LONG-TERM POPULATION TRENDS AND HABITAT PREFERENCES OF WADERS AT STRANDFONTEIN WASTEWATER TREATMENT WORKS

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Dissertation Presented for the Degree of
Master of Sciences
in the Department of Biological Science
UNIVERSITY OF CAPE TOWN

August 2016

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DECLARATION

I know the meaning of plagiarism and declare that all of the work in the dissertation, save for that which is properly acknowledged, is my own.

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November 2016

ABSTRACT

Increasingly, populations of both long-distance migrant and resident wader species are declining at a rapid rate. Migratory waders have complex ecological requirements in their summer and winter ranges, making them particularly susceptible to habitat loss, natural system fluctuations (influencing local seasonal movement), disturbance, effects of climate change, predator impacts at breeding sites, and resource availability. The latter is fundamental in determining species‟ distribution. Selection of a suitable wetland will limit the necessity of local movements and thus optimise resource exploitation. This is particularly relevant for migratory species as replenishing fat reserves at resource-rich sites provisions the birds for the energetically costly flight back to the Tundra after overwintering at southern latitudes.

Various biological and ecological factors affect local invertebrate biomass – including wetland size, substrate type, area of terrestrial or aquatic vegetation cover, water turbidity, wind effects, predator-prey relationships, and extent of littoral edge zone. In turn these influence species richness and diversity of wader abundance at a specific wetland. In artificial wetlands certain of these variables can be controlled to optimise habitat quality for wader populations. Artificial wetlands have the potential to provide partial mitigation from the loss and degradation of natural wetlands, and could play an imperative function in wader conservation. The most productive artificial wetlands in urban areas are typically wastewater treatment works (WWTW). In South Africa, the Cape Flats Waste Water Treatment Works

(CFWWTW) at Strandfontein is a dynamic artificial wetland supporting both abundances and species richness of waders on par with the best natural wetlands nationally.

In order to determine if global declines in wader numbers are evident in wader populations at the southern end of their migratory range, I used long-term historic count data from the literature and medium-term (31 years) count data from Coordinated Water Avian Count (CWAC) records for Strandfontein to analyse trends in abundance, richness, diversity and seasonality of waders.

Although numerous studies have investigated habitat preferences of waders, few have focused on the identification of ecological requirements that influence fine scale distribution of resident and migrant waders in artificial wetlands. This would allow for manipulation of these parameters for effective management of synthetic water bodies to benefit migratory and resident waders. These findings can be utilised to improve the management of other regulated wetlands in the Western Cape Province, South Africa.

The aims of the dissertation are to describe long term migrant and residential wader population trends at Strandfontein, and to identify the potential factors which influence local wader abundance. Once identified, these factors are integrated into a management plan that would create ideal wader habitat, and attract larger numbers of waders to Strandfontein.

Monthly sampling of environmental variables and wader abundance was carried out at Strandfontein, Western Cape Province, South Africa starting in July 2014. This dissertation tentatively identified four major environmental variables that could potentially influence wader abundance at the local scale, namely water level fluctuation, extent of littoral zone, vegetation cover, and invertebrate abundance and biomass. .

Overall, analyses of historic and medium-term data revealed that abundance of migrant waders had declined since 1952, and resident wader numbers had increased over the same period. The same trends continued from 1983 onwards (as reflected in the CWAC count data). This could reflect the global trends in wader populations generally (viz. population decreases in migrants, and stability or increase in resident species). Habitat change and low reproductive success at wintering ranges could explain the oscillating peaks and depressions in abundance seen over the period 1952 to 2014.

Total invertebrate biomass peaked in February and October. Invertebrate abundance and biomass were not correlated, as some numerically abundant species (e.g. chironomid larvae) were orders of magnitude smaller than other larger and less numerically dominant species (e.g. terrestrial invertebrates). Total invertebrate abundance, showed noticeable seasonal fluctuations. Abundance of chironomids and aquatic invertebrates peaked in summer, and that of terrestrial invertebrates and worms peaked in winter. Chironomids were the most abundant component of overall invertebrate abundance; worms had the lowest biomass and abundance over the year.

GLM results identified receding water level, high chironomid abundance and low percentage vegetation cover as the most important variables explaining wader abundance, accounting for 26% of the observed variability in numbers. Greater vegetation cover had a negative association with wader abundance, suggesting that waders avoid heavily vegetated areas when foraging. Access to the mud zone (viz. foraging sites) varied seasonally, when water levels retreated as summer advanced, allowing for access to the invertebrate resources. Chironomid abundance increased in summer, coinciding with wader influx to Strandfontein.

The variables associated with foraging preferences of wader distribution need to be integrated into the conservation management plan for Strandfontein (CFWWTW). Removing dense vegetation on the shoreline (*Typha* reed beds) will open up foraging habitat in the littoral (mud) zone, and limit need for seasonal movement of resident species. Manipulating water levels at intervals throughout the year, and particularly in summer will extend foraging habitat and access to prey, benefitting both resident and migrant species.

CONTENTS

ACKNOWLEDGEMENTS

I am grateful to my supervisor Prof. Mike Picker for his wisdom, encouragement, editorial assistance and for always going above and beyond for me. I would like to thank my co-supervisor Prof. Les Underhill for the statistical assistance and for making the time to help me, even in trying circumstances and to my co-supervisor Dr Dave Whitelaw for moral support and inspiration.

Financial support for this study was provided by the Julie te Groen trust – of the Cape Bird Club. I am truly grateful to the late Julie te Groen herself and the Julie te Groen Trust committee members; Dave Whitelaw, Clifford Dorse, and Richard Barnes for affording me the opportunity to further my education and granting me the chance to benefit the management at Strandfontein.

Thanks must go to the CWAC count team dedicated to Strandfontein for the years of data. Special thanks to Richard Barnes for the collated hard copies.

I appreciate the assistance of the False Bay nature reserve staff and students in field sampling; H. Saal, J. Davids, S. Davids, A. Quisma, M. Bridgens, J. du Plessis, B.Fortuin, T. Marco, R.Morton, K. Anthony, C.Mathews, and R.Isaacs. To my colleagues at the False Bay nature reserve; Mogamat Samsodien, Bronwen Amayo and Victoria Day, thank you for the support when the going got tough. Juggling full time work and a full time degree was not an easy feat, thank you for always listening and for all the advice.

I am appreciative for the editing assistance from Dr. Katherine Hyman, and for your constructive criticism. Amalia Pugnalin is thanked for creating the wonderful maps I needed

for Chapter two, and the South African Weather Service for the wind speed and direction measurements.

And last but by all means not least, I would like to thank my family – Mom, Paul, Nicola and Noekies, for your constant support and love and for cheering me on, until the end.

CHAPTER 1. INTRODUCTION

1.1 Importance of feeding grounds for waders

Waders (Charadriiformes) are a diverse order of small to medium birds, comprising 209 species, 76 of which have been recorded in South Africa (Delany *et al*., 2003; Maclean, 2001). The majority of the South African species are long distance, seasonal visitors (including vagrants) that utilise Southern Africa"s shores and wetlands in their non-breeding period.

