soil reflectance profiles to classify wetlands. You can also examine specific regions of the EM to develop indices that estimate specific physical, chemical, and biological soil properties. Soil chemistry, moisture and substrate type are important when classifying wetlands. However, the protocols are not appropriate for national monitoring since they are based on time-consuming and costly laboratory analysis. This study aimed to develop new methods for estimating wetlands' salt content, bulk density, and soil moisture using remote sensing. Thus, the results suggest new ways for emperically discerning descriptors level six of the South African Wetland classification scheme.

The 6th level of the South African Wetland classification system also uses vegetation similarly to global standards. However, currently, these standards are biased towards soil redox potential and only use vegetation for confirmatory bases by considering whether the

cover of obligate wetland plants reaches 50%. This approach to studying wetland vegetation focuses on species or taxonomy. In recent years, the plant functional trait (PFT) approach is receiving favour over the species approach in vegetation ecology. PFTs are becoming more popular because they overcome the 'inter' and intraspecific confounding plaguing species approaches. The PFT approach overcomes these critical limitations of the species approaches by converting species characteristics and environmental variables into continuous variables. Species can be opportunistic inhabiters of wetlands (facultative) while others are constrained to wetlands (obligate). However, many facultative species inhabit wetlands and *vice versa* with obligate species. On one hand, the PFT approach recognises variation in traits of species and their abundance and occurance. Thus, even in remote sensing, individuals of the same species can be allocated on different ends of the spectrum in the Near-InfraRed region. Therefore, groups of the same species from different sites can also have different mean spectral signatures, which confuses the relationship between species and spectra when using the approach mean species values. Meanwhile, the PFT approach considers differences within and between species based on traits, that are associated with changes in environmental conditions and spectra without confounding. On the other hand, the species approach does not consider variation within as species. As a result the variance of the mean spectra of a species is high, risking overlaps with the spectral ranges of other species. This thesis presents unique methods to delineate and classify wetland boundaries based on the PFT approach and the corresponding Sentinel-2A vegetation indices through two spatially and temporally dependent experiments. The results of this thesis show that we can use remote sensing, edaphic factors and vegetation fucntional traits to sub-classify wetlands to finer detail than HydroGeoMorphic units and delineate boundaries objectively at $\sim$100 m from the wetland water body.

This thesis also discusses the utility of multidimensional scaling (MDS), canonical correspondence analysis (CCA), and gradient analysis for delineating and classifying temperate grassland depressional wetlands in the Mpumalanga Lake District Ecosystem in South Africa. These analytical techniques produce plausible results for delineation and classification that are easy to put into practice at the national level. In summary, this thesis contributes to the delineation and classification of wetlands, as well as to the extraction of soil and vegetation parameters from hyperspectral and multispectral data. There are some studies that use Geographic Information Systems to develop wetland delineation at the national scale, but only use the HydroGeoMorphic level of classification based on subjective desktop heads-up digitisation. It is possible, however, to develop a detailed wetland inventory that goes beyond HydroGeoMorphic level to include vegetation, soil chemistry, and substrate type. The results of these free multispectral remote sensing outcomes are validated and fine-tuned using high precision hyperspectral data. Remote sensing mapping of wetlands over national scales likely depend on improving understanding of the links between soil and vegetation over littoral zones. This thesis paves a new path in the development of wetland buffering protocols for policy development world-wide, as well as wetland classification approaches for developing the Ramsar Convention-required international wetland inventory. The thesis makes a sizable yet crucial contribution to developing better tools for wetland monitoring and conservation.

Chapter 1

Introduction

Wetlands are important ecosystems that provide valuable ecosystem services, critical for human livelihoods and biodiversity (Gardner and Finlayson 2018). Wetlands in temperate grasslands provide critical ecosystem services. They provide 1) supply services, *i.e.* food and water, 2) regulating services, *i.e.* on climate, pollution and floods, 3) esthetic and recreational services, *i.e.* cultural, spiritual and tourism, among others. These ecosystem services support biodiversity and human livelihoods (Tooth and McCarthy 2007). Depressions are unique because they are shallow and have narrow fringe zones. Depressions are strategic water sources because they occur in water-scarce regions (Tiner 2003; De Klerk *et al.* 2016). Thus depressional wetlands are vulnerable to climate variability and change. In part, the Ramsar Convention of 1971 provides the incentive for developing conservation tools for inventorying, monitoring, and protecting these strategic water resources (Turpie 2010; Sieben 2011; Naidoo *et al.* 2019; Van Deventer *et al.* 2020a). However, the limitation to monitoring tools depends on their accuracy, detail and up-to-date information on the extent and composition of the wetland data. These wetland data are incomplete for many temperate grassland regions due to challenging delineation and classification techniques (Dini *et al.* 1999). The delineation and classification of these valuable ecosystems are essential for determining buffer zones that protect them. The same data creates inventories of wetland types for monitoring over large scales (Macfarlane *et al.* 2009). Hence, they help monitor and conserve wetland extents and biodiversity (Taylor *et al.* 1995; Macfarlane *et al.* 2009; Macfarlane *et al.* 2015). Therefore, the buffering and inventorying of wetlands are essential for managing changes in ecosystem services and aquatic biodiversity under a changing climate and dwindling wetland extents (Mutanga *et al.* 2012; Van Deventer *et al.* 2016; Sieben *et al.* 2018).

The changes in wetland ecosystem services due to a changing climate and decline in aquatic biodiversity due to dwindling wetland extents are current societal research challenges (Hails 1996; Finlayson *et al.* 1999; Gopal *et al.* 2000; Aber *et al.* 2012). Wetlands in temperate grasslands are experiencing challenges from intense human activity, *i.e.* agriculture, construction and mining, in addition to changes in patterns of temperature and rainfall (Saunders *et al.* 2014; Mosquera *et al.* 2015; Gandarillas *et al.* 2016). The same ecosystem services support human livelihoods, thus cushioning extreme poverty in many world regions. For example, prominent pressures in the Niger Delta, Mackenzie Delta, Chesapeake Bay, and Bahia Blanca are agriculture and industrial development. In these areas, threats from a) agriculture include animal rearing, *i.e.* fisheries, b) construction for industrial growth, *i.e.* tourism, urban development, shipping, c) mining, *i.e.* dredging, and exploration for minerals, oil and gas. These pressures are causing changes and loss of wetland habitat connectivity, soil water movement, and sediment formation, thus extreme poverty in the long term (Denny 1993; Adekola and Mitchell 2011; Van Wilgen *et al.* 2020; Newton *et al.* 2020). These pressures destroy wetland features and contaminate or pollute wetlands, leading to smaller wetland extents and eventually poverty; people have to buy goods they previously could obtain from the wetlands.

The water body is an essential wetland feature (Slagter *et al.* 2020). However, other wetlands are without a water body. Instead, they have high water saturation in the soil and are called palustrine wetlands (Stolt *et al.* 2001). The term palustrine can also refer to the variably saturated terrestrial region found between the water and upland areas in a "lacustrine" wetland (Simon *et al.* 2001; Leighton *et al.* 2009). Buffer zones may include this palustrine region because these palustrine regions serve as a water body buffer (Basnyat *et al.* 2000). However, strictly speaking, scientifically, a buffer zone ought to be a region of dry upland that protects both this palustrine region and the water body or lacustrine region (Macfarlane *et al.* 2015). This misnomer brings amiss interpretations, likely because political acceptability, not scientific merit, governs declared sizes of wetland buffers (Castelle *et al.* 1994; Macfarlane *et al.* 2009; Dini and Everard 2018). For this thesis, reference is only to the palustrine region of wetlands because this is the current understanding in the field when referencing buffers (Young *et al.* 1980; Schellinger and Clausen, 1992; Castelle *et al.* 1994; Macfarlane *et al.* 2015). This thesis makes a case for recognising the inappropriate use of the word buffer, including the palustrine region as part of the wetland and not as part of the buffer, to protect wetland intactness in its entirety (Ma 2016). Hence, this thesis supports the advancement of the policy on wetland buffering. This thesis presents alternative approaches for determining the width of the wetlands' palustrine region and detecting the wetland boundary to support policies that protect wetlands.

Young *et al.* (1980) reported that a 24.4 m wide vegetation buffer in the temperate region of the United States reduce suspended sediment from a local feedlot by 92%. (Schellinger and Clausen (1992) reported 33% removal of suspended solids from a dairy farm using strips of a constructed wetland of 22.9 m. Castelle *et al.* (1994) review 28 articles on wetland buffering published between 1973 and 1992, mainly focusing on the United States and present a summary diagram on general buffer widths for specific buffer functions. The buffer widths are 30 m, 60 m, 90 m and <100 m for respective functions, *i.e.* water temperature moderation, sediment removal, nutrient removal and species diversity. Macfaclane *et al.* (2015) reviewed respective international literature of more than 150 citations on wetland buffering. From the review in more than 90% of the time, a wetland buffer equal to or greater than 100 m is recommended only for functions related to wildlife (Semlitsch and Bodie 2003). Current wetland delineation approaches report highly variable recommendations for wetland buffers. These variable results could be due to the general practice of buffering to preserve buffer functions or ecosystem services instead of "holistic" ecosystem functioning. Buffer functions seem logical when considering that many approaches to determining wetland buffers are designed for environmental impact assessment (EIA) to guide development decisions around wetlands (Hook 1993). During EIA, the objective is to delineate wetlands to quantify potential negative impacts during the construction of the development and its operational stages (Maltby 1988). Such development might include forestry, buildings, and roads. The objectives of EIA are not the same as the objectives of a wetlands inventory at the national scale. Therefore, an alternative approach proposed by this thesis is to focus on the wetland edaphic factors and botanical functional traits related to climate change. This approach differs from allocating wetland priority and vulnerability status based only on the buffer functions and threats in the general locality.

Current field techniques for delineating wetlands rely on indicators of soil redox potential. The upland side, where the redox potential is the lowest, is the outer boundary of the wetland. Redox potential is oxidation-reduction in a soil column due to its hydric features (Faulkner *et al.* 1989; Faulkner and Patrick 1992). Current delineation techniques assess redox potential and hydric soil features. They interpret soil auger core samples extracted every meter along the littoral gradient to determine where the wetland ends. Each soil core is a wetland sample only if it has; 1) a surface soil horizon high in organic carbon to 1 cm in depth. 2) water must saturate the soil for an extended period than the surrounding landscape or more than half of the year. 3) the soil core has grey decolouration viz, gleying (Hillier *et al.* 2011; Pulley *et al.* 2017). Gleying occurs due to low oxygen conditions resulting from iron (Fe) converting to iron carbonate (Fe++), resulting in a grey soil colour, *i.e.* a low chroma matrix. 4) The soil core must have mottles, small bluish-greenish to yellowish shiny pebble-like and rust-like features, ordinary in clayey soils. 5) The soil core must have a root zone under extended periods of no gaseous exchange with the atmosphere.

All the above five out of six criteria for delineating wetlands do not include vegetation. Even the sixth criterion only considers vegetation cover and not its true diversity (Kotze *et al.* 1996; Pennington and Walters 2006). Therefore, the current criteria for delineating wetlands only have one assessment point on vegetation against five assessment points for redox potential (Megonigal *et al.* 1993). The one point states that a landscape is considered a wetland when obligate and facultative wetland plant species occupy more than 50% of the landscape. Having one assessment point that focuses on vegetation highlights heavy reliance on redox potential. Hence there might be a lack of clarity on other essential vegetation features, *i.e.* vegetation diversity and plant functional traits. Although these rules are not rigid, it is unclear how to allocate the 50% between strata, between dominant and rare species, between obligate and facultative species (Thompson *et al.* 2002; Davidson *et al.* 2006; Ollis *et al.* 2009). The heavy reliance on soil redox potential limits these conventional wetland delineation techniques from using vegetation to delineate wetlands (Vepraskas 2000; Vepraskas and Faulkner 2000; Vepraskas and Lindbo 2012).

Current wetland classification approaches use landscape soil colour, soil moisture movement, morphology, and geomorphological position, *i.e.*, HydroGeoMorphic classification – HGM (Noble et al 2002; Vasilas et al 2005; Wardrop *et al.* 2007; Ollis *et al.* 2015). However, there is no direct link between these HGM approaches and the six-point criteria, mentioned above, used in delineating wetlands. (Nardi *et al.* 2006; Richardson and Vaithiyanathan 2009; Tiner 2016). This lack of linkage makes answering questions that would aid in monitoring and conserving wetlands under climate change difficult. For instance, whether wetland types will change (for example, from depression to valley bottom) or whether changes in climate drive changes in soil moisture (Reed *et al.* 2021). Another question is; will climate changes alter the wetland littoral vegetation if other variables like soil texture and chemistry change, *i.e.* bulk density and soil salinity? To this end, this thesis links paired vegetation and edaphic factor measurements in a coincidence of time and space with Sentinel-2A and hyperspectral remote sensing data. The thesis analyses salinity, bulk density, and soil moisture to understand their patterns and combined effects on vegetation. Hence, the thesis also looks at the response of vegetation to changes in edaphic factors that the HGM approach ignores.

Advances in wetland classification like those proposed by Vepraskas *et al.* (2000), Ollis *et al.* (2015) and others, seek to include wetland descriptors such as soil chemistry, texture, and detail on vegetation in the classification (Ollis *et al.* 2009; Sieben *et al.* 2018; Van der Valk 2020). However, there are no examples of how to implement these wetland descriptors over large scales. Some of these edaphic factors may affect plants simultaneously with soil moisture, *i.e.*, soil chemistry, *i.e.*, salinity and soil texture, *i.e.*, bulk density. Hence, some outstanding critical questions exist regarding the relationship between edaphic factors and vegetation along the littoral gradient (Martorell *et al.* 2021). For instance, which of the edaphic factors do C3 or C4 plants respond to or drive plant diversity, plant cover, or plant height along the wetland littoral zone? These questions include how edaphic factors interact when determining vegetation patterns along the wetland littoral gradient (Niu *et al.* 2021; Martorell *et al.* 2021). Alternatively, does the proportion of facultative to obligate wetland vegetation along the wetland gradient depend on soil moisture alone (Raulings *et al.* 2010)? Therefore, current classification techniques do not incorporate soil edaphic factors and vegetation over large scales (Clairain 2002). These questions include how edaphic factors interact when determining the pattern of vegetation along the wetland littoral zone. In other words, does the wetland threshold depend on soil moisture alone (Raulings *et al.* 2010)? Therefore, current techniques for classifying wetlands do not incorporate details on soil edaphic factors and vegetation over large scales.

Ecologists often use species taxonomy to study wetland vegetation patterns (Hu *et al.* 2015; Van Deventer *et al.* 2017; Van Deventer *et al.* 2019). This species approach has wide use, but similarities in niche environments of different species and the dissimilarity in characteristics within species confound many applications. On the other hand, plant functional traits (PFTs) in plant ecology have gained momentum as a multidimensional analytical alternative (Cornelissen *et al.* 2003; Raulings *et al.* 2010; Soudzilovskaia *et al.* 2013). This PFT approach establishes the relationship between niche environments and species by converting species characteristics into continuous values, thus alleviating the confounding from species characteristics (Cronk and Siobhan Fennessy 2001; Magee and Kentula 2005; Dwire *et al.* 2006). The PFTs have multiple dimensions because they enable the modelling of the interaction of sites, species, and environmental gradients simultaneously without being confounded by interspecific similarities (Garnier *et al.* 2016; Pérez-Harguindeguy *et al.* 2016; Pérez-Ramos *et al.* 2019). Hence PFTs are ideal for understanding plant distributions along the littoral gradient for perceiving niche environments or trait responses at the micro-scale. This understanding would lead to a better comprehension of how anthropogenic and climate changes affect wetlands. Therefore, we should consider PFTs as a proxy for detecting the wetland threshold and for classifying wetlands. There is little progress in implementing PFTs to measure plot level variability for monitoring the temperate grasslands' depressions. Hence this study investigates the likelihood of using PFTs to detect the wetland threshold.

The Geographical Information System (GIS) is useful in wetland monitoring. GIS supports the modelling or automation of wetland classification. GIS is also for outlining the perimeters of wetlands in modern digitising for wetland inventory (Walters *et al.* 2006; Rebelo *et al.* 2009; Melly *et al.* 2017; Van Deventer *et al.* 2020b). GIS modelling includes using contour lines and soil features as ancillary data. The aim is to use the ancillary data to predict areas with a high probability of being inundated after a rainfall event to assist the visual-search methods (Van Deventer *et al.* 2020a). The ocular search methods use eye-balling for differences in tone and texture of the image between suspected wetland and upland areas using aerial images and orthophotos as a backdrop. The idea is to identify areas with a high probability of being wetlands. The observer then traces the perimeters of wetlands using desktop digitisation. To further assist with heads-up digitising, a vectorisation of the backdrop image guides the perimeter line tracing. In desktop digitising, the observer draws a line along the apparent periphery of each wetland using a mouse cursor on a desktop. The GIS approach is adVantageous because very little training is required; hence, it suits developing an inventory in the data-scarce regions. At the same time, GIS approaches are an accessible alternative to manual field mapping conventions. However, GIS approaches alone do not provide a consistent and accurate approach for remotely delineating wetland boundaries and classifying wetland types. GIS also omits depressions on flat landscapes. The reason is that contour lines are not precise enough, *i.e.* not >0.4 m vertical accuracy. GIS can also distort boundaries (Van Deventer *et al.* 2020a). On the other hand, remote sensing is widespread for achieving superior results than GIS approaches alone due to greater detail from spectral and time-series features. GIS data does not have a five-day revisit date like remote sensing. The inferential application of both GIS and remote sensing depends on the coherence between in situ and computed data. Therefore, field data remains essential for validating and testing the reliability of the results from remote sensing data.

Remote sensing mainly analyses spectral reflectance to establish the relationship between spectral information on the image and ground surface (Taylor *et al.* 1995; Dini *et al.* 1999). The need for this type of information underscores the importance of developing remote sensing toolkits such as those from multispectral and hyperspectral remote sensing for ecological monitoring. Such toolkits will enable wetland ecologists to proactively detect instantaneous changes in wetland extents and types annually over regional and national scales. Multispectral and Hyperspectral remote sensing data meet these criteria. Sentinel-2 provides data over large geographical areas at 5-day intervals at plot level (10×10 m). Hyperspectral devices provide data at nanometre levels (1 nm) of spatial detail (Cho and Skidmore 2006; Cho *et al.* 2008; Main *et al.* 2011; Ramoelo *et al.* 2015; Sibanda *et al.* 2015). The objective of using hyperspectral data from edaphic factors in the current study is to test for its correlation with laboratory data from the same edaphic factors. Exploring this relationship tests the probability of substituting these expensive laboratory analyses with hyperspectral measurements as base tests the edaphic factors. Suppose hyperspectral and empirical edaphic factor data measurements correlate and have strong regression relations. In that case, that will mean hyperspectral remote sensing can substitute expensive laboratory analysis. Substituting laboratory analyses with hyperspectral data is desirable since laboratory analyses are expensive. When coupled with a lack of time efficiency, these challenges with laboratory analyses limit the application of level six of the South African wetland classification system. Level six includes classification based on salinity - the amount of salt in the soil, pH - acidity, and substrate - soil type (Ollis *et al.* 2009; Ollis *et al.* 2015). Such substitution or supplementation would make level six of the South African wetland classification system implementable on a national scale for the first time. However, laboratory analyses would remain critical for validation.

Multispectral and hyperspectral products are ideal for linking edaphic factors and PFTs to remote sensing. However, remote sensing studies on wetland delineation utilise the surface information without linking the sub-surface information of the landscape, which is opposite of ground-based approaches (Thenkabail and Nolte 2002; Schmidt and Skidmore 2003; Vaiphasa *et al.* 2005; Adam and Mutanga 2009; Adam *et al.* 2010). Hypothetically, this lack of linkage between surface and sub-surface layers limits the adoption and application of these remote sensing approaches in the mainstream of wetland ecology (Adam *et al.* 2010). Meanwhile, the relationship between soil edaphic factors and vegetation is key to understanding changes in the relationship between soil and primary productivity, plant diversity and vegetation structure (Rheinhardt and Faser 2001; Hájek *et al.* 2013). Rheinhardt and Faserd (2001) and Hájek *et al.* (2013) showed how differences in soil saturation condition lead to differences in the dominance of herbaceous versus shrub vegetation and bryophytes versus vascular plants. Therefore, the literature suggests that it would be hard to produce robust climate-sensitive wetland delineation based on vegetation alone (Adam *et al.* 2010).

Vegetation data alone, without soil information, cannot be the basis for wetland delineation aimed at protecting wetlands against climate changes. Vegetation indices can only predict future changes in wetland vegetation if we understand their relationship with edaphic factors. Therefore, assessing the spatial variability of vegetation at the plot level creates opportunities to link surface (vegetation) and subsurface (soil) data. Furthermore, the high cost of high-resolution satellite data has previously limited the implementation of nationwide monitoring of wetlands using remote sensing. For example, Landsat became the first freely available remote product in 2008, after costing about $200 in the 70s and $4000 in the 90s (Reichhardt 1999). Landsat has a 30-meter spatial resolution, which is not ideal for developing the assessment of wetland vegetation along the littoral gradient or linking vegetation patterns to soil conditions. These 30 m plots have a low spatial resolution to capture and describe the variability of vegetation along the littoral gradient of depressional wetlands. Especially in depressional wetlands, where the entire littoral gradient can start and end within 30 m (De Klerk *et al.* 2016). On the other hand, the variability of soil within a 30 m plot is too high to be considered a single unit.

Nevertheless, high spatial resolution images such as products from WorldView-4 have a 30-50 cm spatial resolution. These are ideal spatial resolutions for monitoring wetland littoral vegetation. However, their acquisition cost is about $22.5 per square kilometre, too expensive for supporting national monitoring (WorldView-4 DigitalGlobe Pty Ltd 2016). Hence, the recently freely available Sentinel-2 satellite data with its 10-20 m spatial resolution presents an opportunity to exploit this spatial resolution and its improved spectral resolution in the visible, NIR and SWIR regions of the electromagnetic compared to Landsat. Hence, there is a need for using Sentinel-2A data to explore its utility in assessing vegetation change along the narrow littoral gradient of depressional wetlands and linking it to edaphic factors. Theoretically, such a linkage could build a strong case for adopting remote sensing approaches in wetland delineation and classification mainstream. The persistence of broadband multispectral data among freely available satellite data has created challenges in adopting the results and recommendations into mainstream ecology and conservation applications. Coarse-resolution data is incoherent with the scale of wetlands with narrow boundaries. To our knowledge, no study has tested the utility of Sentinel-2 and hyperspectral data in characterising the wetland threshold and classify wetlands in temperate grassland regions. Notably, multivariate techniques are required to analyse multiple edaphic factors, PFTs, multiple indices derived from hyperspectral and Sentinel-2 images across sites simultaneously (Pakeman 2011). This thesis aims to examine how multispectral and hyperspectral reflectance correlates with edaphic factors and PFTs when delineating and classifying wetlands.

1.1 Research approach

1.1.1 Wetland thresholds and gradient analysis

This study links to the theoretical perspective of tipping points. The advent of climate change has sparked heightened interest in the boundaries between ecosystem types. The shift in the boundaries of many ecosystems due to climate changes is a consistent prediction, *i.e.* changes in limnetic regions (Figure 1.1). This prediction suggests that boundaries between ecosystems are changing (Neilson 1991; Prado *et al.* 2021). These predictions drive these heightened interest boundary shifts (Chakraborty *et al.* 2013; Duker *et al.* 2015; Piers *et al.* 2020). An even more critical perspective is that of "limited ecosystems shifts" or the concept of Thresholds of Potential Concern (TPCs) (Harley *et al.* 2017; Dakos *et al.* 2019). This concept of TPCs can also frame some of the current research questions in wetland ecology.

For instance, is the boundary between the palustrine region of a; naturally, there would be a gradation between them in a very short distance (Figure 1.1). This concept applies across the vertical cross-section of depressional wetland ecosystems (Figure 1.1). Water or underwater temperature and light thresholds exist along the vertical cross-section of wetlands. Gradient analysis was used in this study to investigate TPCs. The concept of TPCs stipulates that all living organisms have environmental thresholds that delineate the conducive conditions within which they can thrive. The same goes for ecosystems; hence biomes are distributed keenly along with distributions of climatic regions. Thresholds of edaphic factors along the horizontal cross-section of a depressional wetland result in local conditions suitable for specialised species, *i.e.* littoral zone (Figure 1.1).

Along with this distance from the wetland water body to the dry land, there is a threshold beyond which environmental conditions are no longer suitable for wetland vegetation to persist. Dryland vegetation begins to dominate and vice versa. Hence, the belt transect method was preferred because the intention was to sample the resource gradient from the water body to the dryland. This thesis experiments on novel gradient analysis of edaphic factors, vegetation traits, spectroscopy and multispectral data using the belt-transect method, test the critical zone or the TPC of wetland.

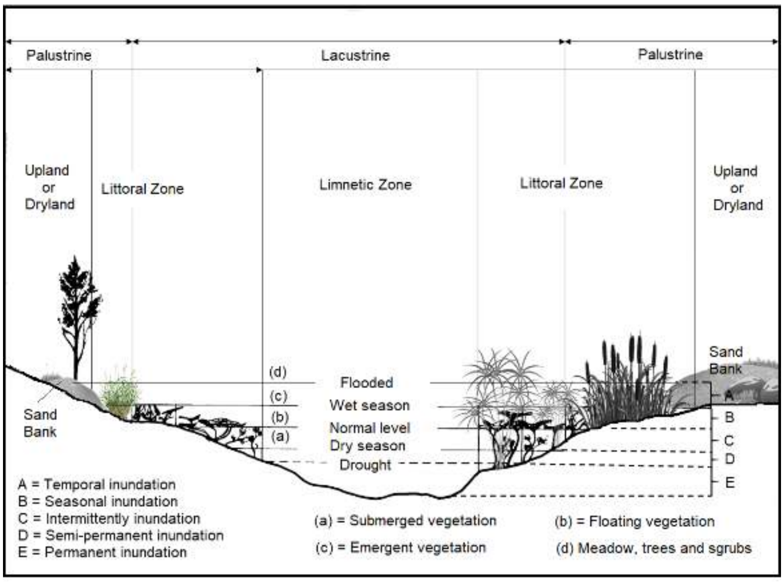

Figure 1.1: Vertical cross section of a depressional wetland ecosystem, showing vertical and horizontal gradients and thresholds.

1.1.2 The Earth's critical zone in wetlands "critical wetland zone"

Earth's critical zone is the complex of environments near the Earth's surface, above and below ground, where biogeochemical cycles interact. It is where living and non-living things regulate each other naturally and sustain life on Earth. (Lin 2010). This fragile skin has become known as the Critical Zone (CZ) because it plays a critical role in natural and managed ecosystems on Earth (Fig 1.2). A mixture of biotic and organic matter and abiotic and other Earth materials characterises the critical zone. The critical zone boosts the endless flows and fluxes of nutrients and energy that sustain the terrestrial ecosystems through catalysis by biotic organisms and chemical reactions of abiotic materials (Chorover *et al.* 2011). The critical zone also has an abundance of environmental gradients that develop owing to high-temperature mineral assemblages in rocks that re-equilibrate with fluids of the critical zone (Anderson *et al.* 2007; Anderson *et al.* 2013). These gradients represent energy and resources that eventually support humans in the form of ecosystem services (Chorover *et al.* 2011). The horizontal and the vertical gradients of the critical zone differ depending on the ecosystem (Fig 1.2).

Similarly, variables of importance also differ depending on the ecosystem. In wetland ecosystems, moisture, salinity and bulk density are critical variables of the critical wetland zone (Banwart *et al.* 2017). The vegetation on the horizontal gradient of the critical wetland zone is also an important variable because it affects the above-mentioned edaphic factors since they affect the vegetation (Brantley *et al.* 2017). Monitoring these interactions between the biotic and abiotic components of the critical wetland zone (Fig 1.2.) can help understand and protect wetland ecosystems and the ecosystem services they provide.

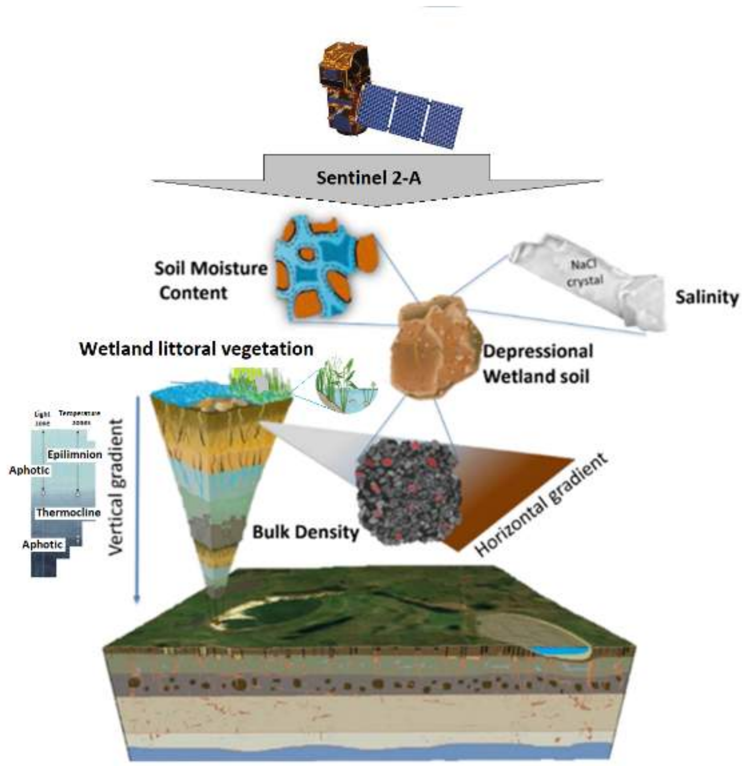

Figure 1.2: Cross-section of the Lake Banagher Farm critical zone. Showing the variables of the vertical and horizontal gradients, including those selected in the study

1.2 Research aims and objectives

This thesis aimed to develop monitoring toolkits for wetlands. The thesis investigates two aspects of monitoring: the delineation of the outermost perimeter of a wetland and its grouping. In this regard, the thesis aims to develop monitoring tools that address critical gaps in linking remote sensing results across two wetland domains related to wetland ecosystem functioning. These domains are the soil and the vegetation domains. This part of the aim requires careful consideration and selection of edaphic factors that drive wetland soil functioning. Careful consideration and selection of plant functional traits because they mediate wetland vegetation functioning in response to changes in the soil. Careful consideration and selection of remote sensing indices related to the selected variable in these two domains of wetland functioning. Furthermore, the thesis aims to address these emerging tools for monitoring wetland ecosystem function to climate change. The delineation and grouping need to monitor the wetlands for response to climate change at a functional level. This part of the aim requires careful consideration and selection of variables mechanistically related to changes in rainfall and temperature — the manuscript links across the three domains.

Moreover, the thesis aims to present experimental evidence to support the hypothesis that remote sensing can delineate and group wetlands at 10 × 10 spatial resolution. This part of the aim requires using the vegetation layer, the main subject of space remote sensing and Sentinel-2 data. At the same time, it requires the use of hyperspectral remote sensing, which can provide the required soil data needed to explain the variations observed from space-borne remote sensing. This part of the aim involves using the time and space coincidence experimental design to synchronise or pair the experimental data across the three domains to build the desired linkages. Furthermore, this part of the aim requires cross-validation of the results incrementally across the experiments. This part of the aim requires validation of the results from the first experiment in the laboratory at high levels of accuracy. Then after that, the results from the second experiment on vegetation undergo cross-validation with the results from the first experiment. Eventually, this aim subjects all the results from the four experiments or remote sensing and the two domains to incremental cross-validation with each other. The synthesis thereof must show sufficient spatial correlation to validate the emerging toolkits (novel approaches) and results and declare them acceptable in achieving the set aim. Therefore, this research aimed to test the utility of wetland plant functional traits, edaphic factors, and remote sensing in objectively delineating wetlands extents and grouping wetland (Ollis *et al.* 2015).

1.2.1 Aim and broad objectives

The aim of this research was to test the utility of wetland vegetation, soil moisture, bulk density and soil salinity (from descriptors at level 6 of the South African classification systems) in the objective delineation of wetlands extents and classification of wetland groups. Therefore the objectives, research questions as pursued in the four main experimental chapters of the thesis were:

1. To investigate the utility of wetland edaphic factors in characterising the wetland threshold, detecting the wetland boundary and depressional grouping wetlands.
2. To investigate the utility of hyperspectral remote sensing indices (proxies for wetland edaphic factors) in characterising the wetland threshold, detecting the wetland boundary and depressional
grouping wetlands.
3. To investigate the utility of wetland littoral vegetation functional traits in characterising the wetland threshold, detecting the wetland boundary and depressional grouping wetlands.
4. To investigate the utility of Sentinel-2A remote sensing indices (as proxies for littoral vegetation) in characterising the wetland threshold, detecting the wetland boundary and depressional
grouping wetlands.

1.2.2 Experimental design aims and narrow objectives

The experimental design was oriented around four general objectives which were investigated as 16 narrow objectives. These were as follows; to investigate the utility of :

1. Wetland edaphic factors (soil moisture, salinity and bulk density)
2. Wetland plant functional traits of littoral vegetation
3. Hyperspectral remote sensing of soil with indices
4. Sentinel-2A remote sensing of vegetation with indices

In:

A. Grouping or classifying wetlands according to variables of edaphic factors and vegetation
B. Characterizing the patterns of the variables along the littoral zone or wetland boundary
C. Detecting the threshold between wetland to dryland
D. Estimating the width of the wetland littoral zone

Objective Matrix	A	B	C	D	Paper Chapter
1	A1	B1	C1	D1	Chapter two
2	A2	B2	C2	D2	Chapter three
3	A3	B3	C3	D3	Chapter four
4	A4	B4	C4	D4	Chapter five

1. Target data sources:

a) Soil data
b) Vegetation data
c) Sentinel-2A
d) Hyperspectral

2. Target variables:

1. Laboratory Soil Bulk Density (SBD)
2. Laboratory Soil Salinity as Electric Conductivity (S/EC)
3. Field – Lab Soil Moisture Content (SMC)
4. Field – Lab Vegetation Moisture Content (VMC)
5. Field – Lab Dry Biomass Weight or Above Ground Biomass (AGB)
6. Field Vegetation Species Richness (VSR)
7. Field Leaf Angle Distribution (LAD)
8. Field Leaf Clumping Index (LCI)
9. Field Leaf Area Index (LAI)
10. Hyperspectral Soil Composition Index (SCI)
11. Hyperspectral Normalised Difference Salinity Index (NDSI)
12. Hyperspectral Misra Soil Brightness Index (MSBI)
13. Hyperspectral Normalised Difference Water Index (NDWI)
14. Multispectral Normalised Difference Vegetation Index (NDVI)
15. Multispectral Normalised Difference Salinity Index (NDSI)
16. Multispectral Red-edge Normalised Difference Vegetation Index (RENDVI)
17. Multispectral Normalised Difference Water Index (NDWI)

1.3 Thesis outline

The main body of the thesis contains four technical chapters, which are presented in paper format. This format means that each of the four chapters has sections, *i.e.* abstract, introduction, methods, results, discussion, conclusion and the reference list. The reference list for the general introduction and concluding chapters is placed at the end of the thesis. The general background, which introduces key concepts, the research approach and objectives, are outlined in the current chapter (**Chapter 1**). **Chapter 2** investigates the utility of wetland edaphic factors (*i.e.* Soil moisture, Soil Salinity and Soil Bulk Density) for delineating the wetland boundary and estimating the size of the wetland threshold using gradient analysis and possibilities of grouping depression wetlands according to similarities and differences in wetland edaphic factors. **Chapter 3** investigates the utility of wetland functional traits of littoral vegetation for delineating the wetland boundary and estimating the size of the wetland threshold using gradient analysis and also possibilities of grouping depression wetlands according to similarities and differences in wetland vegetation functional traits. **Chapter 4** investigates the utility of hyperspectral remote sensing analysis of littoral gradient soils for delineating the wetland boundary and estimating the extent of the wetland threshold using gradient analysis and also possibilities of grouping depression wetlands according to similarities and differences in wetland Hyperspectral remote sensing soil indices. **Chapter 5** investigates the utility of vegetation indices derived from Sentinel-2 images for delineating the wetland boundary and estimating the extent of the wetland threshold using gradient analysis and also possibilities of grouping depression wetlands according to similarities and differences in vegetation indices Chapter 6 is the synthesis chapter. The observations made in the above chapters are president in the context of the whole thesis, and key conclusions are outlined.

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Chapter 2

Determining the wetland-dryland boundary of depressions using littoral gradient analysis of soil edaphic factors

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WETLANDS CONSERVATION

Determining the Wetland-Dryland Boundary of Depressions Using Littoral Gradient Analysis of Soil Edaphic Factors

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Abstract

Depressional wetlands are highly vulnerable to changes in land surface temperature and rainfall. Changes in climate alter the spatial variability of edaphic factors, but this variability in endorheic wetlands is not well-known. This study aimed to assess spatial variability of edaphic factors between wetlands and along their littoral gradients; from the centres of the wetlands to the outer dryland. A sample of 202 paired measurements of three edaphic factors were collected (Soil Moisture Content – SMC-g/g, Bulk Density – BD-g/cm ^{3} and Salinity as Electric Conductivity – EC-dS/m) in 10 m plots along 14 belt transects in eight representative wetlands in the Mpumalanga Lake District, South Africa. In general, there were significant differences between the eight wetlands for SMC and BD but not for EC, at Bonferroni adjusted p–value (0.001). SMC and BD were important in differentiating the eight wetlands. SMC and BD generally showed negative trends along the littoral gradients. The trends occurred over short distances, ranging from 30 to 70 m, reflecting the extent of the wetlands. Understanding of the spatial variability of edaphic factors helps in the management and monitoring of depressional wetlands in the era of climate change.

Keywords: *Africa, Boundary, Depression, Edaphic Factors, Soil Moisture, Threshold, Wetland-Upland*

2.1 Introduction

Wetlands are important, because they are unique ecosystems, fragmented in occurrence, with specially adapted biodiversity, yet crucial to livelihoods and biogeochemical cycles (Euliss *et al.* 2006; Marton *et al.* 2015). Globally, wetlands cover 12.1 million km^2 (1.2 billion ha), and only 6–7% of global land surface (Díaz *et al.* 2019). About 54% (6534 km^2) are permanently inundated and 46% (5.566 km^2) are seasonally inundated (Gardner and Finlayson 2018). About 30% of wetlands globally are in arid and semi-arid areas (Melton *et al.* 2013). Wetlands in arid and semi-arid areas are isolated depressions that are the major sources of water in these areas . Isolated depressional wetland ecosystems provide vital ecosystem services; fulfilling important hydrological and biogeochemical functions that support both biodiversity and human livelihoods (Scholes and Archer 1997; Tooth and McCarthy 2007) (Ramoelo *et al.* 2012). Wherever these dryland wetlands are, they are typically characterised by shallow basins without permanent open water *i.e.* palustrine or with permanent open water *i.e.* lacustrine, or a combination of the two (Verrecchia 2007). Examples of depressional ecosystems include the Mpumalanga Lake District (MLD) in Africa, the Great Lakes region in east Africa, the Prairie Potholes Region (PPR) in the Great Plains of Canada, the Drew Point coast of Alaska, and the area of Orlando in Florida in the United States (US) or the Poyang Lake region in China. Lacustrine depressions typically have a narrow fringe of intermittently inundated vegetation, while palustrine depressions have saturated soil and wetland vegetation that covers the entire depression. Despite their importance and uniqueness, depressional ecosystems are threatened by global environmental change (Junk *et al.* 2006). Due to global change, about 81% of inland wetland biodiversity has been lost since 1970 (Gardner and Finlayson 2018). Meanwhile, global wetland extents are in decline (Díaz *et al.* 2019).