Migratory species rely on more than one locale, namely breeding sites, staging migration sites, and in between stop-overs for roosting and feeding en route. Migrant waders visiting southern African largely use the East Atlantic Flyway (EAF), which links the arctic breeding grounds in the northern hemisphere to their summer range in the southern hemisphere (Birdlife International, 2008). This dependence on three separate sets of resources and multiple habitats makes them vulnerable to environmental impacts at any of these sites and therefore coarse and fine scale wader distribution and abundance is highly irregular (Harebottle *et al*., 2006). Aquatic macroinvertebrates are the primary food source for waders and could be the explanatory variable for fine scale distribution (Manikannan *et al*., 2012). The greatest variation of wader communities present (over an annual basis) can be primarily related to food availability and one of the main factors driving this is water depth (Manikannan *et al*., 2012).

Most wader species express site fidelity and are nomadic, considering the exacting habitat requirements, stop over and seasonal locales need to be predictable in terms of availability of invertebrate prey (Thompson & Hale, 1989; Tomkovich & Soloviev, 1994). Two important wader hotspots recognized in the South-Western Cape, which have global and national significance, viz. the Berg river Estuary and the Langabaan Lagoon, each support approximately 30 000 (largely-migrant) waders in austral summer (Hockey, 1998). Smaller satellite wetland sites like the Strandfontein Section of the False Bay Nature Reserve (commonly known as Strandfontein, or Strandfontein Sewage Works in the literature), Dyer Island and Robben Island are also essential wader (migrant and resident) habitats in South Africa (Summers *et al*., 1977; Hockey, 1998). However, pressure on optimal sites could render the sites unsuitable for wader communities. Environmental hazards across foraging sites include loss of suitable habitat, harsh weather conditions, predation, overharvesting, climate change, and environmental pollution. Moreover, the complex interactions between waders and their prey can also influence breeding success and mortality rates. Together these could explain the global widespread declining numbers of the Charadriformes (Hockey & Turpie, 1999; Kalejta-Summers *et al*., 2001b; Balachandran, 2006; Lagos *et al*., 2008; Rakhimberdiev *et al*., 2011).

Expected and anthropogenic influenced fluctuations apparent at natural wetlands can often affect waders negatively, e.g. both encroachment of vegetation and water level affect access to and presence of food resources. These factors are often unpredictable and will subsequently provide unfavourable wader habitat.

This is of particular significance for sites that have an adaptive management approach to conservation. This methodology acknowledges reservations that are apparent in temporal and spatial dynamics and adopts a responsive and flexible approach to the management of dynamic ecosystems. Insufficient resources and rapid environmental change can compromise management goals and priorities, making adaptive management a necessity (Wiens & Hobbs, 2015).

Zeekoevlei, the largest fresh water body in the Western Cape, South Africa, has historically supported an abundance of wader species. Bird counts of over 20'000 individuals of 15 migratory species were recorded between April 1939 and April 1940 (Broekhuysen & Meiklejohn, 1941). The vlei was monitored regularly from 1939 until the end of 1952 however, in 1953, a permanent weir was built, and thereafter counts subsided. Management practices changed, and resulted in the vlei being primarily utilised for recreational activities, namely yachting and rowing (Winterbottom, 1960). The encroachment of *Typha capensis*, and short annual draw down periods in June, resulted in increased water levels in austral summer. This ultimately led to minimal or no littoral edge and nominal regression of the water edge – potentially directly resulting in the loss of suitable foraging habitat for migrating waders (D.A. Whitelaw, pers. comm). The reserve uses a database tool for monitoring purposes, launched in 2007 (D. Gibbs, pers. comm.), and no migrant waders have been recorded at Zeekoevlei since then (Biodiversity Data Base, 2014). Due to the current management plans, the ecological integrity of the site for wader habitat was compromised resulting in a total loss of all wader species.

Such modifications of habitat are extreme, and smaller natural fluctuations do occur, forcing resident and migratory waders to undergo numerous local and regional movements within a season to search for suitable foraging habitat (Cooper, 1978; Zwarts *et al*., 1998; Plissner *et al*., 2000). Ringed Red Knots (*Calidris canutus*) were used to show that dispersion occurred rapidly throughout the expansive Moray Firth Estuarine Complex in Scotland, United Kingdom, and that the birds consequently frequented wetland habitats elsewhere (Symonds *et al.,* 1984; Symonds & Langslow, 1986).

It is of conservation importance to maintain the ecological quality of wader dependent sites, including non-breeding sites in South Africa where the birds need to both replenish reserves after the long migration south, and build up fat stores for the return flight to their arctic breeding grounds (Hammond &Pearson, 1994). Wader spatial distribution positively correlates with benthic invertebrate populations suggesting that foraging waders use prey abundance to select habitat (Colwell & Landrum, 1993; Safran *et al*., 1997).

Most recent wader research focuses on wader conservation and population demography in light of the worldwide decline in wader numbers, typically focusing on predicting the effect of habitat loss and transformation on wader populations (Durell, 2000; Masero *et al.*, 2000; Murray & Hamilton, 2010). These highlight the need to preserve the more predictable wader hotspot areas for long term conservation.

However, natural wetlands can be unpredictable. The larger natural wetlands in South Africa are generally ephemeral, and small shallow wetlands tend to become flooded with water in winter and dry out in summer months (Dini *et al.*, 1998), providing short lived but high quality habitat and food resources for waders. Waders rely on shallow water and large wet littoral edges for optimum feeding rates, but the summer months in South Africa often escalate the evaporation rates of seasonal wetlands leaving them dry and barren. Whilst receding water levels are optimal feeding habitat for waders (Sanders, 1999), desiccation of shallow pans has negative effects on the invertebrate community. Benthic invertebrates are more readily available to foraging waders in wet or moist wetland areas, since the drying out of the substrate results in the invertebrate community burrowing deeper, hence inaccessible to predators. This instability in natural systems makes them unreliable and only useful to waders for limited window period of water recession.

1.2 Artificial wetlands

Artificial foraging areas offer a potentially more stable feeding ground for waders and their correct management, when aligned to meet the ideal feeding requirements of waders, may have positive spinoffs for wader conservation (Harebottle *et al.*, 2008). From a conservation management perspective, it is important to identify ecological parameters relating to optimal habitat requirements of waders, allowing effective management of degraded artificial systems.

It is now well-documented and recognised by the Ramsar convention that man-made, managed wetlands have the potential to provide partial mitigation of adverse influences resulting from degradation and transformation of natural wetlands, and thereby support the conservation effort of birds (Fasola & Ruiz, 1996; Harebottle *et al*., 2008; Murray & Hamilton, 2010). In this regard the artificial wetland needs to mimic the general availability of resources utilised by waders that natural habitats once provided. Artificial wetlands are wetland areas that are either transformed or created through anthropogenic means, and include salt ponds, sewage works, paddy fields, aqua cultural ponds, and reservoirs.

1.2.1 Wastewater treatment works

The most productive artificial wetlands in most urban areas are wastewater treatment works (WWTW) (Harebottle *et al*., 2008). WWTW offer a rich food resource due to the eutrophic conditions of the water bodies (Harebottle *et al*., 2008) and high biomass or aquatic invertebrates. These attributes could be driving the distribution and abundance of waterbirds at a WWTW (Kalejta-Summers *et al*., 2000b, Ashkenazi 2001, Harebottle, 2012).

The artificially created ponds are considered hypertrophic in comparison to their natural counterparts (Murray & Hamilton *et al.*, 2010). These nutrient rich waters are ideal habitats for benthic invertebrate communities, the preferred food type of most wader species (Safran *et al*., 1997; Sanders, 2000; Colwell, 2010). Waders have a generalised diet of aquatic and terrestrial invertebrates, and some are considered opportunistic feeders implying that they have a wide ranging diet (Rundel, 1982; Hammond & Pearson, 1994). The osesophageal cavities of four wader species in Mingo National Wild-life Refuge, Missouri, were investigated to ascertain generalised diets. Insects (coleopteran and dipteran families) were identified to be the primary prey ingested (Rundel, 1982).

Water levels in WWTW can be manipulated to provide long-term optimal foraging habitat for waders as the water system is more constant and controlled, and may be regulated to transform the habitat to suit individual species needs. Flooding, flushing and draining pans can stimulate germination and regeneration, which provide food and habitat for invertebrates (Safran *et al*., 1997). However WWTW, are subject to major potential problems viz., pathogenic and toxin producing microorganisms, metal and chemical contaminants, human disturbance and physical hazards - most of which are generally less evident in natural systems (Murray & Hamilton, 1998).

In South Africa, Strandfontein is an example of a productive artificial wetland, supporting both bird numbers and species richness comparable with that of the best natural wetlands in the country (Askenkazi, 2001; Kalejta-Summers *et al*., 2001b; Harebottle *et al*., 2008, Marnewick *et al*., 2015). The significance of WWTW for wader conservation is becoming progressively important, as waders utilise them as replacements for natural water bodies lost to anthropogenic change (Rehfisch, 1994). In some cases agricultural lands and commercial salt production pans can similarly provide substantial vital habitats that may be functionally comparable to semi-natural wetlands, for large numbers of waders (Colwell, 2010). The installation of artificial features called foot drains were introduced to create damp habitat when flooded, in grasslands in England, United Kingdom. This provided profitable feeding areas and resource exploitation for breeding Northern Lapwing (*Vanellus vanellus*). Artificial intervention was required to offset the detrimental effects of human interference which diminished breeding sites for foraging waders (Eglington *et al*., 2010).

In some instances artificial sites may be preferred over existing natural wetland systems. Results from a study of wader occurrence at three different salt pans at Guadiana estuary in

South Portugal, suggested a preference by waders to feed at artificially created salt pans compared to tidal flats. In contrast with other salt pan systems most shorebird species in the Guadiana estuary used the salt pans for foraging as opposed to the intertidal flats, even during low tide which is optimum feeding time for waders (Dias, 2009; Dias *et al.,* 2014). Therefore artificial sites can be more attractive than existing natural systems.

While in some occurrences artificial wetlands play an integral role in wader conservation, creating artificial habitat should be viewed as a supplementary measure of conservation. Wetlands are recognised as one of the most essential, productive and diverse habitat types in the world, and a further loss could lead to substantial associative loss of waders (Dini *et al*., 1998; Stroud *et al*., 2008). It is critical to protect the fragmented system in place now, because artificial systems, although offering suitable habitat, are not sustainable for long term conservation. Artificial wetlands generally produce high invertebrate abundance and biomass but low species diversity. Drainage and water regulation can eliminate or degrade the associated insect communities in wetlands (Batzer & Wissenger, 1996). Nelson *et al.* (2000) compared the invertebrate species abundance between wastewater and reference sites (reference wetlands were similar to wastewater wetlands in landscape setting, plant species, and hydrodynamics, however wastewater was not used). Results showed that invertebrate species richness was greater at reference sites.

Linked with loss of natural wetland systems and the further demise predicted (Nicholls *et al.,* 1999; Murray & Hamilton, 2010), is the global increase in wastewater treatment wetlands in developing countries. Murray & Hamilton (2010), show that the development of wastewater treatment works are on the rise, and an ancillary function of this is suitable habitat for waders and associated wetland species alike. However, in many developed countries concerns over water quality and eutrophication have led to the replacement of these artificial wetlands with intensive heavily engineered and chemical treatment technologies which do not provide the valuable resource or habitat for birds that the passive wastewater treatment systems currently do.

1.3 Broad aims and objectives

The aims of the dissertation are to describe long term migrant and resident wader population trends at Strandfontein, and to identify the potential factors which influence local wader abundance. I also determine if maintaining or manipulating an artificial site to create ideal wader habitat, would indeed attract larger numbers of waders to Strandfontein.

The broad aims and objectives of the dissertation are to:

- Analyse a historic (62 year) dataset for long-term trends in seasonality and abundance of migrant and resident waders at Strandfontein WWTW. Mean CWAC count data is used to describe seasonality, diversity, species richness and abundance of migrant and resident waders at Strandfontein.
- Identify biotic and abiotic variables (season, wind speed, water depth/receding water level, invertebrate biomass and abundance, littoral edge size and vegetation cover) associated with fluctuations in wader species abundance at a fine scale (individual water body) using generalised linear models (GLM‟s).
- Track seasonal and spatial variability of invertebrate biomass and abundance at the water body level, and to compare their community structure spatially at defined plots
- To include the variables positively associated with wader biodiversity in a GLM to establish management guidelines for Strandfontein and for managed wetlands in order to provide a framework for sustainable artificial habitats for both local and migrant waders.

CHAPTER 2. STUDY SITES AND SAMPLING METHODS

This chapter provides a background for methods and background relevant to all subsequent chapters. Additional information on Methods provided in subsequent chapters (3 & 4) deals largely with the statistical treatment of data.

2.1 Study area

2.1.1 Strandfontein Section of the False Bay Nature Reserve

Strandfontein (34º 05‟S, 18 º 31‟E) situated 20km south of Cape Town, Western Province, South Africa (Fig. 2.1), is a functional sewage works known as the Cape Flats Waste Water Treatment Works (CFWWTW). Recent surveys (done by the City of Cape Town, unpublished) show that Strandfontein is comprised of approximately 512 ha of aquatic and terrestrial habitats, which covers a larger area than the 59 ha of terrestrial habitats and 319 ha of aquatic habitats Kaletja-Summers *et al*. (2001b) described.

The aquatic habitats encompass approximately 30 interconnected ponds and canals that were historically used as waste maturation pans for the filtering of effluent materials. After introducing chemical cleansing, the pans are now used for emergency water treatment only; however they still contain a high nutrient load. The CFWWTW is an operational sewage works plant that is managed by the Water and Sanitation department of the City of Cape Town Municipality. The area known as the Strandfontein Sewage Works (in the literature), Strandfontein Section (by City of Cape Town), or Strandfontein (the name used in this dissertation) has been managed under the City of Cape Town‟s Biodiversity Management Branch since 2005. Strandfontein forms part of the False Bay Nature Reserve which constitutes a larger network of nature reserves in the City of Cape Town Municipality.

During the study period (July 2014 – June 2015), the aquatic habitat at Strandfontein comprised 30 permanent artificially created pans, five seasonal wetlands, and eight active canals. This excludes the CFWWTW area which falls outside the Strandfontein boundary, comprising largely seasonal wetlands and sludge beds (Figs. 2.1 & 2. 2).