Isolated wetlands in drylands are naturally dynamic, but are also highly vulnerable to global environmental change including climate variability and change (Tiner 2003). For instance, fluctuations in the wetland SMC and water body are naturally occurring, whether between rainfall events, between seasons, or between larger inter-annual cycles (Euliss and Mushet 1996). Climate change may alter these natural fluctuations by increasing land surface temperatures. Global surface heating is causing higher evaporation and transpiration rates in isolated wetlands. In turn, high evapotranspiration reduces wetland inundation and depletes wetland SMC in the narrow terrestrial fringe zones over the long term, especially during drought events. Consequently, different wetland habitats might lose their requisite moisture, and thereby their suitability as habitats to wetland fauna and flora. On the other hand, depressional wetlands can exacerbate climatic warming if they get degraded. The drying and cultivation of wetlands has lowered wetland Carbon (C) sequestration capacity and accelerated carbon oxidation which has increased the atmospheric Carbon dioxide (CO_2) content, thus increasing the greenhouse effect (Euliss *et al.* 2006; Parry 2007; Reeves I 2014; Ronan *et al.* 2020). Explicitly, historical (1976–2004) Methane (CH_4) soil emissions and C sequestration rates were on a steady state that had no radiative forcing on climate (Bartlett *et al.* 1989; Altor and Mitsch 2008; Melton *et al.* 2013; Gardner and Finlayson 2018). However, in the advent of higher global surface heating, the steady state rates in CH_4 emissions have become positive and accelerated, thus increasing the greenhouse effect (Maljanen *et al.* 2002; Chen *et al.* 2013). Hence, the intercontinental monitoring of spatial variability of edaphic factors in these shallow isolated depressional wetland ecosystems with narrow banks is crucial in the era of climate change (Melillo *et al.* 1993; Chattopadhyay and Hulme 1997; Parry 2007).

Despite the vulnerability of depressional wetlands to climate change, monitoring that is specific for the effects of climate change on isolated depressional wetlands is lacking (Euliss and Mushet 1999). Specifically, the spatial variability of edaphic factors in isolated wetlands is not well established (Corwin and Lesch 2005; Bruland and Richardson 2005). Meanwhile, detecting and monitoring the zonation around depressional wetlands over large areas is desirable for effective wetland monitoring of climate change using remote sensing. However, beside hydrology several edaphic factors may be candidates for depressional wetland monitoring *i.e.* BD and salinity. Therefore, interpretation of remote sensing results on depressional zonation requires an understanding of the spatial variability of collective edaphic factors that might drive wetland zonation namely SMC, salinity and BD, and their interactions (Wang *et al.* 2018). Alternative means of understanding the spatial variability and interactions of edaphic factors is by first describing their within and between wetland variability (Niemuth *et al.* 2010). It is therefore specifically important to monitor the pattern of edaphic factors along gradients of wetness in isolated wetlands in the context of climate change.

To achieve effective wetland monitoring at local and regional scales, classifying different zones and habitats in wetland areas is the prerequisite. Hence, they are core of the RAMSAR framework for wetland inventory and ecological character description recommended to RAMSAR signatories. However, accurate delineation of wetland boundaries and zones as well as further grouping of wetlands beyond HydroGeoMorphic type units (HGMs) remain as major challenges for countries that are signatory to the RAMSAR (Euliss and Mushet 1996; Niemuth *et al.* 2010). In the advent of climate change an additional challenge is that monitoring must be relevant to climate change. Owing to strong gradients of micro-elevation, soil texture, soil chemistry, and SMC within isolated wetlands (edaphic factor zonation); commonly a gradual differentiation between wetland plant communities along the zonation gradients can be observed (Castelli *et al.* 2000; Egan and Ungar 2000; Kotze and O'connor 2000; Lyon and Lyon 2011). The spatial variation in moisture regimes is considered a major determining factor of this wetland zonation. Flood-sensitive species disperse further away from the water due to low tolerance to flooding, while flood-tolerant species occur on the fringes of the wetland and inward (Raulings *et al.* 2010). However, despite the importance of hydrological conditions, it is not known how other edaphic factors (salinity and BD) interact with hydrology in determining the zonation (Tieszen *et al.* 1979; Kotze and O'connor 2000; Castelli *et al.* 2000; Egan and Ungar 2000; Kotze and O'connor 2000; Lyon and Lyon 2011; Tiner 2017; Li *et al.* 2018). Hence, two questions arise; how do these edaphic factors interact when determining wetland zones and boundaries? Secondly, can analysis of edaphic factors assist to distinguish lacustrine from palustrine endorheic wetlands within the depressional wetland HGM units?

On the other hand, the baseline patterns of BD along the boundary zones of the depressional wetland HGM unit are not well understood or are unknown for specific areas (Euliss and Mushet 1996). The BD refers to the ratio of the volume of a soil sample to its mass . The pattern of BD along the terrestrial wetland gradient is an important edaphic factor for monitoring changes in wetland function. BD varies with soil structural conditions, and increases with increase in the depth of the soil profile, due to changes in organic matter content, porosity and compaction (Richardson and Vepraskas 2001; Clarkson *et al.* 2003; Rokosch *et al.* 2009; Li *et al.* 2018)). These soil characteristics can be altered by strong water currents. Strong water currents can be caused by flooding events and strong seasonal oscillations in the water levels (Euliss and Mushet 1996). Flooding and oscillations can result from extreme rainfall patterns *e.g.* heavy rain and droughts (Richardson and Vepraskas 2001; Clarkson *et al.* 2003; Rokosch *et al.* 2009; Li *et al.* 2018). To this end, we hypothesize that the BD is highest at the fringes and boundary of depressional wetland zones where most deposition occurs, which could help in the empirical detection of the wetland boundary. Soil salinity is another important edaphic factor that is affected by changes in the amounts and patterns of drought and rainfall (Rogel *et al.* 2000; Xi *et al.* 2016). Heavy rainfall reduces the salinity of the wetland water body. Drought and an increase in evapotranspiration, on the other hand, will increase the salt concentration of the wetland water body (Adams and Bate 1995). Therefore, changes in rainfall and drought patterns would affect the salinity of wetlands (Winter and Rosenberry 1995; Brooks 2000; Rogel *et al.* 2000; Jolly *et al.* 2008; Trites and Bayley 2009). At the fringes and boundary of depressional wetland zones where most deposition occurs, the salt content is expected to be highest, which helps to empirically detect the wetland boundary (Euliss and Mushet 1996). The measurement pattern of EC as a proxy for changes in salinity along depressional wetland gradients can thus be useful for monitoring changes in wetland function in a changing climate (Euliss *et al.* 2001; Bird *et al.* 2013; Sieben *et al.* 2016).

This raises the question, whether the three edaphic factors can be used to investigate and detect the boundary between the endorheic wetlands and upland zones – the "wetland threshold" or "wetland-dryland boundary". Of the few studies that have explored depression wetlands in the African region none of them looks at the variation in edaphic factors such as SMC, salinity and BD among different depressional wetlands in the MLD. We hypothesize that the detection of the wetland boundary can be achieved using edaphic factors. The aim of this study was to establish the within-wetland and between-wetland variability in SMC (g/g), EC (dS/m) and BD (g/cm ^{3}). These three edaphic factors were sampled along several belt transects in eight representative depressional wetlands and used to *(i)* assess potential differences in edaphic factors between wetlands and *(ii)* analyse trends in the edaphic factors from the open water body (center of the wetland) to the outer dryland.

2.2 Material and Methods

2.2.1 General methodology

To ensure repeatability of the research the study was conducted systematically and all the critical steps were recorded. The general methodology (Figure 2.1) includes satellite remote sensing data, which was used to guide the process of selecting sample plots, transects, wetland sites and ecological and data science principles. The alignment of the data collection with remote sensing ancillary data ensured further repeatability because remote sensing data is publicly available (Appendix B). Therefore the exact sample locations where these data were collected can be retrieved by subsequent researchers.

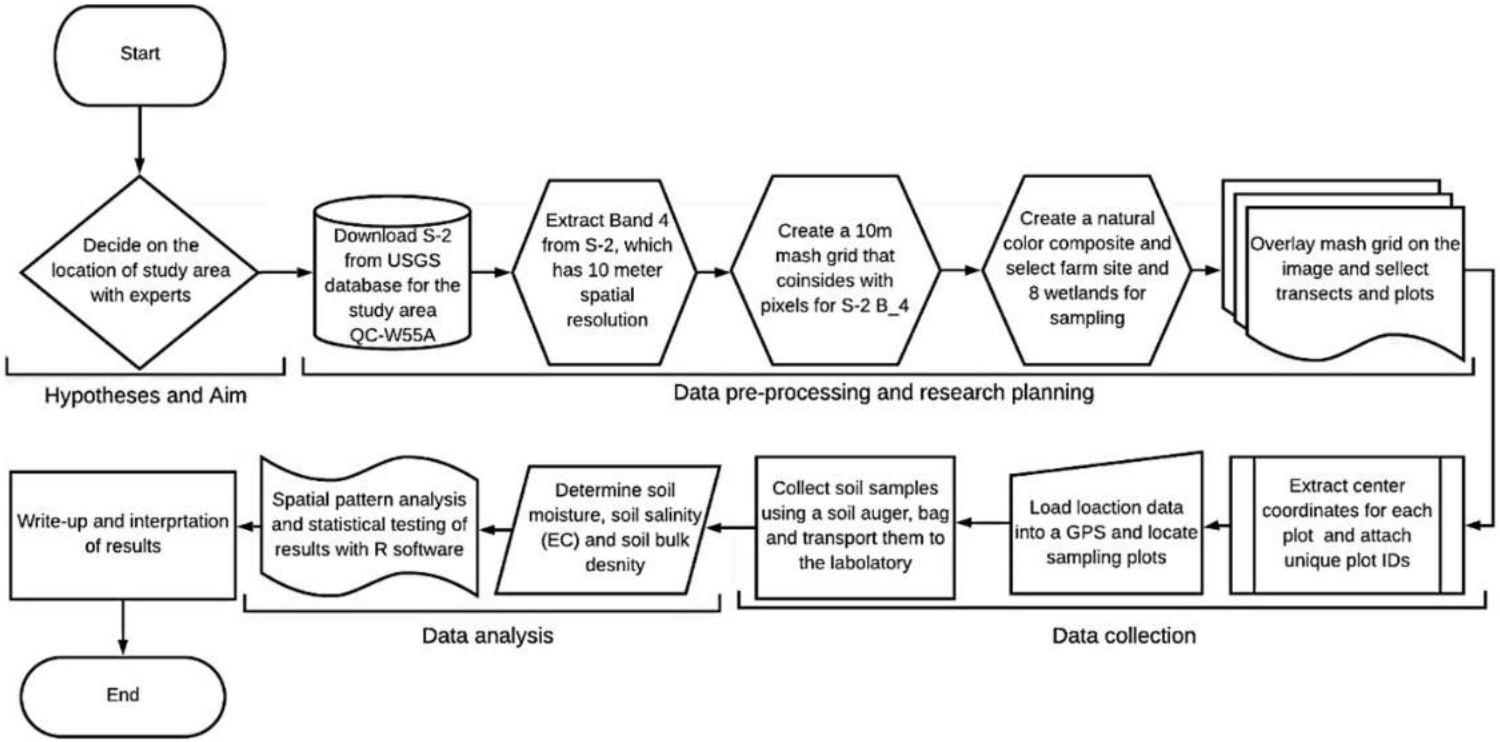

Figure 2.1: The summary overview of the methodology followed in the research. Quaternary Catchment (QC) W55A is a unique number naming conversion used by the former South African Department of Water Affairs and Forestry (DWAFF), now Department of Water and Sanitation (DWS), given to the subject quaternary catchments. Sentinel-2A (S-2), Band 4 (B4), Identification (ID), Global Positioning System Device (GPS), Electric Conductivity (EC)., R is a free software environment for statistical computing and graphics (RCoreTeam, 2019). Landsat 8 data was also download from United States Geographical Survey (USGS) followed by computation of Normalized Difference Vegetation Index that would assist in the selection of sites. The formulation of the hypotheses and aims, with the main activity being deciding on the location of the study site with experts. The second section was on data pre-processing and research planning followed by data collection and then data analysis. The fifth and final step was the interpretation of the results and the writing up.

2.2.2 Selection of study area

The MLD was chosen as a study area because the area is rich and diverse in different types of depressions (and other wetlands) and therefore can be a good case study for isolated wetland ecosystems globally like the PPR of the US. The geology is underlain by a sequence of two strata. First, is the Ecca group; a topping of sedimentary deposits consisting mostly of shale and sandstone and the Dwyka Group below in the stratigraphic position. The catchment receives 767 mm of mean annual precipitation. W55A has over 300 depressional wetlands in just a 20-odd kilometre radius (Goudie and Thomas 1985; Van Deventer *et al.* 2020; Van Deventer *et al.* 2020). Within MLD a subset of depressional wetlands were selected (Appendix A, Lake Banagher Farm, 26°20'11.21"S, 30°21'14.03"E, in the Gert Sibande District, in the Msukaligwa Local Municipality, Mpumalanga Province, South Africa, Figure 2.2). The wetland ecosystem types and the wetland vegetation are very diverse due to variations in elevation, size, shape and the area of the vegetated zones. Therefore, there is a good chance of covering a wide range of habitats in a relatively small area (Watson 1986; Brooks and Hayashi 2002; El-Kawy *et al.* 2011; Wilkinson *et al.* 2016; Vanderhoof *et al.* 2018).

2.2.3 Selecting depression wetlands for sampling

Eight depressional wetlands (Appendix A) were selected to represent the range of depressional habitats. The diversity of wetlands was observed in terms of (a) extent of the water body, (b) extent of vegetation cover (c) shape and (d) size.

2.2.4 Field surveys design and the belt-transect method

Two field surveys were conducted in order to sample wetland SMC, soil salinity and soil BD. The first survey was conducted in March 2018 and the second was conducted in November 2018. The first survey focused on two wetlands and the second one focused on six wetlands. Only two wetlands had data collected in both sampling periods.

Field survey design

The sampling procedure was based on the belt transect method according to the Sentinel-2A pixels scheme. Sentinel-2A provides data with global coverage in a cycle of about once every 5 days from above the equator. In addition to near infrared and shortwave infrared bands, it has three red-edge bands (Bands 5-7 with the center of the bands at 705, 740 and 783 nm respectively) which has been proven useful in vegetation classification and possibly for edaphic factors. The intention was to cover the range of variation in the visible vegetation physiognomy from the edge of the vegetated part of the wetland up to the dryland area that is bordering the wetland (Figure 2.2). The best location for transects was considered to be the region of the wetland-dryland gradient that had the highest turnover in pixel tone (colour variation). A high variation of pixel tone was considered to reflect higher turnover in species or vegetation structure or both. At each wetland, a field survey was conducted using the belt-transect method. The belt transect method was preferred because the intention was to sample a longitudinal gradient from the water body to the dryland. The width of transects was 10 m as determined by the spatial resolution of Sentinel-2A. Transects had varying lengths, dependent on the width of the wetland zone (30 m - 130 m).

Setting up the belt transects

A mash grid made up of contiguous 10 m plots following the rows of Sentinel-2A pixels was generated in ArcGIS (ArcMap 10.5). This grid was projected on a true colour composite of the Sentinel-2A in order to identify the best locations for transects, following the approach by Goodman (1990).

For wetlands greater than 0.2 km^2 (which were the three largest wetlands, Appendix A), three transects were selected around each wetland (Figure 2.2). For the smaller wetlands only a single transect was sampled. Plots with similar vegetation structure and composition as the one preceding them were not repeated. The purposeful sampling ensured that the sampling maximised the efficiency of representative sampling of landscape features and avoided repetitive sampling. The length of a transect was limited by the fence or by reaching dry ground.

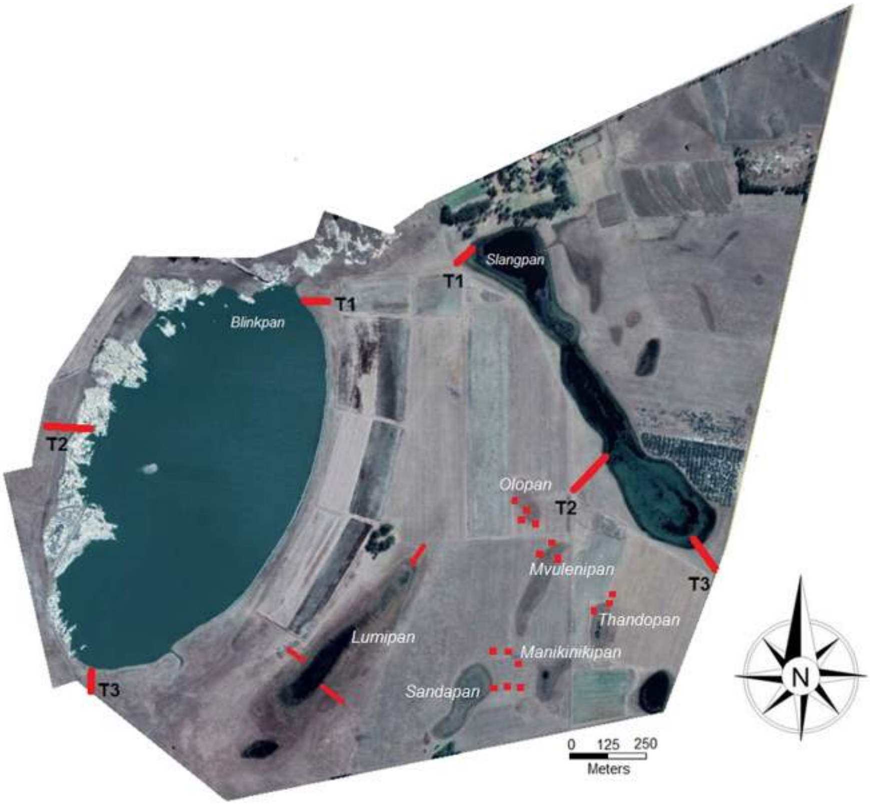

Figure 2.2: A map showing the positioning of the sampled transects numbered T1 to T3 (red coloured 10 × 10 m belt transect of plots 100 m^2). The image has been clipped to the shape of the current boundary of the Lake Banagher farm. The map was created using World Imagery DigitalGlobe sub-meter resolution (0.5 m), which provides low spatial resolution 15 m, high spatial resolution 60 cm and high spatial resolution 30 cm TerraColor imagery on Red-Green-Blue colour composite at small and mid-scales (1:591m down to 1:288km).For more information and terms of use, visit http://goto.arcgisonline.com/maps/WorldImagery. Dataset created December 12, 2009 and last updated June 2018.

Sampling at plot level

In each plot two subplots were sampled. We selected random plots by tossing a 0.25 m^2 quadrat into the main 100 m^2 quadrant from two of its corners. In each subplot samples of three edaphic factors were collected, EC, SMC and BD. During sampling, caution was taken to avoid roots and any roots that were found in the soil sample were removed in order to prevent them having an effect on the SMC measurement.

Sampling at subplot level

Inside each 0.25 m^2 quadrat, one sub-sample was collected from the top soil (top 15-20 cm) using a Johnson's Soil Auger. The volume of the soil core was calculated from the dimensions of the cylinder; 8.5 cm in diameter and 16 cm in height. The two sub-samples were sealed in zip-lock bags and transported to the laboratory in order to determine:

- Wetland SMC in the topsoil (first 15-20 cm)
- Wetland soil salinity as determined by EC using water saturated soil waste extractions.
- Soil BD using the ratio of soil weight to soil volume of a soil core of known dimensions.

2.2.5 Selecting edaphic factors

Three edaphic factors were selected (Appendix E), namely; SMC, soil salinity and BD.

2.2.6 Measuring soil moisture, salinity, bulk density

Soil moisture content (g/g)

We applied the gravimetric method, which aims to measure the mass difference between wet mass and dry mass of a soil sample. The mean SMC of two or three sub-samples was calculated and reported in gram per gram (g/g) as the plot level (100 m^2 quadrat) SMC. The gravimetric method is the only direct method of measuring SMC (Reynolds 1970). At the laboratory, each wet subsample was removed from the zip lock bag into a foil tray and reweighed, to obtain the lab wet-weight relative to field wet-weight to assess the quality of transportation and storage of samples and to guarantee null processing effect. The foil trays were then placed in an oven at 80 ºC to the dry soils for 48 hours (Labotec oven, the Term-o-mat model with a temperature range of 30-250 ºC). After drying each sub-sample was weighed again to obtain the dry weight before sieving. During sieving the amount of root mass was determined and found to be negligible although its mass was subtracted from the mass of sieved samples for greater accuracy of BD.

Soil bulk density (g/cm^3)

Dry BD is the mass of soil particles per unit bulk volume of soil (Avnimelech *et al.* 2001; Chaudhari *et al.* 2013). The two variables used to calculate BD are the volume and the dry mass of an undisturbed soil core. The volume of the soil core was calculated from sieved soil samples. The dry mass of the soil core was obtained using a balance scale after drying the samples in the oven, before sieving. A precision ruler on the side of the cylinder was used to measure the height of the soil core (Jeffrey 1970; Harris *et al.* 2003).After oven drying the conventional equilibrium BD was corrected by removing coarser particles (clods and gravel larger than 2 cm) following the advice of (McKenzie *et al.* 2002) who observed over estimations of BD ranging 2.2–3.0 g/cm^3 due to coarse fragments. The use of standard or maximum compaction state to deal with the problem of porosity has proven useful. However, interpretation and measurements of absolute soil compaction is prone to error as opposed to expressing it in relative terms (Blackwell and Soane 1981). Sieving and free relative compaction were important to control porosity (Six *et al.* 2001). The soil was ground and passed through a small sieve (2 mm) using the standard method, resulting in pre and post sieving dry weight. Care was taken to ensure that there was no soil in the refuse after sieving. The refuse was checked to ensure that it included only roots, rocks or twigs. Furthermore all the soil samples were treated the same way *i.e.* sieved with the same size of sift using the same protocol (Carter 1990). Each soil core was weighed at free total field gravitational acceleration of 9.79041m/s2 for the location of City of Tshwane (25°45'10.78"S, 28°16'36.34"E) at 1320 meters above sea level (masl), since the center of gravity changes with change in geographical position.

Soil salinity as electric conductivity (dS/m)

Soil samples were analysed at the Agricultural Research Council for total dissolved salts using EC as a proxy for salinity with the soil saturated paste (SP%) extract methodology (McKenzie and Chomistek 1989; Kargas *et al.* 2018). For the saturation percentage determination (SP%), oven dried samples were ground and passed through a 2 mm sieving using the standard method. The 1:10 soil over water ratio was used to create a paste, where 1000 ml of distilled water was added to 100g of the sieved soil. The mixture was shaken for 1 minute by hand, 4 times at 30 min-intervals. The soil pastes were then left for 24 hrs to reach equilibrium. Subsequently, the suspended clear liquid extract was collected and tested for EC.

2.2.7 Data analysis

Density plots as well as box and whiskers plots were used to visualise and assess the variability of the three edaphic factors. Significance tests were conducted to assess the statistical validity of the results. All analyses (Table 2.1) were conducted using R version 3.6.2 (R Development Core Team, Contributors 2019). The Tukey Honest Significant Difference test, accounting for the Bonferroni effect was used to control for Type I errors in multiple comparisons. In order to test the significance of the hypothesis at = 0.05, the possible number of combinations or Bonferroni coefficient (m) for eight wetlands was m=28. The m value and new alpha level of 0.001 were calculated using the combination formula (Eq. 5). Where the default alpha level (0.05) is divided by m *i.e.* (nCr = n / r * (n - r)). Where n represents the total number of items *i.e.* 8, and r represents the number of items being compared at a time *i.e.* 2, to calculate the Bonferroni adjustment alpha level. Maps in this paper were created using ArcGIS® software by Esri. ArcGIS® and ArcMap™ used herein under intellectual property license, Copyright © Esri, unless otherwise stated. For more information about Esri® software, please visit www.esri.com (ESRI 2019).

Table 2.1: Summary table of analysis conducted in the study

Research objective	Objective of analysis	Visual Presentation	Statistical Test	Dataset used
A. To investigate differences in edaphic factors across wetlands	To assess whether there is variability in the means of edaphic factors across the eight sampled wetlands	Density plots using the R ggplot2 package	Bonferroni $\binom{n}{r}$ adjusted 8 Group ANOVA	Matrix of wetland ID by edaphic factor
	To assess whether there are significant differences in mean and median of the observed variability of edaphic factors across wetland sites	Box and whisker plots using the R base graphics package	Bonferroni $\binom{n}{r}$ adjusted Turkey´s HSD	Matrix of wetland ID by edaphic factor arranged as a matrix of 28 paired combinations of 8 groups
A. To investigate the interaction of edaphic factors in determining wetland function and their potential for wetland classification	To assess whether differences in edaphic factors across the wetland can be used to group the wetlands	PCA Ordination in R	Data modelling with Constrained Canonical Correspondence Eigen distance values	2 matrices; one on site ID by mean plot level edaphic factors
	To show the interaction among the three edaphic factors, and test the type and strength of the associations among the variables	R Scree and Factor plots	Correlation and Factor analysis	Matrix of edaphic factor constrained by site
B. To investigate the patterns of a edaphic factors along the wetland gradient and the existence of the wetland threshold	To assess whether the pattern of response for soil moisture and bulk density varies and for electric conductivity a positive slope along the wetland and gradient	R Line graphs and R poly lines	Polynomial regression coefficients	Matrix of wetland ID by site by edaphic factor
	To understand whether the response patterns of a edaphic factors show any thresholding that could suggest the position of the wetland threshold	Vertical line extraction R "abline´´ function	Error-based Threshold extracting	Matrix of wetland ID by site by edaphic factor
B. To investigate the differences in edaphic factors between the two sides of the wetland gradient threshold	To assess whether wetland edaphic factors have a flat slope across all wetlands or slope = to zero	Box and whisker plots using the R base graphics package	Hierarchical clustering with 3 edaphic factors	Matrix of wetland ID by edaphic factor by transect
	To assess whether there are difference among transects that have been plotted along the longitudinal gradient of the same wetland	Box and whisker plots using the R base graphics package	Student T-test	Matrix of wetland ID by edaphic factor by plot

2.3 Results

2.3.1 Differences and groupings in edaphic factor between wetlands

The mean values (Appendix H) of the eight wetlands in were significantly different in SMC ($F_{7,111} = 16.63$, $p< 0.000$) and BD ($F_{7,111} = 6.468$, $p< 0.000$) at a Bonferroni adjusted alpha level ($p<0.001$, One-way Analysis of Variance (ANOVA)) and not in EC ($F_{7,111} = 2.25$, $p>0.001$), as it can be observed in Figure 2.3 (box plots, results significance presented alphabetically). Tukey's Honestly Significant Difference (HSD) multiple comparisons of 28 comparable pairs showed that the significant difference ($p<0.001$) can be attributed to four pairs for SMC namely Blinkpan-Lumipan, Lumipan-Manikinikipan, Lumipan-Slangpan, Blinkpan-Thandopan and two pairs for BD namely, Blinkpan-Lumipan and Blinkpan-Slangpan (Appenndix F). The results from ordination analysis, conducted using all three variables, revealed three groups of wetlands (Principal Component analysis (PCA)). Group A biased towards high BD, group B biased towards high BD and SMC and group C biased towards high EC. The first two PCA axes were the most important latent variables that were highly correlated (78.71%) to SMC and BD. Therefore, ordination results further support the ANOVA findings of the importance of SMC (45.46%) and BD (21.29%) in differentiating the wetlands from one another (Figure 2.4).

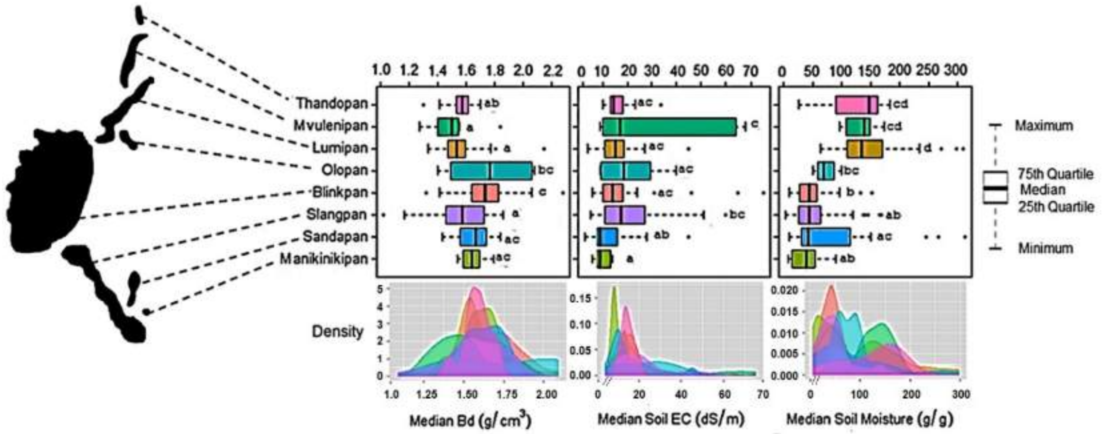

Figure 2.3: Median SMC, salinity and BD across eight sampled wetland sites (data that is combined by site). Data have been ordered according to the descending abundance of SMC. Wetlands that were sampled in the first survey are Blinkpan and Slangpan, while the rest were sampled in the second survey. Sample density distributions of the edaphic factors of the eight wetland appear below following the same colour scheme of the horizontal boxplots.

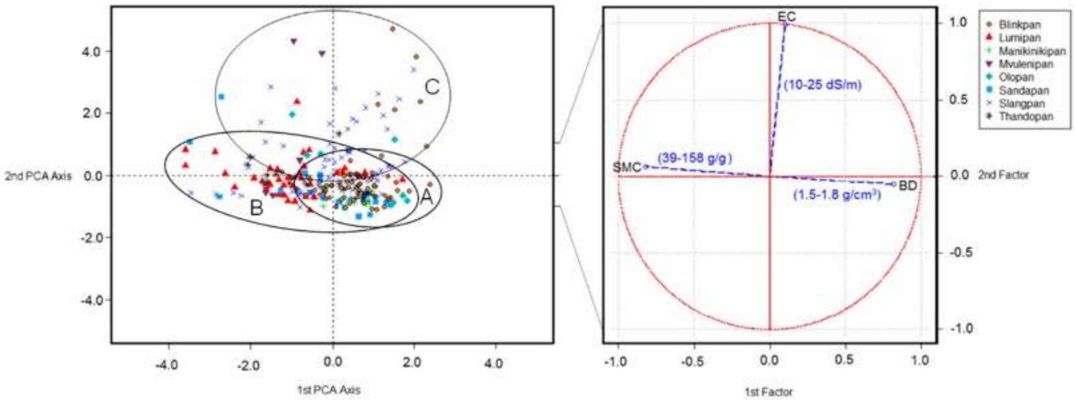

Figure 2.4: Eigenvalues of the correlation matrix for the three active variables shown as percentage influence of each variable (left). Projection of the variables on a 1×3 factor-plane (right).

2.3.2 Trends in the edaphic factors from the open water body to the outer dryland

Generally, there were negative trends in the response of SMC ($r^2$2=0.99-0.71) and BD (r^2=0.84-0.29) along the gradient from the centres of the wetlands to the outer dryland boundary, while EC had a positive trend – r^2=0.92-0.24 (Figure 2.5). However, at relatively short distances, ranging from 30 to 70 m, this probably reflected the extent of the palustrine section of the depressional wetlands (Figure 2.5 and Appendix G). When the data from the edaphic factors were combined across the gradients of all wetlands to drylands, BD, SMC and EC maintained their general trends (Figure 2.6). The end of the patterns of edaphic factors, which probably reflects the mean seasonal maximum extent of the wetland, at 70 m on average for combined data. The wetlands were significantly different in SMC from dryland (Appendix I) and the variance (Appendix N) in this difference might reflect a difference in wetland function. Site-specific comparisons (Appendix M – t-test) showed that this significant difference can be attributed to one of the three wetlands in comparison (Appendix F – Tukey's HSD and Appendix G – coefficients of the polynomial regression models).

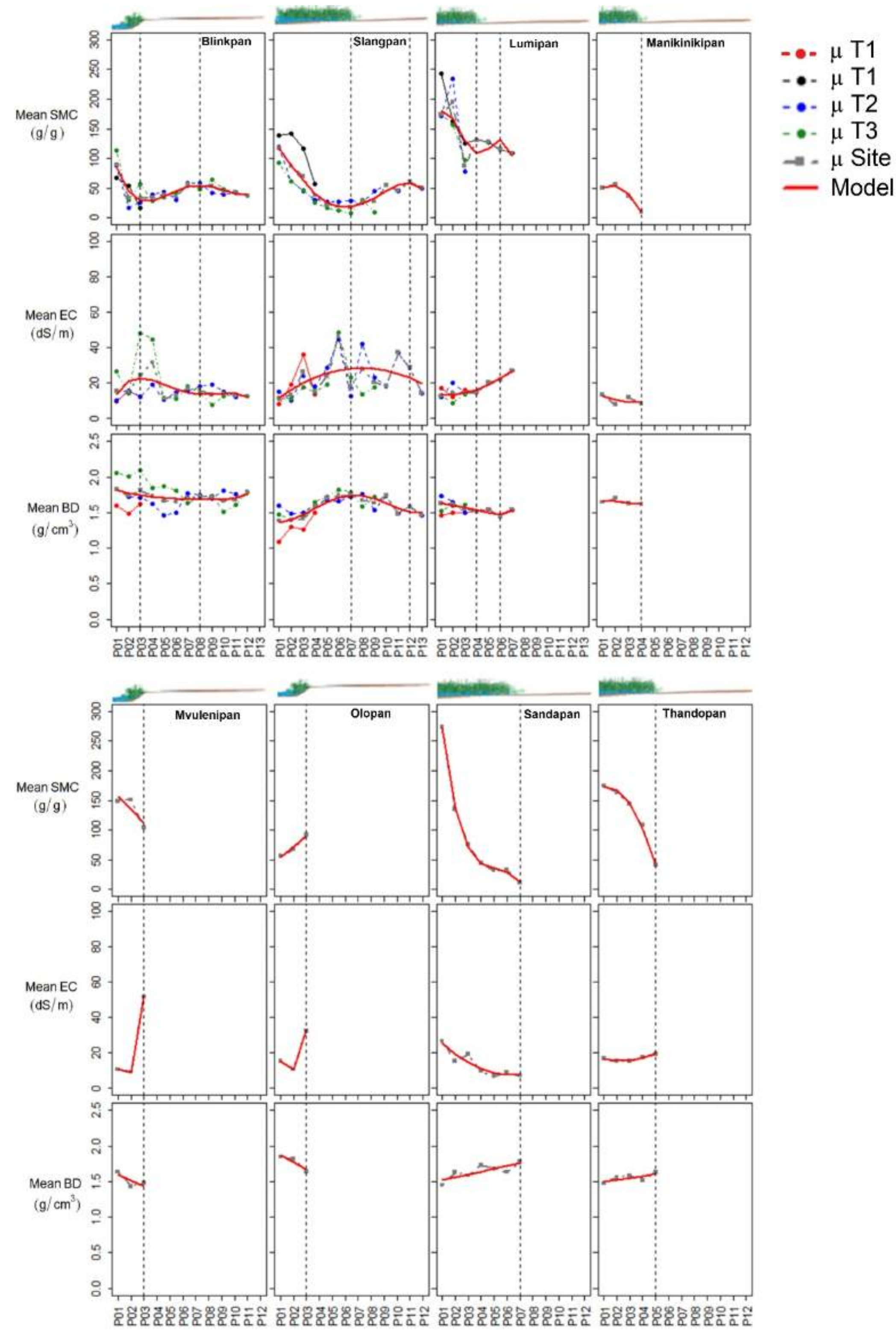

Figure 2.5: Statistics in Appendix G - Response of edaphic factors to increasing distance from the edge of the wetland water body. The vertical dotted lines represent points of drastic change in the pattern of edaphic factors along the gradient of increasing distance from the wetland water body.

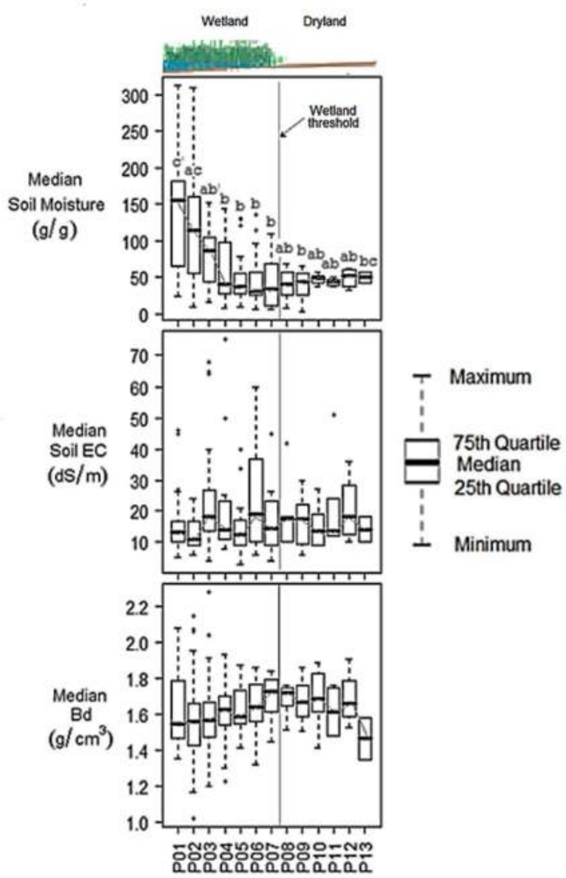

Figure 2.6: Median values of edaphic factors aggregated by plot number across the littoral gradients of the eight depressional wetlands. Showing differences in edaphic factors between the two sides of the wetland threshold. Plots with the same alphabets are not significantly different in edaphic factors and were plotted on the same branch or cluster of the hierarchical dendrogram.

2.4 Discussion

2.4.1 Differences in edaphic factors between depressional wetland

In this study, we investigated differences in edaphic factors among eight depressional wetlands in a temperate grassland biome. Our results showed that significant site level differences were detected in SMC (3/8 wetlands) and BD (2/8 wetlands), but not in EC (p<0.001). In a similar study, (Rogel *et al.* 2000) in the Mediterranean region of Southeast Spain used canonical correspondence analysis to relate the species distribution with certain soil conditions to classify wetlands into dry salt marshes and wet salt marshes. They found that the edaphic variable that best explained the data was maximum SMC. Their study however dealt only with sampled salt marshes, hence, salinity was an important criterion differentiating between the two groups of wetlands in the Mediterranean region. Our study showed that BD was important in grouping wetlands. Although nutrient distribution was not included in the study, it may play an important role as well because (Bai *et al.* 2005) found a significant relationship between nutrient distribution and edaphic factors *i.e.* SMC and BD. (Bai *et al.* 2005) found that SMC and BD (clay content) were important determinants of soil organic matter (SOM) and total nitrogen (TN), hence SOM and TN could be left out on the variable selection without perceived prejudice on the research outcome (Six *et al.* 1998; Six *et al.* 2001; Six *et al.* 2004). These results can be relied upon because these results were produced with data that were collected over two growing seasons and the results were consistent between the sampling periods. This means that insignificant statistical differences in the values of edaphic factors among the wetlands showed no effect of sampling time. There is a need for further research on the seasonal time-series of edaphic factors in the littoral zone of depressional wetlands (Niemuth *et al.* 2010). We used principal component analyses to assess the statistical differences in edaphic factors across different scales; site and plot levels. In ordination the orthogonality of latent variables achieves direct multiple comparison of similarities between edaphic factors across all plots and sites simultaneously.

We tested similarities in edaphic factors across the wetland sites. The results on differences in edaphic factors across the eight wetlands showed that although the wetlands differ in characteristics that partly affect or drive the edaphic factors, there is still convergence or grouping in the edaphic factor characteristics. Meaning similar edaphic factor conditions are present within formed groups and these groups are more likely to result in wetland groups with similar functioning. This group therefore is very useful for managing wetlands over large scales where there is a need to know where the same management can be repeated or where methodologies would need to differ within the same HydroGeoMorphic unit. Other regions can therefore use similarity in edaphic factor based grouping to discern monitoring and management regimes across many wetland sites anywhere in the world.