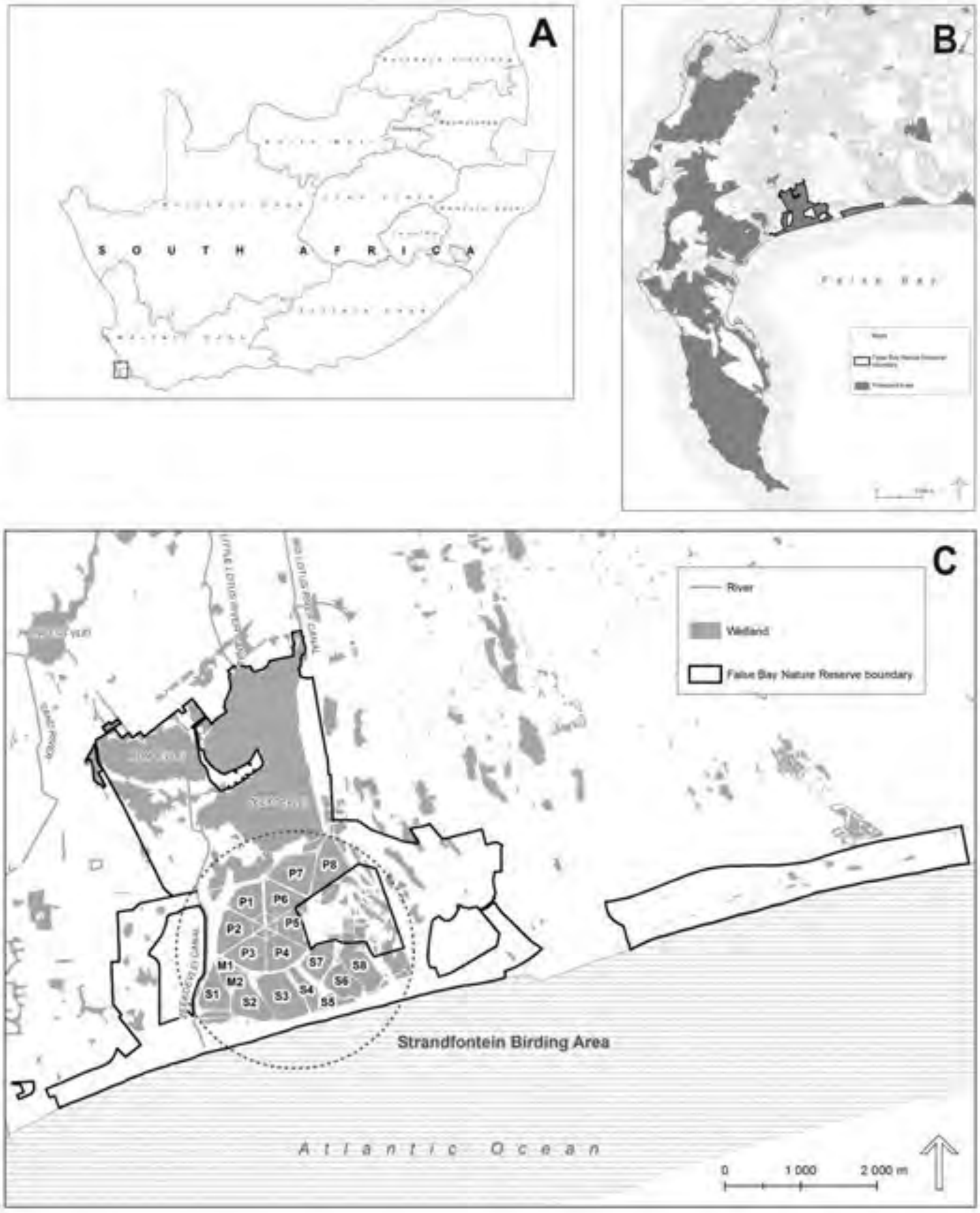

Fig. 2.1. Map of South Africa (A) showing position of Cape Peninsula (B) and False Bay Nature Reserve (C, black outline), containing Strandfontein section (dashed circle). P1-P8, M1 and M2, and S1 – S8 are pan names.

The permanent ponds are either deep (up to 1.5m) and fringed with vegetation (*Typha capensis* and *Phragmites australis),* or shallow with large bare sandy littoral edges or shallow edged with aquatic or terrestrial vegetation (Kalejta-Summers *et al*., 2001b).In addition, the canals are bare or overgrown with aquatic macrophytes, reed beds (largely *T. capensis* and *Phragmites*) and sludge beds (Kalejta-Summers *et al*., 2001b).

2.1.2 The history of Tamatievlei

Until 1921, Strandfontein comprised a 1 ha coastal, seasonally flooded, temporary marsh known as Tamatievlei (Winterbottom 1960; Ashenkazi, 2001). In 1922, the first two irrigation paddocks were built which expanded the water surface area to cover 3.2 ha. Historically these paddocks were used to hold sewage (Ashenkazi, 2001), and formed the basis of the Cape Flats Waste Water Treatment Works (CFWWTW) as it is known today. In the 1950"s, Winterbottom (1960) described the area as „permanent in part", with increases in size due to flood water in winter, having a diameter of 365 m. This was supported by the construction of an additional nine ponds in the same decade. In 1962 the outlet from the CFWWTW was fabricated so that treated water could run into the Zeekoevlei canal and subsequently into the sea (Bickerton, 1982). Expansion and development continued at the sewage works and toward the end of the 1960s, Strandfontein boasted 28 artificially constructed pans, varying in size from 18 to 365 m (diameter), used to treat sewage (Ashenkazi, 2001; Harebottle, 2012). The large number of shallow ponds used a maturation process, whereby treatment of effluent functioned primarily by „algal decomposition". The 1970"s saw an additional six ponds created, bringing the total number of water bodies to 34, with a total water surface area of 306 ha. One of the biggest changes at Strandfontein was the introduction of a new activated sludge plant which began operating in 1980. This system, which remains the incumbent technology, chemically treats sewage in the plant and then releases by-products into the pans as a second phase in filtering the treated water.

2.1.3 Pan primary one at Strandfontein Section

Primary Pan 1 (P1) was used for the study. It was selected from the 30 permanent waterbodies as it supported high wader abundance in summer months, and generally high observational counts of resident migrants in winter months (pers. obs.). The pan forms part of the primary water system for the area, and is a segment of the „wagon wheel‟ which is fed directly from the operational wastewater treatment ponds. The water levels for this pan have been manipulated in the past years, i.e. water flow is shut off so that water levels are kept at a low constant. Water is fed equally to the pans in the ox wheel through the centre feeding station. The circular centre cement sluice is designed for water management, in that wooden boards can be placed into the sluices for each water channel feeding the different segments of the wheel. This enables control of the inflow of additional water or shutting off the water entering a specific pan. The water levels fluctuate seasonally, but are aided by the water operating system from the centre point (Fig. 2.2).

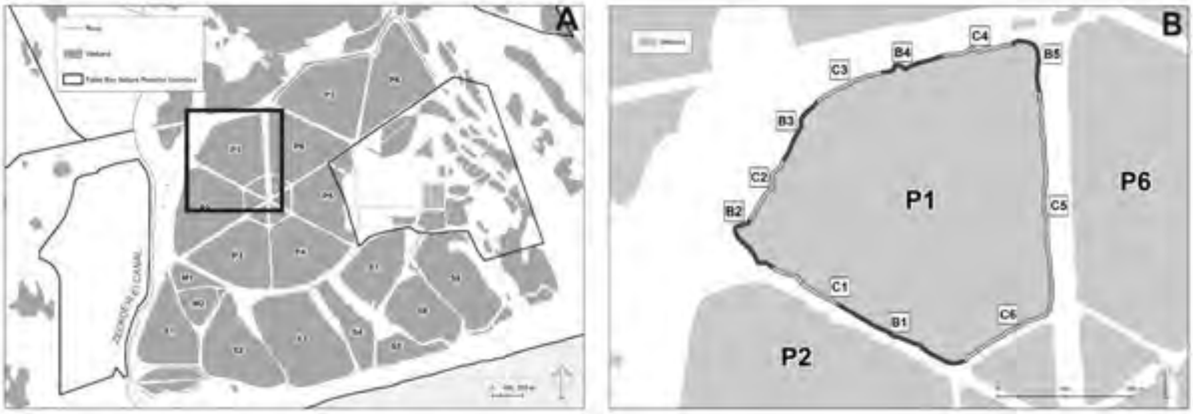

Fig. 2.2. Map of the Strandfontein section of the False Bay Nature Reserve (A), with emphasis on the study site P1 (B) showing the pan divided into 11 labelled plots for sampling.

The pan was previously infested with water hyacinth (*Eichhornia crassipes*) , which was eradicated in 2012. The immediate effects were an increase in bird species, particularly wader migrants during the summer months (D. Whitelaw, pers. comm.). Pan P1 has a perimeter of approximately 1200 m. The pan was divided into 11 separate plots, all 100 m in length (apart from B1 (200 m) and C5 (400 m)), each having similar vegetation and topography. The eastern edge of the pan was lined with dense reed beds (plots C4, C5 and C6 were overgrown with *T. capensis.)* and the remaining edges were either bare or had some marginal aquatic or terrestrial invasive or indigenous vegetation. A long arm excavator was used to remove all marginal vegetation, and the banks of six of the plots were graduated to provide a controlled variation in bank profile. The rest of the plots were left as controls. The plots were labelled B1-B5 (bulldozed) and C1-C6 (control). Each plot had a sampling subplot every 10-20 m where the environmental sampling took place in a 2 m by 2 m plot, once monthly (Fig. 2. 2).

2.2 Sampling

2.2.1 Benthos Sampling

Invertebrates were sampled once monthly between July 2014 and June 2015. A soil sample was taken randomly from the mud zone (the moist seepage area above the high water mark) and also from the substratum underwater (referred to as sub), within a 2 m x 2 m area of the water level. Invertebrates were sampled at each subplot by taking two samples per zone (mud or sub), which were combined into one representative sample per zone. Samples were taken using a stainless steel cylindrical prawn pump with a 5 cm circumference, equipped with a plunger. This was inserted into the soil (fine mud) and rotated so that it cut neatly into the substratum. The plunger was withdrawn from the prawn pump, which sucked the soil into the cylindrical tube. Soil was ejected from the plunger and the top 10cm was measured with a ruler and only this section was utilised. Occasionally there was difficulty in using the steel cylinder pump, and a hand trowel was used instead. This was rotated into the soil at roughly 10cm deep to „cut" out a section of substratum as a subsample. In drier months, thick sludge formed on the littoral edges, and there was some difficulty to attain soil samples. In this instance a plastic container measuring a circumference of 5cm and 10cm in height was used to „scoop" out the soil/sludge sample. A 10 cm sampling depth was used as to include general foraging restrictions of most waders, invertebrate biomass can occur below 10 cm, however is mostly inaccessible to most waders due to morphological, and behavioural traits (Sanders, 1999; Sanchez *et al*., 2006).

This was transferred to labelled plastic bags, which depicted the plot, subplot and zone sampled, e.g. B1-A-Mud (B1 plot, subplot A, sampled in the Mud zone). These samples were then refrigerated and processed within 48 hours.

Bagged samples were transferred to a plastic sampling tray and live invertebrates were retrieved using fine forceps. Samples were preserved in 90% alcohol and identified using a Leica Wild M8 microscope. After sorting and identifying to genus or taxonomic level, samples were stored in 90% ethanol in labelled eppendorfs.

The invertebrates collected in the substratum and mud samples were combined to one representative subplot sample and then sorted into five categories worms, snails, larvae of aquatic insects, aquatic insects and terrestrial insects. The invertebrates were dried in an oven over night at 50 degrees Celsius. The dried invertebrates were then weighed on a three decimal point scale and mass and number were recorded.

2.2.2 Trial sampling

The extraction method was chosen after a pilot sampling session. The pilot sampling involved removing 12 soil samples from a selected area and applying two different invertebrate extraction techniques to the samples - swirl and sieve methods. For the „swirl" method two cored soil samples per zone (sub or mud) of a sub-station were placed in a 20 l bucket filled with approximately 10 l water. The water and soil were swirled by hand to break up the core soil samples, and suspend the soil and invertebrates as well as organic matter in the water. The suspended particles and water were passed through a 1 mm diameter mesh net, and the detritus and invertebrates that were trapped decanted into a plastic container containing 100 ml of water. The sieve method involved placing the cored and measured soil samples directly into the net over a 20 l bucket, and running water through the net until only invertebrates and larger detritus remained. A two sample t-test assuming unequal variance showed no significant difference ($t = 0.89$, $p = 0.2$, $df = 9$) between total number of invertebrates in swirl ($\bar{x}$ 13.5, SD 7.12) and sieve ($\bar{x}$ 9.33, SD 9.0) sorting methods. The swirl method was far easier in practise and less time consuming compared to the sieve method, and was used for the rest of the invertebrate sampling

2.2.3 Environmental variables

2.2.3.1 Water levels

At the time of initial sampling (June 2014) a 60 cm wooden pole was driven into the soil at the exact point that the water level reached the bank in order record the water level mark. This stake was used as the comparison for all future water level measurements. In the monthly sampling that followed, the waters receding (negative values) or escalating levels (positive values) would be measured from the pole.

2.2.3.2 Vegetation Cover

Percentage vegetation cover was estimated visually from a 2 m^2 area at each subplot. The area used for vegetation cover was taken at the water mark, 1 m into the water, and 1 m onto the bank. The area was visually divided and estimated, and the total vegetation coverage per area was approximated and recorded. The total area in the 2 m^2 quadrant that was covered in vegetation was expressed as a percentage, 100% being 100% vegetation coverage of the whole block.

2.2.3.3 Mud/wet zone area

At each subplot, five measurements of the wet (seepage) zone were taken. The wet zone was considered to be the area of seepage of water into the littoral edge. The measurement was taken from the water"s edge and included the total wet area of the sand bank. In cases where this zone was not evident, removal of the top layer of sand usually revealed the seepage zone. The core soil samples were all taken from this measured area, as well as from an area 1 m from the shoreline, into the submerged zone. In the cases were no wet area was measured, the mud soil samples were taken right at the water edge, at the start of the littoral zone.

2.2.3.4 Wind speed and direction

Hourly readings of wind speed and direction were obtained from the South African Weather Service for the period July 2014 to June 2015, and means obtained for the day on which bird counts were taken.

2.2.4 Bird counts

The birds included in this study were all small to medium sized waders (Table 2.2), and the direct-count method was used. Bird counts were executed a week before the environmental sampling, on the day of the environmental sampling, and a week after the environmental sampling. All counts were conducted in daylight hours before noon (between 08:00 and 11:59) using a pair of 40 x 12 magnification Nikon binoculars. Counts were conducted from a vehicle, where all surface area and most edges are visible, following the same route around the pond. All wader species and their abundances were recorded per plot. Birds" behaviour was categorized as either foraging or roosting, and only foraging counts were used in analyses. It was possible to identify each bird accurately to species as a relatively low number of species occur at the site. Only birds which were found within 1 m of the water"s edge, in the benthos sampling area were counted. Waders that foraged deeper were excluded from the count. In the drier times of year central islands developed, and birds in these areas were also excluded.