2.4.2 Patterns of edaphic factors along the wetland littoral gradient

The edaphic factors we observed along the wetland littoral gradient showed a negative trend for SMC and BD and a positive trend for salinity. These trends are related to field capacity, which is the amount of SMC or water content held in the soil after excess water has drained away and the rate of downward movement has decreased (Colman 1947; Castelli *et al.* 2000). The available water capacity is as well important in explaining these patterns, and it refers to the ability of soil to hold water from infiltrating to the lower levels of the soil profile but yet making it available to plants. Therefore it is the capacity of a soil to store water for use by plants (Cassel and Nielsen 1986). It is the water held between field capacity and the wilting point. This is where BD and salinity become important, because an increase in both reduces the available water capacity. Increase in BD reduces field capacity and increase in salinity lowers the wilting point (Figure 2.7 panel A). These patterns are consistent across the eight endorheic wetlands. The exception was in Olopan where SMC was higher outside the wetland compared to within the wetland. The high BD and EC within Olopan, suggest that soils within the wetland are sandy, allowing a quick draining of water as a result of low water holding capacity, and probably salinity and high litter cover, which might be unpropitious for growth of most plants. It was confirmed during the survey that Olopan had sandy soil and is dominated by a mono specific stand of *Eleocharis* species, which is a sedge with high litter turn over. (Paul 2016) also found anomalous low water holding capacity in soil samples from within a wetland with extremely high organic matter compared to clay surrounding the wetland. This pattern suggests that Olopan's BD levels could be limiting to plant growth. While the species *Eleocharis* might be specialised physiologically and morphologically, to grow in saline sandy soils. However, it is yet unknown

$$\left(\frac{\left(Plot X_1 of i^{th} transect \in j^{th} site\right)}{\left(Plot X_2 of i^{th} transect \in j^{th} site\right)}\right) \times \left(\frac{100}{1}\right)$$

how and why *Eleocharis* species monopolise depressions with high salt and sand content. In this study, an unexpected result was that patterns of EC and BD along the wetland gradient showed highest abundance at the edge of the wetland water body. This result overall is in agreement with the findings of (Raulings *et al.* 2010) who found that internal topographical variation generates a mosaics of water regimes at fine spatial scales that allow plant species with different water regime requirements to co-exist over small distances in the temperate grassland of Australia.

2.4.3 The wetland-dryland threshold boundary for delineating endorheic wetlands

In this research we tested whether edaphic factors can be used to delineate the boundary of endorheic wetlands by thresholding these edaphic factors; similarly to studies in the PPR, situated in the temperate grasslands of the US (Wu and Lane 2016). However, the delineation of thresholds of endorheic wetlands from (Wu and Lane 2016) are based on micro elevation that is determined using Light Detection and Ranging and do not specify the distance from the wetland water body. Probability density analyses of the distribution of SMC in (Vivoni *et al.* 2008) conducted in Valles Caldera, New Mexico, also indicated distinct moisture regimes along the semiarid vegetation gradients, similarly to the current study but do not specify the distance from the wetland water body or a threshold that can be tested elsewhere. In this study, the empirically derived threshold of the maximum extent of individual wetlands ranged between 30 m and 70 m. However the aggregate threshold for all eight depressional wetlands, based on the three median edaphic factor values, was 70 m, hence, we recommend the use of a maximum buffer of a 100m, in order to add a precautionary vegetation buffer of 30 m to accommodate the ferralitic zone of subsurface incoming seepage. The buffer width should be based on site specific recommendation using the percentage change threshold, hence the 100 m is a policy recommendation, not a scientific result. (Ma 2016) suggests a minimum buffer width of 20 m (Semlitsch and Russell Bodie 1998). Wetland buffering is important for wetland management in the water protection, flooding control, groundwater storage, habitat for wild species, recreation, aesthetic and removal of sediment and pollutants (Castelle 1992; Correll 1996; Wenger and Fowler 2000)(Castelle 1992; Correll 1996; Wenger and Fowler 2000; Gleason *et al.* 2003). In theory, for generalisation of a percentage change threshold can be used in the place of a distance measure. This theoretical approach allows the results of our study to be applied to other wetlands globally and can therefore be theoretically represented (equation 6 simplified as equation 7) for determining the wetland threshold using empirical measurements of edaphic factors.

From our results, this theory offers the hypothesis that the wetland threshold is found where the median percentage change of three consecutive plots is between 0% and –5% for BD and between 0% and +5% for SMC. Most literature estimated the required buffer to be much lower than this (Macfarlane *et al.* 2015). Therefore, this result is crucial for the South African policy framework and environmental impact assessments (Macfarlane *et al.* 2015). Recently however, a buffer of 100 m has been recommended and this corresponds with the findings of this study (Wilkinson *et al.* 2016). (Keller *et al.* 1993) suggested a100m-wide vegetated buffer around the inundated zone of valley bottom wetlands. Meanwhile the South African National Wetland Monitoring Programme makes mention of a 100 m buffer area, but as a recommendation rather than a necessity and merely for determining land cover around wetlands; and not as a rule explicit for declaring wetland extents as recommended in the current study (Wilkinson *et al.* 2016). Our results suggest that current legislation on wetland buffering might be underestimating or overestimating the wetland threshold for other HGM units (Appendix O). Designating buffers that are too narrow might result in unsustainable abstraction of wetland moisture by permitting compounding, overuse or plantations within the wetland extent. This is even more important if the wetlands is perched than when groundwater dependent Microtopography is an important factor to consider when studying the wetland threshold. Microtopography is important because it predicts the drainage patterns. In the current study, we did not make detailed measurements of stratigraphic position and microtopography as the findings are linked to the Sentinel-2 satellite imagery. The variation in stratigraphic position and resulting microtopography in different wetlands or different sides of the same wetland may offer different resistance to water flow. Hence stratigraphic position and associated microtopography of the type of vegetation could be driving observed variation (30 -70 m) in the width of thresholds.

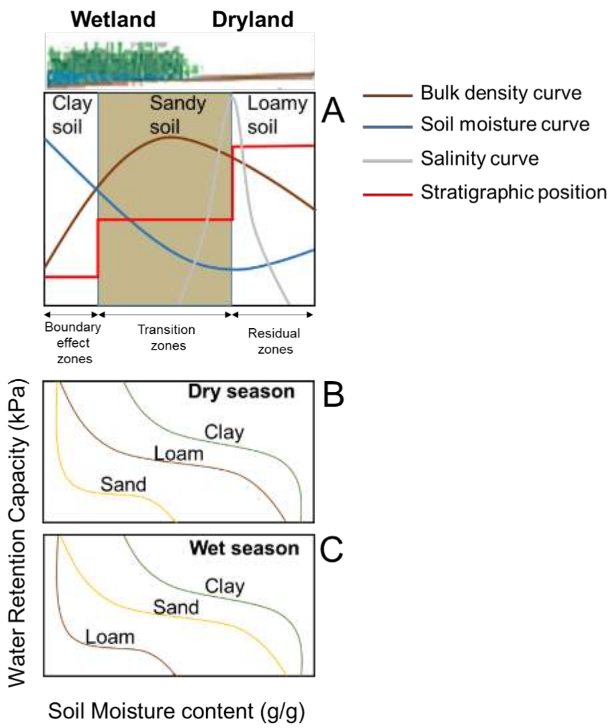

Figure 2.7: The conceptual model of the relationship between edaphic factors along the littoral zones of depressional wetlands as well as stratigraphic position and distance from the wetland water body from our results.

In addition to stratigraphic position due to microtopography, the principles of water retention capacity may be circumstantial in depressional wetlands where the fringe zones are periodically saturated, and this may have important implications for the seasonal shifts on the wetland threshold. During the wet season sandy soil has a higher field capacity than loamy soil, however, during the dry season the available water capacity of the sandy soil drops to below that of loamy soil (Figure 2.7 panel B&C). Similarly the different regions along the wetland littoral gradient have different levels of available water capacity because their level of saturation depends on the distance from the wetland water body and their stratigraphic position in the micro-elevation. In other words available water capacity in these depressional littoral zones is not mainly driven by rainfall or days after rainfall alone. Therefore, this study highlights the importance of the 1) stratigraphic positioning and 2) relative distance from the wetland water body in the understanding the field capacity and hydrology of depressional wetlands. These two variables; the stratigraphic positioning and relative distance from the wetland water body, prevail over the principles of 1) available water capacity, 2) field capacity and 3) water retention capacity in determining levels of edaphic factors. Hence further research should consider adding wetland characteristics in addition to the edaphic factors when grouping depressional wetlands or studying depressional wetland vegetation in other temperate grassland regions globally. This means the results of the study are applicable in other international regions because the stratigraphic positioning and relative distance from the wetland water body in our study site are similar to those as in all depressions. Therefore, the results suggest that substrate type might have marginal effects.

2.5 Conclusion

This study showed that depressional wetlands that occur in the temperate grassland biome as represented by a sample of eight depressions in the MLD ecosystem are significantly different in edaphic factors and are related to the differences in sensitivity to climate change. However similarities are present among some of the wetlands depressions are related to the differences in sensitivity to climate change This study also revealed consistent horizontal trends in the edaphic factors from the open water to the outer dryland, characterised by a declining trend in SMC and increasing trends for salinity and BD. This study demonstrated that for depression wetlands within the MLD the wetland threshold (threshold between dryland and wetland) can be empirically detected at a relatively short distances of about 30 to 70 metres; a threshold where the trends of edaphic factors change to opposite directions with a percentage change that is greater than 5%. This threshold can potentially inform the delineation of the outer edge of endorheic wetlands, which are poorly mapped globally and are under threat.

We therefore conclude that: Depressional wetlands are characterised by narrow littoral zones possible as universal characteristic. Depressional wetlands are dynamic and are poised to suit a high diversity of floral and faunal species. Depressional wetlands are vulnerable to climate changes and if not monitored this system might disappear slowly and unnoticed under climate change. The minimum depressional wetland buffering in legislation should be considered at 100 m in order to protect this HGM unit. Current wetland buffering legislation might be allowing (legally) farming and construction within wetlands in practise, while denouncing it in sentiment. We can now detect the wetland boundary using edaphic factors, especially those that are retrievable with remote sensing. We can now detect wetlands that were previously extremely hard to detect, for example when using remote sensing, such as wetlands that do not have a permanent water body. We have made progress in demonstrating that the objective delineation of the wetland threshold and its associated permanently, seasonally and temporally inundated regions of the littoral zone is achievable.

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Chapter 3

Determining the wetland-dryland boundary of depressions using functional traits of littoral vegetation

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Abstract
Depressional wetlands are highly vulnerable to changes in land surface temperature and rainfall. Climate changes alter the spatial variability of littoral wetland vegetation. However, the spatial variability of wetland Plant Functional Traits (PFTs) or variables for productivity and structure in the wetland littoral zone is not well understood. This study aims to assess the spatial variability of PFTs between wetlands and along their littoral gradients, from the water body through the inundated zone with vegetation to Upland. Pairs of 202 PFT variables sampled vegetation structure. These PFTs were Species Richness, Leaf Angle Distribution, Leaf Clumping, PFTs for wetland vegetation productivity. Leaf Area Index, Above Ground Biomass (AGB) and Vegetation Moisture Content (VMC). Fourteen belt transects of contiguous 10 m plots sampled eight depressions in the Mpumalanga Lake District, South Africa. In general, there were significant differences between the eight wetlands for functional traits of vegetation structure but not for functional traits of vegetation productivity, at Bonferroni adjusted p-value (0.001). However, PFTs of vegetation productivity were significant in differentiating the eight wetlands. At the same time, structural traits were important in detecting the wetland boundary. VMC and AGB generally showed negative trends along the littoral gradients. The trends occurred over short distances, ranging from 30 to 90 m, reflecting the extent of the saturated zone of the wetland. Understanding the spatial variability of vegetation functional traits helps manage and monitor depressional wetlands in a time of climate change.

Keywords: *Depression wetland, Richness threshold, Soil edaphic factors, South Africa, Vegetation Ecology, Wetland-dryland boundary*

3.1 Introduction

Can the plant functional traits be used to detect the boundary between the wetland and Upland - the "wetland threshold" or the "wetland-dryland boundary"? This question is vital because wetlands are important ecosystems. Wetlands are essential and have specially adapted biodiversity and functional traits, crucial for biogeochemical cycles and livelihoods (Euliss *et al.*, 2006; Marton *et al.*, 2015). Despite being fragmented, they cover 12.1 million km^2 (1.2 billion ha), about 6-7% of the global land surface (D´ıaz *et al.*, 2019). Approximately 30% of wetlands occur in arid and semi-arid areas, and of these, many are isolated depressional wetlands (Van Deventer *et al.*, 2020). About half of these wetlands in temperate regions have a permanent water body, whereas the other half is seasonally inundated (Denny, 1993). Many of these wetlands are the primary water source in areas where they occur (Gardner and Finlayson 2018). As a result, the water and biogeochemical functions provide critical support for biodiversity and human livelihoods (Scholes and Archer 1997; Tooth and McCarthy 2007; Ramoelo *et al.* 2012). Their unique characteristics are what makes wetlands provide these vital ecosystem services (Verrecchia 2007).

Some of these unique characteristics include very shallow depressions without permanent open water. Instead, they have saturated soil and wetland vegetation covering the entire depression. They may also have a permanent open water body and a narrow fringe of intermittently inundated vegetation. Alternatively, they may have a combination of permanent open water and a wide saturated and vegetated area. The study site for the research, Mpumalanga Lake District (MLD), southern Africa, is an isolated depressional ecosystem. Upland surrounds isolated wetlands hence "isolated". However, they may still have intermittent inflow or underground recharge. This paper is of interest to the international audience because the MLD ecosystem is similar to other international systems. They are similar because they are all iconic international examples of predominantly depressional catchments. These examples of iconic depressional systems include the Great Lakes region in eastern Africa, the Prairie Potholes Region (PPR) in the Great Plains of Canada, the Drew Point coast of Alaska, and the area of Orlando in Florida, in the United States (US) or the Poyang Lake in Asia. These listed ecosystems are part of the wetlands of international importance listed under the RAMSAR convention. They are under threat from global environmental change (Junk *et al.* 2006). Global environmental change has reduced global wetland extents. About 81% of inland wetland biodiversity have been extinct since 1970 (Gardner and Finlayson 2018).

Plant functional traits (PFTs) are characteristics of plants that relate to structure, functioning, and temporal biological changes. The PFTs represent plant ecological strategies in response to environmental factors (Cornelissen *et al.*, 2003; Pérez-Harguindeguy *et al.*, 2016; Togashi 2016). The two major functional plant trait groups important for wetland ecology are primary production and vegetation structural traits. These trait groups can be plot-level values as aggregate trait values. Some important aggregate structural plant functional traits include Plant Species Richness (PSR), Vegetation Leaf Clumping (VLC) and Leaf Angle Distribution (LAD). In contrast, some important aggregate traits of primary productivity include Leaf Area Index (LAI), Above-Ground Biomass (AGB) and Vegetation Moisture Content (VMC). Nilson (1971) proposed the first vegetation metrics that express the spatial dispersion of foliage. Nilson (1971) modifies the Markov chain analysis of Beer's law, which defines the probability of transmitting a beam of light at the zenith angle through the canopy, *i.e.* (Hagemeier and Leuschner 2019; Wu *et al.* 2019). Its fundamental assumptions are a random distribution of leaf angles; and LAI. Zou *et al.* (2018) define LAI as half of the total (all-sided) leaf area per unit ground surface area. This manuscript hypothesises that these traits would change as the climate changes. Their current pattern varies along the wetness gradient of depressional wetlands. Henceforth, the link between environment and functional traits for vegetation structure and those of primary production can assist in monitoring the response of wetland littoral vegetation to climate change (Roscher *et al.* 2018).

Despite this background, no studies have provided comprehensive data and synthesis needed to understand changes in the patterns of the range functional traits of vegetation structure and productivity along the wetness gradient of depressional wetlands. Drought prediction is a consistent projection of climate change for many areas of Southern Africa and specifically temperate grassland regions where depressional wetlands occur. Drought is a major abiotic stressor for many terrestrial ecosystems, including wetlands, particularly in South Africa, with an arid to semi-arid climate, and has devastating effects on the persistence of wetland biodiversity. It is unclear whether the abundance of soil moisture on the fringe of depressional wetlands results in a higher VMC in wetland vegetation than dryland vegetation (Browne *et al.* 2020). As wetland scientists, we need to assess the baseline pattern of LAD along the littoral gradients of depressional wetlands to monitor its changes in response to changes in temperature and rainfall. We also need to test the hypothesis that VLC declines as one moves from the water body to Upland. This manuscript theorises that vegetation that receives lesser soil moisture reduces the radiation in its canopy by increasing the gaps between leaves.

The manuscript also hypothesises that an associated increase in leaf area is partly due to an increase in the proportion of lateral leaves as one moves away from the wetland. We theorise that LAD decreases as one moves away from the edge of the wetland water body, with flat aquatic macrophytes to erect variations. This decrease is from more vertical leaf angles with fewer lateral leaves dominated by sedges to less vertical angles with more lateral leaves dominated by grass. We hypothesise that VMC and AGB increase with species richness as one moves away from the wetland water body. We postulate that species richness increases with an increase in the distance from the wetland wate-rbody. We also postulate that wetland vegetation keeps less VMC because it grows in an abundance of water. The manuscript posits that different species create multiple vegetation strata with a net increase in AGB at the outer fringe of the vegetated saturated soil between the water-body and Upland. Therefore, a question emerges; can these plant functional traits be used to 1) characterise the vegetation zones between dryland and the wetland water-body, 2) detect the outer edge of the saturated vegetated regions and 3) group wetlands? Studies in literature do not explore depression wetlands to examine the variation in plant functional traits of structure and productivity to answer these same three questions. Our central hypothesis is that plant functional traits can detect the wetland boundary can achieve 1) characterising the vegetation zones between dryland and the wetland water-body, 2) detecting the outer edge of the saturated vegetated regions and 3) grouping wetlands. This manuscript aims to establish the within-wetland and between-wetland variability in wetland plant functional traits focusing on the littoral gradient of depressional wetlands.

3.2 Material and Methods

3.2.1 General methodology

This conceptual framework of methodology aligns the data collection with remote sensing ancillary data. This alignment allows subsequent researchers to retrieve the exact sample locations for these data for future replication. In conjunction with the remote sensing data, applying data science principles ensures systematical recording of all critical steps and selection of sample plots, transects and wetland sites. The general methodology included five general sections and shows how repeatability is ensured (Figure 3.1).

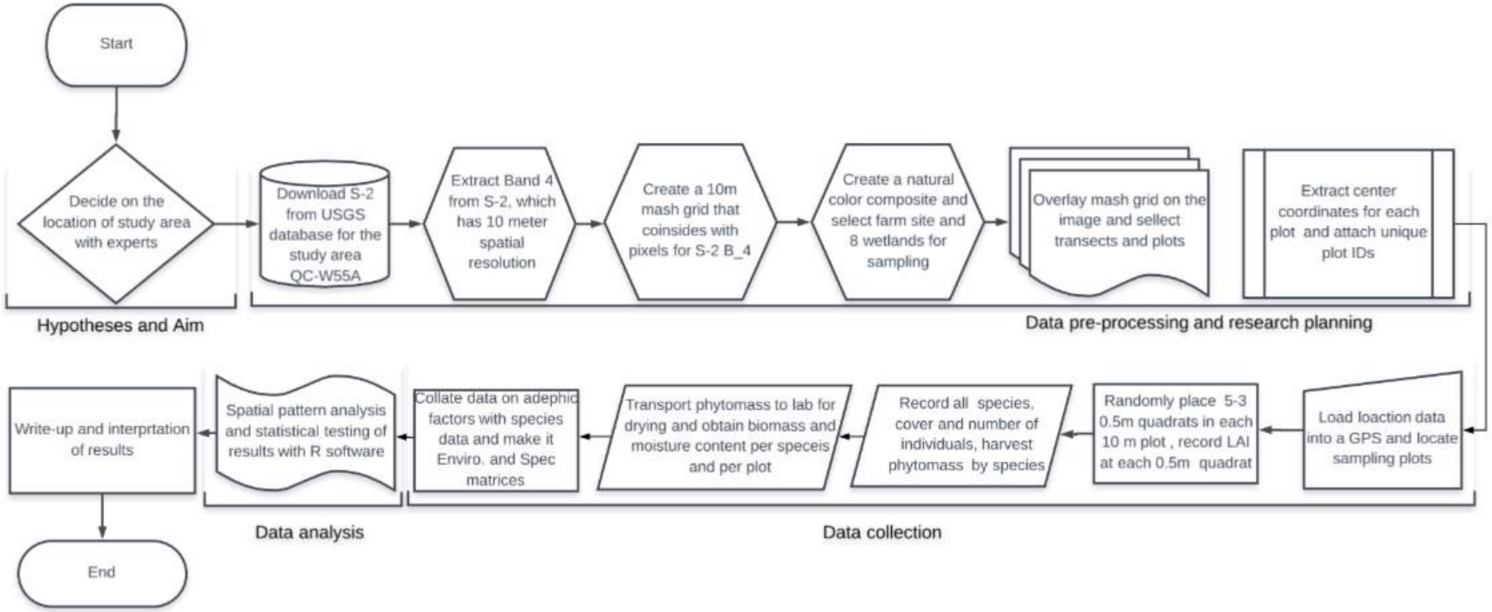

Figure 3.1: Conceptual model workflow of the methodology. United States Geological Survey (USGS), Quaternary Catchment (Q.C.), W55A is a unique number naming conversion used by the Department of Water Affairs and Forestry (DAFF) South Africa was given to the subject quaternary catchment. Sentinel-2A (S-2A), Band 4 (B4), Identification (I.D.), Global Positioning System Device (GPS), Electric Conductivity (E.C.), R is a free software environment for statistical computing and graphics.

3.2.2 Study area

The study area was in the Mpumalanga Lake District for its rich and diverse flora and fauna. The Mpumalanga Lake District (MLD) has different depressions (and other wetlands). Therefore the MLD can be a good case study for globally isolated wetland ecosystems. A sequence of two strata underlies the geology. First is the Ecca group, a topping of sedimentary deposits mainly consisting of shale and sandstone. Then there is the Dwyka Group below in the stratigraphic position (Bell *et al.* 2001; Foster *et al.* 2015). The catchment receives 767 mm of mean annual precipitation. W55A has over 300 depressional wetlands in just a 20-odd kilometre radius (Goudie and Thomas 1985; van Deventer *et al.* 2020a; van Deventer *et al.* 2020b; van Deventer 2021). Within the Mpumalanga Lake District, a subset of depressional wetlands was selected (Lake Banagher Farm, 26°20'11.21" S, 30°21'14.03" E, in the Gert Sibande District, in the Msukaligwa Local Municipality, Mpumalanga Province, South Africa, Appendix A). The wetland ecosystem types and the wetland vegetation vary due to elevation, size, shape, and the vegetated zones' area. Therefore, there is a good chance of covering a wide range of habitats in a relatively small area (Watson 1986; Brooks and Hayashi 2002; El-Kawy *et al.* 2011; Wilkinson *et al.* 2016; Vanderhoof *et al.* 2018).

3.2.3 Selecting depressional wetlands for transect sampling

Eight wetlands (Table 3.1) that represent the diversity of wetlands in the MLD were selected. The diversity was observed in terms of (a) extent of the water body, (b) extent of vegetation cover, (c) shape and (d) size. Figure 3.2 shows that eight sites adequately represent the actual species composition or the sampled wetland communities. To prove sufficient sampling had occurred, the number of new species encounters had reach zero (Figure 3.2), As more wetlands are added to the dataset, the number of new species encounters approaches zero, as shown by the flattened curve. The flattening curve indicates that no new species have emerged; new species encounters have reached saturation.

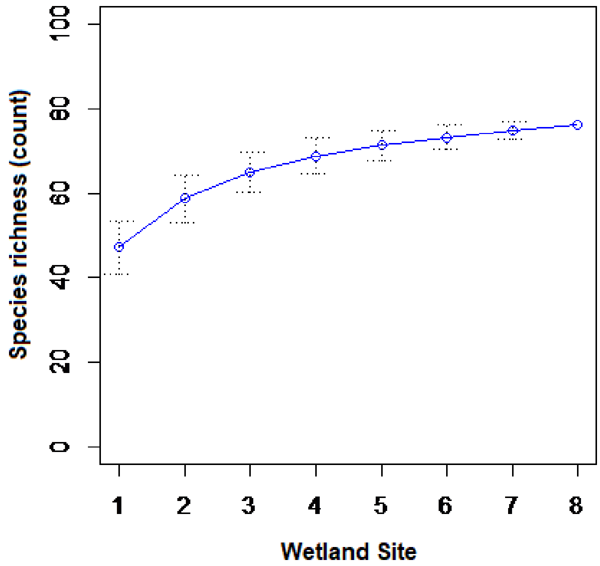

Figure 3.2: Species richness curve (blue line) semi-variogram, showing an increase in species richness (y-axis) with an increase in the number of sites sampled (x-axis). Note the levelling out of species richness at 80 species after vegetation sampling of six (6) to eight (8) plots. This graph shows the rate of encountering new species while adding one wetland at a time to the dataset. As is evident from the image, the encounter rate began to flatten after sampling three wetlands. By the time the sampling was on eight wetlands, the encounter rate was virtually zero. The error bars show the variance within sites.

Setting up the belt transects

A mash grid consisting of contiguous 10 m plots following the rows of Sentinel-2A pixels was generated in ArcGIS (ArcMap version 10.5) using only the red band. Furthermore, the location of the transects was focusing on regions with high species turnover. A grid overlaid on a colour composite of the three red edge bands of the Sentinel-2A was an aid for identifying subtle changes in tone and texture of vegetation (Goodman 1990). For wetlands more significant than 0.2 km^2 (the three most extensive wetlands, Appendix C), sampling involved three transects around each wetland (Figure 3.3). For the smaller wetlands, sampling was involving only one transect. The length of a transect was at the fence line for the pilot sites. After learning from the pilot samples how to determine the start of upland, the transect length would end at the start of upland. Therefore transects had varying lengths that depended on the width of the wetland zone (30 m - 130 m). The assumption was that variation in tone and texture of image spectra were variations in the visible vegetation physiognomy (Figure 3.3).

A mash grid made up of contiguous 10 m plots following the rows of Sentinel-2A pixels was generated in ArcMap 10.5. Following an approach, the grid was overlaid on a true colour composite of the Sentinel-2A to identify the best transect locations (Nondlazi *et al.* 2021). The sampling procedure followed the framework provided by the Sentinel-2A pixels and used a belt transect method. This study sampled only functional traits in contiguous plots with dissimilar vegetation structures and compositions. The purposeful sampling ensured that the sampling maximised a representative sampling of landscape features and avoided repetitive sampling. Sentinel-2A provides data with global coverage snapshots after five days. In addition to near-infrared and short-wave infrared bands, it has three red-edge bands (Bands 5-7 with the centre of the bands at 705, 740 and 783 nm, respectively). These bands are instrumental in vegetation classification. The width of transects was 10 m as determined by the spatial resolution of the Sentinel-2A images.

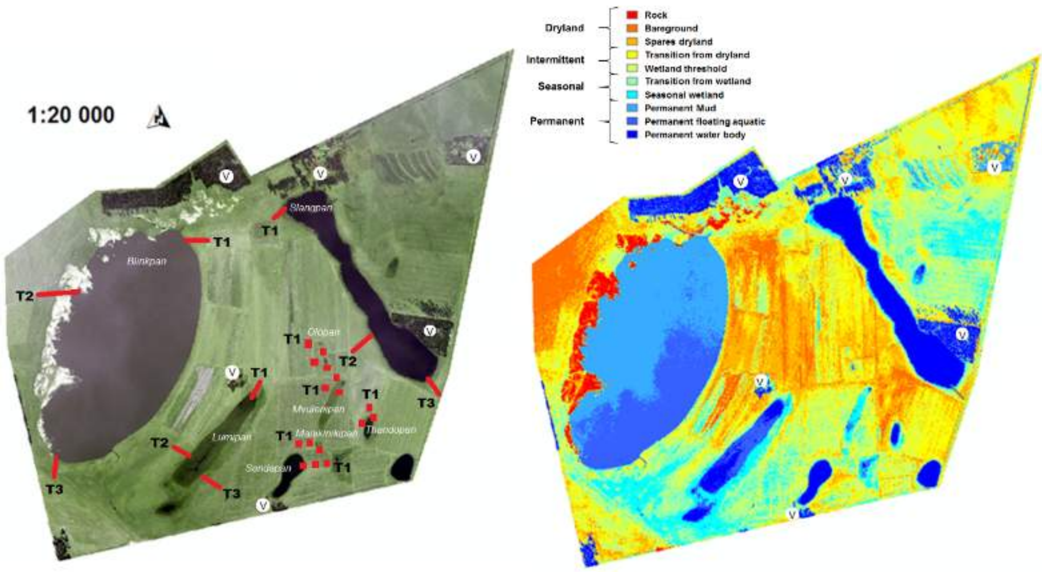

Figure 3.3: A map showing the positioning of the sampled transects numbered T1 to T3 (red coloured 10 m × 10 m belt transect of plots 100 m^2. The image clipping is to the shape of the region of Interest (ROI) boundary. The maps, unless stated otherwise, are from Digital Globe sub-meter resolution (0.5 m) from the World Imagery dataset created 12 December 2009 and last updated June 2018, which provides low spatial resolution 15 m, high spatial resolution 60 cm and high spatial resolution 30 cm imagery. The scale was 1:70.53 on and Red-Green-Blue colour composite. The V-symbols represent areas where VMC is confounding the land surface moisture signal.

3.2.4 Sampling at plot level

In each 10 × 10 m plot, at least three subplots could adequately sample vegetation species composition. We selected random plots by tossing a 0.25 m^2 quadrat over the head into the main 100 m^2 quadrants from two corners facing away from the quadrant. In each subplot, samples of the six plant functional traits include Plant Species Richness (PSR), Leaf Area Index (LAI), Leaf Angle Distribution (LAD), Apparent Clumping Factor (ACF), Vegetation Moisture Content (VMC) and Above Ground Biomass (AGB). The tape measure was used to determine the width of transects, as well as measuring pixel dimensions. Distance from the wetland water body was the basis for analysing differences in vegetation patterns and soil variables along the littoral zone. The distance from the water body determined whether proximity to water correlates with target variables or functional traits. These same plots used in the study are those in (Nondlazi *et al.* 2021), which is the first experimental chapter of this thesis.

3.2.5 Selecting experimental variables

Six wetland vegetation characteristics are experimental variables, namely Plant Species Richness (PSR), Apparent Clumping Factor (ACF), Leaf Angle Distribution (LAD), Above Ground Biomass (AGB), Vegetation Moisture Content (VMC), Leaf Area Index (LAI). These variables affect the gap fraction and, therefore, stimulate new seedlings. There are fewer gaps between leaves per unit increase of LAI for clumped canopies compared to canopies with a more random leaf arrangement (Schulze 1982; Asner 1998; Gower *et al.* 1999). Secondly, they affect both radiation interception and distribution within the canopy. The radiation within the canopy controls the behaviour of stomata and thus respiration and water loss (Chen *et al.* 2003). Foliage clumping LAD and LAI are considered mechanisms to optimise plant growth across species by manipulating within-crown radiative regimes (Cescatti 1998). Clumping in canopies LAD and LAI vary across vegetation species (Beadle 1993; Schulze 1982). Plant species differ in their ability to control radiation within the canopies (Daynard 1969). The species that cannot withstand the increases in climate change's radiation force will face exclusion from that depressional wetlands ecosystem. Therefore, understanding the basic pattern of species turnover along the littoral gradient of depressional wetlands is essential for monitoring the impacts of climate change on wetlands (Güsewell *et al.* 2005; Huikkonen *et al.* 2020).

3.2.6 Sampling traits at subplot level

Inside each 0.25 m^2 quadrat, collection of two readings using a LICOR canopy analyser machine, one above the canopy and another below the canopy of the grass swards. The data were downloaded and analysed at the laboratory. Species richness is the count of different species within a defined area (Huston and Huston 1994; Magurran and McGill 2011; Magurran 2013). Inside each 0.25 m^2 sub-plot quadrat assessment, we sampled 1) the three dominant species in terms of cover-abundance using the Braun-blanket scale were recorded together with 2) the average height of vegetation, 3) the recording of the rest of the species is to attain total species composition and 4) the relative abundance of species. The species data, *i.e.* species composition, were collected to conduct a power statistic semi-variogram through a species-area curve. The curve determines the adequacy of sample size concerning species diversity. The relative abundance, together with the cover and height of vegetation, validated and supported which three species were the most physiognomically dominant. All the aboveground phytomass within each 0.25 m^2 quadrat was collected separately for each species. This separation facilitates confirming whether the three most physiognomically dominant species were also dominant in terms of AGB. These data support the analysis of AGB and VMC. Hence analysis was derived through these methods. These vegetation properties are important structural, functional characteristics of vegetation productivity that we should value for at least two reasons. During sampling, the observer cared to avoid non-living organic matter such as dung, sticks and twigs. The inclusion of fresh litter of the current season was vital to ensure the accuracy of measurements.

Table 3.1: Methods, equations and definations for deriving the six variables used in the research

Variable	Source	Method	Definition	Eq
Categorical variables				
Wetland ID/Site	From South African map of local names or arbitrary names	Place hold names	A unique name and three-letter abbreviation using the firsts three letters of a given name	
Position along a transect (m)/Plot-in-transect number $Plot\ distance\ i = \sum_{i=10\ m}^{n}$	Horizontal distance	Measuring tape reading	The sequential numbering of 10 m plots along a transect	(1)
Transect	GPS coordinates	Denuding a line along the littoral gradient	Contiguous 10×10 m squares that are sampling the same vicinity of a littoral gradient	
Functional Traits of Productivity				
Leaf Area (index) $L = 2\sum_{i=1}^{5} \overline{K}_i W_i$	LAI-2200C Plant Canopy Analyzer	It calculates the amount of blue light (320-490 nm) intercepted at five zenith angles, based on readings taken above and below the canopy.	Calculates the fraction of diffuse radiation that passes through the canopy (gap fraction) as a function of the zenith angle.	(2)
Grass Biomass (g./g)	Vegetation harvesting	Observer cuts and weighs plant foliage after oven drying to a stable weight.	Differences in biomass across locations (plots, along the wetland gradient and sites)	
Vegetation Moisture Content (g/g)	Balance scale	Weighing biomass before and after weighing	Differences in biomass weight between pre and post- drying weight	
Functional Traits of Structure				
Foliage Diversity/Species Richness (count) $D = \sum_{i=1}^{s} p_i^0$	Vegetation survey, Species inventory and listing	In 10×10 meter plots + across wetlands. Count of the number of individuals of each species encountered	The number of species found in each 10×10 m plot. Differences in species composition across locations (plots along the wetland gradient and sites)	(3)
Apparent Clumping Factor $ACF = \frac{LAI_{\log(avg(T))}}{LAI_{avg(\log(T))}}$	LAI-2200C Plant Canopy Analyzer	The device calculates the interception of blue light (320-490 nm) at five zenith angles from readings taken above and below the canopy.	The closeness of leafy foliage and branches of a vegetation canopy	(4)
Leaf Angle Distribution $G(\theta) = \frac{-\ln P(\theta)}{\mu S(\theta)}$	LAI-2200C Plant Canopy Analyzer	The device calculates the interception of blue light (320-490 nm) at five zenith angles from readings taken above and below the canopy.	It is the dominant angle at which vegetation holds its leafy foliage	(5)

3.2.7 Sampling structural vegetation functional traits

Six plant functional traits were part of sampling along several belts transects in eight representative depressional wetlands to analyse trends in the plant functional traits. The sampling was aiming to: *(i)* assess potential differences in plant functional traits between wetlands and *(ii)* analyse trends in the plant functional traits from the palustrine region to the outer dryland *(iii)* test the utility of plant functional traits.

Plant species richness (PSR)

Species richness refers to the number of species found within a location, *e.g.* plot (Gotelli and Chao 2013). Identifying and recording all observed plant species within each subplot was the basis for assessing plant species richness. The assumption was that the species recorded from all the subplots that belong to the same plot were adequate to represent the species composition of each plot. Five subplots were sufficient to sample the species richness of a 10 $\times$ 10 m plot. In comparison, three subplots were the bare minimum to represent species composition within a 10 $\times$ 10 m plot at the study site. It was essential to record the spatial distribution of species to understand how different locations differ in the set of species present. The comparison of species across different locations entails a comparison of species richness. The idea was to understand whether directional and co-occurrence patterns exist within and across locations (wetland sites, transects and plots). Another hypothesis of this thesis is that depressional grouping wetlands according to variations in their species richness was possible. For example, Depressions that have a homogenous vegetation stratum will have lower richness levels. Reed-beds can be either monospecific or with a rich mixture of subordinate species but are often very species-poor. Wetlands can also have various vegetation layers of many different species occupying the different strata.

Apparent clumping factor (ACF)

A key characteristic of wetland vegetation is its unique vertical stature. The vertical statuer is charateristic of rhizomatous helophytes plants. The erect vegetation also characterises the stature of vegetation structure in the terrestrial region of the lacustrine depression wetland region. Hypothetically, the erectness of vegetation in the wetland region is crucial because it facilitates the quick emergence of the green foliage components above the water level as flooding conditions change over the seasons. Another significant characteristic of this stature is the limited amount of lateral leaves. This thesis proposes that the clumping of vegetation in the wetland zone is presumably different from the dryland zone due to these characteristics. Therefore, the foliage clumping becomes an important vegetation property for monitoring spatial shifts in the wetland-dryland transitional zone, represented on maps as a single boundary line. However, the baseline pattern of vegetation clumping along the wetland littoral zone. To measure the Apparent Clumping Factor (ACF), the observer collects one reading from the centre of each sub-plot. The Apparent Clumping Factor (ACF) was measured to account for the non-random distribution of foliage within the canopy (Ryu *et al.* 2010). It is equivalent to the apparent clumping index Wapp defined by Ryu *et al.* (2010). To measure ACF, the observer collects one reading of ACF from the centre of each subplot using the LI-COR LAI-2000. (Cutini *et al.* 1998).

Leaf angle distribution (LAD)

The characteristic distribution of the angle of leaves within a plant canopy is species-specific (Anten and Hirose 1999; Deckmyn *et al.* 2000). Foliage clumping, LAI and LAD, are important functional traits of plant species to consider when studying the impacts of climate change on the littoral vegetation of wetlands. Species richness is an index of species diversity (Adams 2010). The LAD is one of the traits through which plant species regulate radiation within a canopy. Changes in the dominant LAD are also expected, along with the climatic changes predicted and associated with changes in species composition along the littoral gradient of depressional wetlands. Therefore, temporal variations in LAD might be helpful in monitoring changes in that wetland littoral vegetation due to climate change. For this reason, the observer employed an LI-COR LAI 2000 meter to collect a value at the centre of each subplot (Table 3.1).

3.2.8 Sampling productivity functional traits

Above ground biomass (AGB)

The amount of AGB that a plant produces depends on the number and size of tillers that the plant community produces (Johnson and Tieszen 1976; McWilliam *et al.* 1993). Therefore the amount of AGB within a unit area of the ground surface depends on the number of plant individuals of each species present within that ground and their characteristic tiller production (García *et al.* 1993; Litton and Boone Kauffman 2008). Changes in climate that affect species composition would likely affect primary production. Sedge species have no-tillers (Li *et al.* 2021). Therefore, as one moves away from the water body and grass dominates over sedges increasing AGB. However, changes in species composition do not necessarily translate to changes in AGB production because different species might produce the same amount of AGB. Nevertheless, we hypothesised that the species that grow closer to the wetland have less tiller production than some species that grow further away from the wetland. This low tillering rate is because sedges dominate the plant species that grow closer to the wetland. Sedges have few, short to no-tillers. In contrast, tiller producing grasses dominate the terrestrial wetland region further away from the wetland water-body. At each subplot, the AGB was harvested at ground level using a pair of scissors and weighed separately for each species. Later calculation of AGB at the lab. It calculated using the average of all subplots within each plot was calculated as a plot level value of AGB.

Leaf area index (LAI)

The leaf area index is the one-sided green leaf area per unit ground surface area (Cutini et al. 1998; Ryu *et al.* 2010). The C_4 photosynthetic pathways may outperform C_3 photosynthetic pathways under ambient conditions because C_4 have a higher quantum yield. A quick increase in the initial slope of the photosynthetic rate characterises C_4 plants. In contrast, C_3 plants have a flatter initial curve that increases with temperature (Reeves *et al.* 2014). Therefore, a rise in carbon dioxide and temperature may affect the threshold of negative returns for C_4 plants while continuously accelerating the photosynthetic rates for C_3 plants. This hypothesis is even more probable for C_3 plant species that have a high variance in leaf area (Martorell *et al.* 2021). The C_3 plant species achieve higher respiration rates because they have greater leaf area. Thus, achieved growth will enable them to out-compete coexisting C_4 plants both because of competition for space and accelerated growth rates from increased photosynthetic potential (Niu *et al.* 2021). Hence, shifts in the proportion of C_3 to C_4 along the wetness gradient of depressional wetlands may occur because of climate changes (Reeves *et al.* 2014). There is a need to understand better the baseline leaf area patterns along the littoral gradient of depressional wetlands. This understanding is essential for understanding the impacts of climate change on these highly endangered yet crucial ecosystems. The use of the LI-COR LAI 2000 meter was for measuring LAI. Leaf Area Index (LAI) measurements were collected at the centre of each subplot and were averaged at plot level and aggregated at site level.

Vegetation moisture content (VMC)

When water stress affects plants, it will act directly on plant growth and development (Martorell *et al.* 2021). That action will result in reduced photosynthetic potential as well as dry mass production and seed production (Washburn *et al.* 2015). Ultimately the plants will die, and the specific populations less tolerant to water stress will completely disappear (Hu *et al.* 2015). To this end, the response of water content within a plant canopy can be an important indicator of water stress in vegetation (Vivoni *et al.* 2008). Some of the critical variables for measuring vegetation water content include leaf equivalent water thickness and VMC. However, these two are less desirable because equivalent water thickness is not a direct measurement. At the same time, VMC is direct but more laborious than equivalent water thickness. On the other hand, active developments in vegetation indices include estimations of vegetation water content and, therefore, vegetation health and stress (Nouri *et al.* 2014). However, direct measurements of VMC are requisite for developing accurate remote sensing models (Hu *et al.* 2015). It was, therefore, essential to ascertain the baseline pattern of VMC along the littoral gradient of depressional wetlands. Three random subsamples of 0.25 m^2 from each 100 m^2 quadrant were collected and weighed to obtain AGB samples. The samples went through the recording and weighing before and after drying to calculate direct measurements of VMC. .

3.3 Data analysis

All the statistical analyses (Table 3.2) were conducted in R software. We used a principal component analysis (PCA) to assess the statistical differences in PFTs across different scales, site and plot levels. In ordination, the orthogonality of latent variables achieves multiple direct comparisons of similarities between PFTs across all plots and sites simultaneously. On the other hand, climate changes might lead to the expansion of the wetland zone into the dryland or *vice versa*.

Table 3.2: The application of statistical analytic methods to the different objectives at different scales

Research objective	Objective of analysis	Visual Presentation	Statistical Test	Dataset used
A. To investigate differences in functional traits of vegetation structure and productivity	To assess whether there is variability in the means of functional traits of vegetation structure and productivity across the eight sampled wetlands	Density plots using the R ggplot2 package	Bonferroni $\binom{n}{r}$ adjusted 8 Group ANOVA	Matrix of wetland ID by functional traits of vegetation structure and productivity edaphic factor
	To assess whether there are significant differences in mean and median of the observed variability of functional traits of vegetation structure and productivity across wetland sites	Box and whisker plots using the R base graphics package	Bonferroni $\binom{n}{r}$ adjusted Turkey's HSD	Matrix of wetland ID by functional traits of vegetation structure and productivity arranged as a matrix of 28 paired combinations of 8 groups
A. To investigate the interaction of functional traits of vegetation structure and productivity in determining wetland function and their potential for wetland classification	To assess whether differences in functional traits of vegetation structure and productivity across the wetland can be used to group the wetlands	PCA Ordination in R	Data modelling with Constrained Canonical Correspondence Eigen distance values	Two matrices; one on-site ID by mean plot level functional traits of vegetation structure and productivity
	To determine whether functional differences in the structure and productivity of vegetation could be used to group wetlands	R Scree and Factor plots	Correlation and Factor analysis	Matrix of functional traits of vegetation structure and productivity constrained by site
B. To investigate the patterns of functional traits of vegetation structure and productivity along the wetland littoral gradient and the existence of the wetland threshold	To assess whether the pattern of response for functional traits of vegetation structure and productivity have a positive slope along the wetland and gradient	R Line graphs and R polylines	Polynomial regression coefficients	Matrix of wetland ID by site by functional traits of vegetation structure and productivity tor
	To understand whether the response patterns of functional traits of vegetation structure and productivity shows any threshold that could suggest the position of the wetland boundary	Calculation of percentage change	Successive plot level subtraction	Matrix of wetland ID by site by functional traits of vegetation structure and productivity
B. To investigate the differences in edaphic factor functional traits of vegetation structure and productivity between the two sides of the wetland gradient threshold	To assess whether wetland functional traits of vegetation structure and productivity have a flat slope across all wetlands or slope = to zero	Box and whisker plots using the R base graphics package	Hierarchical clustering with 3 edaphic factors	Matrix of wetland ID by functional traits of vegetation structure and productivity by transect
	To assess whether there are the difference among transects that have been plotted along the longitudinal gradient of the same wetland	Box and whisker plots using the R base graphics package	Student T-test	Matrix of wetland ID by functional traits of vegetation structure and productivity

3.4 Results

3.4.1 Variation in functional traits of vegetation communities along the wetlands-upland gradient

The eight wetlands were significantly different in LAI ($F_{8,75}$=3.73, $p<$ 0.001), LAD ($F_{8,75}$=15.9, $p<$ 0.001) and ACF ($F_{8,75}$=27.1, $p<$ 0.001), at a Bonferroni adjusted alpha level (p<0.001, One-way ANOVA) and not in AGB ($F_{8,75}$= 0.398, p= 0.918), VMC ($F_{8,83}$=2.21, p=0.034), PSR ($F_{7,71}$ = 2.524, p= 0.0224) as it can be observed in Figure 3.4 (boxplots, results including significance are presented in alphabetical order). Multiple comparisons (p<0.001, Turkey's HSD) between the 28 comparable pairs showed that the significant difference (p<0.001) is contributed by three pairs with significant differences in LAI, including Blinkpan-Lumipan (p<0.00), Lumipan-Sandapan (p<0.001), Lumipan-Thandopan (p<0.01). The significant difference (p<0.001) could also behave a contribution to 12 of the 28 comparable pairs that were significantly different in LAD (Table 3.3). The significant difference (p<0.001) could also be attributed to 12 of the 28 comparable pairs that were significantly different in ACF (Table 3.4).

The results from ordination analysis (Figure 3.5), conducted using all six vegetation characteristic variables, revealed three groups of wetlands (–resulting from the PCA). Group A had high values for the traits on vegetation characteristics for primary production. In contrast, groups B and C were biased towards high vegetation characteristics for vegetation structure. Group C had some functional traits for productivity. However, the 1st and 2nd PCA axes were the most critical latent variables highly correlated (75.87%) to functional traits of productivity and structure, respectively. Therefore, ordination results further support the ANOVA findings of the importance of functional traits for productivity (41.27%) and functional traits for structure (34.60%) in differentiating the wetlands from one another (Figure 3.5). There was an interaction between traits for leaf structure and those of plant production.

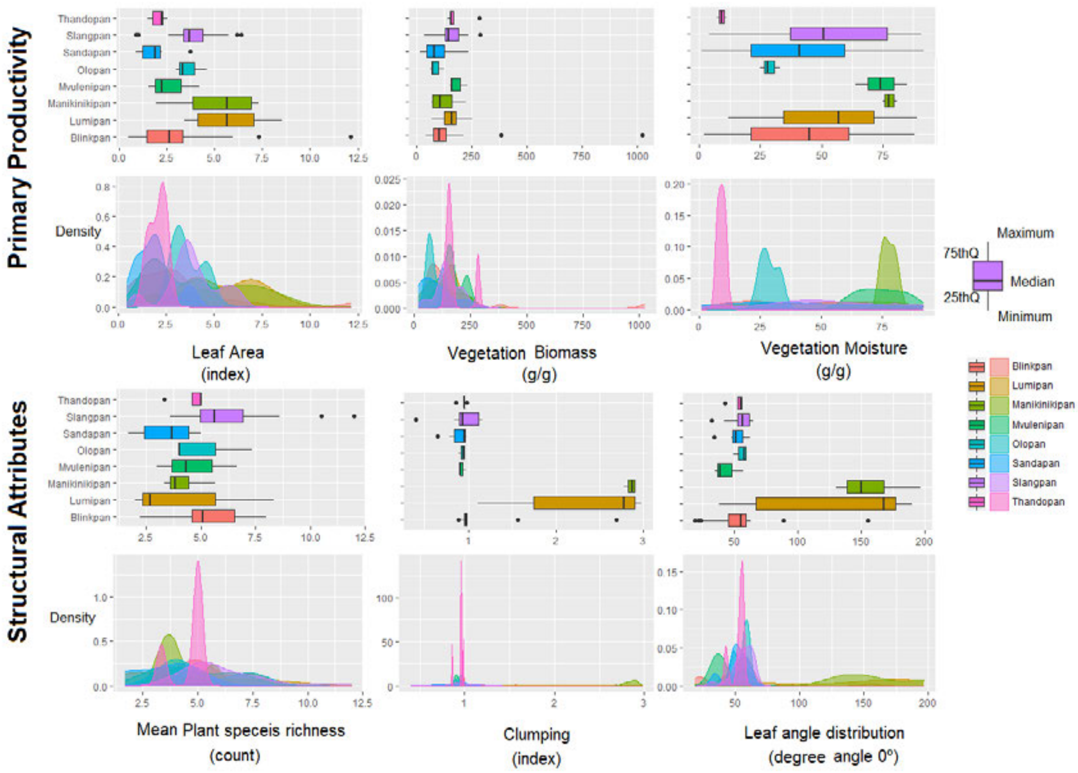

Figure 3.4: Median values of vegetation functional traits for structure and productivity across eight-sampled wetland sites

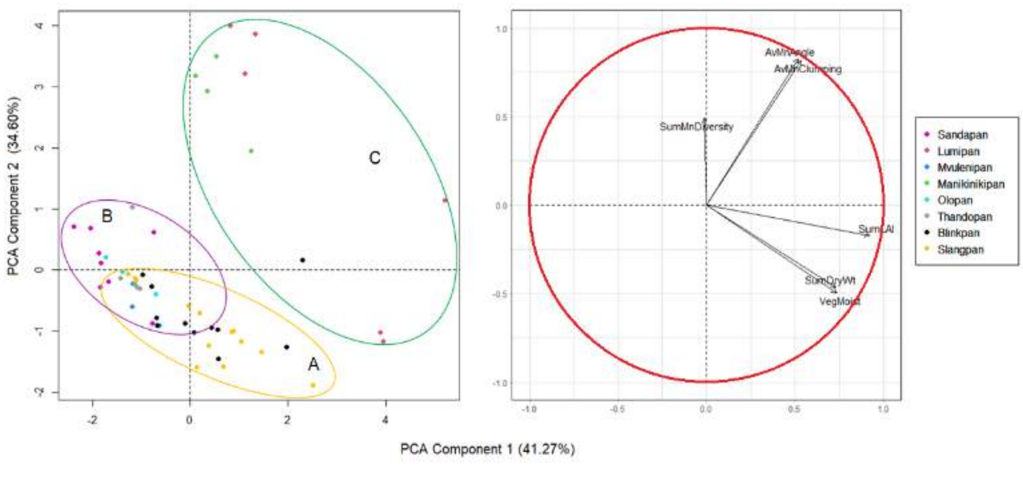

Figure 3.5: Eigenvalues of the correlation matrix for all sampled active variables shown as percentage surpasses influence of each PCA component (PCA 1 and 2)

Table 3.3: Post-hoc tests for LAD by Site (using method = pairwise)

Site Contrast	Estimate	SE	DF	*t*-ratio	*p*.value
Blinkpan - Lumipan	-81.7837	9.7828	76	-8.36	***
Blinkpan - Manikinikipan	-101.8051	15.001	76	-6.7866	***
Blinkpan - Mvulenipan	11.0257	17.0204	76	0.6478	1
Blinkpan - Olopan	-0.9504	17.0204	76	-0.0558	1
Blinkpan - Sandapan	3.5824	11.9118	76	0.3007	1
Blinkpan - Slangpan	-1.589	7.8789	76	-0.2017	1
Blinkpan - Thandopan	2.1764	13.6467	76	0.1595	1
Lumipan - Manikinikipan	-20.0214	16.0828	76	-1.2449	1
Lumipan - Mvulenipan	92.8094	17.9811	76	5.1615	***
Lumipan - Olopan	80.8333	17.9811	76	4.4955	***
Lumipan - Sandapan	85.3661	13.2483	76	6.4436	***
Lumipan - Slangpan	80.1948	9.7828	76	8.1976	***
Lumipan - Thandopan	83.9601	14.8276	76	5.6624	***
Manikinikipan - Mvulenipan	112.8308	21.2755	76	5.3033	***
Manikinikipan - Olopan	100.8547	21.2755	76	4.7404	***
Manikinikipan - Sandapan	105.3875	17.4598	76	6.036	***
Manikinikipan - Slangpan	100.2162	15.001	76	6.6806	***
Manikinikipan - Thandopan	103.9815	18.6865	76	5.5645	***
Mvulenipan - Olopan	-11.9761	22.7445	76	-0.5266	1
Mvulenipan - Sandapan	-7.4433	19.2226	76	-0.3872	1
Mvulenipan - Slangpan	-12.6147	17.0204	76	-0.7412	1
Mvulenipan - Thandopan	-8.8493	20.3433	76	-0.435	1
Olopan - Sandapan	4.5328	19.2226	76	0.2358	1
Olopan - Slangpan	-0.6386	17.0204	76	-0.0375	1
Olopan - Thandopan	3.1268	20.3433	76	0.1537	1
Sandapan - Slangpan	-5.1713	11.9118	76	-0.4341	1
Sandapan - Thandopan	-1.406	16.3109	76	-0.0862	1
Slangpan - Thandopan	3.7653	13.6467	76	0.2759	1

Signif. codes: 0 '*' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1**

Table 3.4: Post-hoc tests for ACF by Site (using method = pairwise)

Site Contrast	Estimate	SE	Df	*t*-ratio	*p*-value
Blinkpan - Lumipan	-1.3015	0.1295	76	-10.0478	***
Blinkpan - Manikinikipan	-1.806	0.1986	76	-9.0929	***
Blinkpan - Mvulenipan	0.1247	0.2254	76	0.5535	1
Blinkpan - Olopan	0.1189	0.2254	76	0.5275	1
Blinkpan - Sandapan	0.1679	0.1577	76	1.0647	1
Blinkpan - Slangpan	0.084	0.1043	76	0.8056	1
Blinkpan - Thandopan	0.108	0.1807	76	0.5977	1
Lumipan - Manikinikipan	-0.5046	0.2129	76	-2.3695	0.3256
Lumipan - Mvulenipan	1.4262	0.2381	76	5.9905	***
Lumipan - Olopan	1.4203	0.2381	76	5.9659	***
Lumipan - Sandapan	1.4694	0.1754	76	8.3767	***
Lumipan - Slangpan	1.3855	0.1295	76	10.6966	***
Lumipan - Thandopan	1.4094	0.1963	76	7.1793	***
Manikinikipan - Mvulenipan	1.9307	0.2817	76	6.8541	***
Manikinikipan - Olopan	1.9249	0.2817	76	6.8333	***
Manikinikipan - Sandapan	1.9739	0.2312	76	8.5388	***
Manikinikipan - Slangpan	1.8901	0.1986	76	9.5161	***
Manikinikipan - Thandopan	1.914	0.2474	76	7.7361	***
Mvulenipan - Olopan	-0.0059	0.3011	76	-0.0195	1
Mvulenipan - Sandapan	0.0432	0.2545	76	0.1697	1
Mvulenipan - Slangpan	-0.0407	0.2254	76	-0.1806	1
Mvulenipan - Thandopan	-0.0167	0.2694	76	-0.0621	1
Olopan - Sandapan	0.049	0.2545	76	0.1927	1
Olopan - Slangpan	-0.0348	0.2254	76	-0.1545	1
Olopan - Thandopan	-0.0109	0.2694	76	-0.0404	1
Sandapan - Slangpan	-0.0839	0.1577	76	-0.5318	1
Sandapan - Thandopan	-0.0599	0.216	76	-0.2774	1
Slangpan - Thandopan	0.024	0.1807	76	0.1326	1
Signif. codes: 0 '*' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1**					

3.4.2 Trends of plant functional traits along the wetland gradient

Generally, the vegetation structure showed a negative trend as the distance from the water body increased, excluding vegetation moisture content. Meaning, there are more horizontal leaves and a higher number of species per area as the distance from the water body increased. In contrast, vegetation productivity showed a positive trend. The PFTs of vegetation productivity increased along the distance from the water body to the outer dryland boundary. These patterns occur at relatively short distances, ranging from 30 to 100 m. Their end probably reflectes the extent of the wetlands (Figure 3.6).

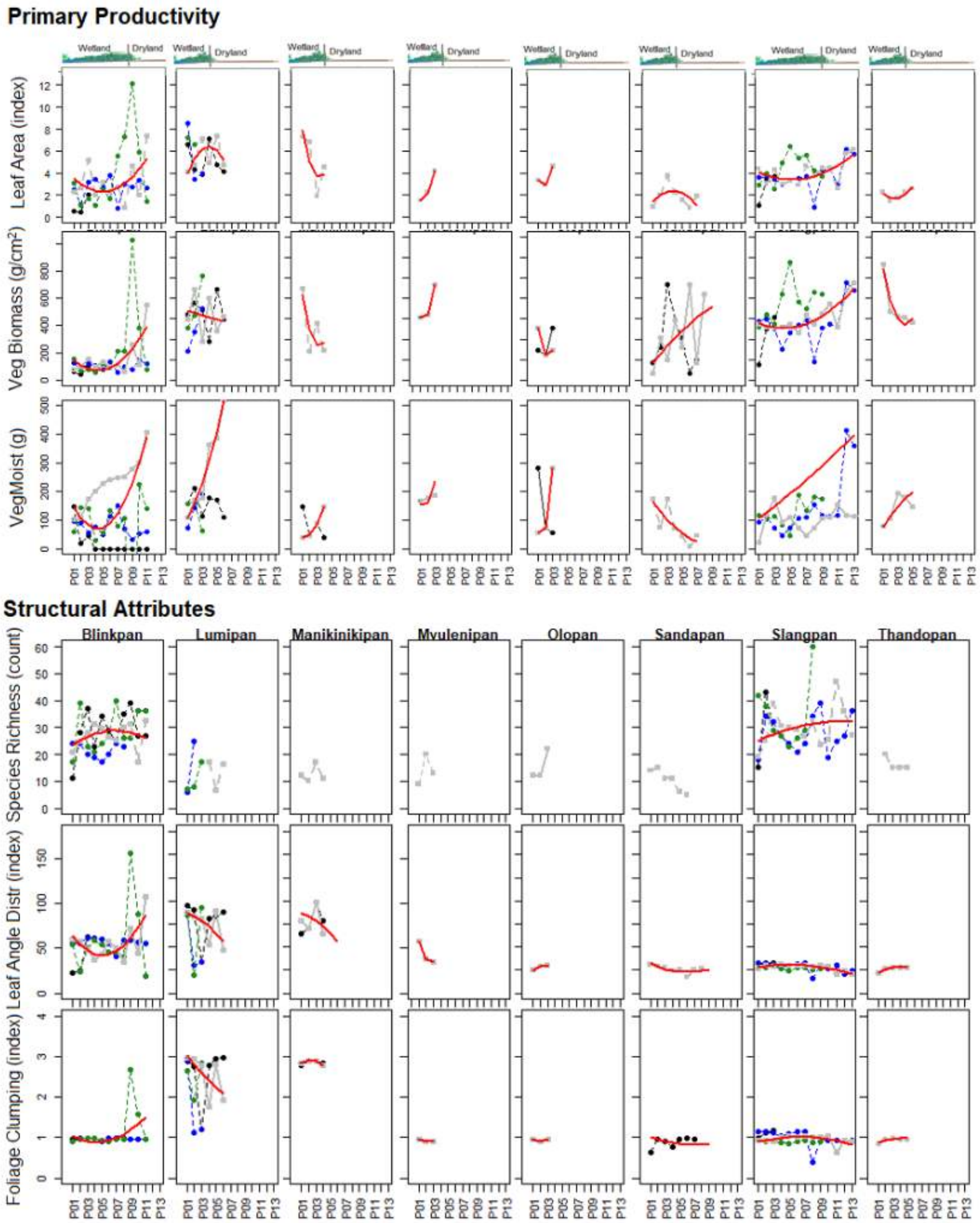

Figure 3.6: Polynomial regression models (red lines) of the response of edaphic factors to increasing distance from the centre of the wetland water body (red lines, mathematical model iterations and statistics).

3.4.3 Vegetation functional traits as indicator of the wetland threshold

Combining the data from vegetation characteristics across the littoral gradients of all wetlands shows that all vegetation characteristics maintain their general trends (Figure. 3.7). The end of the patterns of plant functional traits, which probably reflects the mean seasonal maximum extent of the wetland, at 90 – 100 m on average for combined data.

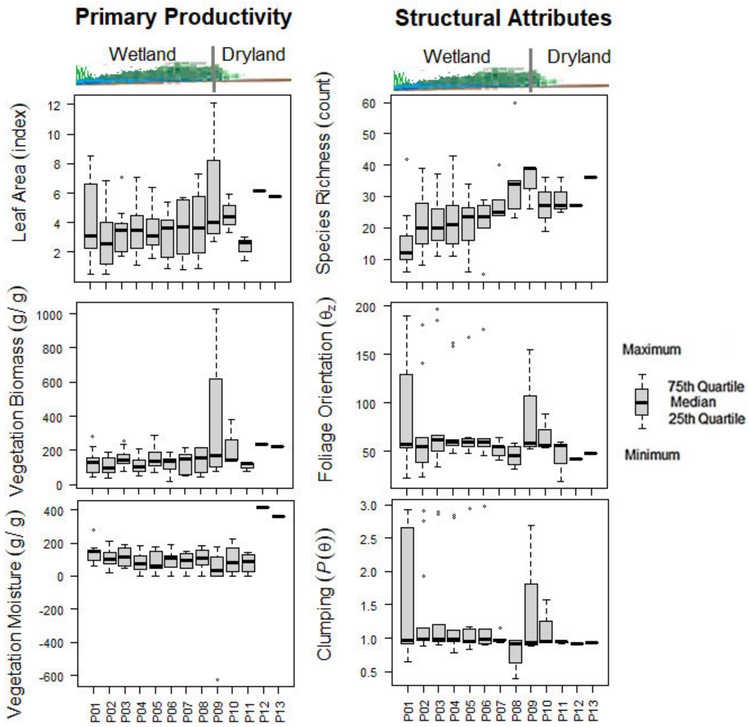

Figure 3.7: Median values of plant functional traits per plot number across the wetland-dryland gradient of increasing distance from the wetland water body.

3.5 Discussion

3.5.1 Differences in vegetation functional traits between depressional wetland

These results show that significant (p<0.001) site-level differences were detected in structural, functional traits but not in the functional traits for plant productivity. Our results showed significant site-level differences among pairs of LAD (12/28 wetland pairs), ACF (12/28 wetland pairs) and LAI (3/28 wetland pairs), but not in PSR; AGB; VMC. This result suggests that wetland vegetation structure is more diverse than wetland productivity between wetlands. Meaning traits of productivity might be more similar across wetlands. The data and results come from data collection over two growing seasons. Outcomes were consistent between the sampling periods. Although the two growing seasons are still too few to draw a definite conclusion, these results are consistent, and the differences between the two seasons are insignificant for this dataset. There is a need for further research on the seasonal time-series of plant functional traits in the littoral zone of depressional wetlands (Niemuth *et al.* 2010; Stegen and Swenson 2009; Lewis and Wang 2010). This thesis contributes methods to achieve time-series changes in plant functional traits along the littoral gradient of depressional wetlands. It is worth noting that; excluding fresh litter would distort the comparisons of seasonal primary production since fresh litter is part of primary production. Excluding fresh litter would affect the difference between plots with low litter production and those that produce more litter early in the season.

3.5.2 Patterns of vegetation functional traits between depressional wetland

The patterns of vegetation functional traits along the wetland littoral gradient were as expected: a negative trend for structural vegetation traits and a positive trend for functional productivity traits. This result could suggest that vegetation is getting denser and with more biomass along the wetness gradient. The exception was one structural, functional trait, plant species richness (PSR), which had a positive trend from wetland to upland. These patterns are consistent across the eight depressions. The exceptions for the positive trend of functional productivity traits were Manikinikipan for LAI, Lumipan, Manikinikipan, Olopan and Thandopan for AGB and Sandapan for VMC. In this study, an unexpected result was that plant species richness changed with functional productivity and structural-functional traits. Similarly, LAI is a leaf trait belonging to structural-functional traits.

The LAI is also a key indicator of primary productivity. Hence, the high clumping and vertically acute leaf angle trait combination result in species dominance or monodominance. For instance, Bai *et al.*. (2005) suggest that the relationship between productivity and species richness are positive in agreement with the literature, *e.g.* Dodson *et al.* (2000), Mittelbach *et al.* (2001) and Wang (2017). In this study, the relationship between primary productivity and diversity was not a direct positive relationship. The data aggregated to site-level data showed a similar pattern of PSR to structural, functional traits rather than productivity traits. In Wu *et al.* (2016), functional traits and plant species diversity were important in explaining grassland productivity. However, Wu *et al.* (2016) did not demonstrate the relationship between species diversity and vegetation structure. These trends were over relatively short distances (30 – 90 m). The soil properties could drive these trends (Nondlazi et al 2021), especially if there is a correlation in the patterns of soil edaphic.

3.5.3 Detection of a wetland-dryland threshold

The recommendation of a 100 m wide threshold was supported by the analysis of changes in plant function. This result is very important for wetland buffering since none of published literature derives an empirical threshold, especially one that is derived from the functional traits of vegetation. Keller *et al.*. (1993) suggested that 100 m wide vegetated buffer was important for birds but did not make the distinction of the important of a wetland threshold since not all vegetation is wetland vegetation. Meanwhile the National Wetland Monitoring Programme currently also makes mention of a 100 m buffer. This study showed that wetland vegetation has functional traits that are uniquely different from the trait of upland vegetation. Therefore highlighting the knowledge gap of the impact of wetland vegetation buffer verses vegetation buffer that constitutes both upland and wetland vegetation. So, the suitable size of the upland vegetation buffer remains a question. This result highlights the crucial need for legislative framework that addresses the size of the upland vegetation buffer separately from the vegetation buffer of the wetland water body. Our results suggests that that current legislation on wetland buffering might be unclear on this issue and might be leading to wetland degradation as a result. Microtopography could be driving observed variation (30 -100 m) in the width of the functional trait based wetland thresholds.

3.6 Conclusion

This study showed that depressional wetlands in the Mpumalanga Lake District ecosystem are significantly different in the measured structural and productivity plant functional traits. These trait differences can derive empirical subgroups of depressional wetlands that reveal similar sensitivities to climate change and other effects on wetland vegetation. This study shows the horizontal trends in the measured vegetation characteristics from the edge of the open water in the depression to the surrounding upland. These trends are characterised by a declining trend in structural PFT and increasing product characteristics. This study also showed that depression wetlands within the Mpumalanga Lake District have distinct wetland vegetation thresholds. These are the threshold between dryland and wetland that can detect empirically at a relatively short distance of about 30 to 100 metres. This threshold is a point where the patterns of plant functional traits change in opposite directions.

This study showed that depressional wetlands in temperate grassland biome such as the Mpumalanga Lake District ecosystem are significantly different in plant functional traits. However, similarities in vegetation characteristics are present among some of the depressions wetlands This study demonstrated that detecting the wetland threshold of depression wetlands in relatively short distances of about 30 to 100 metres in the Mpumalanga Lake District is achievable. This threshold between wetland and uplands can be empirically detected using trends of plant functional traits along the littoral gradient. A change to opposite directions is more remarkable than 5% denotes the threshold.

We, therefore, conclude that narrow vegetation littoral zones characterise depressional wetlands. Depressional wetlands are dynamic and are poised to suit a high diversity of floral and faunal species. Depressional wetlands are vulnerable to climate change. If not monitored, this system might disappear slowly and unnoticed under climate change. The legislation should specify the minimum depressional wetland buffering at 100 m. This specification would protect this HydroGeoMorphic unit in South Africa for these areas and for testing elsewhere in the world. Current wetland buffering legislation might be allowing (legally) farming and construction within wetlands in practise while denouncing it in sentiment. We can now detect the wetland boundary using plant functional traits, especially those detectable with remote sensing, *e.g.* LAI and AGB. We have made progress in demonstrating that the objective delineation of the wetland threshold is achievable. Vegetation with rare plant functional traits on the littoral zones of depressional wetland are critically endangered.

3.7 References for chapter 3

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Chapter 4

Determining the wetland boundary of depressions using Hyperspectral remote sensing analysis of littoral gradient soils

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Abstract

Depressional wetlands are highly vulnerable to changes in land surface temperature and rainfall. Changes in climate alter the spatial variability of edaphic factors, but this variability has not been studied using hyperspectral remote sensing as an alternative to laboratory analysis. This study aimed to assess spatial variability of hyperspectral remote sensing soil indices between wetlands and along littoral gradients of wetland banks. A set of 784 paired sub-sample measurements of four soil remote sensing soil indices were collected from 80 samples of 10 × 10m plots along 14 belt transects in eight representative wetlands in the temparate grassland region, South Africa. In general, there were significant differences between the eight wetlands for Soil Composition Index (SCI), Normalised Difference Salinity Index (NDSI), Misra Soil Brightness Index (MSBI) and Normalised Difference Water Index (NDWI) at Bonferroni adjusted p –value (<0.001). SCI and NDWI were important in differentiating the eight wetlands. NDWI and SCI generally showed negative trends over short distances, ranging from 30 to 70 m, along the littoral gradients. MSBI and NDSI generally showed positive trends. Understanding of the spatial variability of remote sensing soil indices helps in the management and monitoring of depressional wetlands in the era of climate change.

Keywords: *Depression wetland, Hyperspectral, Remote Sensing, Soil indices,Salinity, NDSI, MSBI, NDWI, SCI*

4.1 Introduction

The unique soil properties found on the littoral zones of depressional wetlands support specially adapted littoral vegetation but, might change due to climate changes (Vepraskas and Craft 2016; McKenzie *et al.* 2002). The potential impacts are depletive to the total wetland extent while catastrophic for biodiversity and ecosystem services. Meanwhile, wetlands are among the most diverse and valuable ecosystems on earth (Bowen 1994; DeFauw 2020; Max Finlayson *et al.* 2015). The global value of wetland ecosystems was 14.9 trillion USD in 2014 (Acharya 2020; Valk and van der Valk 2012; Winning 2009).The Jagadishpur Reservoir a Ramsar Site in Nepal has an estimated total annual economic value 94.5 million (Baral *et al.* 2016). Unchecked, the decline in wetland extents can affect human health detrimentally. For instance, wetlands can reduce pathogens as part of their water quality ecosystem service, inaddition to provision of food, water and leisure (Istenič *et al.* 2009; Wilcock *et al.* 2012). *E. coli* infections are an international human health problem (Ault and Morris 1997; McCarthy 1996). The World Health Organization estimates that 1 million illnesses, resulting in more than 100 deaths and nearly 13 000 life adjusting disabilities worldwide per year are due to *E.coli* (Pires *et al.* 2019). Changes in climate, such as increase in rainfall create flood conditions that potentially reduce soil salinity and increase sand fraction or soil moisture on the littoral zones of wetlands (Corwin and Lesch 2005; Bruland and Richardson 2005). These changes can degrade habitat suitability for many biodiversity and disable characteristics that are important for provision of ecosystem service. As a result, changes in wetland soil properties are a persistent prediction due to climate changes (Pires *et al.* 2019; Bohn and Lettenmaier 2013; Indoria *et al.* 2020). Therefore, monitoring spatiotemporal changes in soil properties relative to wetland boundaries is important, timely and of interest to the international research community and cannot be ignored (Zamorano *et al.* 2018; Chervan *et al.* 2019).

the concerns of monitoring wetland extents, numerous research efforts have been conducted to assess the boundary of wetlands, however challenges persist (Adam *et al.* 2010; Van Deventer *et al.* 2020). In the last few decades, researchers have followed different techniques to estimate soil properties *e.g.* aerial photography, multispectral and hyperspectral sensors, LiDAR (light detection and ranging), and SAR (synthetic aperture radar), InSAR (interferometric synthetic aperture radar), and other microwave systems, (*e.g.* Gravity Recovery and Climate Experiment - GRACE (Naidoo *et al.* 2015, Gangat textitet al. 2020; Cho *et al.* 2010). A key gap according to RAMSAR is international 'status and trends in wetland ecosystem extent' including endangered depressional wetland ecosystems (Chenery *et al.* 2015, Jones *et al.* 2011). Researchers have realized that the unique features of wetlands *i.e.* lack of a single uniform land-cover feature; high dynamism and constantly changing spectral signature; and steep environmental gradients that result in narrow ecotones are challenging the capacity of remote sensors (Knipling 1970; Huete and Jackson 1988; Qi *et al.* 1995; Asner *et al.* 2000; Cho *et al.* 2010; Adam *et al.* 2010). Overcoming most of these challenges requires training of satellite data using ground observations and correction for soil background. However the determination of soil properties requires laboratory analysis that involves procedures like saturated paste extraction (Slavich and Petterson 1993; Hossain *et al.* 2020). The cost of these laboratory procedures is a current limitation in wetland monitoring. Hence, the question on the use of hyperspectral data to reproduce the patterns that are obtainable from laboratory procedures such as SPE is a critical research question.

Aerial photography is time-consuming and expensive, while freely available multispectral data is limited to low resolution (10 m). Lidar, like laboratory analysis, is highly challenging to use for regular monitoring at large scales due to cost (Cho *et al.* 2012). Additionally for Lidar there is also difficulty in field sampling of grass and wet chemistry (Naidoo *et al.* 2012). Compared with most approaches, microwave RS is promising when fused with multispectral, but is yet to be proven in detecting the boundary of depression wetlands. Hyperspectral remote sensing technology has the advantages of simple, time-saving and labor-efficiency and therefore could be useful in soil monitoring compared to saturated pasty extractions (Mashimbye *et al.* 2012). Owing to its continuous sampling or hyperspectral sampling at high spectral differentials (<5nm) in detail it has been widely applied to wetland research, but more on vegetation than wetland soils (Mutanga and Skidmore 2004; Skidmore *et al.* 2010; van Deventer *et al.* 2015; Cho *et al.* 2007; Cho and Skidmore 2006). For small, grassy wetlands, hyperspectral canopy measurement that could be useful as training data still encounter soil background attenuation of the signal and dependency on weather. However, there are no studies that demonstrate the ability of closerage hyperspectral laboratory data in detecting trends in soil properties along the littoral gradients of depressional wetlands (Cho *et al.* 2010).

Hewson *et al.* (2008) found that spectral indices Soil composition Index (SCI), NDSI, MSBI, based on satellite thermal infrared (TIR) (region = 300-1400 nm), discriminate clay mineral-rich soil from mostly coarser quartz-rich sandy soil and to a lesser extent from the silty quartz-rich soil (Hewson and Cudahy 2013). Bulk density is among important wetland soil edaphic factors. While it can be altered by changes in climate such as high rainfall and floods, it is an important determinant of species distribution along wetland littoral zones(Sieben *et al.* 2016; Sieben *et al.* 2016). Remote sensing of bulk density is therefore desirable and possible (Sieben *et al.* 2017). However, thermal IR imagery is difficult to interpret and process because there is absorption by moisture in the atmosphere that creates windows with no data in parts of the TIR region. On the other hand, hyperspectral data offers a continuous wavelength spectrum of TIR. Therefore it might be possible to improve the TIR region of satellite data using the TIR region from hyperspectral data. However we first need to establish the usefulness of the TIR region from hyperspectral data for detecting the wetland boundary.

Of the few studies that have explored depression wetlands in the African region none of them looks at the variation of remote sensing of wetland soil among different depressional wetlands in the MLD (De Klerk *et al.* 2016; Bird *et al.* 2013; Bird and Day 2014; Foster *et al.* 2015; Burger *et al.* 2018; Sieben *et al.* 2017). We hypothesize that the detection of the wetland boundary can be achieved using remote sensing indices. The aim of this study was to establish the within-wetland and between-wetland variability in MSBI, NDSI and SCI. These three remote sensing indices were sampled along several belt transects in eight representative depressional wetlands and used to (i) assess potential differences in remote sensing soil indices between wetlands and (ii) analyse trends in the remote sensing soil indices from the open water body (centre of the wetland) to the outer dryland.

4.2 Material and Methods

4.2.1 General methodology

To ensure repeatability of the research the study was conducted systematically and all the critical steps were recorded. The general methodology (Figure 4.1) includes satellite remote sensing data, which was used to guide the process of selecting sample plots, transects, wetland sites and ecological and data science principles. The alignment of the data collection with remote sensing ancillary data ensured further repeatability because remote sensing data is publicly available. Therefore the exact sample locations where these data were collected can be retrieved by subsequent researchers.

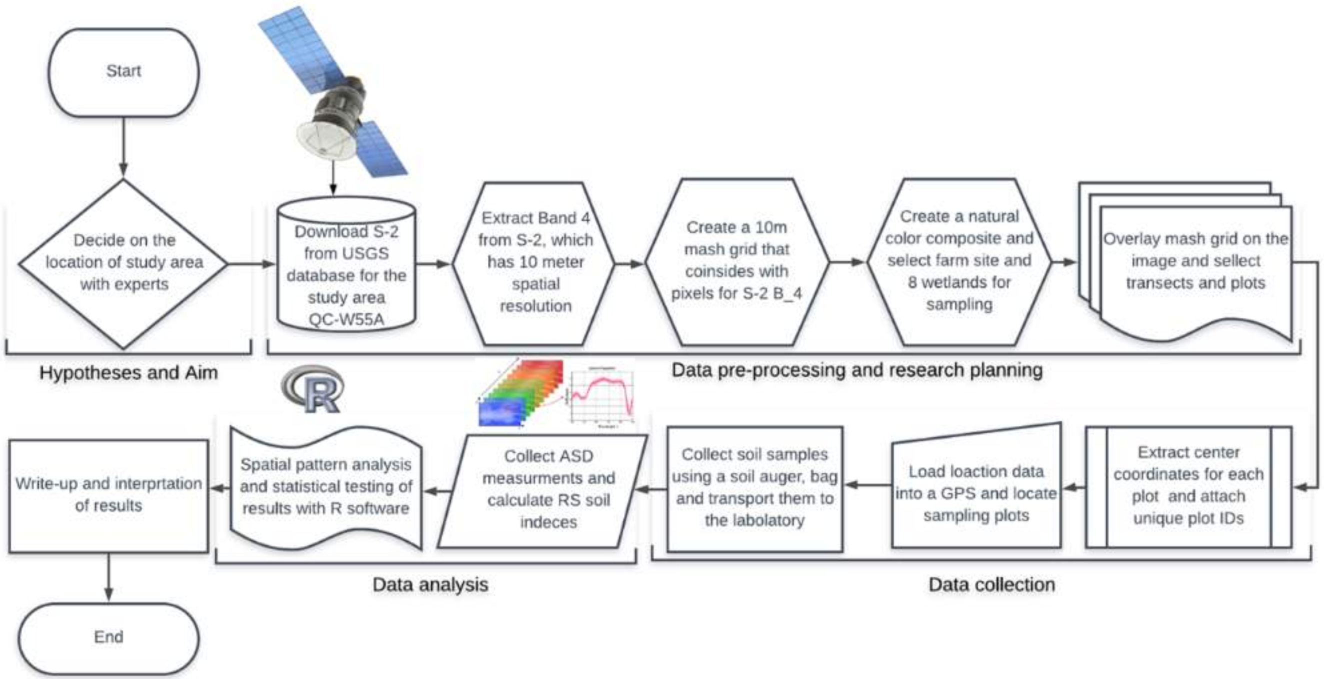


Figure 4.1: The summary overview of the methodology followed in the research. W55A is a unique number naming conversion used by the former South African Department of Water Affairs and Forestry (DAFF) now Department of Water and Sanitation (DWS) given to the subject quaternary catchments.Sentinel-2A (S-2), Band 4 (B4), Identification (ID), Global Positioning System Device (GPS), United States Geographical Survey (USGS) .