Table 2.2 Migrant and residential wader composition at Strandfontein, South Africa derived from the Coordinated Water Avian Count data, surveyed by the Cape Bird Club.

Scientific Name	Common Name	Resident / Migrant
Actitis hypoleucos	Common Sandpiper	Migrant
Arenaria interpres	Ruddy Turnstone	Migrant
Calidris alba	Sanderling	Migrant
Calidris canutus	Red Knot	Migrant
Calidris ferruginea	Curlew Sandpiper	Migrant
Calidris melanotos	Pectoral Sandpiper	Migrant
Calidris minuta	Little Stint	Migrant
Charadrius hiaticula	Ringed Plover	Migrant
Limosa lapponica	Bartailed Godwit	Migrant
Limosa limosa	Black-tailed Godwit	Migrant
Numenius arquata	Eurasian Curlew	Migrant
Numenius phaeopus	Whimbrel	Migrant
Phalaropus fulicaria	Grey Phalarope	Migrant
Phalaropus lobatus	Red-necked Phalarope	Migrant
Philomachus pugnax	Ruff	Migrant
Pluvialis squatarola	Grey Plover	Migrant
Tringa glareola	Wood Sandpiper	Migrant
Tringa nebularia	Common Greenshank	Migrant
Tringa stagnatilis	Marsh Sandpiper	Migrant
Xenus cinereus	Terek Sandpiper	Migrant
Burhinus capensis	Spotted Thick knee	Resident
Burhinus vermiculatus	Water Thick knee	Resident
Charadrius marginatus	White-fronted Plover	Resident
Charadrius pallidus	Chestnut-banded Plover	Resident
Charadrius pecuarius	Kittlitz's Plover	Resident
Charadrius tricollaris	Three-banded Plover	Resident
Gallinago nigripennis	African Snipe	Resident
Haematopus moquini	African Black Oystercatcher	Resident
Himantopus himantopus	Black Winged Stilt	Resident
Recurvirostra avosetta	Pied Avocet	Resident
Rostratula benghalensis	Greater Painted-snipe	Resident
Vanellus armatus	Blacksmith Lapwing	Resident
Vanellus coronatus	Crowned Lapwing	Resident

2.2.5 Coordinated Water Avian Counts (CWAC)

The method used in CWAC is very similar to the bird count method described above, and is carried out once monthly by members of the Cape Bird Club (CBC). All counts were conducted in daylight hours, and counts start between 08:00 and 08:30, depending on season. The entire reserve („Strandfontein") is divided into four segments, north, south, east and west, and four teams cover these simultaneously. This way, each segment is counted at the same time, and overlap or double counting is limited. Counts were conducted from a vehicle, and binoculars or a scope are used to view bird species. Observational counts of abundance and richness of all waterbird species in the vicinity of the Strandfontein, within each segment are recorded and the data submitted to the University of Cape Town"s (UCT) Animal Demography Unit (ADU) and CBC. The CWAC at Strandfontein is one of the longest running bird monitoring surveys in the Western Cape. The counts started in July 1983 and all data is presently available from the ADU, however the original raw count data for all waterbird species was obtained from the CBC. CWAC is a good indicator for species that show seasonal variation in abundance (Thomas *et al*., 2015).

CHAPTER 3. HISTORICAL TRENDS OF WADER DIVERSITY & ABUNDANCE AT STRANDFONTEIN

3.1 Introduction

3.1.1 Global decline in wader abundance

Globally, wader populations are generally considered to be declining (Wetlands International, 2002; Wetlands International, 2006; Zöckler *et al.,* 2003; Birdlife International, 2008; Wetlands International, 2012; Zöckler, 2012; Risley *et al*.,2013; Convention of Migratory Species, 2014; Simmons *et al*., 2015; van Rommen *et al*., 2015).

While cyclical fluctuations occur in all species; the decline of migratory wader numbers is more extreme, and greater than general population fluctuations, which tend to recover over a certain time period (Birdlife International, 2008). Over a 30 year study period (1970 to 2000), of 119 Afro-Palearctic migrant species, 40 (33%) declined significantly during the period 1970–1990, and between 1990–2000, 20 (17%) migrant species showed significant declines in number (Sanderson *et al*., 2006; Birdlife, 2008) More recent trend data of coastal waterbird populations of the East Atlantic Flyway, showed that of the 47 coastal water birds assessed, 14 were waders which occurred at Strandfontein. For these 14 species, 57% were in decline and 43% stable or increasing between 1980 and 2015 (van Rommen *et al*., 2015).

The Wetlands International Waterbird Population Estimates (2002) indicated a decline on a global level of 44% of the known population trends for waders (Delany *et al*, 2003; Gosbell & Clemens, 2006). This estimate is similar to the evaluation of migratory wader populations and their status in Africa and Western Eurasia by Wetlands International, between the 1980s and 1990s (Stroud *et al*., 2008; Delany *et al*, 2009). Of the 115 populations of African and Western Eurasia migratory species (49), reliable estimates indicated a decline or possible decline in 45% of the populations (Delany *et al*., 2003). However, such declines are not always detectable on a local scale. Data from 40 years of autumn migration (1964 - 2003) of 17 wader species passing Blåvandshuk in Western Denmark showed that only one species exhibited a negative population trend, with the remaining 16 Palearctic wader species populations on the East Atlantic flyway (EAF) showing fluctuating or increasing trends (Meltofte *et al.,* 2006).

Analyses of population trends for waders require consistent, high quality, long-term data sets, which are not always readily available. The majority of South African waders are migratory; with numbers fluctuating spatially and temporally, in response to dynamic environmental conditions. Migrants are vagile, and analyses of single sites will not provide reliable trends at the population or species level, globally. This (coupled with the fact that international and local factors contribute to the fluctuations in abundance and populations) poses a challenge to determine the actual extent of decline. Fluctuations in species number and population trends can be attributed to a manifold of global and local factors including land use change (e.g. habitat loss and transformation), illegal hunting, recreational disturbance, climate change, natural system fluctuations (local seasonal movement), and prey interactions (Blomqvist *et al*., 2002; Zöckler *et al*, 2003; Stroud, 2004; Burton *et al*., 2006; Sanderson *et al*., 2006; Rakhimberdiev *et al*., 2011; Aharon-Rotman *et al*., 2014; Convention of Migratory Species, 2014).

Apart from the impacts of habitat degradation and transformation, global changes in shore bird number so can also be attributed primarily to climate change and lemming cycles. Lemming abundance is regulated by predators and resource availability, and population trends show three year cycles. The alternative prey hypothesis suggests that arctic foxes and other lemming predators (birds of prey), will switch food source based on the avaliabilty of lemmings, shifting to wader or geese chicks and eggs when lemmings are infrequent, subsequently lowering wader breeding success (Underhill, 1987; Underhill *et al*., 1989; Blomqvist *et al*., 2002; Aharon-Rotman *et al*., 2014). However, recent research has argued that lemming cycles are „fading out" and that there was no strong correlation found between breeding success in Australia (using the East Asian-Australian Flyway) and predation (in the Arctic) (Aharon-Rotman *et al.*, 2014). Nevertheless, other studies have shown that lemming cyclicity coincides with wader population fluctuations (Summers, 1986; Underhill, 1987; Blomqvist *et al.*, 2002; Aharon-Rotman *et al.*, 2014), and it is suggested that this signal is transmitted to the southern hemisphere through the EAF which results in declines or fluctuations in wader numbers in the Southern Africa (Underhill, 1987; Blomqvist, *et al.*, 2002).