4.2.2 Selection of study area

The MLD was chosen as a study area because the area is rich and diverse in different types of depressions (and other wetlands) and therefore can be a good case study for isolated wetland ecosystems globally like the PPR of the US. The geology is underlain by a sequence of two strata. First, is the Ecca group; a topping of sedimentary deposits consisting mostly of shale and sandstone and the Dwyka Group below in the stratigraphic position. The catchment receives 767 mm of mean annual precipitation (Department of Water Affairs (DWA), 2003). W55A has over 300 depressional wetlands in just a 20-odd kilometre radius (Goudie and Thomas 1985; De Klerk *et al.*, 2016; Van Deventer *et al.*, 2020a; Van Deventer *et al.*, 2020b). Within MLD a subset of depressional wetlands were selected (Appendix C, Lake Banagher Farm, 26°20'11.21"S, 30°21'14.03"E, in the Gert Sibande District, in the Msukaligwa Local Municipality, Mpumalanga Province, South Africa, Appendix A). The wetland ecosystem types and the wetland vegetation are very diverse due to variations in elevation, size, shape and the area of the vegetated zones. Therefore, there is a good chance of covering a wide range of habitats in a relatively small area (Watson, 1986; Brooks and Hayashi, 2002; El-kawy *et al.*, 2010; Wilkinson *et al.*, 2016; Vanderhoof *et al.*, 2018).

4.2.3 Selection of sampling locations – the depressional wetland sites

Eight depressional wetlands (Appendix C) were selected to represent the range of depressional habitats. The diversity of wetlands was observed in terms of (a) extent of the water body, (b) extent of vegetation cover (c) shape and (d) size.

4.2.4 Field surveys design

Two field surveys were conducted in order to sample wetland SMC, soil salinity and soil BD. The first survey was conducted in March 2018 and the second was conducted in November 2018. The first survey focused on two wetlands and the second one focused on six wetlands. Only two wetlands had data collected in both sampling periods.The sampling procedure was based on the belt transect method according to the Sentinel-2A pixels scheme. Sentinel-2A provides data with global coverage in a cycle of about once every 5 days from above the equator. In addition to near infrared and shortwave infrared bands, it has three red-edge bands (Bands 5-7 with the centre of the bands at 705, 740 and 783 nm respectively) which has been proven useful in vegetation classification and possibly for edaphic factors. The intention was to cover the range of variation in the visible vegetation physiognomy from the edge of the vegetated part of the wetland up to the dryland area that is bordering the wetland (Figure 4.2). The best location for transects was considered to be the region of the wetland-dryland gradient that had the highest turnover in pixel tone (colour variation). A high variation of pixel tone was considered to reflect higher turnover in species or vegetation structure or both. At each wetland, a field survey was conducted using the belt-transect method. The belt transect method was preferred because the intention was to sample a longitudinal gradient from the water body to the dryland. The width of transects was 10 m as determined by the spatial resolution of Sentinel-2A. Transects had varying lengths, dependent on the width of the wetland zone (30 m - 130 m).

Setting up the belt transects

A mash grid made up of contiguous 10 m plots following the rows of Sentinel-2A pixels was generated in ArcGIS (ArcMap 10.5). This grid was projected on a true colour composite of the Sentinel-2A in order to identify the best locations for transects, following the approach by Goodman (1990). For wetlands greater than 0.2 km^2 (which were the three largest wetlands, Appendix C), three transects were selected around each wetland (Figure 4). For the smaller wetlands only a single transect was sampled. Plots with similar vegetation structure and composition as the one preceding them were not repeated. The purposeful sampling ensured that the sampling maximised the efficiency of representative sampling of landscape features and avoided repetitive sampling. The length of a transect was limited by the fence or by reaching dry ground.

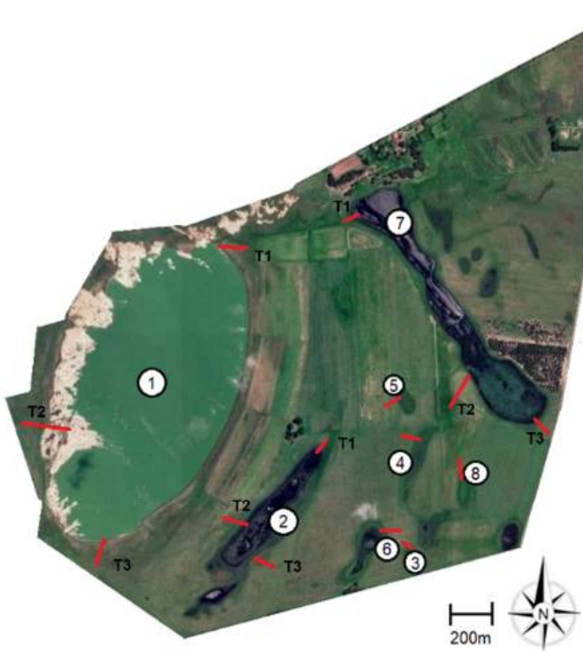


Figure 4.2: A map showing the positioning of the sampled transects numbered T1 to T3 (red coloured 10×10 m belt transect of plots 100 m^2) lines are for illustration purposes. The image has been clipped to the shape of the current boundary of the Lake Banagher farm. The map was created using Sentinel-2A at spatial resolution of 10 m, natural colour composite.

Sampling at plot and subplot levels

In each plot two sub-plots were sampled. We selected random sub-plots by tossing a 0.25 m^2 quadrat into the main 100 m^2 quadrat from two of its corners. To ensure a random selection of the subplots, the observer tossed the quadrant over the head while facing away from the plot. Inside each 0.25 m^2 quadrat, one sub-sample was collected from the top soil (top 15-20 cm) using a Johnson's Soil Auger. The two sub-samples were sealed in zip-lock bags and transported to the laboratory in order to collect the hyperspectral images and determine spectral variation of the soil samples. During sampling, caution was taken to avoid roots and any roots that were found in the soil sample were removed in order to prevent them having an effect on the spectral measurements.

4.2.5 Imagery Collection

Soil reflectance spectra was collected by conducting measurements in a dark room using the ASD FieldSpec FR spectrometer with a spectral range of 350-2500nm. A 200 W halogen lamp built in the ASD was illuminated 2 cm above the soil sample at about a 15° zenith angle. While collecting the measurements the probe was placed at an 8° field angle and 2 cm away vertically at the top of the target sample. A white board with 20 cm × 200 cm was used for standardization with reference reflectance. While a black smooth surfaced non-shiny plastic boat was used as the casting surface of the sample during measurement. One sample was measured 10 times, and an average soil spectral reflectance was obtained (505 samples). High-frequency noise smoothing was not adopted in this study because the focus was less of the nature of spectral profiles but of the trends of reflectance measurements along the littoral gradient of depressional wetlands (Figure 4.3).

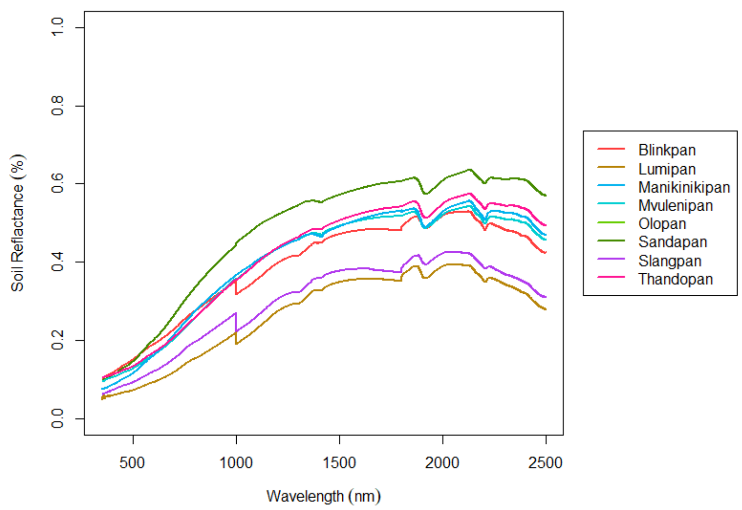


Figure 4.3: The spectra curves of soil samples varied. The variation and therefore the spectral separability was mainly high in the region beyond the thermal Infrared region (350-1400 nm).The moisture absorption regions located near 1400 nm, 1900 nm and 2100 nm were evident even though the soil samples were oven dried before sampling. The spectral reflectance between 350-1800 nm had a general increase or upward trend with increase in wavelength. The soil spectral reflectance curve became flat in 1800-2200nm. It showed a decreasing trend from about 2200- 2500 nm.Thus, the soil spectral curves can be divided three regions; 1) increasing, 2) flattening and 3) decreasing.

4.2.6 Spectral Data Processing and selecting remote sensing soil indices

Spectral pre-treatments of soil samples for the eight sampled wetlands were evaluated using ViewSpecPro version 6.0 (19 Jan 2011) which is a program modified for post-processing operation on ASD hyperspectral data. These included untransformed spectra, first derivatives with no gaps. Once the indices were calculated, operating within the Excel (Ver. 2010) environment, using mean-centered spectral and analyte data fields samples (505 samples) each of these spectral measurements was applied to remote sensing (RS) indices for all eight wetland sites (Table 4.1). Both the spectral data and the RS soil indices were aggregated from subplot to plot and site levels before further analysis. The number of factors used in each PCA analysis was chosen based on the 14 available soil indices form the international database for remote sensing indices (https://www.indexdatabase.de). Nine of which included vegetation components and were excluded. From the five relevant ones, two had multiple interpretations and were thus excluded (Main et al 2011). Testing for outliers was not performed, and all observed values were included in the analysis. The indices were calculated by mimicking Sentinel-2A through resampling of hyperspectral data to the radiometric resolutions of S-2A. The 13 spectral bands of Sentinel-2A range from the Visible (VNIR) and Near Infra-Red (NIR) to the Short Wave Infra-Red (SWIR): 4 × 10 m Bands: the three classical RGB bands ((Blue (493 nm), Green (560nm), and Red (665 nm)) and a Near Infra-Red (833 nm) band; 6 × 20 m Bands: 4 narrow Bands in the VNIR vegetation red edge spectral domain (704 nm, 740 nm, 783 nm and 865 nm) and 2 wider SWIR bands (1610 nm and 2190 nm) for applications such as snow/ice/cloud detection, or vegetation moisture stress assessment.

Table 4.1: Formulas for remote sensing indices and the associated summary statistics

Name	Abbrev.	Mean Formula	Band Centre Formula
Normalized Difference Salinity Index	NDSI	$\frac{1600:1700 - 2145:2185}{1600:1700 + 2145:2185}$	$\frac{1650 - 2165}{1650 + 2165}$
Soil Composition Index	SCI	$\frac{1600:1700 - 760:860}{1600:1700 + 760:860}$	$\frac{1650 - 810}{1650 + 810}$
Misra Soil Brightness Index	MSBI	500:600+600:700+700:800+800:1100	550+650+750+950
Normalized Difference Water Index	NDWI	$\frac{1070 - 1200}{1070 + 1200}$	$\frac{1070 - 1200}{1070 + 1200}$

Reflectance summary	Minimum	1st Quartile	Median	Mean	3rd Quartile	Maximum
NDWI	-2.2672	-0.5096	-0.317	0.5775	-0.2016	0.6226
NDSI	-0.82124	-0.219073	-0.009457	0.070926	0.115664	0.973037
SCI	-0.8016	0.1349	0.2591	0.1996	0.3351	0.976
MSBI	0.0282	0.7066	0.9072	0.8845	1.0845	2.7007

Reference NOTES: Remote sensing indices database the remote sensing indices data base and the radiometric resolution guide (https://www.indexdatabase.de and https://sentinel.esa.int/web/sentinel/user-guides/sentinel-2-msi/resolutions/radiometric)

Soil Composition Index (SCI)

The soil composition index (SCI) was developed from the Advanced Space borne Thermal Emission and Reflection Radiometer (ASTER-4) satellite launched 1999-12-18 (IDB - Index DataBase). ASTER is a 15 band satellite with a spatial resolution of 15 m, 30 m and 90 m and wavelengths ranging from visible near infrared (VISNIR : 520 nm) to the thermal infrared – TIR:10950 nm (Survey and U.S. Geological Survey 2008). The SCI is purported to have the potential to detect chemical soil composition such as nitrogen or iron dioxides (Al-Khaier 2003). The SCI can be modelled to both Sentinel-2A as well as hyperspectral sensors using Hyperion as a reference (IDB - Index DataBase). The SCI has not been widely tested in its application to soil remote sensing. The limitation of freely available remote sensing data in the thermal infrared and the obstruction of soil by vegetation represent some major drawbacks in application. However, its use in laboratory remote sensing to compensate for or replace laboratory analysis as potential but has not been tested. The high cost of laboratory analyses makes hyperspectral remote sensing an attractive alternative for deriving trends in soil composition such as important edaphic factors *e.g.* bulk density. The SCI uses the red edge to near infrared region (7 60-800 nm) and the short wave infrared region (1600-1700 nm). These religions are often used to analyse vegetation. These regions are known to interact with the internal structure of vegetation tissues. It was therefore expected that these regions might also be useful in retrieving information about soil particle structure.

Normalized Difference Salinity Index (NDSI)

The natural interaction between salty sea water and soils along the coastline has driven wide application remote sensing indices of soil salinity (Chi *et al.* 2019; Das *et al.* 2010; Abdel-Kader 2013; Land *et al.* 2011). The monitoring of salinity intrusion has been a key application area for soil salinity indices (Nguyen *et al.* 2020). The normalised different salinity index (NDSI) has an accurate detection for overall salinity and is applicable on exposed to soils (Al-Khaier 2003) . The application of the salinity index to detect changes in soil chemical salt conditions has been widely used in literature despite the limitation in areas where vegetation covers the soil. The idea of collecting soil samples underneath vegetation over geo-referenced special skills and analysing it using laboratory spectroscopy has potential to solve the challenge of vegetation cover however has not been widely tested. Variations in the salt content of the soil underneath vegetation poses another unique challenge when correcting soil background attenuation of the remote sensing signal when analysing vegetation.

Soil Brightness Index (SBI)

Misra and Wheeler (1977) performed PCA and computed the Misra Soil Brightness Index (MSBI) along with two vegetation indices using Landsat data (Xue and Su 2017). Unlike other that are ratio based soil indices the MSBI is based on band addition in the Near ultraviolet, Visible (NUV) region (100 – 1100 nm) without any subtraction or division, *i.e.* orthogonal indices (Bannari *et al.* 1995). Specifically, this band addition math uses the green to yellow visible range - (500 – 600), yellow to red visible range (600 – 700 nm), red edge to near infrared 700 – 800 nm and near infrared to extreme infrared (800 – 1100 nm) wavelengths (IDB - Index DataBase). The MSBI can be traced back to the foundational works of Kauth and Thomas (1976) based on the tasselled cap approach. Changes in the soil brightness index can be influenced by changes and sorry moisture soil organic matter as well as salinity.

Normalised Difference Water Index (NDWI)

The Normalized Difference Water Index (NDWI) is derived from a band ratio of Near-Infrared (NIR) and Short Wave Infrared (SWIR) channels (Gao 1996; Gao 1995). The traditional application such as that of Tucker (1980) emanates from the premise of the response of SWIR reflectance to changes in both the vegetation water content and the spongy mesophyll structure in vegetation canopies, and the response of the NIR reflectance to leaf internal structure and leaf dry matter content (Ceccato *et al.* 2001; Ceccato *et al.* 2002; Jackson 2004; Huang *et al.* 2009). Against this background we hypothesized the non-traditional use on dry soil samples (Delbart *et al.* 2005). We hypothesise that these two wavelength regions (SWIR and NIR) should also respond to variations in soil structure (Gu *et al.* 2007; Gu *et al.* 2008; Wardlow and Egbert 2008). We proposed that its response to the structures of spongy mesophyll cells would also interact with the different soil structural compositions that emanate from differences in waterlogging characteristics of soil along the wetland gradient (Delbart *et al.* 2006; Jackson 2004).

4.3 Data analysis

Density plots as well as box and whiskers plots were used to visualise and assess the variability of the three soil remote sensing indeces. Significance tests were conducted to assess the statistical validity of the results. All analyses (Appendix) were conducted using R version 3.5.0 (R Core Team 2019). The Tukey Honest Significant Difference test, accounting for the Bonferroni effect was used to control for Type I errors in multiple comparisons. In order to test the significance of the hypotheses at α 0.05, the possible number of combinations or Bonferroni coefficient (m) for eight wetlands was m=28. The m value and new alpha level of 0.001 were calculated using the combination formula (Eq. 5). Where the default alpha level (0.05) is divided by m *i.e.* *(nCr = n / r * (n - r))*. Where n represents the total number of items *i.e.* 8, and r represents the number of items being compared at a time *i.e.* 2, to calculate the Bonferroni adjustment alpha level. Maps in this paper were created using ArcGIS® software by Esri. ArcGIS® and ArcMap™ used herein under intellectual property license, Copyright © Esri, unless otherwise stated. For more information about Esri® software, please visit www.esri.com.

4.4 Results

4.4.1 Differences and groupings in soil indices between wetlands

The mean values (Appendix K) of the eight wetlands in were significantly different in SCI ($F_{7,775} = 16.7234$, $p < 0.001$), NDSI ($F_{7,775} = 31.5774$, $p < 0.001$), NDWI ($F_{7,775} = 51.4653$, $p < 0.001$) and), MSBI ($F_{7,775} = 74.2812$, $p < 0.001$) at a Bonferroni adjusted alpha level ($p<0.001$, One-way Analysis of Variance (ANOVA)), as it can be observed in Figure 4.4 (Results of multiple comparisons with Turkey's Honestly Significant Difference (HSD) are presented in Appendix L). The THSD multiple comparisons of 28 comparable pairs showed that the significant difference ($p<0.001$) can be attributed to 17 pairs for NDWI, 14 pairs for MSBI and 11 pairs for SCI and NDSI (Appendix L). The results from ordination analysis, conducted using all four variables, revealed three groups of wetlands (Principal Component analysis - PCA, Figure 4.5). Group A biased towards NDWI and MSBI, group B biased towards SCI and group C biased towards NDSI. The first two PCA axes were the most important latent variables that were highly correlated (85.42%) to the four variables. Therefore, ordination results further support the ANOVA findings of the importance of spectral difference in differentiating the wetlands from one another (Figure 4.4).

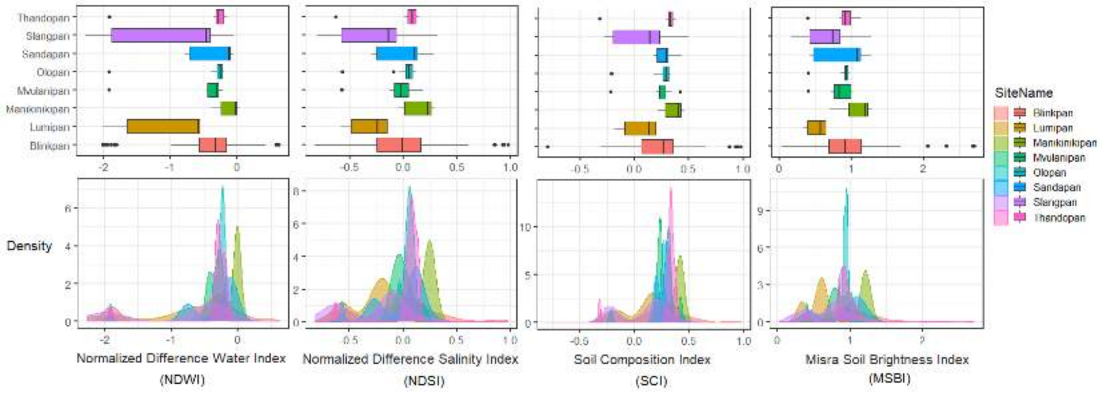


Figure 4.4: Median NDWI, NDSI, SCI and MSBI across the eight sampled wetland sites (data that is combined by site). Data have been ordered alphabetically by site name. Sample density distributions of the four soil remote sensing indices of the eight wetlands appear below following the same colour scheme of the horizontal boxplots.

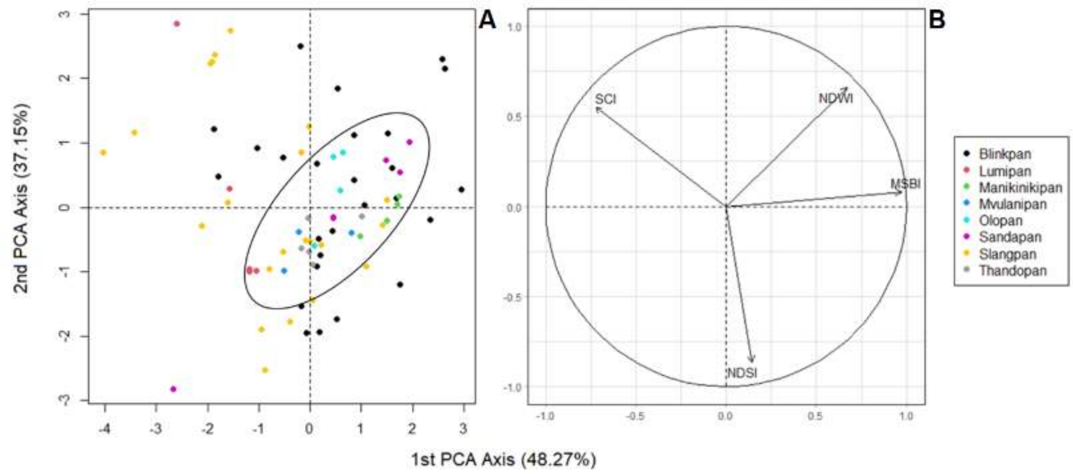
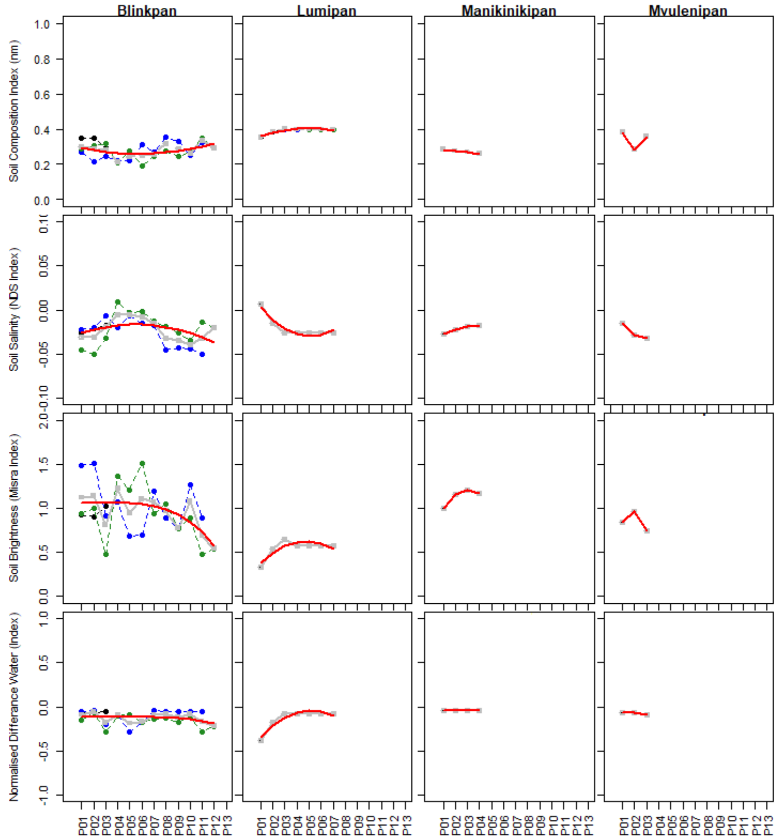


Figure 4.5: PCA ordination diagram of the Lake Banagher Wetland Soil Dataset with data points representing plots grouped by site (colours). The correlation circle plot (panel B) based on eigenvalues of the correlation matrix for the four active variables. Projection of the variables on a 1 × 4 factor-plane (right). Data point in each on the four regions of the ordination space are influenced be the variable that is correlated with the specific region of the ordination space.

4.4.2 Trends in the edaphic factors from the open water body to the outer dryland.

Generally, there were negative trends in the response of SCI (r^2=0.98-0.34) and NDWI (r^2=0.99-0.20) along the gradient from the centres of the wetlands to the outer dryland boundary, while NDSI and MSBI had positive trends (r^2=0.99-0.25 and r^2=0.99-0.21 respectively). However, at relatively short distances, ranging from 30 to 70 m, this probably reflected the extent of the palustrine section of the depressional wetlands (Figure 4.6 and Appendix K). When the data from the remote sensing indices were combined across the gradients of all wetlands to drylands, the indices maintained their general trends, SCI and NDWI = negative, NDSI and MSBI = positive (Figure 4.7). The end of the patterns of edaphic factors, which probably reflects the mean seasonal maximum extent of the wetland, at 70 m on average for combined data with the 8th plot (80 m) showing a change in the direction of the pattern to the opposite direction.



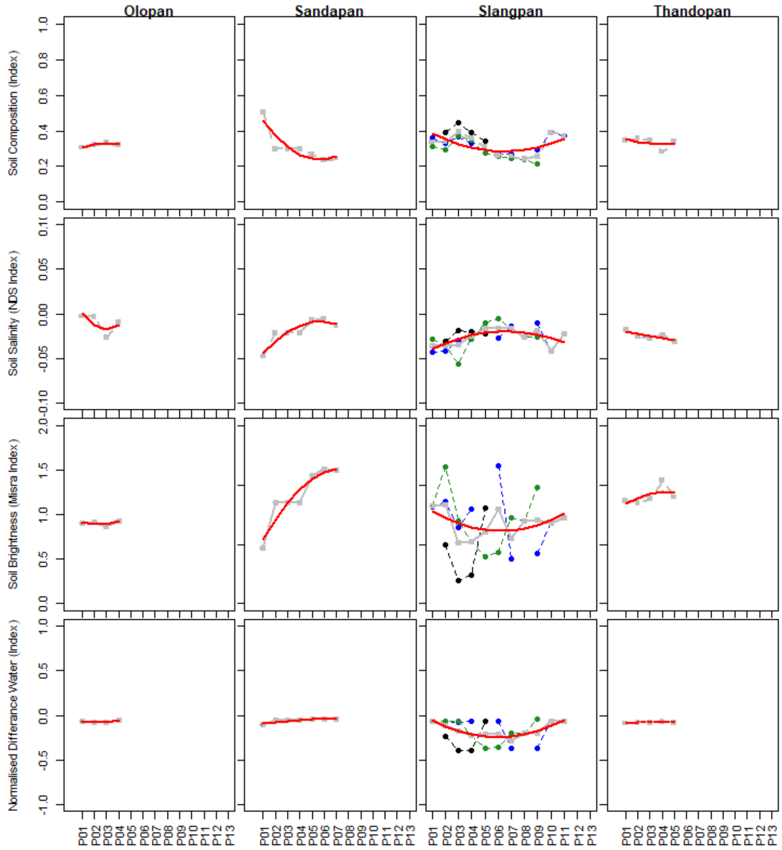


Figure 4.6: Polynomial regression models (red solid lines, mathematical model iterations and statistics in appendix) of soil remote sensing indeces to increasing distance from the edge of the wetland water body.

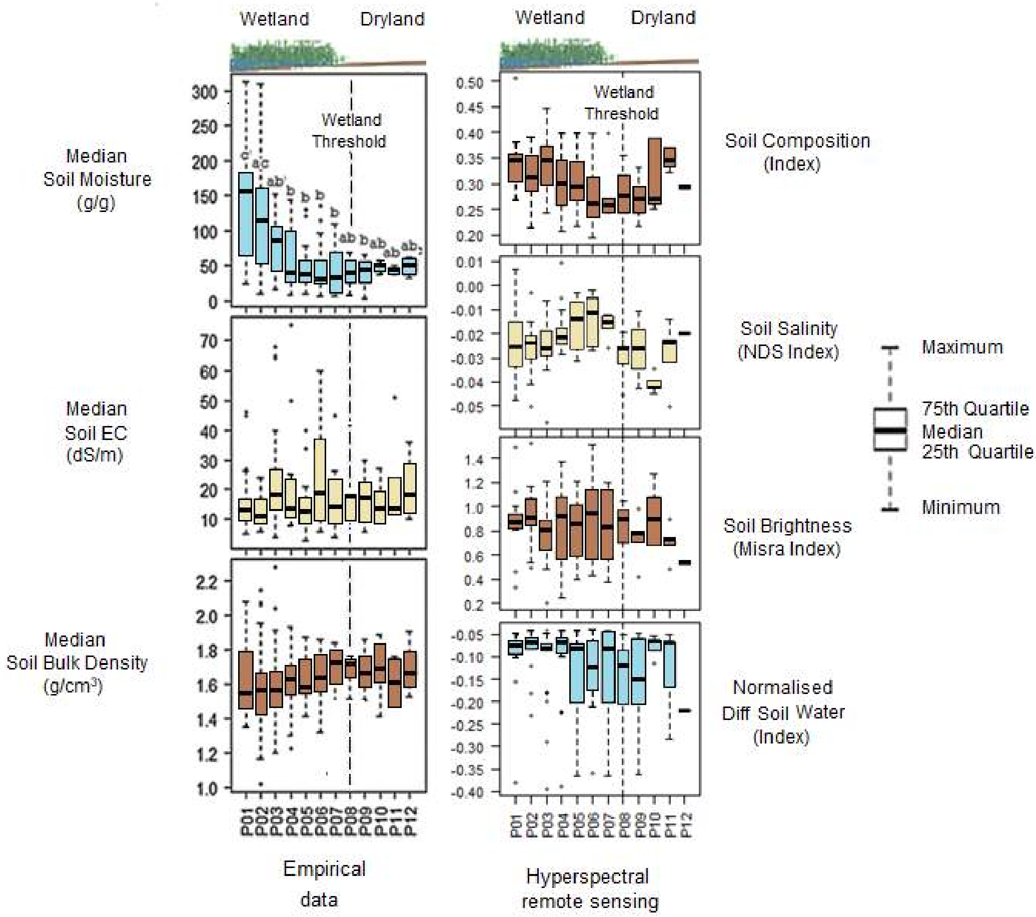


Figure 4.7: Mean values of soil spectral indices aggregated by plot number across the literal gradient of a depressional wetlands; showing adjacent groups of plots being similar spectral indices and differences in spectral indices between the two sides of the wetland threshold

4.5 Discussion

4.5.1 Differences in edaphic factors between depressional wetland

In this study, we investigated differences in soil remote sensing indices among eight depressional wetlands in a temperate grassland biome. Our results showed that significant site level differences were detected in SCI (11/28), NDWI (17/28), NDSI (11/28), and MSBI (14/28), wetland pairs (p<0.001). Due to the convenience of spectral indices in highlighting specific types of land covers many indices have been developed for application in remote sensing. However, fuel indices have been developed to enhance soil information. A major reason beside the complexity of soil properties such as soil moisture texture as well as physical and chemical composition, is the obstruction by vegetation. Fortunately, the laboratory hyperspectral approach proposed in our empirical method from this paper may serve as a better alternative to address this difficulty. The detailed nature of hyperspectral information effectively avoids the spectral variability problem. The aforementioned soil indices have been frequently used in mapping soil properties in previous studies (Al-Khaier 2003; Chi *et al.* 2019; Das *et al.* 2010; Abdel-Kader 2013; Land *et al.* 2011;Misra and Wheeler 1977; Gao 1996; Gao 1995; Delbart *et al.* 2005; (Delbart *et al.* 2006; Jackson 2004). Due to the novelty of our study there are no previous studies that have applied these remote sensing indices to the delineation of wetlands or grouping wetland sites (Wilson and Norman 2018). These results can be relied upon because these results were produced with data that were collected from repeated measurements of spectral data. This means that the variability within symbols and with insides was adequately repeated to arrive at a reliable mean spectral value. There is a need for further research on the seasonal time-series of spectral indices in the littoral zone of depressional wetlands (Li *et al.* 2015; Chi *et al.* 2019) . We used principal component analyses to assess the statistical differences in spectral indices across different scales; site and plot levels. In ordination the orthogonality of latent variables achieves direct multiple comparison of similarities between spectral indices across all plots and sites simultaneously. We tested similarities in spectral indices across the wetland sites. The results on differences in spectral indices across the eight wetlands showed that although the wetlands differ in characteristics that partly affect or drive the spectral indices, there is still convergence or grouping in the trends spectral indices. Meaning similarity in spectral indices are present within formed groups and these groups are more likely to result in wetland groups with similar functioning. This group therefore is very useful for managing wetlands over large scales where there is a need to know where the same management can be repeated or where methodologies would need to differ within the same HydroGeoMorphic unit. Other regions can therefore use similarity grouping based on spectral indices to discern monitoring and management regimes across many wetland sites anywhere in the world.

4.5.2 Patterns of edaphic factors along the wetland littoral gradient

The spectral indices were observed along the wetland littoral gradient showed a negative trend for NDWI and SCI and a positive trend for NDSI and MSBI. These trends are related to field capacity, which is the amount of water content held in the soil after excess water has drained away and the rate of downward movement has decreased (Colman 1947, Castelli *et al.*, 2000, Twarakavi *et al.* 2009). The available water capacity, is as well important in explaining these patterns, and it refers to the ability of soil to hold water from infiltrating to the lower levels of the soil profile but yet making it available to plants (Cassel and Nielsen 1986). It is the water held between field capacity and the wilting point.

4.5.3 The wetland-dryland threshold boundary for delineating endorheic wetlands

In this research we tested whether remote sensing soil indices can be used to delineate the boundary of endorheic wetlands by thresholding these edaphic factors; similarly, to studies in the PPR, situated in the temperate grasslands of the US (Wu and Lane, 2016). However, the delineation of thresholds of endorheic wetlands from Wu and Lane (2016) are based on micro elevation that is determined using Light Detection and Ranging and do not specify the distance from the wetland water body. In this study, the empirically derived threshold of the maximum

$$\left(\frac{\left(Plot X_1 of i^{th} transect \in j^{th} site\right)}{\left(Plot X_2 of i^{th} transect \in j^{th} site\right)}\right)\times\left(\frac{100}{1}\right)$$

extent of individual wetlands ranged between 30 m and 70 m. However the aggregate threshold for all eight depressional wetlands, based on the three median soil remote sensing indices, was 70 m, hence, we recommend the use of a maximum buffer of a 100m, in order to add a precautionary vegetation buffer of 30 m to accommodate the ferralitic zone of subsurface incoming seepage. The buffer width should be based on site specific recommendation using the percentage change threshold, hence the 100 m is a policy recommendation, not a scientific result. Ma (2016) suggests a minimum buffer with of 20 m (Semlitsch and Bodie 2013). Wetland buffering is important for wetland management and water protection, flooding control, groundwater storage, habitat for wild species, recreation, aesthetic and removal of sediment and pollutants (Castelle *et al.*, 1992; Correll 1996; Wenger and Fowler 2000; Gleason *et al.*, 2003). In theory, for generalisation of a percentage change threshold can be used in the place of a distance measure. This theoretical approach allows the results of our study to be applied to other wetlands globally and can therefore be theoretically represented (equation 6 simplified as equation 7) for determining the wetland threshold using empirical measurements of edaphic factors (Nondlazi et al in review).Therefore, this result is crucial for the South African policy framework and environmental impact assessments (Macfarlane *et al.* 2015).

4.6 Conclusion

This study showed that depressional wetlands that occur in the temperate grassland biome as represented by a sample of eight depressions in the MLD ecosystem are significantly different in soil remote sensing indices and might be related to the same indices from Senitnel-2A, at plot pixel level but this remains to be tested. However, similarities are present among some of the wetlands depressions are related to the differences in sensitivity to climate change This study also revealed consistent horizontal trends in the soil remote sensing indices from the open water to the outer dryland, characterised by a declining trends for SCI and NDWI and increasing trends for NDWI and MSBI. This study demonstrated that for depression wetlands within the MLD the wetland threshold (threshold between dryland and wetland) can be empirically detected at a relatively short distances of about 30 to 70 metres; a threshold where the trends of soil remote sensing indecs change to opposite directions with a percentage change that is greater than 5%. This threshold can potentially inform the delineation of the outer edge of endorheic wetlands, which are poorly mapped globally for wetlands that are under threat. We therefore conclude that: Depressional wetlands are characterised by narrow littoral zones possible as universal characteristic that is detectable using remote sensing. Depressional wetlands are dynamic and are poised to suit a high diversity of floral and faunal species and can be potentially detected from satellite remote sensing using Senitnel-2A. The minimum depressional wetland buffering in legislation should be considered at 100 m in order to protect this HGM unit. Current wetland buffering legislation might too narrow. We can now detect the wetland boundary using remote sensing soil indices, especially those that are retrievable with soil data and potentially vegetation data as well. We can now detect wetlands that were previously extremely hard to detect, for example when using remote sensing, such as 10 m spatial resolution data that do not have a permanent water body. We have made progress in demonstrating that the objective delineation of the wetland threshold and its associated permanently, seasonally and temporally inundated regions of the littoral zone is achievable.

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Chapter 5

Determining the wetland boundary of depressions using gradient analysis of littoral vegetation with Sentinel-2A indices

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Abstract

Depressional wetlands are highly vulnerable to changes in land surface temperature and rainfall. Changes in climate alter the spatial variability of wetland littoral vegetation, but this variability has not been studied using Sentinel-2MSI remote sensing indices as an alternative to soil assessment with a soil auger. This study aimed to assess spatial variability of Sentinel-2MSI remote sensing vegetation indices as an alternative to soil assessment with a soil auger. We studied remote sensing vegetation indices between wetlands and along littoral gradients of wetland banks. A set of 85 paired sub-sample measurements of four vegetation remote sensing vegetation indices were collected from 10x10m plots along 14 belt transects in eight representative wetlands in the Mpumalanga Lake District, South Africa. In general, at Bonferroni adjusted p –value (0.001) there were significant differences between the eight wetlands for Normalised Difference Vegetation Index (NDVI) and Normalised Difference Salinity Index (NDSI) but no Red-edge Normalised Difference Vegetation Index (RENDVI) Normalised Difference Water Index (NDWI), $0.001<p<0.05$. NDWI and RENDVI were important in differentiating the eight wetlands. NDSI, NDWI and NDVI generally showed negative trends over short distances, ranging from 30 to 70 m, along the littoral gradients. RENDVI generally showed positive trends. Understanding of the spatial variability of remote sensing vegetation indices helps in the management and monitoring of depressional wetlands in the era of climate change.

Keywords: *Depression wetland, Sentinel, Remote Sensing, vegetation indices, Salinity, NDSI, NDVI, NDWI, RENDVI*

5.1 Introduction

Wetlands have international importance (McInnes 2013; Goodwin 2017). They play an integral role in the ecology of the watersheds where they are located, and supply ecosystem services beyond the boundaries of their watersheds (Laidig and Zampella 1999; McKinney and Charpentier 2009). Many faunal and floral species of commercial importance, such as fish, reeds and papyrus are harvested from wetlands (Chapman 1999; Mnaya *et al.* 2007; Small 2017; Zolfaghari 2018). Wetlands are a wildlife refuge and nursery hence they are biodiversity hotspots and among the most productive ecosystems worldwide (Pantshwa *et al.* 2019; Lofgren 2020). Wetland biodiversity, especially vegetation, is crucial for wetland ecosystem services (Pantshwa *et al.* 2019). Water filtration, sediment trapping, floodwater retention, and carbon (C) storage are some of the ecosystem services owed to wetland littoral vegetation (Uwimana 2019). Wetlands also control the source, amount, and distribution of sediment and nutrients. Hence, they influence the temporal and spatial distribution of other floral and faunal organisms (Han and Park 2014). Wetlands facilitate mitigation of climate change through wetland carbon sequestration that outweighs its methane (CH4) emissions despite being Earth's largest natural source of 185±21TgCyr1 atmospheric flux (Melton *et al.* 2013; Zhu *et al.* 2015; Saunois *et al.* 2016). Wetlands have low decomposition rate hence they are estimated to store 4% to 30% of Earth's 2500Pg soil C pool (Ji *et al.* 2020; Rejmánková and Sirová 2007). Therefore, wetland degradation through drainage, overgrazing and changes in vegetation needs to be monitored to avoid decline and extinction of biodiversity, increased carbon and methane emissions and reduction in sequestration capacity.

Wetland vegetation health is a critical issue of wetland biophysics (Zhang *et al.* 2019); responsible for sustainability of wetland ecosystem services (Milne 2018; Yang 2020). Without properly functioning biophysical processes, wetlands would be unable to provide the valuable ecosystem services that they are popular for (Ollis *et al.* 2015). Tall reeds found along the fringes reduce the speed of water currents (Yano *et al.* 2017). Reduction in water currents increases sand and sediment deposition, which improves water quality (Fennessy *et al.* 1994; Stubblefield *et al.* 2006; Qiu and McComb 2000). During water drawdown floating chemicals get trapped along the sandbank and get exposed to radiation under low moisture conditions (Karcz and Mackiewicz 2009). This allows pathogens to be sterilised out of the water. Specialised annual vegetation is able to cover this sandbank during some periods of the winter months to limit excessive drying of the soil (Karcz and Mackiewicz 2009). Hence, healthy wetland vegetation is so crucial for the sustainability of depressional wetlands (Evans and Freeland 2000). The loss of wetlands has gained considerable attention over the past few decades, up to 50% loss since 1900 (Davidson 2014). That is why wetland monitoring under changing climatic conditions has to focus on the wetland littoral vegetation. Climate change threatens the persistence of vegetation species along wetland littoral zones (Carrington *et al.* 2001). Monitoring using quantitative ecological surveys and nonparametric approaches are incapable of delivering nationwide monitoring of depressional wetlands (Ovaskainen *et al.* 2016). Remote sensing has become the popular option for monitoring and mapping wetlands at national scales especially (Rebelo *et al.* 2009; Adam *et al.* 2010; Mahdavi *et al.* 2018; Adeli *et al.* 2020). However regarding vegetation on the littoral banks of depressional wetlands; remote sensing data with spatial scales that are small enough to quantify spatial changes in every meter are still not freely available *e.g.* World View and aerial images (Baetz 2000; Aroma and Raimond 2015). Optical remote sensing is one of the most attractive options because it offers vegetation indices and some data have been distributed free of charge (Verrelst *et al.* 2015; Jackson and Huete 1991; Huete 2012; Alam *et al.* 2017). The opportunities to obtain optical remote sensing data have improved due to the Sentinel-2A satellite launch on June 23, 2015 (Djamai and Fernandes 2018). Now, it is collecting multispectral data including 13 bands covering the visible, shortwave infrared bands (SWIR) wavelength regions that are freely available (Huang *et al.* 2016; Du *et al.* 2016). However, sufficient consideration has not been given to the potential of vegetation indices that could quantify the wetland boundary despite coarse resolution (10m vs 1m). Sentinel-2A provides various vegetation spectral indices that can be extracted including in the SWIR region (Sonobe *et al.* 2018). These indices are influenced by plant properties *i.e.* pigments, leaf water contents, biochemical, physiological and biophysical properties that vary at fine resolution <1m (Cho *et al.* 2007; Main *et al.* 2011; Cho *et al.* 2008). There is specific interest in the normalized difference vegetation index (NDVI), red-edge normalized difference vegetation index (RENDVI), normalized difference salinity index (NDSI), and the normalized difference water index (NDWI) for studying wetland vegetation (Pettorelli 2013; Fernández-Manso *et al.* 2016; Wang *et al.* 2019; Zhang *et al.* 2019). However, while NDVI values < 0.20, are known to represent non-vegetative surfaces, and values <-0.0 are known to represent water or very moist surfaces; but NDVI value between -0.0 – -0.4, and corresponding ranges in RENDVI, NDWI and NDSI have not been declared or tested for delaminating the wetland threshold. We hypothesized that the detection of the wetland boundary can be achieved using Sentinel-2A VIs *i.e.* NDVI, RENDVI, NDWI and NDSI.

However, most literature focuses on regression and classification models built based on reference data from random field plots of Sentinel-2 satellite images or comparing these capabilities when using different acquisition dates (Bué *et al.* 2020; Mateo-Garcia *et al.* 2018; Lo 2008; Zhang and Du 2019; Xian 2007; Wang *et al.* 2010; Dube *et al.* 2015; Feng *et al.* 2018) . Most interest focuses on estimating vegetation properties and classification mapping. These include biomass and leaf area index and performance of machine-learning algorithms *e.g.* k-nearest neighbours vs random forest vs support vector machines (Jung and Lee 2019; Kumar and Mutanga 2019). Compared to SPOT, RapidEye, Landsat and MODIS; Sentinel-2 provides the red-edge spectral bands that extend its potential usefulness for analysis of vegetation, but free of charge and at competitive spatial resolution (10–20 vs 30 m or 250 m) (Koutsias and Pleniou 2015; Haddad 2018). For these reasons, it justifiable to investigate the capability of Sentinel-2A data on VIs in delimiting the wetland boundary using gradient analyses and parametric statistics of data derived from systematic transect sampling; which is a novel approach. The usefulness of Vegetation Indices (VIs) derived from Sentinel-2 data for estimation of the threshold between wetland and dryland has not been investigated before to our knowledge. Based on the experience of other researchers who investigated a variety of satellite sensors in wetland vegetation, we see Sentinel-2 images as a valuable source of data for such applications. There is also limitation in the number of studies that do not use machine-learning or mapping methods in the context of Vis in wetlands. Within this framework, the aim of the present study was to evaluate the potential of Sentinel-2 data for delineating the wetland threshold. The objectives were to sample vegetation indices along several belt transects in eight representative depressional wetlands and used to (i) assess potential differences in vegetation indices between wetlands and (ii) analyse trends vegetation indices from the open water body (centre of the wetland) to the outer dryland.

5.2 Material and Methods

5.2.1 Summary methodology

To ensure repeatability of the research the study was conducted systematically and all the critical steps were recorded. The general methodology (Figure 5.1) includes satellite remote sensing data, which was used to guide the process of selecting sample plots, transects, wetland sites and ecological and data science principles. The alignment of the data collection with remote sensing ancillary data ensured further repeatability because remote sensing data is publicly available. Therefore the exact sample locations where these data were collected can be retrieved by subsequent researchers.

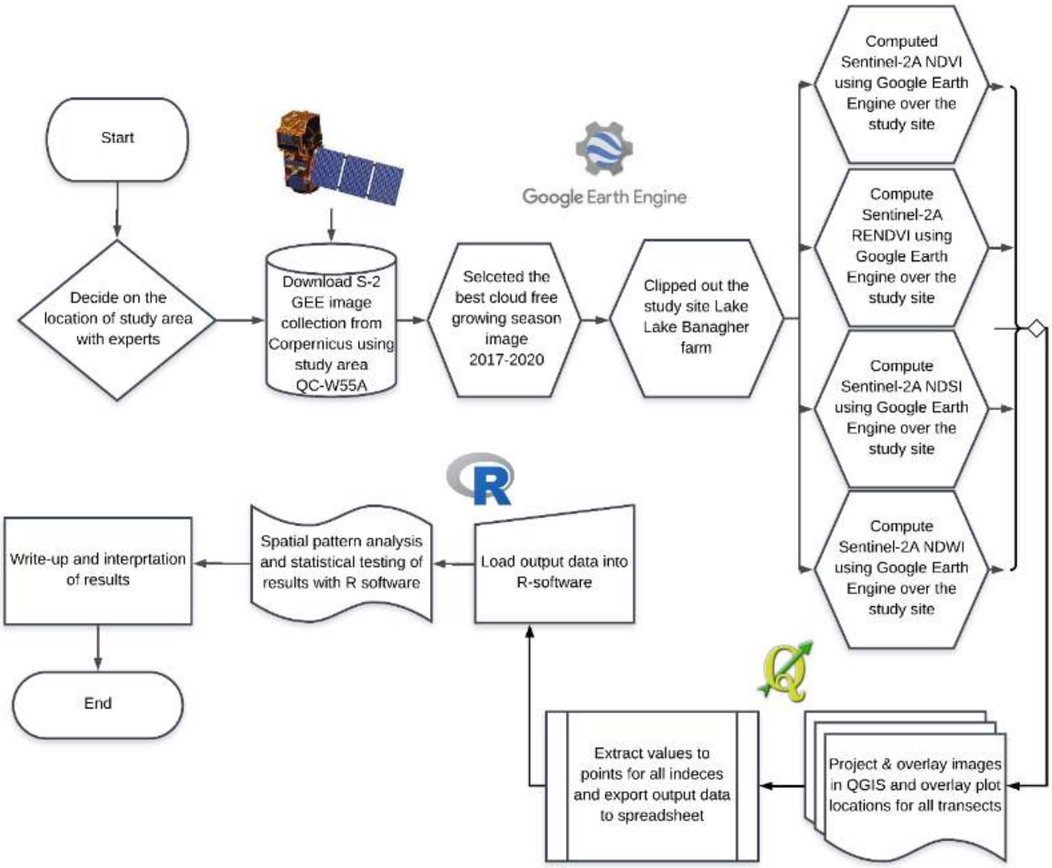


Figure 5.1: The summary overview of the methodology followed in the research.

5.2.2 Selection of study area

The MLD was chosen as a study area because the area is rich and diverse in different types of depressions (and other wetlands) and therefore can be a good case study for isolated wetland ecosystems globally like the PPR of the US. The geology is underlain by a sequence of two strata. First, is the Ecca group; a topping of sedimentary deposits consisting mostly of shale and sandstone and the Dwyka Group below in the stratigraphic position. The catchment receives 767 mm of mean annual precipitation (Department of Water Affairs (DWA), 2003). W55A has over 300 depressional wetlands in just a 20-odd kilometre radius (Goudie and Thomas 1985; De Klerk *et al.*, 2016; Van Deventer *et al.*, 2020a; Van Deventer *et al.*, 2020b). Within MLD a subset of depressional wetlands were selected (Appendix C, Lake Banagher Farm, 26°20'11.21"S, 30°21'14.03"E, in the Gert Sibande District, in the Msukaligwa Local Municipality, Mpumalanga Province, South Africa, Appendix A). The wetland ecosystem types and the wetland vegetation are very diverse due to variations in elevation, size, shape and the area of the vegetated zones. Therefore, there is a good chance of covering a wide range of habitats in a relatively small area (Watson, 1986; Brooks and Hayashi, 2002; El-kawy *et al.*, 2010; Wilkinson *et al.*, 2016; Vanderhoof *et al.*, 2018).

5.2.3 Selection of sampling locations – the depressional wetland sites

Eight depressional wetlands (Appendix C) were selected to represent the range of depressional habitats (Figure 5.2). The diversity of wetlands was observed in terms of (a) extent of the water body, (b) extent of vegetation cover (c) shape and (d) size.

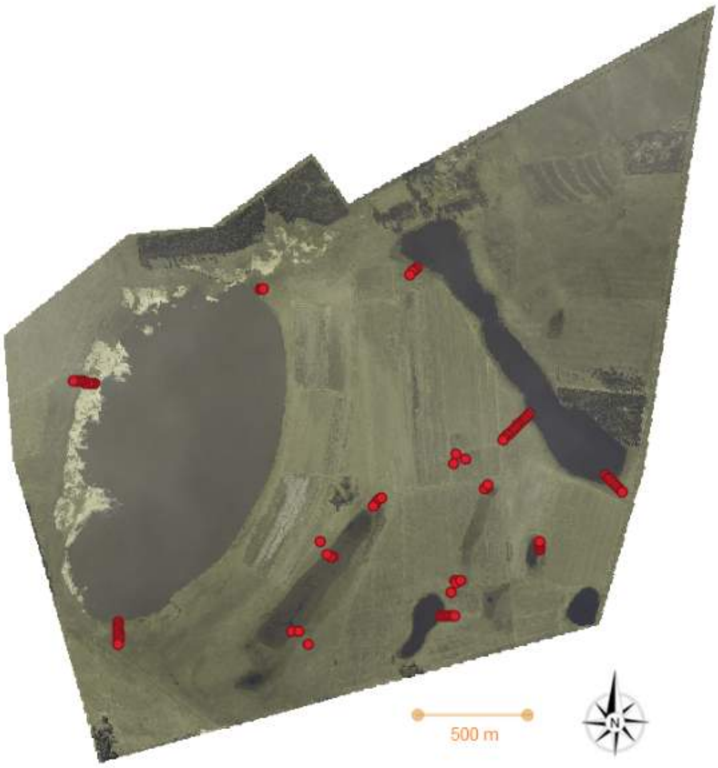


Figure 5.2: A map showing the positioning of the sampled transects numbered T1 to T3 (red coloured 10 m × 10 m belt transect of plots 100 m^{2}) lines are for illustration purposes. The image has been clipped to the shape of the current boundary of the Lake Banagher farm. The map was created using Sentinel-2A at spatial resolution of 10m, natural colour composite.

5.2.4 Field surveys design

Two field surveys were conducted in order to sample wetland NDVI, RENDVI,NDSI and NDWI. The first survey was conducted in March 2018 and the second was conducted in November 2018. The first survey focused on two wetlands and the second one focused on six wetlands. Only two wetlands had data collected in both sampling periods.The sampling procedure was based on the belt transect method according to the Sentinel-2A pixels scheme. Sentinel-2A provides data with global coverage in a cycle of about once every 5 days from above the equator. In addition to near infrared and shortwave infrared bands, it has three red-edge bands (Bands 5-7 with the centre of the bands at 705, 740 and 783 nm respectively) which has been proven useful in vegetation classification and possibly for edaphic factors. The intention was to cover the range of variation in the visible vegetation physiognomy from the edge of the vegetated part of the wetland up to the dryland area that is bordering the wetland (Figure 5.3). The best location for transects was considered to be the region of the wetland-dryland gradient that had the highest turnover in pixel tone (colour variation). A high variation of pixel tone was considered to reflect higher turnover in species or vegetation structure or both. At each wetland, a field survey was conducted using the belt-transect method. The belt transect method was preferred because the intention was to sample a longitudinal gradient from the water body to the dryland. The width of transects was 10m as determined by the spatial resolution of Sentinel-2A. Transects had varying lengths, dependent on the width of the wetland zone (30 m - 130 m).

Setting up the belt transects

A mash grid made up of contiguous 10 m plots following the rows of Sentinel-2A pixels was generated in ArcGIS (ArcMap 10.5). This grid was projected on a true colour composite of the Sentinel-2A in order to identify the best locations for transects, following the approach by Goodman (1990). For wetlands greater than 0.2 km^2 (which were the three largest wetlands, Appendix C), three transects were selected around each wetland (Figure 5.2). For the smaller wetlands only a single transect was sampled. Plots with similar vegetation structure and composition as the one preceding them were not repeated. The purposeful sampling ensured that the sampling maximised the efficiency of representative sampling of landscape features and avoided repetitive sampling. The length of a transect was limited by the fence or by reaching dry ground.

5.2.5 Imagery collection, data pre-processing and selecting vegetation indices

Concurrent availability of 10-day Sentinel-2 data provides an unprecedented opportunity to gather high-resolution (10) data for national mapping of wetland ecosystems. Sentinel-2 data became available for South Africa in the middle of 2015 and its capabilities to map wetland ecosystem types, boundaries and species still requires adequate assessment. The easy and simultaneous access to entire archive of Sentinel-2 products, as well as the fast and scalable computational tools through GEE, makes GEE an essential and powerful tool wetland monitoring and assessment. Processing cloud-free imagery composition can be a challenging task, however the combined used of coding computations in R and GEE offer seamless alternatives to expensive software such as ARCGIS and ENVI. The study location (Lake Banagher farm) was identified and delineated with a polygon using four vertices. A Sentinel-2 MultiSpectral Instrument (MSI), Level-2A image collection was downloaded from the European Union - Copernicus (ESA). Images that fall within the interval of the target dates ("2017-07-01", "2020-09-30") were filtered from the downloaded collection of images using the "filterDate" function algorithm. The resulting subset collection was then sorted in ascending order by the metadata property, cloud cover from the least cloud cover to the highest, using the sort function. This function uses the S-2A cloud probability is created with the Sentinel 2 cloud detector library (using LightGBM). All bands are up-sampled using bilinear interpolation to 10m resolution before the gradient boost base algorithm is applied. The resulting 0..1 floating point probability is scaled to 0..100 and stored as a UINT8. Areas missing any or all of the bands are masked out. Higher values are considered to be clouds or highly reflective surfaces. The first image out of this collection - *i.e.* the most cloud free image in the date range, was selected and used for the analysis (COPERNICUS/S2 SR/20191004T074749 20191004T080733 T36JTR). Define visualization parameters were defined in a JavaScript dictionary for the rendering of a true colour composite as bands 4,3 and 2 as RGB respectively. Normalised Different Salinity Index (NDSI) was computed as" (SWIR1 - SWIR2) / (SWIR1 + SWIR2)". Where SWIR1="B11" (1610 nm) and SWIR2 = "B12" (2190 nm) at a spatial resolution of 20 m. Normalised Difference Water Index (NDWI) was computed as" (GREEN - NIR) / (GREEN + NIR)". Where GREEN="B03" (560 nm) and NIR= "B08" (842 nm) at a spatial resolution of 10 m. Normalised Difference Vegetation Index (NDVI) was computed as" (NIR - Red) / (NIR + Red)". Where NIR="B08" (842 nm) and Red= "B04" (665 nm) at a spatial resolution of 10 m. Red-edge Normalised Difference Vegetation Index (RENDVI) was computed as" (VRE1 - VRE2) / (VRE1 + VRE2)". Where VRE1="B05" (705 nm) and VRE2= "B06" (740 nm) at a spatial resolution of 20 m (Figure 5.3).

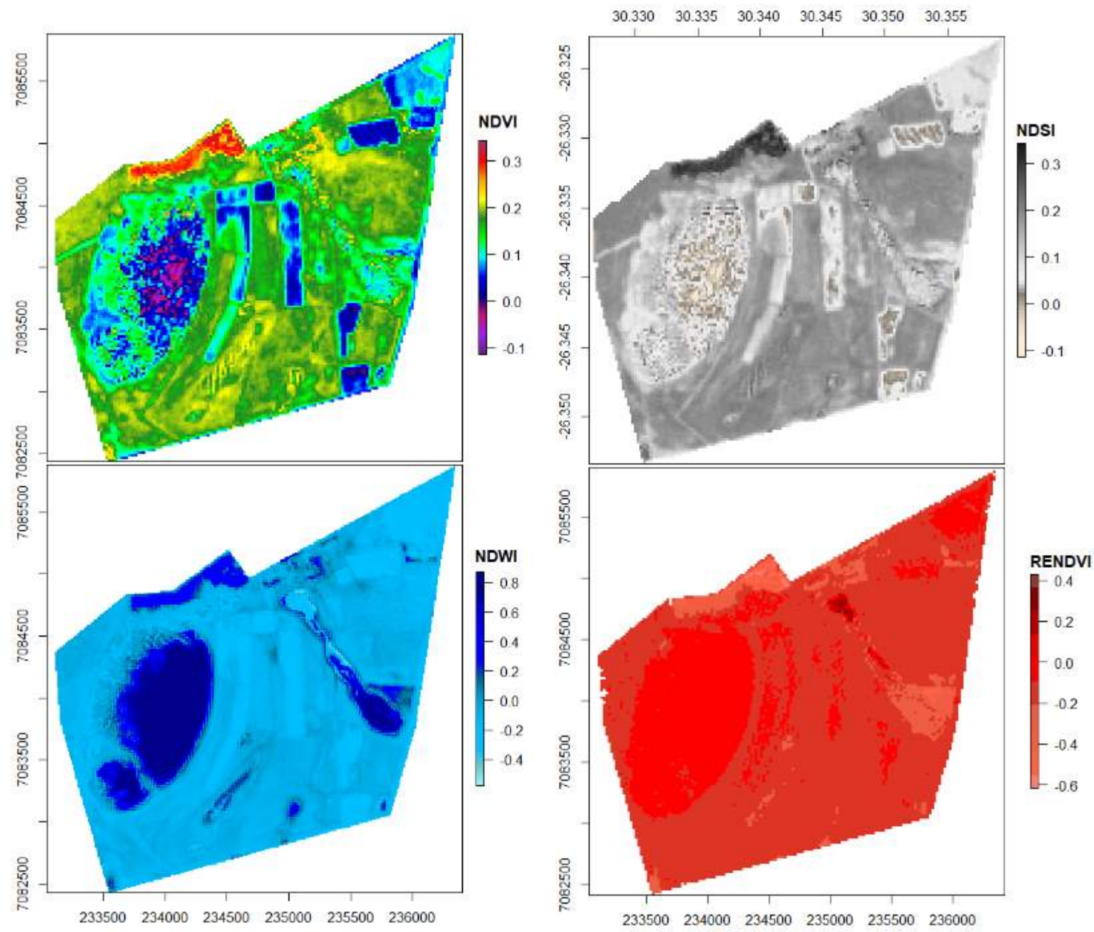


Figure 5.3: Maps of indices from which the data were extracted . The maps were produced using Google Earth Engine.

Normalised différence (NDVI) and Red-edge Normalised différence vegetation index (RENDVI)

The normalized difference vegetation index was one of first satellite vegetation indices and is strongly correlates with canopy cover ($r2 = 0.84$), photosynthesis and primary production of vegetation. For, Sentinel 2 it is calculated using the visible red and the Near Infrared (NIR) bands. These religions are often used to analyse vegetation. These regions are known to interact with the internal pigment and chemistry of vegetation tissues. The red-edge NDVI is less prone to saturation because it penetrates the vegetation canopy. This means that it can measure the variation of leaf foliage that is not exposed, that is located at the bottom of the canopy. This mean while NDVI would decline when the species composition changes to more sedge species that have less leaves and therefore less chlorophyll, which would confound the same declining response when vegetation biomass declines.

Normalized Difference Salinity Index (NDSI)

The natural interaction between salty sea water and soils along the coastline has driven wide application remote sensing indices of soil salinity (Chi *et al.* 2019; Das *et al.* 2010; Abdel-Kader 2013; Land *et al.* 2011). The monitoring of salinity intrusion has been a key application area for soil salinity indices (Nguyen *et al.* 2020). The normalised different salinity index (NDSI) has an accurate detection for overall salinity and is applicable on exposed to soils (Al-Khaier 2003) . The application of the salinity index to detect changes in soil chemical salt conditions has been widely used in literature despite the limitation in areas where vegetation covers the soil. The idea of collecting soil samples underneath vegetation over geo-referenced special skills and analysing it using laboratory spectroscopy has potential to solve the challenge of vegetation cover however has not been widely tested. Variations in the salt content of the soil underneath vegetation poses another unique challenge when correcting soil background attenuation of the remote sensing signal when analysing vegetation.

Normalised Difference Water Index (NDVI)

The Normalized Difference Water Index (NDWI) is derived from a band ratio of Near-Infrared (NIR) and Short Wave Infrared (SWIR) channels (Gao 1996; Gao 1995). The traditional application such as that of Tucker (1980) emanates from the premise of the response of SWIR reflectance to changes in both the vegetation water content and the spongy mesophyll structure in vegetation canopies, and the response of the NIR reflectance to leaf internal structure and leaf dry matter content (Ceccato *et al.* 2001; Ceccato *et al.* 2002; Jackson 2004; Huang *et al.* 2009). Against this background we hypothesized the non-traditional use on dry soil samples (Delbart *et al.* 2005). We hypothesise that these two wavelength regions (SWIR and NIR) should also respond to variations in soil structure (Gu *et al.* 2007; Gu *et al.* 2008; Wardlow and Egbert 2008). We proposed that its response to the structures of spongy mesophyll cells would also interact with the different soil structural compositions that emanate from differences in waterlogging characteristics of soil along the wetland gradient (Delbart *et al.* 2006; Jackson 2004).

5.3 Data analysis

Density plots as well as box and whiskers plots were used to visualise and assess the variability of the three vegetation remote sensing indices. Significance tests were conducted to assess the statistical validity of the results. All analyses were conducted using R version 3.5.0 (R Core Team 2019). The Tukey Honest Significant Difference test, accounting for the Bonferroni effect was used to control for Type I errors in multiple comparisons. In order to test the significance of the hypotheses at = 0.05, the possible number of combinations or Bonferroni coefficient (m) for eight wetlands was m=28. The m value and new alpha level of 0.001 were calculated using the combination formula (Eq. 5). Where the default alpha level (0.05) is divided by m *i.e.* (nCr = n / r * (n - r)). Where n represents the total number of items *i.e.* 8, and r represents the number of items being compared at a time *i.e.* 2, to calculate the Bonferroni adjustment alpha level. Maps in this paper were created using ArcGIS® and ArcMap™ software by Esri; used herein under intellectual property license, Copyright © Esri, unless otherwise stated. For more information about Esri® software, please visit www.esri.com.

5.4 Results

The mean values (Figure 5.4) of the eight wetlands in were significantly different in NDVI ($F_{7,77}$ = 4.3539, $p<$ 0.001) and NDSI ($F_{7,77}$= 7.0765, $p<$ 0.001) but no significant difference in NDWI ($F_{7,77}$= 3.135, p=0.0058), RENDVI ($F_{7,77}$ = 3.1995, p=0.005) at a Bonferroni adjusted alpha level (p<0.001, One-way Analysis of Variance (ANOVA)), as it can be observed in Figure 5.4. The results from ordination analysis, conducted using all four variables, revealed three groups of wetlands (Principal Component analysis (PCA)). Group A biased towards NDVI and NDSI, group B biased towards NDWI and group C biased towards RENDVI. The first two PCA axes were the most important latent variables that were highly correlated (88.38%) to the four variables. The first PCS axis were biased towards RENDVI (18.78%) and the second PCA axis were biased towards NDWI (69.51%).Therefore, ordination results further support the ANOVA findings of the importance of spectral difference in differentiating the wetlands from one another (Figure 5.5).

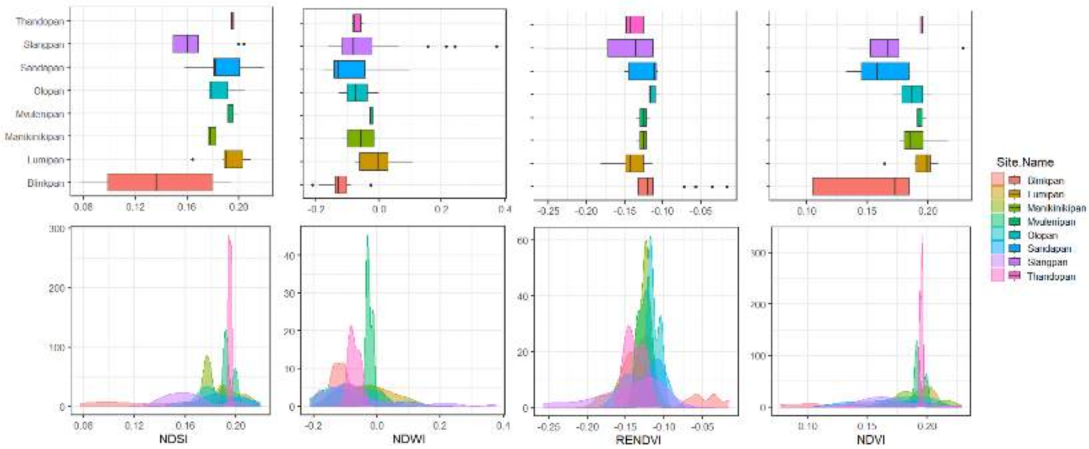


Figure 5.4: Median NDWI, NDSI, RENDVI and NDVI across the eight sampled wetland sites (data that is combined by site). Data have been arranged in descending alphabetic ordered by site name. Sample density distributions of the four vegetation remote sensing indices of the eight wetlands appear below respective horizontal boxplots following the same colour scheme of the horizontal boxplots.

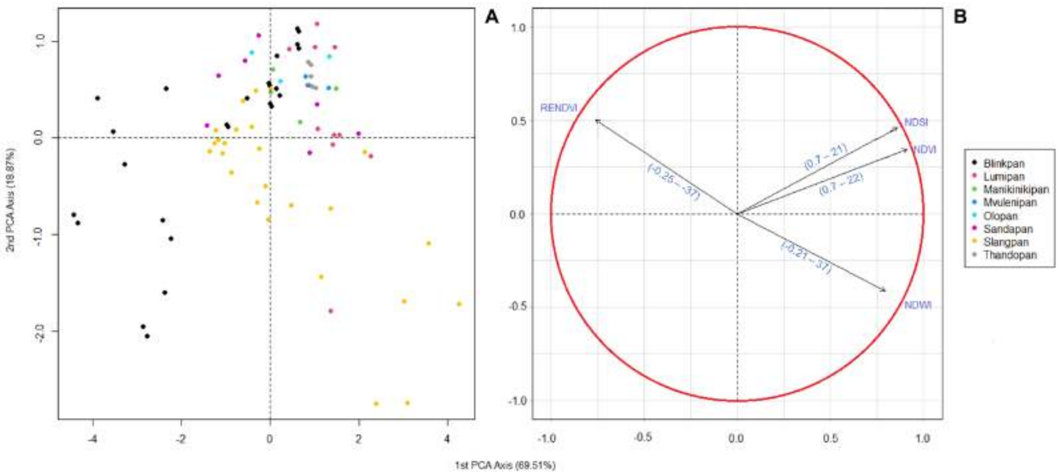
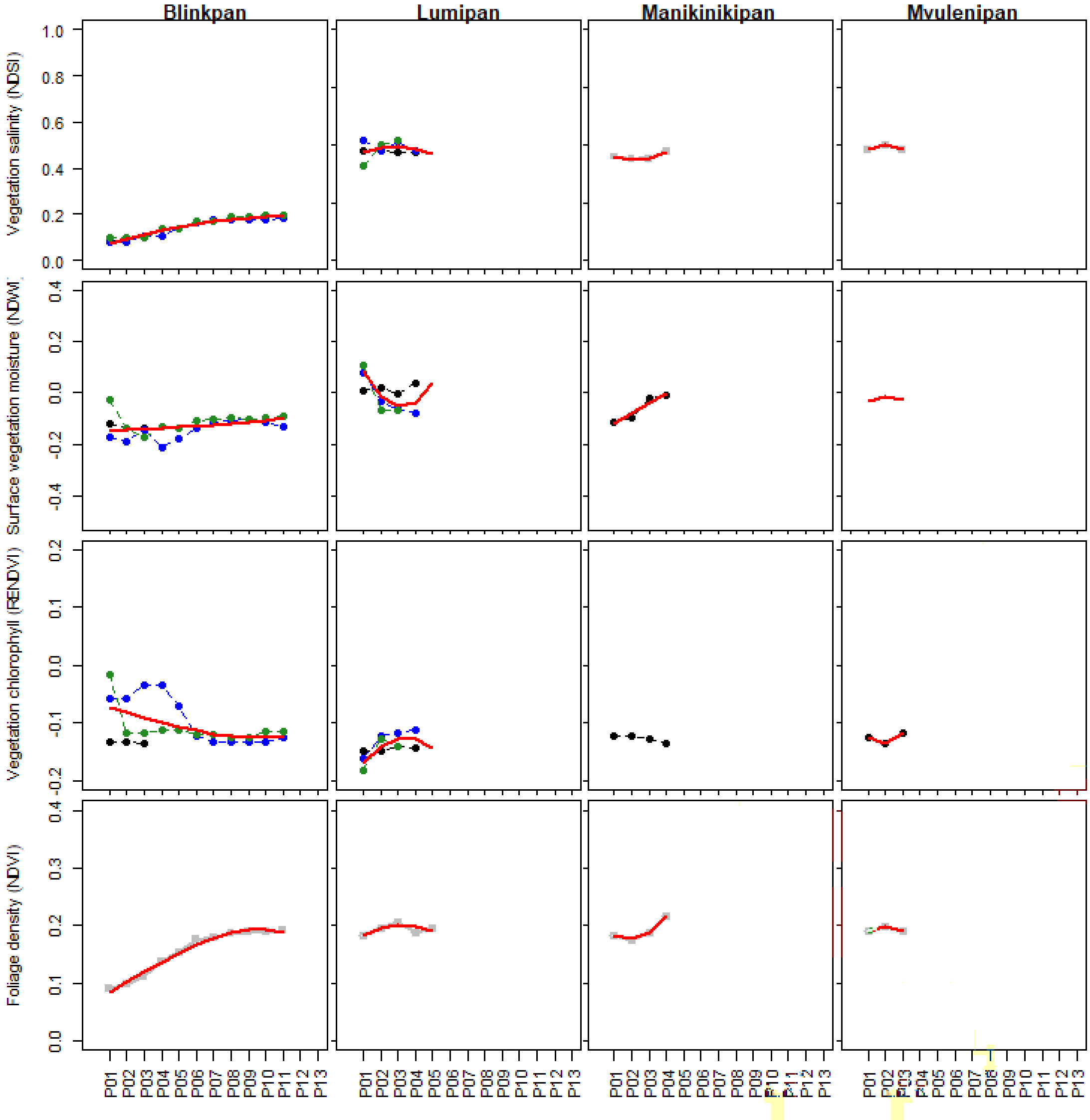


Figure 5.5: PCA ordination diagram of the Lake Banagher Wetland remote sensing vegetation indices dataset with data points representing plots grouped by site (colours). The correlation circle plot (panel B) based on eigenvalues of the correlation matrix for the four active variables. Projection of the variables on a 1 × 4 factor-plane (right). Data point in each on the four regions of the ordination space are influenced be the variable that is correlated with the specific region of the ordination space.

5.4.1 Trends of edaphic factors along the wetland gradient

Generally, there were negative trends in the response of NDSI (r^2=0.96-0.34), NDVI (r^2=0.99-0.20) and NDWI (r^2=0.95-0.20) along the gradient from the centres of the wetlands to the outer dryland boundary, while RENDVI had positive trends (r^2=0.95-0.25). However, at relatively short distances, ranging from 30 to 70 m, this probably reflected the extent of the palustrine section of the depressional wetlands (Figure 5.6). When the data from the remote sensing indices were combined across the gradients of all wetlands to drylands, the indices maintained their general trends, NDSI, NDVI and NDWI = negative, and RENDVI = positive (Figure 5.7). The change of the patterns of Sentinel-2A indices, which probably reflects the mean seasonal maximum extent of the wetland, at 70 m on average for combined data with the 8th plot (80 m) showing a change in the direction of the pattern to the opposite direction.



Blinkpan
Lumipan
Manikinikipan
Mvulenipan
Vegetation salinity (NDSI)
Surface vegetation moisture (NDWI)
Vegetation chlorophyll (RENDVI)
Foliage density (NDVI)

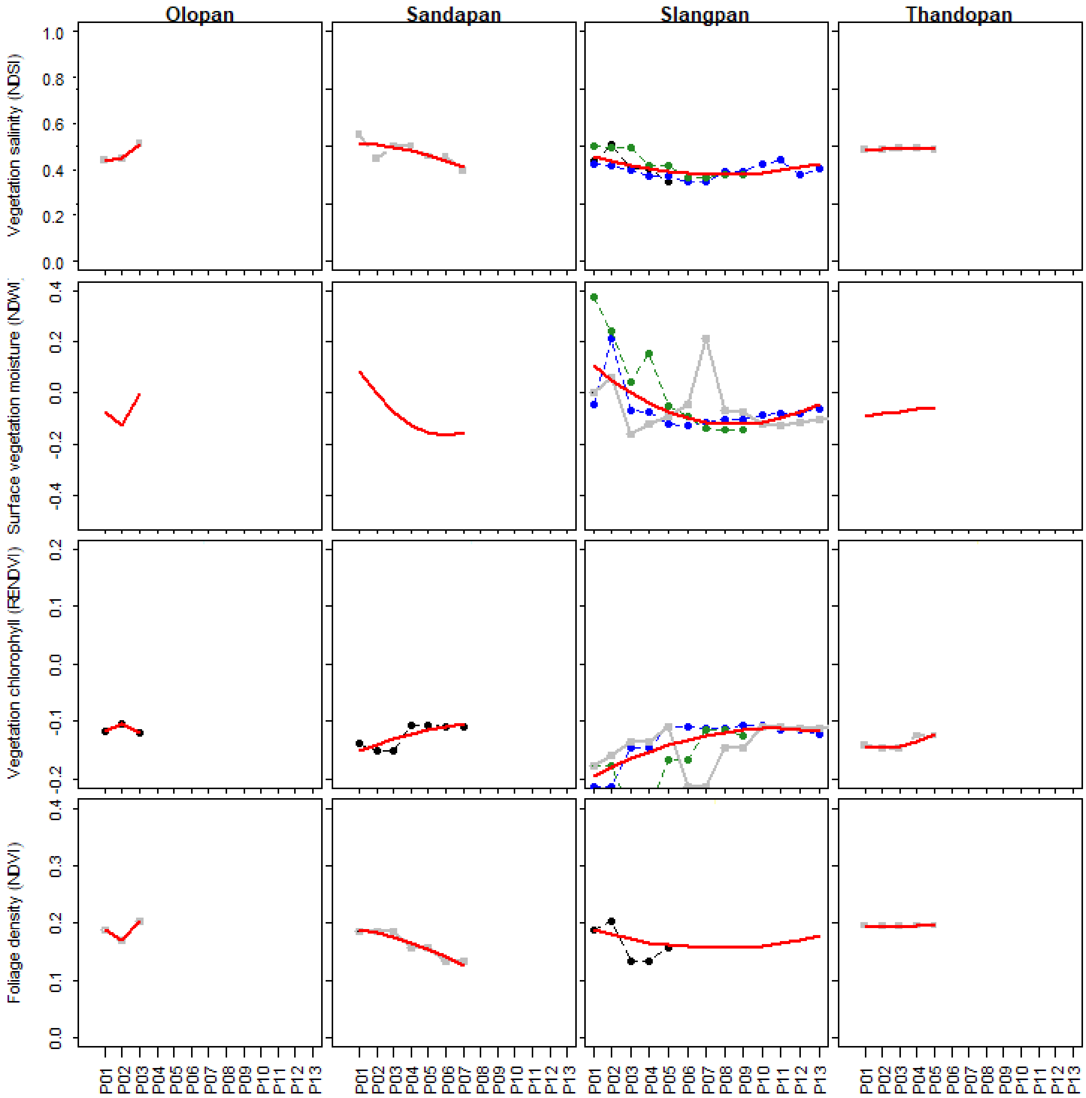


Figure 5.6: Polynomial regression models (red solid lines, mathematical model iterations and statistics in apecdix) of the response of remote sesning vegetation indeces to increasing distance from the edge of the wetland water body.

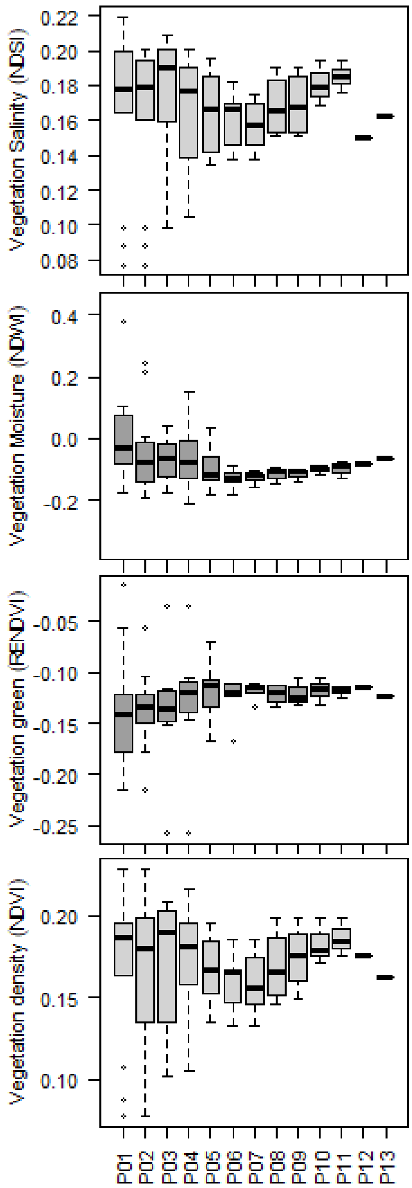


Figure 5.7: Mean values of vegetation spectral indices aggregated by plot number across the literal gradient of a depression and Wetlands; showing adjacent groups of plots being similar in spectral indices and differences in spectral indices between the two sides of the wetland threshold.