Climate change will impact wader populations globally, as it has been shown to alter timing and pattern of migration (predominately by altering the timing of physiological changes related to migration). The collative impacts of climate change, particularly sea level, will alter habitats, affect competition between species, and change the distribution and availability of water (Convention of Migratory Species, 2014) which would results in certain wader species could be at a risk of extinction if they are unable to adjust (Rehfisch & Crick, 2003). Currently there is evidence showing reduced survival rates in wader species that over-winter in Africa (Sanderson *et al.*, 2006), possibly brought about by exposed, desiccated habitats in the sub-Saharan, due to reduced rainfall. This could explain these observed depressed over-winter survival rates, which subsequently project a current negative trend for EAF migratory species (Birdlife, 2008; van Rommel *et al.*, 2015).

Less severe impacts on species abundance and richness of waders, in South Africa, can be attributed to hunting and human disturbance. The nature and magnitude of human disturbance on birds can be associated with energetic costs, suggesting that Palearctic waders will have a greater response to disturbance when habitat quality (food resource and high predation) is lower (Yasué, 2006). Long-term census data showed that human disturbance was implicated as a potential threat to declining local shorebird abundance on Plymouth beach, United Kingdom (Pfister *et al.*, 1992). Monitoring of coastal bird community structure in the Western Cape, South Africa, between 1980 and 2010, suggests marked decreases in wader numbers (greatest decrease in Scolopacidae and Charadriidae), where the coastal areas surveyed are significantly impacted by human disturbance (Ryan, 2013).

Habitat transformation and degradation (generating a lack of resources) could explain the demise in wader numbers globally and locally. In all continents, continuing loss and degradation of viable habitats have been widespread and serious threat to migratory birds as it has the potential to compromise the viability of wader populations and breeding success, and seems only likely to intensify as economic expansion negatively impacts the remaining wetlands (Convention of Migratory species, 2014). As resources are often the major driver of species distribution, a degraded site lacking resources will be abandoned by waders, who move seasonally and locally between wetlands.

On Robben Island, Western Cape, South Africa, Palearctic wader populations fluctuated considerably between the 1977 and 2001. While nine of the 14 waders monitored showed substantial declines, it is more likely exaggerated and can possibly be explained by seasonal movement (Underhill, 1987; Underhill *et al*., 2001). The remote island is conceivably only inhabited in certain years, when optimum resources in other sites have been over exploited (Underhill 1987; Underhill *et al*., 2001). This leads to the assumption that declines of larger populations of waders at more pronounced feeding sites in South Africa (viz. Langebaan Lagoon, Swartskop Estuary and the Berg River Estuary) may be caused by the redistribution of populations into smaller flocks, which dispersed over larger areas and at remote satellite sites (viz. Robben island and Strandfontein) (Underhill *et al*., 2001; Zöckler *et al*., 2003).

It remains a challenge to address the causes of declines in wader population numbers, which could be explained by international and local influences often interacting (Sanderson *et al*., 2006; Thomas *et al*., 2015). Wader distribution is highly dynamic, and varies spatially and temporally. When combined with the migratory component of their ecology (geographical coverage, change of preferred sites), the collection of long term, viable data records and projection of global trends becomes a formidable challenge for biologists.

3.1.2 Strandfontein, a reliable feeding site

Geographically, South Africa is the final destination for migrant waders which use the EAF. Migratory waders begin their journey in the Arctic tundra and drop off en route south at reliable feeding sites. Banc d‟Arguin, Mauritiana has had records of over 2.35 million waders, comprising 30% of all waders using that flyway. Other wader hotspots along the route include the Bijagos archipelago in Guinea-Bissau (ca. 700 000 waders), while closer to South Africa, Walvis Bay Ramsar Site and Sandwich Harbour in Namibia have had records of 150 000 and 99 000 individuals respectively (Delany *et al*., 2009). The waders that end up reaching South Africa aggregate at larger wetlands along the west coast viz. Langabaan Lagoon, Swartskop River Estuary and the Berg River Estuary (Summers *et al*., 1977; Velásquez *et al*., 1991; Delany *et al*., 2009).

The Strandfontein Section of the False Bay Nature Reserve (which is approximately 512 ha in size), is described as one the most important waterbird sites, in terms of bird abundance and energy consumption in the Western Cape, on par with substantially larger natural systems such as the Langebaan Lagoon (4500 ha) and Berg River Estuary (7300 ha) (Harebottle, 2012). Strandfontein is one of the southernmost sites of the EAF, and it is remarkable that the birds actually fly this far south considering the larger, more suitable sites en route and to the west.

A large volume of count data is available for Strandfontein, and the earliest recorded surveys date back to 1952 (Winterbottom, 1960). Bird monitoring programmes or „Cape Bird Club vlei counts‟, were originally set up in over 15 wetlands in the Western Cape by the Cape Bird Club (CBC) members. These counts, possibly the first recorded instance of volunteers coordinating a waterbird census (Harebottle, 2012), were initiated in an attempt to gain some understanding on the distribution and degree of movements of Palearctic migrants waders in the Western Cape (Winterbottom, 1960; Harebottle, 2012). After this, monthly counts at Strandfontein, Rondevlei and Rietvlei (1961 to 1967), detailed the abundance of bird species across the wetlands (Blaker & Winterbottom, 1968). From these surveys, Strandfontein was identified as an important breeding and foraging ground for resident and migrant waterbirds. Coordinated Water Avian Counts (CWAC), initiated in 1983, and are still carried out by members of the CBC today. The Strandfontein dataset in particular is one of the longest uninterrupted surveys at an individual site in Africa, and has been utilised by researchers to analyse long term population and breeding trends (Ashenkazi, 2001 Kalejta-Summers *et al*., 2001b).

With widespread global declines of migrant and resident waders, there is a greater need to maintain and develop reliable wader habitat in South Africa. This chapter analyses historic population count data of waders at Strandfontein to identify long-term trends in the small populations of resident and migrant waders.

3.2 Methods

3.2.1 CWAC Data

Raw census data for wader species from July 1983 to June 2014 was obtained from the CBC. Resident and migrant wader species monthly counts were extracted from the records. Overall 338 monthly counts were made by the CBC between July 1983 and June 2014 (34 months were thus not assessed by the CBC). A total of 184171 resident species were counted and 54621 migrants were counted during the duration of the 31 year period.

3.2.2 Historic trends, seasonal variation and change in abundance

Five time periods (1952-1958, 1962-1966, 1983-1993, 1993-2004, and 2004-2014) ranging between four and 11 years each, were utilised for the analysis of historic trends and seasonal variation at Strandfontein. For analyses a year was defined from July to June in the following year (the first CWAC count was initiated in July of 1983). For migrants data was only analysed for the non-breeding months (March to October), as opposed to July to June as for the residents. It was also useful to use this method of defining a year over two years as migrant waders generally arrive in the summer of one year (e.g. November 1983) and stay for a period of up to six months (not including over wintering juveniles) and migrate the following year (e.g. March 1984). By defining a year as July to June or October to March, the count data reflects actual numbers of birds that have migrated for a single summer season.