5.5 Discussion

5.5.1 Differences in remote sensing vegetation indices across depressional wetland

In this study, we investigated differences in remote sensing indices of vegetation among eight depressional wetlands in a temperate grassland biome. Our results showed that significant site level differences were detected for two of the indices NDVI (1/28) and NDSI (1/28) between wetland pairs ($p<0.001$). Four other wetlands showed significant difference at $p<0.05$. Wilson and Norman (2017) analysed spatial and temporal trends in vegetation greenness and soil moisture but used normalized difference infrared index (NDII) instead of normalized difference water index NDWI and normalized difference vegetation index (NDVI) from Landsat instead of Sentinel-2A. To our knowledge there are no previous studies that have applied these remote sensing indices to the delineation of wetlands or grouping wetland sites wetland site in the MLD (Wilson and Norman 2018). The effect of grazing was also noted by Wilson and Norman (2017) from NDVI performance due it its effects on canopy cover while NDII was better at tracking changes in areas with continued grazing. Which could explain only one out of 28 pairs being significate as all the other camps where the other wetlands are found were grazed. Lumipan was in a rested camp that was going to be grazed following Blinkpan which was being grazed during the time of sampling. These results can be relied upon because these results were produced with data that were collected from cloud free images and the indices were manually calculated using a java script in Google Earth Engine. This means that the image quality, location reference and calculations were based on standard protocol that were collected together with management information and are repeatable. There is a need for further research on the seasonal time-series of vegetation spectral indices in the littoral zone of depressional wetlands (Li *et al.* 2015; Chi *et al.* 2019). We tested similarities in spectral indices across the wetland sites. The results on differences in spectral indices across the eight wetlands showed that although the wetlands differ in characteristics that partly affect or drive the spectral indices, there is still high convergence or grouping in the trends of vegetation spectral indices. Meaning similarity in spectral indices are present within formed groups and these groups are more likely to result in wetland groups with similar functioning. Two wetland were distinct from each other (Blinkpan and Slangpan) while other wetlands were similar.

5.5.2 Patterns of remote sensing vegetation indices along the wetland littoral gradient

The trends in vegetation spectral indices are related to field capacity, and edaphic factors, including the amount of water content held in the soil after excess water has drained away and the rate of downward movement has decreased (Colman 1947, Castelli *et al.*, 2000, Twarakavi *et al.* 2009). Furthermore, NDVI can be strongly influenced by changes in surface and groundwater availability (Aguilar *et al.* 2012; Fu and Burgher 2015; Sims and Colloff 2012). The available water capacity, is as well important in explaining these patterns, and it refers to the ability of soil to hold water from infiltrating to the lower levels of the soil profile but yet making it available to plants (Cassel and Nielsen 1986). It is the water held between field capacity and the wilting point.

$$\left(\frac{\left(PlotX_1 of i^{th} transect \in j^{th} site\right)}{\left(PlotX_2 of i^{th} transect \in j^{th} site\right)}\right) \times \left(\frac{100}{1}\right)$$

5.5.3 The wetland-dryland threshold boundary for delineating endorheic wetlands

In this research we tested whether remote sensing vegetation indices can be used to delineate the boundary of endorheic wetlands by thresholding these edaphic factors; similarly, to studies in the PPR, situated in the temperate grasslands of the US (Wu and Lane, 2016). However, the delineation of thresholds of endorheic wetlands from Wu and Lane (2016) are based on micro elevation that is determined using Light Detection and Ranging and do not specify the distance from the wetland water body. In this study, the empirically derived threshold of the maximum extent of individual wetlands ranged between 30 m and 70 m. However the aggregate threshold for all eight depressional wetlands, based on the three median vegetation remote sensing indices, was 70 m, hence, we recommend the use of a maximum buffer of a 100m, in order to add a precautionary vegetation buffer of 30 m to accommodate the ferralitic zone of subsurface incoming seepage. The buffer width should be based on site specific recommendation using the percentage change threshold, hence the 100 m is a policy recommendation, not a scientific result. Ma (2016) suggests a minimum buffer with of 20 m (Semlitsch and Bodie 2013). Wetland buffering is important for wetland management and water protection, flooding control, groundwater storage, habitat for wild species, recreation, aesthetic and removal of sediment and pollutants (Castelle *et al.*, 1992; Correll 1996; Wenger and Fowler 2000; Gleason *et al.*, 2003). In theory, for generalisation of a percentage change threshold can be used in the place of a distance measure. This theoretical approach allows the results of our study to be applied to other wetlands globally and can therefore be theoretically represented (equation 6 simplified as equation 7) for determining the wetland threshold using empirical measurements of edaphic factors (Nondlazi et al in review). Therefore, this result is crucial for the South African policy framework and environmental impact assessments (Macfarlane *et al.*, 2015).

5.6 Conclusion

This study showed that depressional wetlands that occur in the temperate grassland biome as represented by a sample of eight depressions in the MLD ecosystem are significantly different in vegetation remote sensing indices and might be related to the same indices from field measurements, at plot pixel level but this remains to be tested through regression. However, similarities are present among some of the wetlands depressions are related to the size on the littoral zones. This study also revealed consistent horizontal trends in the vegetation remote sensing indices from the open water to the outer dryland, characterised by a declining trends for NDSI , NDVI, NDWI and increasing trends for RENDVI. This study demonstrated that for depression wetlands within the MLD the wetland threshold (threshold between dryland and wetland) can be empirically detected at a relatively short distances of about 30 to 70 metres; a threshold where the trends of vegetation remote sensing indices change to opposite directions with a percentage change that is greater than 5%. This threshold can potentially inform the delineation of the outer edge of endorheic wetlands, which are poorly mapped globally for wetlands that are under threat.

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Chapter 6

Synthesis: Determining the wetland boundary using ecological remote sensing under climate change

6.1 Introduction

Wetlands are critically important; they are rapidly declining in extent and biodiversity integrity (Van den Broeck *et al.* 2015; Chatanga *et al.* 2020). Yet, they provide many important wetland ecosystems services including water in scares countries, like South Africa. Therefore, we cannot overstate the need for wetland monitoring and conservation. The wetland water body is an essential wetland feature (Slagter *et al.* 2020). However, other wetlands are without a water body but still provide other ecosystem services, including a refuge for endangered biodiversity. Instead, they have high water saturation in the soil and are called palustrine wetlands (Stolt *et al.* 2001). The palustrine can also refer to the water-saturated soil between upland and lacustrine wetlands. Buffer zones may include this palustrine region because they serve as a water body buffer (Basnyat *et al.* 2000).

Nevertheless, a buffer zone should be a dry upland region that protects this palustrine region and the water body (Castelle *et al.* 1994; Macfarlane *et al.* 2009; Dini and Everard 2018). This manuscript focuses on only the palustrine region of wetlands since this is the current topic for wetland delineation. This thesis supports the advancement of the policy on wetland buffering. It presents alternative approaches for detecting the position and determining the width of the wetlands' palustrine region and outer boundary to base wetland buffering policies that protect wetlands. The manuscript focuses only on the subgrouping of depressional wetlands as a HydroGeoMorphic group and the outer boundary of the palustrine region of wetlands. Therefore, this manuscript contributes to developing complete and accurate inventories by proposing new approaches to subtyping wetlands beyond the HydroGeoMorphic classification and delineating the outer boundary of the palustrine region objectively and remotely. The manuscript applied soil chemistry substrate type, botanical functional traits, and remote sensing of plants and soil to achieve the objectives of delineation and classification.

HydroGeomorphic approaches for monitoring wetlands have long been insufficient, becoming increasingly apparent as climate change intensifies (Gwin *et al.* 1999). Therefore, scientists have proposed various ways of advancing wetland classification beyond the HydroGeoMorphic approach to subtypes (Stander and Ehrenfeld 2009). Subtyping HydroGeoMorphic units is often not possible in practice due to limitations in delineation and classification (Ollis *et al.* 2015). There was a need to develop new approaches to subtype wetlands to monitor, conserve, and protect them under a changing climate. To this end, scientists and policy practitioners have long recognised remote sensing as a tool that fits this purpose.

However, a fundamental limitation has been the coarseness of spatial resolution offered by freely available datasets (Gxokwe *et al.* 2020). Recent improvements in the availability of free data sets to incorporate improved spatial and spectral resolutions have reignited the quest for the subtyping of wetlands (Ramoino *et al.* 2017). This manuscript contributes to this objective of subtyping wetlands (Sieben *et al.* 2018). It uses the depressional group of the HydroGeoMorphicunits as an example due to its diversity and importance for water provision in arid and dryland areas. Therefore this manuscript had set an objective of using soil moisture, soil chemistry, substrate type, vegetation functional traits, hyperspectral remote sensing indices and finally Sentinel-2 remote sensing indices to delineate and group Hydrogeomorphic subtypes. The basis for determining the decision rule of significant differences between depressional wetlands was a Bonferroni adjusted p-value of 0.001. In addition to soil moisture and soil chemistry, the comparison was including substrate type, vegetation functional traits, hyperspectral remote sensing, and Sentinel-2 data. This paper provides an overview of how ecological remote sensing can be assessed using multivariate analysis and multidimensional scaling (Kenkel and Orloci 1986; Dixon 2003). There is an attempt here to define and subtype depression wetlands and potentially other hydrogeological wetland types. The data to conduct these subtypes is collected on the vegetated edges of wetlands and used to characterise the wetlands. Hence the ability to delineate wetlands accurately becomes pivotal to these new capabilities.

In the introduction of this manuscript, the literature review demonstrates how current approaches to wetland delineation are biased to soil characteristics and against vegetation (Vepraskas and Faulkner 2000). Even with soil characteristics, other characteristics such as salt chemistry and soil type are effectively not considered appropriately, if at all. This inappropriate consideration is probably also influenced by the high emphasis on soil hydrology and even using wetland vegetation cover only as a confirmation factor (Thompson *et al.* 2002). Furthermore, this approach is field-based, highly subjective, and has room for political influence on the width of wetland boundaries and buffer zones (Lynagh and Urich 2002). The field approaches used to validate desktop heads-up digitisation of wetland boundaries still produce inaccurate boundary lines up to 40 metres from the actual wetland boundary (Appendix O). Due to the continued decline in wetland ecosystems, scientists have long known that buffering approaches are not enough to protect the wetland ecosystems. More accurate objective and seamlessly implementable approaches over national scales are required. In particular, there is a knowledge gap in developing empirical evidence supporting a wetland buffering exercise independent of wetland ecologists performing the delineation process. It is not uncommon for experienced wetland ecologists to disagree on the actual wetland boundary, even when together at the same site. In addition to determining the actual wetland boundary along the littoral gradient of depressional wetlands, this paper would provide objective, non-observer dependent and empirical evidence of how to do so.

Furthermore, the manuscript seeks to describe the pattern of environmental variables of the wetland littoral zone. Describing these patterns would be necessary as compounding evidence to any proposed wetland boundary positioning and providing climate change impacts. In this regard, this manuscript presents the gradient analysis approach applied with the beltline transect method to detect the actual wetland boundary to the accuracy of 10 m. In the Mpumalanga Lake District in South Africa, the palustrine region of depressions is roughly 80 metres in width. There is a narrow band of saturated soil between the edge of the water bodies and the uplands. The narrowband can be better sampled by transects for a smaller spatial scale, *e.g.* worldview's pixels, albeit the cost. Among freely available remotely sensed data, the spatial scale of 10 m is the smallest. These specific remotely sensed data is essential for monitoring wetlands at the national level on a seasonal and annual basis. Additionally, this manuscript presents evidence of the declining pattern of soil moisture. A formula for the rate of decline when developed as a basis for machine learning classifiers such as random forests.

Furthermore, this manuscript presents the first evidence that uses remote sensing indices to detect the wetland boundary using the ecological approach off gradient analysis with belt transects. This manuscript's application of hyperspectral remote sensing indices is an alternative to the costly laboratory procedures to collect empirical data on soil chemistry and substrate type. These findings indicate that we may identify wetlands at the national level using rapid, accurate, and empirically sound methods and remote sensing approaches. Therefore, the implications of the results from this manuscript promise rapid improvements in our ability to protect and conserve wetland ecosystems and their critical ecosystem services. Further research is required to clarify number one the utilities of these methodologies at hi special resolution such as those provided by worldview for, number two On determining the drivers of the week of the wetland littoral zone along the entire Periphery of individual wetland units. In the following paragraphs, the manuscript discusses these results and the clear implications for research and Society.

6.2 Major findings

Understanding the spatial variability of edaphic factors, vegetation functional traits, remote sensing soil indices, and vegetation indices can help manage and monitor depressional wetlands in the era of climate change.

6.2.1 Edaphic factors, aggregate plant functional traits, hyperspectral soil indices and Sentinel-2A vegetation indices can group wetlands

The significant differences between the eight wetlands for Soil Moisture Content and Soil Bulk Density represent the classification of depression wetlands relevant for monitoring under climate change. The significant differences between the eight wetlands for Functional vegetation Traits based on vegetation structure represent a classification of depression wetlands relevant for monitoring changes in wetland functional structure under climate change. The significant differences between the eight wetlands for Soil Composition Index (SCI) and Normalised Difference Water Index (NDWI) represent a new approach to classifying wetlands. This classification approach is relevant for monitoring the effects of climate change on wetland soils. The significant differences between the eight wetlands for S-2A vegetation indices represents a new approach to classifying wetlands using remote sensing.

6.2.2 Edaphic factors, aggregate plant functional traits, hyperspectral soil indices and Sentinel-2A vegetation indices can detect the outer boundary of wetlands

Soil Moisture Content generally showed a negative trend. Soil Bulk Density had a negative trend of variance but a positive trend along the littoral gradients. Therefore, the wetland threshold is at the highest median bulk density and lowest median soil moisture. The trends occurred over short distances, ranging from 30 to 70 m, reflecting the extent of the wetlands, and the changes in these inflexion points can help monitor the wetland boundary under climate change. Vegetation Moisture Content and Above Ground Biomass generally showed inflexion points along the littoral gradients, probably representing the wetland boundary. The point was over short distances of 90 m, probably reflecting the extent of the wetlands. Changes in the positioning of the inflexion point over time could monitor changes in the vegetation structure of the wetland boundary under climate change. NDWI and SCI generally showed negative trends along the littoral gradients, suggesting that the wetland threshold is the lowest median NDWI and SCI. However, for Misra and NDSI, the changes in the wetland boundary would be monitored through changes in the position of the highest median NDSI.

6.3 Discussion

6.3.1 Remotely sensed data can estimate the width and position of the wetland threshold

It is challenging to outline the outer edges of the saturated part of depressional wetlands because it has an edge that smoothly and sharply curves in and out. What makes it specifically difficult are the dissimilarities in soil and vegetation characteristics between the areas curving inwards and those curving outwards along the whole circumference of wetlands? Difficulty ensues when aiming to digitise the edges accurately on a desktop. The same applies when using a soil auger survey. This manuscript suggests that soil proxies result in a narrower threshold than when using vegetation proxies. This result suggests that using a soil auger or desktop digitising to approximate the wetland threshold will probably lead to intrusion into the ecotone. This intrusion is why detecting the outer edge of the wetland-dryland ecotone using more objective, accurate and autonomous methods is desirable. Autonomous methods for detecting and delineating the wetland boundary would be more reliable than current methods. They change in response to changes in environmental conditions and do not rely on the researcher's experience. It is detecting the point where soil moisture from the wetland water body can no longer be detected. Another way is to detect where wetland vegetation can no longer grow due to unfavourable edaphic conditions. In this research, we took it a step further. We realise that even within vegetation species that grow on grassland, some prefer growing closer to the edge of the ecotone.

Therefore, instead of ending at plant-level functional types (sedge, grass, and herb), we also considered leaf-level functional traits applicable across species. These three autonomous approaches for detecting the wetland boundary demonstrate high utility by precisely detecting the point where the ecotone's outer edge; detecting within a range of 30-100 meters (Fig. 6.2). This result suggests that the true ecotone is generally 80 m wide. The sampled wetlands were representative of the wetlands in the MLD. Van Deventer (2021) and Nondlazi *et al.* (2021) show using NDVI and heads-up digitising that more than half (53.2%) of the wetlands in the MLD are depressions. Blinkpan is about 1.5 km^2 in area and the top 15 most enormous depressions in the about 83 979 depressions in the MLD. Depressions are the most abundant type of wetlands in South Africa.

A total of five of the seven soil-based proxies used to detect the wetland boundary concurred on the outer edge of the ecotone being at 60 to 80 m for these eight depression wetlands (Fig. 6.2). It is noteworthy that two of these proxies were from field data measurements of soil augmented with laboratory techniques, while three were from hyperspectral data measurements of soil. This coincidence means both field measurements and remote sensing measurements corroborated this threshold. Furthermore, the NDWI derived from Sentinel-2A also supported the 70-80 m threshold. This support makes sense since both soil moisture content measured from field samples and NDW indicate that the point where vegetation soil moisture had an inflexion is at 60-80 meters. It is unlikely that the coincidence between Sentinel-2 normalised difference water index and soil moisture is due to the vegetation moisture content (Fig. 6.2). This unlikely event would imply that vegetation that grows closer to the wetland has more vegetation moisture content.

Nevertheless, the pattern of vegetation moisture content along the wetland littoral gradient is known. This pattern suggests that vegetation that grows further away from the wetland keeps more moisture because moisture is less abundant. In comparison, vegetation that grows closer to the wetland keeps less moisture because moisture is readily available. Hence, the most logical conclusion to this observation of coincidence between Sentinel-2 NDWI and soil moisture would be that S-2A NDWI is biased towards soil moisture than vegetation moisture. However, this does not imply that vegetation is not part of the signal. Instead, considering the entire surface of the littoral zone, the signal follows the same pattern as that of field soil moisture.

Detecting the wetland threshold using normalised different salinity index and normalised difference vegetation index from Sentinel-2A yielded a 70-80 m threshold. Both bulk density and hyperspectral soil composition index suggest a 60-80 m threshold. A closer look at the relationship between bulk density and vegetation variables revealed that bulk density was a strong driver of species diversity than either soil moisture or soil salinity. Therefore, the subtext here is that the normalised difference vegetation index is more biased towards changes in species diversity along the wetland littoral zone. Normalised Difference Salinity Index (NDSI) impacts species diversity through direct interactions with bulk density and direct effects. The traditional laboratory techniques are time inefficient and cost-prohibitive for monitoring these edaphic factors nationwide. Therefore, it should be possible to substitute or supplement these traditional laboratory techniques with more cost-effective remote sensing alternatives like SCI and Normalised Difference Salinity Index (NDSI). The wetland threshold from edaphic soil factors (Figure 6.1 and Table 6.1) is closer (6-7 m) to a wetland water body than the in situ expression from vegetation variables (70-80 m). Detecting the wetland threshold (Figure 6.1) using vegetation characteristics yielded the widest threshold (90-100 m). All vegetation characteristics used were plot-level plant functional traits, i.e., above-ground biomass, vegetation moisture content, leaf area index, plant species richness, vegetation leaf angle, and clumping in 10 X 10 m plots. These in situ functional trait data have no interference from soil background, as is the case with Sentinel-2A vegetation proxies. It is, therefore, clear that plant functional traits provide the most conservative autonomous threshold extraction. Future research efforts should focus on using remote sensing to model these functional traits.

Table 6.1: Synthesis table of thresholds among sampled variables from respective biophysical strata and data types.

Biophysical Strata	Data Types in 10 m plot	Variable Abbreviation	60-70 m	70-80 m	90-100 m
Top soil (0.3 m)	Field/Laboratory measurement	SMC	X		
		EC	X		
		BD		X	
	Hyperspectral samples	SCI		X	
		Hyp-NDSI	X		
		SBI	X		
		Hyp-NDWI	X		
Vegetation Canopy	Field plant harvesting	VMC			X
		AGB			X
		PSR			X
	Field / Canopy analyser	LAI			X
		VLA			X
		VLC			X
	Sentinal-2A	NDSI		X	
		NDWI	X		
		RENDVI		X	
		NDVI		X	

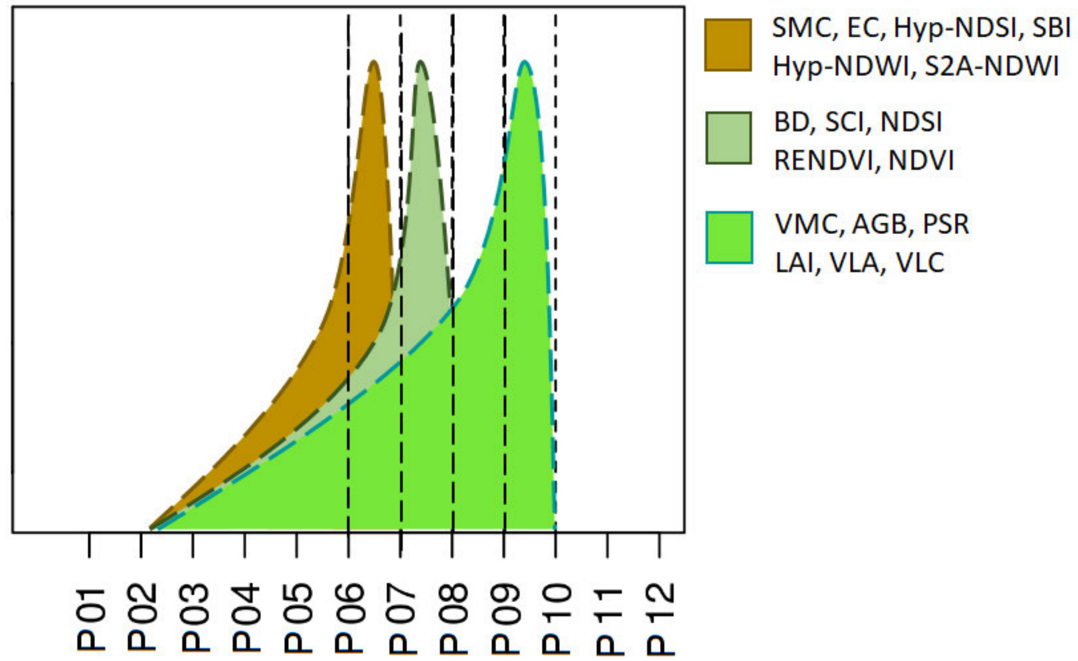

Figure 6.1: Synthesis conceptual of the thresholds along the wetland ecotone exhibiting wetland zonation.

From studying the different variables, we note that the threshold from plant functional traits represents the most conservative distance from the wetland water board (Figure 6.2). It is also noteworthy that prioritising the functional trait threshold means that Wetlands' productivity and structure can be conserved. While prioritising the threshold at 60 - 80 m would only ensure that the adaptor conditions and the species composition a preserved. It is noteworthy that while edaphic factors and species are critical functional traits and show many ecosystem services. Therefore, it will be more advisable to use the functional trait threshold and declare the minimum wetland buffer of 100m.

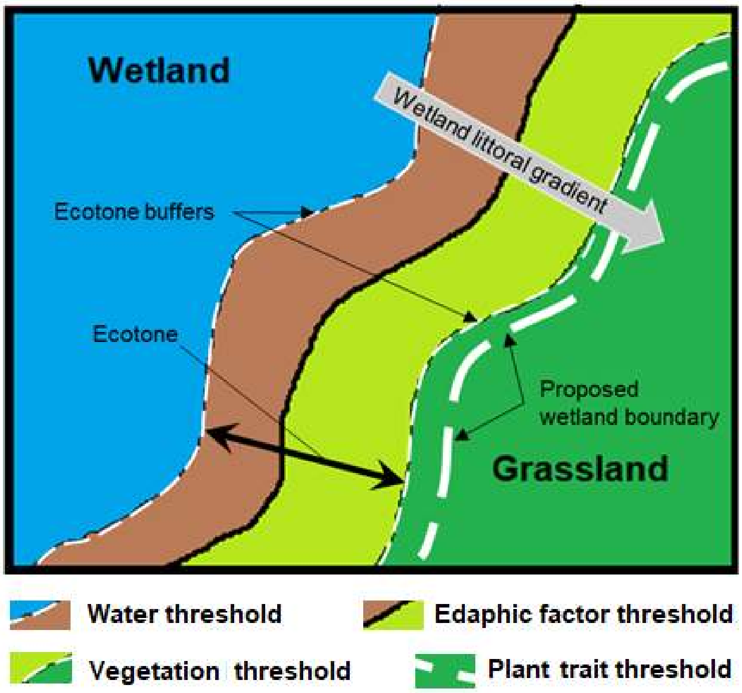

Figure 6.2: Illustration of the wetland ecotone and the application of the boundary thinking.

6.3.2 Remotely sensed data can decern the pattern of edaphic factors and plant functional traits along the littoral gradeint

Successive thresholds of edaphic factors, vegetation and functional traits characterise the wetland ecotone. The data presented in this manuscript show that a unique characteristic of the wetland threshold is the relationship pattern between soil and vegetation. An increase or decrease in the patterns of edaphic factors along the littoral gradient does not necessarily reflect the actual pattern of the variable. The pattern also depends on the spatial nature of the variable and the approach or measurement technique. However, the dominant environmental gradient is the moisture gradient, and its pattern decreases from the water body to the upland. Sentinel-2A measurements of normalised different salinity indexes show a declining trend. When measured with hyperspectral devices, it has an increasing pattern.

In contrast, the hyperspectral measurement was a pure soil signal. This change in these specific conditions tends to continue beyond the outer buffer. However, when wetland meets uplands, the pattern of the wetland and remote sensing variables becomes opposite that seen within the wetland. These emerging characteristic conditions continue for a long distance into the dryland. Beyond the outer buffer, they are characteristically homogeneous. However, for vegetation, changes in plant functional traits continue to characterise the ambient environment. The ambient environment is not dependent on soil alone but other ecological interactions between plant species, flora and fauna, and the physical environment, including microtopography. Most literature alludes to this relationship by explaining the gradation of vegetation along the littoral gradient of the wetland ecotone. The literature further explains this to be due to the variation of environmental conditions along the ecotone. However, it has been unclear as to which environmental variables exactly affect which component of the vegetation. Part of the results in the preceding section provides some clues about these unknown drivers between soil and vegetation along the wetland threshold.

In both the hyperspectral and Sentinel-2A multispectral measurements, the normalised difference water index (NDWI) decreases as you move away from the wetland water body (Figure 6.3). The decreasing pattern in normalised difference water index is consistent with soil moisture content measured using the gravimetric method along the littoral gradient. Of the six variables that showed a threshold at 60-70 m, all supported the hypothesis that the moisture gradient has a decreasing pattern along the littoral zone. The other three variables, i.e. Electric conductivity (EC), normalised difference salinity index (NDSI) and soil brightness index, had an increasing pattern from water body to upland. The results of this study support the hypothesis that NDSI will show the same pattern as electrical conductivity since electric conductivity is a proxy for salinity. Furthermore, soils with salt crust have a bright colour due to the salt crystals on top of the soil. Therefore, the soil brightness index would have the same pattern as soil salinity. Patterns of saltiness that are inversely proportional to soil moisture characterise the wetland littoral gradient. These two characteristics indicate a permanent wetland zone or a wetland ecotone within a buffer zone. They may exhibit the wetland boundary within which obligate wetland species are most abundant.

Only the soil bulk density threshold at 70-80 m and had an increasing pattern similar to the red-edge normalised difference vegetation index. The rest of the variables were remote sensing proxies and had decreasing patterns (Figure 6.3). Two of these variables from Sentinel-2A data, i.e. Normalised difference salinity index (NDSI) and normalised difference vegetation index (NDVI). Others from hyperspectral data, i.e. Soil composition index (SCI). The high NDVI closer to the wetland water board might suggest that the vegetation closest to the internal buffer has a high abundance of mesophyll cells. We know that vegetation that grows on saturated soils benefits from the abundance of mesophyll cells for respiration. We know that the normalised difference vegetation index uses the red light. At the same time, healthy mesophyll cells strongly reflect the near-infrared region. In contrast, high chlorophyll pigment strongly absorbs red light. When NDVI is lower further away from the wetland, the vegetation has low chlorophyll pigment and rate of respiration, thus low cellulose. Higher cellulose aligns to perennial species with a more extended life strategy like grass. In contrast, low cellulose would align with species such as sedges with a short life strategy. The pattern of RENDVI provides further evidence of the wetland ecotone to the characterisation of the pattern of NDVI. Low RENDVI closer to the wetland water body means that the vegetation closer to the wetland water body has low maturity, which aligns with annual vegetation. Furthermore, low red edge NDVI means the vegetation is paler in colour due to low chlorophyll thus low cellulose content. Herbaceous vegetation's Red-Edge NDVI may change depending on the cover as the cover influences the region of the electromagnetic spectrum used for computation of Red-Edge NDVI. As the first threshold of the wetland gradient describes changes in edaphic factors, the second threshold of the gradient could define changes in species composition.

Changes in vegetation functional traits define the last threshold of the wetland ecotone. These vegetation functional traits are related to vegetation structure and vegetation productivity. They all have an increasing pattern along the wetland ecotone (Figure 6.3). However, what is noteworthy is that belief level traits that directly respond to two light and temperature as variables of the ambient environment tend to exhibit both an increase and a decrease. Interrogating the data on these functional traits, it becomes evident that they show a threshold at both the 60-70 and the 90-100m thresholds.

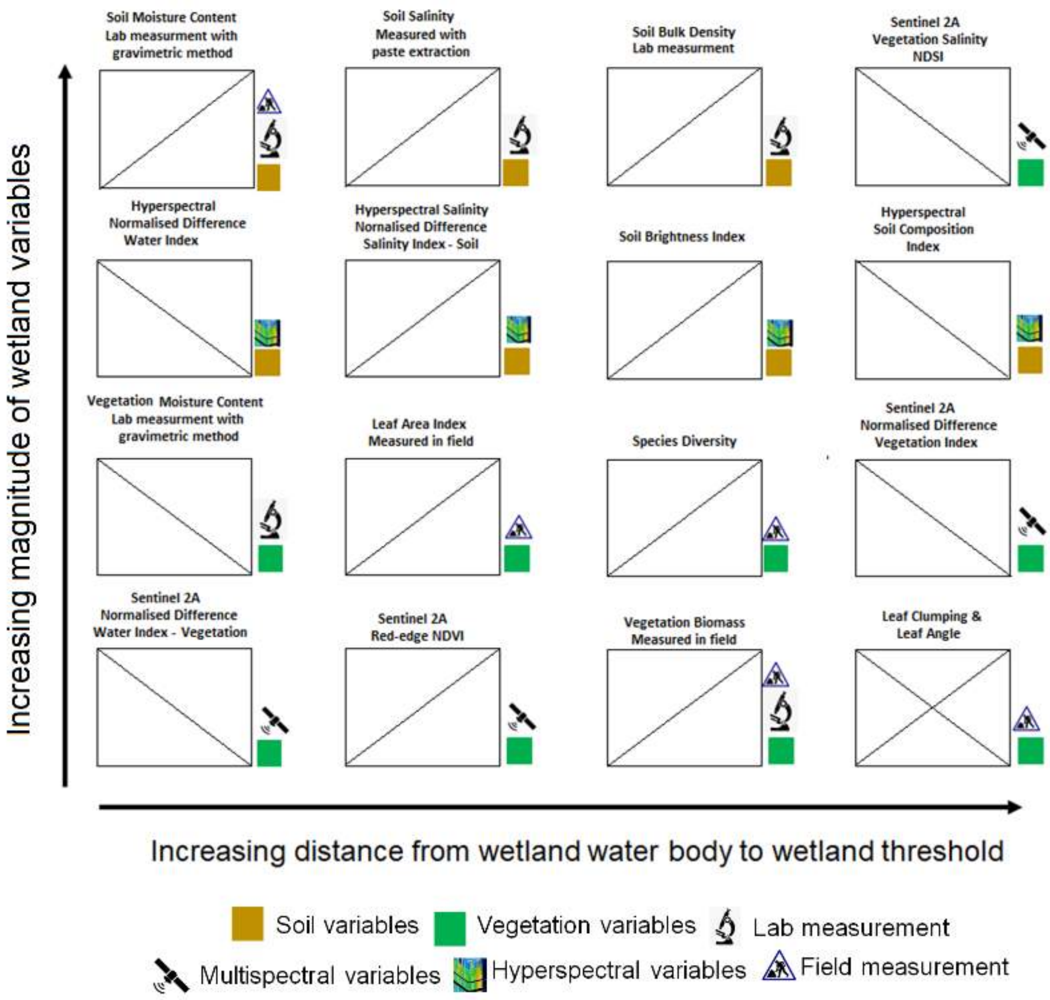

Figure 6.3: Synthesis of the trends along the wetland ecotone exhibiting wetland zonation.

6.3.3 Remotely sensed data can objectively group depressional wetlands

Characterising the differences between wetlands using HydroGeoMorphiccharacteristics has it has been worthwhile. However, we have been unable to apply levels 5 and 6 of our national wetland classification scheme. We need to apply these levels as a wall-to-wall to the same extent that we apply HydroGeoMorphicunit classification. Level 5 of the national wetland classification system requires information on the hydroperiod (Ollis *et al.*, 2013; 2015). NDWI coincides with soil moisture content in detecting the wetland threshold and its distribution pattern along the wetland littoral gradient. The manuscript presents a use case for classifying wetlands using edaphic factors and plant functional traits. Using a normalised difference water index based on the soil moisture content, the classification of the wetlands within Lake Banagher is possible using remote sensing. To categorise wetlands based on remote sensing, we must consider proxies related to surface type, pH, and vegetation salt content.For proxies of change in salt content and vegetation traits, this manuscript also uses the normalised difference salinity index (NDSI), the red-edge normalised difference vegetation index (RENDVI), and the normalised difference vegetation index (NDVI).

The use of procedures and proxies such as Soil Composition Index (SCI) and Soil Brightness Index (SBI) to monitor substrate type is also beneficial. There is a requirement for more work to understand the relationship between vegetation functional traits and available indices and possibly develop new indices for functional traits. Furthermore, a seventh level of the national wetland classification system that continues from vegetation cover types might be necessary to incorporate functional traits systematically in wetland monitoring. The descriptors in level six of the South African Wetland Classification System that focus on vegetation would not capture essential changes in vegetation functional traits. Its usefulness is limited to capturing vegetation cover types, such as the proportion of sedges to grass to forbes. Therefore, descriptors level six wetland classify wetlands into functional types, *e.g.*, grassy wetlands, herb wetlands, and herb or forb wetlands. Hence, a proposal for vegetation descriptors levels 7 of the wetland classification system for vegetation functional traits is here, thus a submission (Figure 6.4).

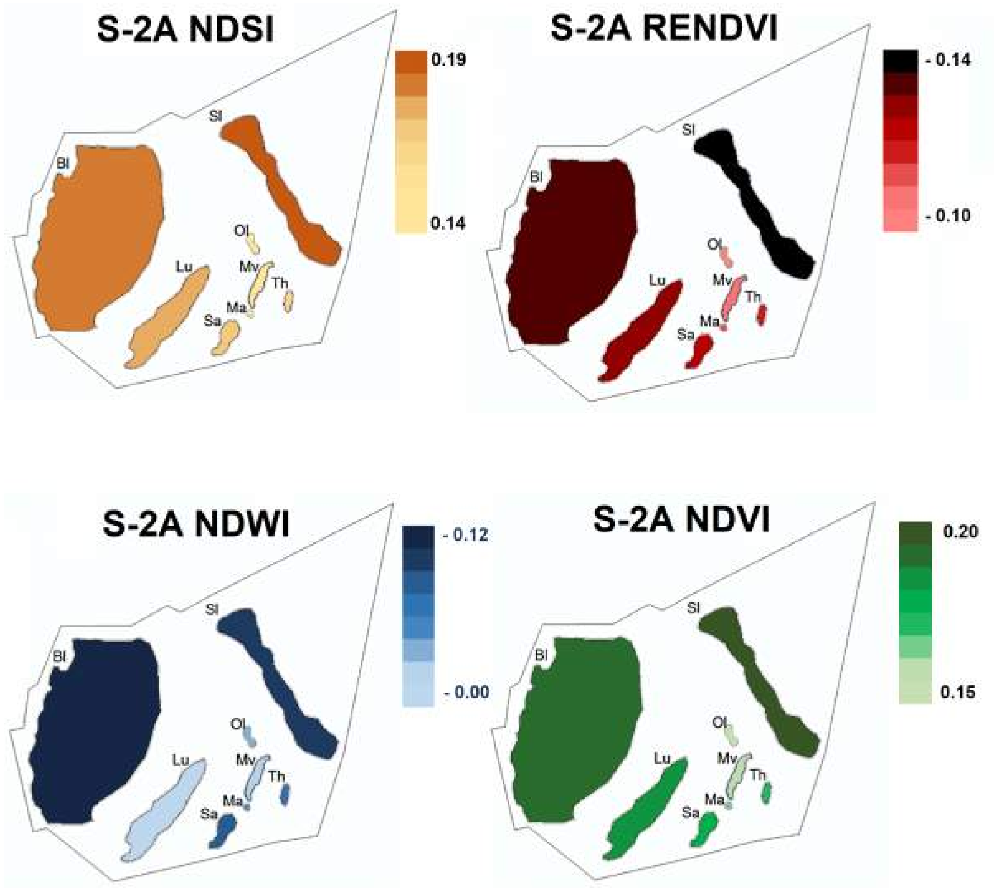

Figure 6.4: Use cases for implementing the descriptors level 5 & 6 classification system of south Africa.

6.3.4 Drivers of wetland threshold

There was a close relationship between drivers of the wetland threshold, *e.g.* soil moisture content, elevation and the width of threshold, and wetland size, bulk density, and wetland depth (Figure 6.5). There might be some other variable that correlates with salinity which drives wetland littoral vegetation.

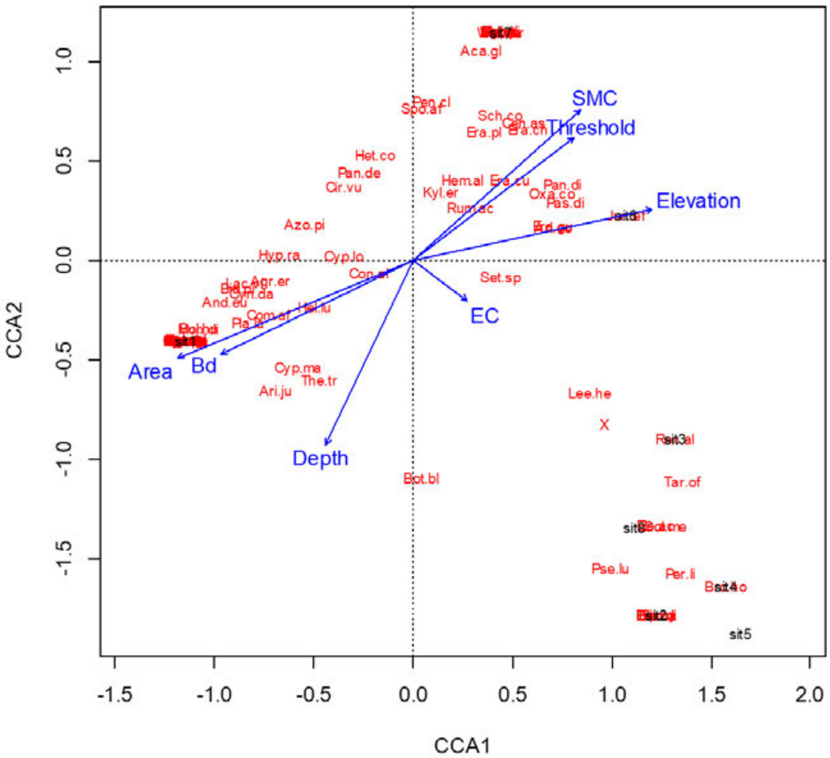

Figure 6.5: Proposed drivers of wetland littoral zone

6.3.5 Limitations of the study

While this study highlighted some key drivers of the wetland littoral zone, there are also gaps and limitations about these drivers. The study's main limitation is that it did not cover long-term seasonal and annual time series to see changes in the drivers of the threshold over time.

6.4 Conclusion

These results have important implications for long-term monitoring of wetlands functioning in response to climate change. For instance, current national wetland monitoring relies on GIS modelling techniques (van Deventer *et al.* 2020). The study's main limitation is that it did not cover long term seasonally and annual time series. While this study highlighted some key drivers of the wetland littoral zone, there will also be gaps and limitations about these drivers. There was a relationship between soil moisture content and the width of the threshold. There might be some other variable that correlates with salinity which drives wetland littoral vegetation. It is also broader than standard guidelines of 30 -32 m specified in the South African National Environmental Management Act (NEMA) of 1998 that was gazetted on 13 April 2017 (South African Environmental Act 107 of 1998), depending on microtopography. To improve wetland protection, the South African Environmental Management Act needs review.

Rather than extrapolating the results of this manuscript to wetlands throughout the country or the world, it provides recommendations based only on the study's outcomes. At the same time, the sampled wetlands cannot represent every difference present among the Wetlands of the MLD. They, however, represents the more minor depressions 1.5 km^2 which are the dominant size. About 77% of the HydroGeomorphic wetlands typed in South Africa are depressions. Based on the context, a 100 m guideline is recommended, with a 30 m extension to the field proof. The 30 m is to accommodate the region of the incoming drainage or the ferritic zone, which is a unique ecosystem on the edge of wetlands zones. Current wetland buffering legislation might be allowing (legally) farming and construction within wetlands in practice while denouncing it in sentiment. Wetland plant functional traits further support the 100 m threshold. These results can be relied upon because these results are from data that were collected over two growing seasons (annual time series), and the results were consistent between the sampling periods. Regarding the effects of seasonality (seasonal time series) and precisely the effect of season on the trends patterns of the variables;

The methods do not use magnitudes to detect the threshold but rather the empirical pattern of variables and their inflexion point. This non-reliance of the magnitude means that insignificant statistical differences in the values of edaphic factors among the wetlands showed no effect of sampling time on the position of the inflexion point. s There is a need for further research on the seasonal time-series of edaphic factors in the littoral zone of depressional wetlands (Niemuth *et al.*, 2010). We used principal component analyses to assess the statistical differences in edaphic factors across different scales, site and plot levels. In ordination, the orthogonality of latent variables achieves multiple direct comparisons of similarities between edaphic factors across all plots and sites simultaneously. Research in other environments is needed to verify that the wetland boundary derived from vegetation functional traits represents the wetland's outermost edge. There is a need for further research into the implications of the soil threshold being shorter than the vegetation threshold and the functional trait threshold being the widest. There is a need for further research on the inner edge of the wetland ecotone that is as detailed as the current research. There is a need for further research on the available width of the wetland threshold in other environments. There is a need for further research on changes in the magnitude of the time series of the differences in soil and vegetation variables. The hypothesis that shorter boundaries have steep gradients needs further testing. Their practices that the methodology used in this research is adequate for investigating the inner edge of the wetland ecotone needs to be tested.

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Chapter 7

Appendices

Appendix A

Map of study site and its location

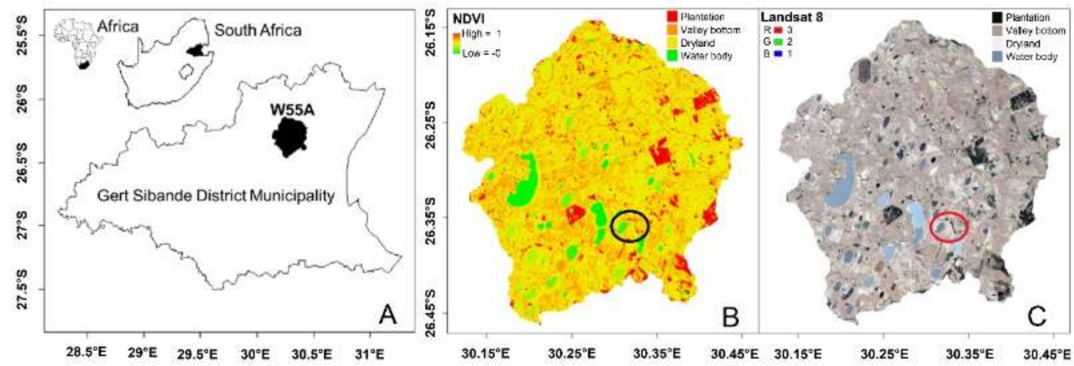

Figure A.1: Spatial variations NDVI (A) and Landsat 8, 2017 actual colour composite image (B), in the Quaternary Catchment W55A and the Gert Sibande District of the Mpumalanga Province, South Africa. In Lake Banagher farm (red and black oval shape (B&C)), wetland ecosystem types (valley-bottom wetlands in orange and depressional wetlands in light green (B)) are observable within the quaternary catchment W55A.

Appendix B

Center coordinates of 10 m sampling plots

Site Name	Site Code	Transect	Plot	lat/X	long/Y
Blinkpan	BLK	T1	P01	30.337750	-26.33400
Blinkpan	BLK	T1	P02	30.337850	-26.33400
Blinkpan	BLK	T1	P03	30.337940	-26.33400
Blinkpan	BLK	T2	P01	30.330120	-26.33780
Blinkpan	BLK	T2	P02	30.330020	-26.33780
Blinkpan	BLK	T2	P03	30.329930	-26.33780
Blinkpan	BLK	T2	P04	30.329850	-26.33780
Blinkpan	BLK	T2	P05	30.329710	-26.33770
Blinkpan	BLK	T2	P06	30.329650	-26.33770
Blinkpan	BLK	T2	P07	30.329530	-26.33770
Blinkpan	BLK	T2	P08	30.329460	-26.33770
Blinkpan	BLK	T2	P09	30.329350	-26.33770
Blinkpan	BLK	T2	P10	30.329240	-26.33770
Blinkpan	BLK	T2	P11	30.329130	-26.33770
Blinkpan	BLK	T3	P01	30.330990	-26.34760
Blinkpan	BLK	T3	P02	30.330990	-26.34770
Blinkpan	BLK	T3	P03	30.330980	-26.34780
Blinkpan	BLK	T3	P04	30.330990	-26.34790
Blinkpan	BLK	T3	P05	30.330980	-26.34800
Blinkpan	BLK	T3	P06	30.331000	-26.34810
Blinkpan	BLK	T3	P07	30.330980	-26.34810
Blinkpan	BLK	T3	P08	30.330980	-26.34820
Blinkpan	BLK	T3	P09	30.330990	-26.34830
Blinkpan	BLK	T3	P10	30.330980	-26.34840
Blinkpan	BLK	T3	P11	30.330970	-26.34850
Lumipan	LUM	T1	P02	30.342820	-26.34300
Lumipan	LUM	T1	P03	30.342910	-26.34290
Lumipan	LUM	T1	P04	30.343030	-26.34280
Lumipan	LUM	T1	P05	30.343130	-26.34270
Lumipan	LUM	T2	P01	30.340860	-26.34510
Lumipan	LUM	T2	P02	30.340760	-26.34510
Lumipan	LUM	T2	P03	30.340560	-26.34500
Lumipan	LUM	T2	P04	30.340290	-26.34450
Lumipan	LUM	T3	P01	30.338920	-26.34810
Lumipan	LUM	T3	P02	30.339210	-26.34810
Lumipan	LUM	T3	P03	30.339610	-26.34870
Manikinikipan	MAN	T0	P01	30.346200	-26.34660
Manikinikipan	MAN	T0	P02	30.346360	-26.34620
Manikinikipan	MAN	T0	P03	30.346450	-26.34630
Manikinikipan	MAN	T0	P04	30.346660	-26.34620
Mvulenipan	MVU	T0	P01	30.347940	-26.34240
Mvulenipan	MVU	T0	P02	30.347830	-26.34240
Mvulenipan	MVU	T0	P03	30.347960	-26.34220
Olopan	OLO	T1	P01	30.346980	-26.34120
Olopan	OLO	T1	P02	30.346560	-26.34100
Olopan	OLO	T1	P03	30.346450	-26.34140

Site Name	Site Code	Transect	Plot	lat/X	long/Y
Sandapan	SAN	T0	P01	30.345720	-26.34760
Sandapan	SAN	T0	P02	30.345820	-26.34760
Sandapan	SAN	T0	P03	30.345920	-26.34760
Sandapan	SAN	T0	P04	30.346030	-26.34760
Sandapan	SAN	T0	P05	30.346120	-26.34760
Sandapan	SAN	T0	P06	30.346220	-26.34760
Sandapan	SAN	T0	P07	30.346330	-26.34760
Slangpan	SLA	T1	P01	30.344970	-26.33330
Slangpan	LUM	T1	P01	30.342710	-26.34300
Slangpan	SLA	T1	P02	30.344840	-26.33330
Slangpan	SLA	T1	P03	30.344740	-26.33340
Slangpan	SLA	T1	P04	30.344640	-26.33350
Slangpan	SLA	T2	P01	30.349930	-26.33940
Slangpan	SLA	T2	P02	30.349820	-26.33950
Slangpan	SLA	T2	P03	30.349730	-26.33960
Slangpan	SLA	T2	P04	30.349610	-26.33970
Slangpan	SLA	T2	P05	30.349510	-26.33980
Slangpan	SLA	T2	P06	30.349410	-26.33980
Slangpan	SLA	T2	P07	30.349310	-26.33990
Slangpan	SLA	T2	P08	30.349210	-26.34000
Slangpan	SLA	T2	P09	30.349090	-26.34010
Slangpan	SLA	T2	P10	30.348990	-26.34020
Slangpan	SLA	T2	P11	30.348890	-26.34030
Slangpan	SLA	T2	P12	30.348800	-26.34040
Slangpan	SLA	T2	P13	30.348700	-26.34040
Slangpan	SLA	T3	P01	30.353380	-26.34200
Slangpan	SLA	T3	P02	30.353460	-26.34210
Slangpan	SLA	T3	P03	30.353550	-26.34220
Slangpan	SLA	T3	P04	30.353660	-26.34230
Slangpan	SLA	T3	P05	30.353760	-26.34230
Slangpan	SLA	T3	P06	30.353850	-26.34240
Slangpan	SLA	T3	P07	30.353960	-26.34250
Slangpan	SLA	T3	P08	30.354050	-26.34260
Slangpan	SLA	T3	P09	30.354140	-26.34270
Thandopan	THA	T0	P01	30.350290	-26.34500
Thandopan	THA	T0	P02	30.350290	-26.34490
Thandopan	THA	T0	P03	30.350290	-26.34480
Thandopan	THA	T0	P04	30.350290	-26.34470
Thandopan	THA	T0	P05	30.350300	-26.34460

Appendix C

Depressional wetlands sampled

Site Name *Coordinates*	Visual	Shape	Code	Area (km^2)	Width (m)	Length (m)	Endorheic type	Samples (*n*)
				Threshold (m)	*Perimeter (m)*	*Depth (m)*	*Elevation (m)*	
Blinkpan ***E 30.333639°*** ***S -26.340425°***			Bl	**1.299**	**583-836**	**1692**	**Lacustrine**	51
				30	*4687*	*~ 4*	*1660-1664*	
Slangpan ***E 30.349214°*** ***S -26.336879°***			Sl	**0.310**	**73-304**	**1438**	**Lacustrine**	37
				70	*3554*	*~ 10*	*1677-1667*	
Lumipan ***E 30.339689°*** ***S -26.346721°***			Lu	**0.211**	**47-128**	**1038**	**Lacustrine**	12
				40	*2462*	*~ 3*	*1668-1665*	
Mvulenipan* ***E 30.347461°*** ***S -26.343843°***			Mv	**0.036**	**54-82**	**406**	**Palustrine**	08
				30	*992*	*~ 2*	*1671-1669*	
Sandapan ***E 30.344849°*** ***S -26.347959°***			Sa	**0.040**	**50-136**	**349**	**Lacustrine**	08
				70	*887*	*~ 4*	*1673-1669*	
Olopan ***E 30.346827°*** ***S -26.340838°***			Ol	**0.011**	**59-76**	**193**	**Palustrine**	21
				30	*476*	*<0.5*	*1670-1670*	
Thandopan ***E 30.349984°*** ***S -26.345125°***			Th	**0.013**	**57-62**	**185**	**Lacustrine**	51
				50	*473*	*0.5*	*1671-1671*	
Manikinikipan ***E 30.346764°*** ***S -26.346141°***			Ma	**0.003**	**14-30**	**55**	**Palustrine**	15
				40	*200*	*< 0.5*	*1671-1671*	

Figure C.1: The following wetland features were estimated using Google Earth Pro; area, width, depth, length and elevation.

Appendix D

Illustration of a belt transect plot along a littoral gradient

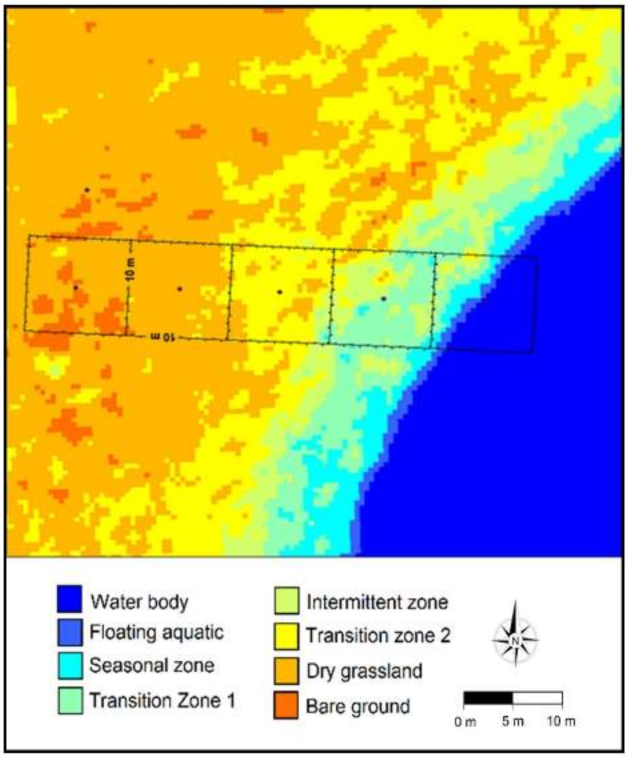

Figure D.1: Selection of the grid cells that sampled the gradient of wetland moisture zones. The transect is not precisely perpendicular to the aquatic zone because the transects were selected from a template created with Sentinel-2A pixels; therefore, the path and orientation could not be changed. The intention was to cover the range of variation in the visible vegetation physiognomy along the banks of wetlands (Fig. 3). The length of the transect was limited by the camp's fence, where the wetland is enclosed within a camp. Maps were developed using ArcGIS (ArcMap 10.5).

Appendix E

An overview of the edaphic factors sampled, their source, the derivation method, and their formula

Variable	Source	Method	Method formula	#Eq.
Position along a transect (m)	Horizontal distance	Measuring tape reading	$Plot\ distance\ i = \sum_{i=10\ m}^{n}$	(1)
Soil Moisture Content (g /g)	Mass lost by an oven dried soil sample	Scale mass reading	$\boldsymbol{Soil\ MC_{\theta Gw}} = \frac{m_{water}}{m_{moist\ soil} - m_{tray}}$	(2)
Soil Salinity (dS/m)	Total dissolved solids in water extracted from a mixed soil-water saturated-paste solution.	Electric conductivity – EC	1:10 liquid extraction method	
Bulk Density (g/m^3)	The ratio of the weight of a sample soil core to the cubic dimensions of a soil core.	Unstructured soil	***Step 1:*** Volume of Soil Core in grams per centimeter cubed (cm^3) $\boldsymbol{V_{soil\ core}}\ cm^3 = \pi \times \frac{1}{2} r^2 \times Ht_{soil\ core}$	(3)
			Step 2: Bulk Density (D_b) $\boldsymbol{SoilD_{b_{g/cm^3}}} = \frac{m_{dry\ soil}}{V_{soil\ core}cm^3}$	(4)

Figure E.1: Showing the summary of the detailed methods for deriving the experimental variables used in the research. EC= Electrical Conductivity, SMC=Soil Moisture Content, BD=Bulk Density

Appendix F

Resulting p-values for pairwise comparisons of edaphic factor between wetlands using Turkey's HSD

Soil Moisture Content *(g/g)*														
	Mean Difference							**P-values**						
	Bl	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**	**Bl**	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**
Lu	96.30							0.000						
Ma	-8.30	-105.00						1.000	0.000					
Mv	87.80	-8.50	96.00					0.006	1.000	0.031				
Ol	25.00	-71.00	33.30	-62.80				0.960	0.067	0.950	0.493			
Sa	39.50	-57.00	47.80	-48.30	14.50			0.212	0.024	0.505	0.615	0.999		
Sl	4.81	-91.00	13.10	-83.00	-20.00	-35.00		0.999	0.000	0.998	0.011	0.988	0.367	
Th	79.50	-17.00	87.70	-8.31	54.50	39.90	74.70	0.001	0.988	0.021	1.000	0.533	0.644	0.002
Soil Electrical Conductivity *(dS/m)*														
	Mean Difference							**P-values**						
	Bl	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**	**Bl**	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**
Lu	0.20							1.000						
Ma	-6.23	-6.43						0.902	0.907					
Mv	7.24	7.05	13.47					0.893	0.920	0.544				
Ol	2.91	2.71	9.14	-4.33				0.999	1.000	0.898	0.999			
Sa	-3.29	-3.48	2.94	-10.53	-6.20			0.988	0.989	1.000	0.715	0.977		
Sl	5.78	5.58	12.01	-1.47	2.87	9.07		0.119	0.358	0.214	1.000	1.000	0.253	
Th	0.15	-0.04	6.38	-7.09	-2.76	3.44	-5.62	1.000	1.000	0.968	0.965	1.000	0.998	0.903
Soil Bulk Density *(g/cm^3)*														
	Mean Difference							**P-values**						
	Bl	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**	**Bl**	**Lu**	**Ma**	**Mv**	**Ol**	**Sa**	**Sl**
Lu	-0.18							0.000						
Ma	-0.08	0.1						0.940	0.867					
Mv	-0.21	-0.03	-0.13					0.133	1.000	0.890				
Ol	0.04	0.22	0.12	0.25				1.000	0.131	0.928	0.269			
Sa	-0.09	0.09	-0.01	0.12	-0.13			0.727	0.763	1.000	0.868	0.843		
Sl	-0.16	0.02	-0.08	0.05	-0.2	-0.07		0.000	1.000	0.938	0.998	0.179	0.877	
Th	-0.18	0	-0.1	0.04	-0.22	-0.09	-0.02	0.098	1.000	0.957	1.000	0.321	0.944	1.000
Lu=Lumipan, Ma=Manikinikipan, Mv=Mvulenipan, Ol=Olopan, Sa=Sandapan, Sl=Slangpan, Th=Thandopan, Bl=Blinkpan														

Figure F.1: Pairwise analysis of differences between wetlands in soil moisture, salinity and bulk density across eight sampled wetland sites (data combined by site) Turkey's HSD . The decision rule was set at Bonferroni adjusted alpha level <0.001. A negative mean difference value shows that for the compared edaphic factor, the site in the column is greater than the site in the row for the given pair of sites in comparisons. In contrast, the opposite is valid for a positive value.

Appendix G

Summary data regarding ensembles of polynomial regression models fitted to edaphic factors for each wetland site.

Site	Res. *Error*	*DF*	M r^2	A r^2	*F* value	Intercept *p*-value	Overall *p*-value	The formula for polynomial regression
Soil Moisture Content *(g/g)*								
Blinkpan	6.57	4,7	0.89	0.83	14.48	0.000	0.001	$y = 0.0779x^4 - 2.398x^3 + 25.08x^2 - 100.69x + 163.77$
Slangpan	6.355	4,8	0.96	0.95	61.68	0.000	0.000	$y = -0.0362x^4 + 0.7805x^3 - 2.612x^2 - 25.699x + 145.28$
Lumipan	41.85	4,2	0.58	-0.23	0.7154	0.013	0.653	$y = -1.2377x^4 + 19.074x^3 - 95.33x^2 + 157.86x + 99.774$
Manikinikipan	4.099	2,1	0.98	0.95	36.57	0.033	0.116	$y = -8.25x^2 + 27.417x + 32.083$
Olopan	4.899	1,1	0.96	0.92	26.01	0.025	0.123	$y = 17.667x + 36.667$
Sandapan	5.248	4,2	0.99	0.99	461.9	0.001	0.002	$y = 0.2348x^4 - 6.9798x^3 + 70.083x^2 - 300.15x + 510.33$
Thandopan	4.797	2,2	0.99	0.99	248.6	0.000	0.004	$y = -9.0238x^2 + 22.043x + 159.6$
Mvulenipan	19.87	1,1	0.71	0.43	2.527	0.054	0.357	$y = -22.333x + 179.44$
Soil Electrical Conductivity *(dS/m)*								
Blinkpan	5.777	4,7	0.39	0.04	1.131	0.000	0.414	$y = -0.0186x^4 + 0.544x^3 - 5.4178x^2 + 19.931x - 1.5949$
Slangpan	9.73	2,10	0.24	0.09	1.624	0.000	0.245	$y = -0.3376x^2 + 5.3865x + 6.6416$
Lumipan	1.126	2,4	0.96	0.95	62.1	0.000	0.000	$y = 0.4127x^2 - 1.0397x + 13.595$
Manikinikipan	3.801	2,1	0.33	-0.98	0.2564	0.117	0.813	$y = 0.5833x^2 - 4.0167x + 15.917$
Olopan	10.89	1,1	0.56	0.13	1.317	0.200	0.456	$y = 13.333x^2 - 44.5x + 46.167$
Sandapan	3.41	2,4	0.85	0.78	11.91	0.000	0.020	$y = 0.6071x^2 - 7.8214x + 32.333$
Thandopan	0.5087	2,2	0.95	0.90	19.23	0.000	0.049	$y = 0.5714x^2 - 2.6952x + 18.433$
Mvulenipan	18.03	1,1	0.72	0.44	2.606	0.263	0.353	$y = 22.083x^2 - 67.75x + 56.167$
Soil Bulk Density *(g/cm³)*								
Blinkpan	0.04843	4,7	0.56	0.32	2.305	0.000	0.157	y = 9E-05x4 - 0.0022x3 + 0.021x2 - 0.1014x + 1.9089
Slangpan	0.07071	4,8	0.82	0.73	9.443	0.000	0.004	y = 0.0003x4 - 0.0073x3 + 0.0585x2 - 0.0957x + 1.4103
Lumipan	0.04927	4,2	0.79	0.39	1.98	0.000	0.362	y = 0.001x4 - 0.0129x3 + 0.0571x2 - 0.1292x + 1.7186
Manikinikipan	0.03665	1,2	0.29	-0.05	0.8439	0.000	0.455	$y = -0.0151x + 1.6853$
Olopan	0.06017	1,1	0.84	0.69	5.579	0.012	0.255	$y = -0.1005x + 1.9704$
Sandapan	0.07066	1,5	0.63	0.56	8.674	0.000	0.032	$y = 0.0393x + 1.4824$
Thandopan	0.04942	1,3	0.50	0.34	3.088	0.000	0.177	$y = 0.0275x + 1.4686$
Mvulenipan	0.1001	1,1	0.54	0.09	1.203	0.024	0.470	y = 0.1226x2 - 0.568x + 2.0787

Figure G.1: Summary statistics and associated coefficients of the polynomial regression models fitted for each of the three edaphic factors across the eight sampled depressional wetlands are presented in Figure 2.5

Appendix H

Presentation of summary statistics data used in the manuscript

Plot#	EC Mean±SD (dS/m)	SMC Mean±SD (g/g)	BD Mean±SD (g/cm^3)	N
P01	15.25±09.62	142.45±75.66	1.63±0.21	33
P02	12.56±04.35	115.43±70.84	1.58±0.24	35
P03	21.92±15.57	078.68±41.48	1.59±0.21	34
P04	18.86±15.53	061.91±47.23	1.60±0.17	21
P05	15.19±09.82	050.74±37.29	1.63±0.13	19
P06	23.43±16.88	047.86±39.12	1.63±0.18	14
P07	17.07±10.36	047.21±38.92	1.70±0.12	14
P08	19.17±11.81	040.17±22.98	1.68±0.09	6
P09	16.75±08.24	040.50±21.75	1.67±0.12	8
P10	15.17±07.05	048.00±07.38	1.69±0.17	6
P11	21.00±15.39	044.00±04.56	1.62±0.15	6
P12	20.50±11.27	049.50±14.18	1.69±0.16	4
P13	14.00±05.66	050.00±11.31	1.46±0.17	2

EC= Electrical Conductivity
SMC=Soil Moisture Content
BD=Bulk Density

Figure H.1: Summary statistics and associated coefficients of the polynomial regression models fitted for each of the three edaphic factors across the eight sampled depressional wetlands are presented in Figure 2.5

Appendix I

Comparing Soil Moisture between wetland and upland across the three wetlands

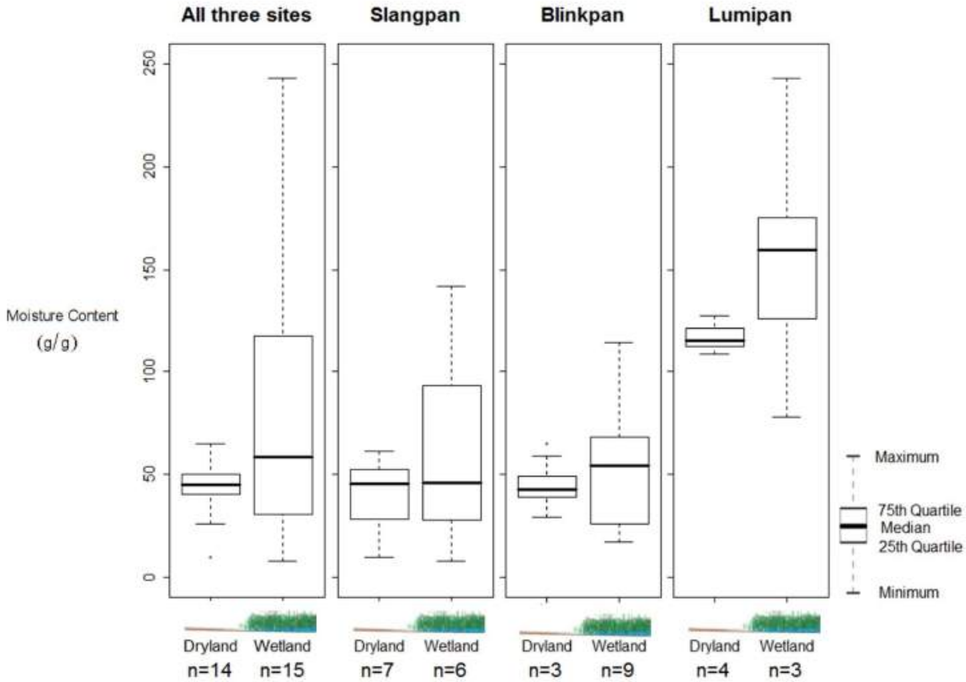

Figure I.1: Differences in SMC between the wetland and dryland zones for the three most extensive wetlands (Blinkpan, Slangpan and Lumipan). Results of statistical significance can be seen in Appendix I

Appendix J

Coefficients for polyline models fitted on Hyperspecral indeces

Variable	Site	Err	R^2	R	*F*-Stat	Df	DF	P-Val	Order
SCI	Blinkpan	2979	0.3413,	0.1949	2.331	2	9	0.1528	2nd
	Lumipan	0.007248	0.8821,	0.8231	14.96	2	4	0.01391	2nd
	Manikinikipan	0.00191	0.9898,	0.9693	48.28	2	1	0.1012	2nd
	Mvulenipan	0.06994	0.05662,	-0.8868	0.06001	1	1	0.8471	1st
	Olopan	0.003775	0.9574,	0.8722	11.24	2	1	0.2064	2nd
	Sandapan	0.04933	0.8034,	0.7051	8.173	2	4	0.03865	2nd
	Slangpan	0.05119	0.354	0.1925	2.192	2	8	0.1742	2nd
	Thandopan	0.03404	0.335	-0.33	0.5037	2	2	0.665	4th
NDSI	Blinkpan	0.01175	0.253	0.08697	1.524	2	9	0.2692	2nd
	Lumipan	0.00459	0.9064	0.8596	19.37	2	4	0.008762	2nd
	Manikinikipan	0.000346	0.9978	0.9933	223.4	2	1	0.04726	4th
	Mvulenipan	0.003962	0.9004	0.8007	9.037	1	1	0.2044	1st
	Olopan	0.01348	0.5183	-0.445	0.5381	2	1	0.694	3rd
	Sandapan	0.00711	0.8347	0.752	10.1	2	4	0.02733	4th
	Slangpan	0.007831	0.4486	0.3107	3.254	2	8	0.09244	2nd
	Thandopan	0.003863	0.686	0.372	2.185	2	2	0.314	2nd
MSBI	Blinkpan	0.15	0.5786,	0.4849	6.178	2	9	0.02047	4th
	Lumipan	0.06305	0.7236,	0.5854	5.235	2	4	0.07641	2nd
	Manikinikipan	0.009645	0.9963,	0.9889	134.2	2	1	0.06093	2nd
	Mvulenipan	0.1409	0.2087	-0.5826	0.2637	1	1	0.698	1st
	Olopan	0.03026	0.6052,	-0.1845	0.7663	2	1	0.6284	4th
	Sandapan	0.09909	0.8817,	0.8226	14.91	2	4	0.01399	2nd
	Slangpan	0.1121	0.2508,	0.06345	1.339	2	8	0.3151	2nd
	Thandopan	0.08708	0.3215,	-0.357	0.4738	2	2	0.6785	2nd
NDWI	Blinkpan	0.05251	0.2069,	0.0307	1.174	2	9	0.3523	4th
	Lumipan	0.04087	0.9163,	0.8745	21.9	2	4	0.007002	2nd
	Manikinikipan	0.000469	0.9876,	0.9629	39.97	2	1	0.1111	2nd
	Mvulenipan	0.00713	0.7969,	0.5938	3.923	1	1	0.2976	1st
	Olopan	0.000644	0.999,	0.9971	517.4	2	1	0.03107	2nd
	Sandapan	0.009748	0.8534,	0.7801	11.64	2	4	0.02149	2nd
	Slangpan	0.03208	0.8564,	0.8204	23.85	2	8	0.000426	2nd
	Thandopan	0.006281	0.6823,	0.3646	2.148	2	2	0.3177	2nd

Figure J.1: Coefficients of fitted polynomial regression models

Appendix K

Data summary for Hyperspectral data

SCI								
	Site_Name	***n***	**Mean**	**Median**	**Min**	**Max**	**SD**	**Variance**
	Blinkpan	248	0.3208	0.3156	0.1911	0.5183	0.0764	0.0058
	Lumipan	195	0.3151	0.3209	0.1825	0.451	0.06	0.0036
	Manikinikipan	66	0.3693	0.4066	0.2112	0.4546	0.0823	0.0068
	Mvulanipan	42	0.2704	0.2329	0.1988	0.4205	0.0685	0.0047
	Olopan	46	0.2912	0.2944	0.2518	0.3321	0.0283	0.0008
	Sandapan	101	0.2795	0.2793	0.1948	0.3895	0.0527	0.0028
	Slangpan	15	0.2641	0.1829	0.172	0.4374	0.1267	0.0161
	Thandopan	70	0.3353	0.3331	0.2923	0.3838	0.0256	0.0007
NDSI								
	Site_Name	***n***	**Mean**	**Median**	**Min**	**Max**	**SD**	**Variance**
	Blinkpan	248	0.3208	0.3156	0.1911	0.5183	0.0764	0.0058
	Lumipan	195	0.3151	0.3209	0.1825	0.451	0.06	0.0036
	Manikinikipan	66	0.3693	0.4066	0.2112	0.4546	0.0823	0.0068
	Mvulanipan	42	0.2704	0.2329	0.1988	0.4205	0.0685	0.0047
	Olopan	46	0.2912	0.2944	0.2518	0.3321	0.0283	0.0008
	Sandapan	101	0.2795	0.2793	0.1948	0.3895	0.0527	0.0028
	Slangpan	15	0.2641	0.1829	0.172	0.4374	0.1267	0.0161
	Thandopan	70	0.3353	0.3331	0.2923	0.3838	0.0256	0.0007
NDWI								
	Site_Name	***n***	**Mean**	**Median**	**Min**	**Max**	**SDd**	**Variance**
	Blinkpan	248	-0.1704	-0.1943	-0.5228	0.143	0.1504	0.0226
	Lumipan	195	-0.3359	-0.3297	-0.4886	-0.1822	0.0835	0.007
	Manikinikipan	66	-0.0739	-0.0206	-0.3063	0.0376	0.1176	0.0138
	Mvulanipan	42	-0.3307	-0.2992	-0.4586	-0.2094	0.0864	0.0075
	Olopan	46	-0.2589	-0.2513	-0.363	-0.1896	0.0521	0.0027
	Sandapan	101	-0.31	-0.1779	-0.7796	-0.0671	0.2359	0.0557
	Slangpan	15	-0.4967	-0.5388	-0.5948	-0.3563	0.1052	0.0111
	Thandopan	70	-0.2631	-0.2777	-0.3485	-0.1523	0.0611	0.0037
MSBI								
	Site_Name	***n***	**Mean**	**Median**	**Min**	**Max**	**SD**	**Variance**
	Blinkpan	248	1.1765	1.1565	0.6479	1.6956	0.243	0.059
	Lumipan	195	0.8648	0.8761	0.6982	1.1025	0.0974	0.0095
	Manikinikipan	66	1.1521	1.1951	0.9168	1.2849	0.1234	0.0152
	Mvulanipan	42	0.8821	0.8645	0.7375	1.0188	0.1076	0.0116
	Olopan	46	0.9295	0.9429	0.8482	0.9724	0.0372	0.0014
	Sandapan	101	0.8826	0.9814	0.422	1.1612	0.2464	0.0607
	Slangpan	15	0.7047	0.6832	0.592	0.8402	0.1053	0.0111
	Thandopan	70	0.9442	0.9255	0.8375	1.1309	0.0836	0.007

Figure K.1: Summary statistics

Appendix L

Results for multiple comparisons of sites on basis of hyperspecrtal indices

Table L.1: Results of Tukey's Honestly Significant Difference Post Hoc test (THSD)

Paired Comparisons	SCI		NDSI		NDWI		MSBI	
Site Name	Diff	*p*-val	Diff	*p*-val	Diff	*p*-val	Diff	*p*-val
Lumipan-Blinkpan	-0.0057	0.9857	-0.0886	0	-0.1655	0	-0.3116	0
Manikinikipan-Blinkpan	0.0485	0	0.0571	0.0042	0.0965	0	-0.0243	0.9766
Mvulanipan-Blinkpan	-0.0504	0.0001	-0.1267	0	-0.1603	0	-0.2944	0
Olopan-Blinkpan	-0.0295	0.0977	-0.075	0.0005	-0.0885	0.0014	-0.247	0
Sandapan-Blinkpan	-0.0413	0	-0.1124	0	-0.1395	0	-0.2939	0
Slangpan-Blinkpan	-0.0567	0.0274	-0.189	0	-0.3262	0	-0.4718	0
Thandopan-Blinkpan	0.0146	0.7289	-0.0413	0.0968	-0.0927	0	-0.2322	0
Manikinikipan-Lumipan	0.0542	0	0.1457	0	0.262	0	0.2873	0
Mvulanipan-Lumipan	-0.0447	0.0018	-0.0382	0.444	0.0052	1	0.0172	0.9992
Olopan-Lumipan	-0.0238	0.3471	0.0135	0.9951	0.077	0.0129	0.0646	0.3463
Sandapan-Lumipan	-0.0356	0.0003	-0.0239	0.6307	0.026	0.772	0.0178	0.9924
Slangpan-Lumipan	-0.051	0.0759	-0.1005	0.0144	-0.1607	0.0003	-0.1602	0.019
Thandopan-Lumipan	0.0203	0.3472	0.0472	0.041	0.0728	0.0031	0.0794	0.0314
Mvulanipan-Manikinikipan	-0.099	0	-0.1839	0	-0.2568	0	-0.2701	0
Olopan-Manikinikipan	-0.0781	0	-0.1321	0	-0.185	0	-0.2227	0
Sandapan-Manikinikipan	-0.0899	0	-0.1695	0	-0.2361	0	-0.2695	0
Slangpan-Manikinikipan	-0.1052	0	-0.2461	0	-0.4228	0	-0.4475	0
Thandopan-Manikinikipan	-0.034	0.0546	-0.0984	0	-0.1892	0	-0.2079	0
Olopan-Mvulanipan	0.0209	0.8146	0.0517	0.3404	0.0718	0.2047	0.0474	0.9181
Sandapan-Mvulanipan	0.0091	0.9953	0.0143	0.9966	0.0208	0.9911	0.0005	1
Slangpan-Mvulanipan	-0.0063	1	-0.0623	0.5533	-0.1659	0.0013	-0.1774	0.022
Thandopan-Mvulanipan	0.065	0	0.0854	0.0017	0.0676	0.1742	0.0621	0.6306
Sandapan-Olopan	-0.0118	0.9734	-0.0374	0.5326	-0.051	0.4051	-0.0469	0.8195
Slangpan-Olopan	-0.0271	0.8634	-0.114	0.011	-0.2377	0	-0.2248	0.0007
Thandopan-Olopan	0.0441	0.0104	0.0337	0.7338	-0.0042	1	0.0147	0.9999
Slangpan-Sandapan	-0.0153	0.9906	-0.0766	0.1819	-0.1867	0	-0.1779	0.008
Thandopan-Sandapan	0.0559	0	0.0711	0.0008	0.0468	0.3397	0.0616	0.3401
Thandopan-Slangpan	0.0712	0.0038	0.1477	0.0001	0.2335	0	0.2396	0.0001

Appendix M

Results of t-test between wetland and dryland

Site	T-value	DF	P-value	Dryland Mean SMC	Wetland Mean SMC
Between sites	-3.863	58.2	0.0002	44.0	78.9
Within Slangpan	-1.527	23.8	0.1399	40.3	58.7
Within Blinkpan	-0.686	8.7	0.5102	44.6	52.4
Within Lumipan	-2.263	10.5	0.0458	117.3	157.5

EC= Electrical Conductivity
SMC Soil Moisture Content
BD Bulk Density

Figure M.1: Results of t-test between wetland and dryland zones for each of the three sampled wetlands and all three wetlands together

Appendix N

Variance between sites in wetland and dryland

Edaphic factor	Within Wetland	F statistic	Degrees of Freedom	Ratio of variances	P-value	Minimum variance	Maximum variance
SMC	Blinkpan	0.086544	16,8	0.086	0.000	0.021	00.270
(g/g)	Lumipan	0.032897	2,9	0.032	0.064	0.005	01.295
	Slangpan	0.14966	7,17	0.149	0.016	0.047	00.676
BD	Blinkpan	0.32069	16,8	0.320	0.050	0.078	01.002
(g/cm^3)	Lumipan	0.44553	2,9	0.445	0.692	0.077	17.548
	Slangpan	0.37082	7,17	0.370	0.186	0.117	01.676
EC	Blinkpan	0.42973	16,8	0.429	0.143	0.105	01.342
(dS/m)	Lumipan	1.1913	2,9	1.191	0.695	0.208	46.923
	Slangpan	0.84524	7,17	0.845	0.867	0.267	03.821

EC= Electrical Conductivity
SMC=Soil Moisture Content
BD=Bulk Density

Figure N.1: Variance in the three edaphic factors of interest across the three biggest wetlands.

Appendix O

Sample ground truth of National Wetland Map 5

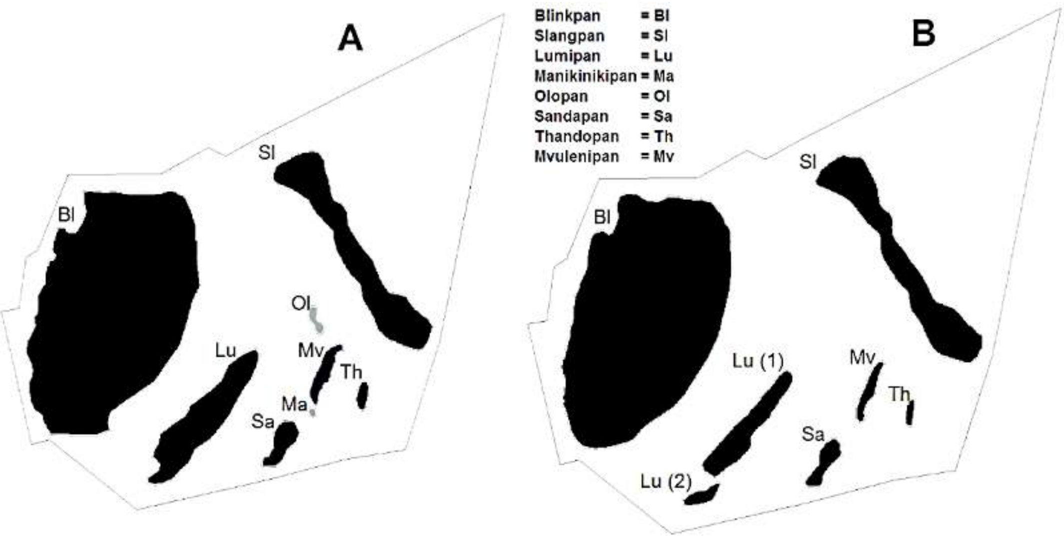

Figure O.1: The boundaries of the eight sampled wetlands, digitised with assistance of ground truth Global Positioning System (GPS) tracks with minimum accuracy of 5 m Panel A. Panel B = the same wetlands as published in the National Wetland Map 5 (Van Deventer *et al.* 2020)