To analyse the historic trends at Strandfontein, available data from published journals was used in combination with CWAC data (Table 3.1). Median bird count was used for this analysis as the median is a more appropriate and robust measure of the „middle" of the data (central tendency) than the mean, when the data includes a number of outliers (waders occasionally arrive in very large flocks). The monthly median bird count data for the 1950s and 1960s were extracted directly from the journal articles of Winterbottom (1960) and Blaker & Winterbottom (1968). From 1983 to 2014, raw CWAC count data were used and divided into three, ten to 11 year time periods (namely 1983-1993, 1993-2004, and 2004-2014). Ten year periods were used as these were more likely to give better reflection of trends compared to shorter periods which would be heavily influenced by outlier years. Monthly median bird counts were calculated per species per year, and then averaged over the 10 – 11 year periods to represent each of the three time frames.

The data set used for the change in abundance graph for Strandfontein (July 1983 - June 2014) was also drawn from the original raw CWAC data. Monthly CWAC counts were carried

Table 3.1 The five time periods, range of years and data source of the counts used for the historic trend graph to analyse population trends of waders at Strandfontein between 1952 and 2014.

Time period	Years	Available data	Reference
1952 to 1958	6	Median	Winterbottom (1960)
1962 to 1966	4	Median	Blaker & Winterbottom (1968)
1983 to 1993	10	Count (converted to median)	Unpublished CWAC database
1993 to 2004	11	Count (converted to median)	Unpublished CWAC database
2004 to 2014	10	Count (converted to median)	Unpublished CWAC database

out by the CBC members on most months during this time period, a minority of surveys did not take place (due to poor viewing conditions, or lack of volunteers) thus there were occasionally less than 12 count totals per year (Harebottle, 2012).

At the species level, the four most abundant migrant wader species (Little Stint, Curlew Sandpiper, Ringed Plover and Ruff) found at Strandfontein during the 31 year CWAC data (based on annual totals) were analysed to review trends.

3.2.3 Monotonic regression

Monotonic regression (sometimes also known as isotonic regression) produces a step function through a scattergram of points. Here, the scattergram consisted of the annual counts of wader abundance data (see 3.2.2 above). Each step in the monotonic regression line is the average of the set of points which it covers. The monotonic regression of a set of points is the best line to fit the points in such a way that the value at each step is higher than the value before it. It is the best line that the sum of squares of the distances between the step function and the points is minimized. The calculations were performed using the "Monotonic" directive in Genstat (Release 17) (VSN International 2014). In this case what is needed is a downwards step function. To achieve this, each count was subtracted from the maximum count, the monotonic regression calculated and back-transformed. Monotonic regression is a critical component of non-metric multidimensional scaling, and was apparently first introduced by Kruskal (1964).

The monotonic regression was compared with an overall trend computed by linear regression. The monotonic regression tests the idea that changes in population size have been in steps rather than as a gradual trend through time. Approximate F-tests were performed based on the standard likelihood ratio principle. The test consists of considering the reduction in the residual sums of squares from fitting the models, generating an analysis of variance (ANOVA) table. F values were calculated as:

$$F = \frac{\left(\frac{RSS_1 - RSS_2}{p_2 - p_1}\right)}{\left(\frac{RSS_2}{n - p_2}\right)}$$

Where RSS stands for Residual Sums of Squares, n = 31 (number of years of data) and the P2 (Monotonic regression) and P1 (normal regression) are the numbers of parameters in the models.

3.2.4 Species richness and diversity

Species richness for migrant and resident waders (July 1983 - June 2014) was extracted from CWAC count data. The data was analysed using the software PRIMER 6, version 6.1.15 with PERMANOVA+ add-in, version 1.0.5 (Anderson *et al.*, 2008) to generate monthly and annual species richness scores as well as Simpson and Shannon-Wiener (log_e) diversity indices. Months having a species richness value of two or less were removed from the analyses as abundances were too low.

3.3 Results

3.3.1 Historic trends and seasonal variation

Although expected for migrants, seasonal variation in abundance was also evident in resident waders, peaking during the summer months when migrant wader numbers were also at their peak (Fig. 3.1) - this pattern in seasonal abundance was consistent across 62 years.

Although resident waders were present throughout the year, with abundances peaking between January and February, a small population typically remained during winter months. Migrants first appeared in substantial numbers in October and November, apart for the period 1962-1966 when numbers increased as early as August. Numbers of migrants peaked in mid- to-late summer, and May was consistently the departure month for all migrants across the 62 year period of review.

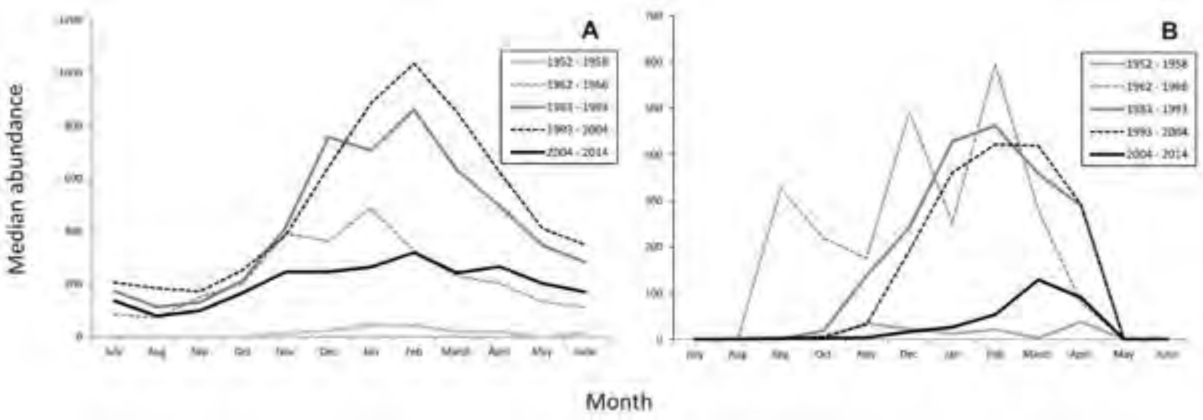

Fig. 3.1. Seasonal abundance of resident (A) and migrant (B) waders at Strandfontein during five bracketed time periods (1952-1958, 1962-1966, 1983-1993, 1993-2004, and 2004-2014). Median values given.

Both migrant and resident waders initially had their lowest abundances in the 1950s with progressive increases in numbers in the following decades, until the 1993-2004 period when numbers started dropping off. Resident waders showed a steady increase in abundance from 1955-2004, whilst migrant wader numbers were consistently high and stable between 1962 and 2004. After 2004, migrants showed a decrease from the consistently high levels observed between 1960 and 2004, and resident wader abundances decreased in the 2004-2014 periods after peaking in 1993-2004.

3.3.2 Change in abundance and monotonic regression

When analysed by year, numbers of residents fluctuated erratically between 1983 and 2014, with population numbers remaining relatively stable over the 31 year period of observation (Fig 3.2). Resident abundances peaked over the 2006/2007 period (1162 individuals), although numbers dropped back to original counts thereafter and continued to remain stable up to 2014. In contrast, migrant numbers declined progressively over the three decadal periods, with a sharp drop after 1999/2000. The summer of 2002 marked the most extreme decrease in migrant waders, with an 84% decrease in abundance from 2000/2001 to 2001/2002. The minimum mean abundance count over the whole time period however was in 2008/2009 (n =12.40). Numbers did not recover to original levels after this, and continued to remain relatively low, with a slight recovery after 2013 (a mean increase from 54 to 259).

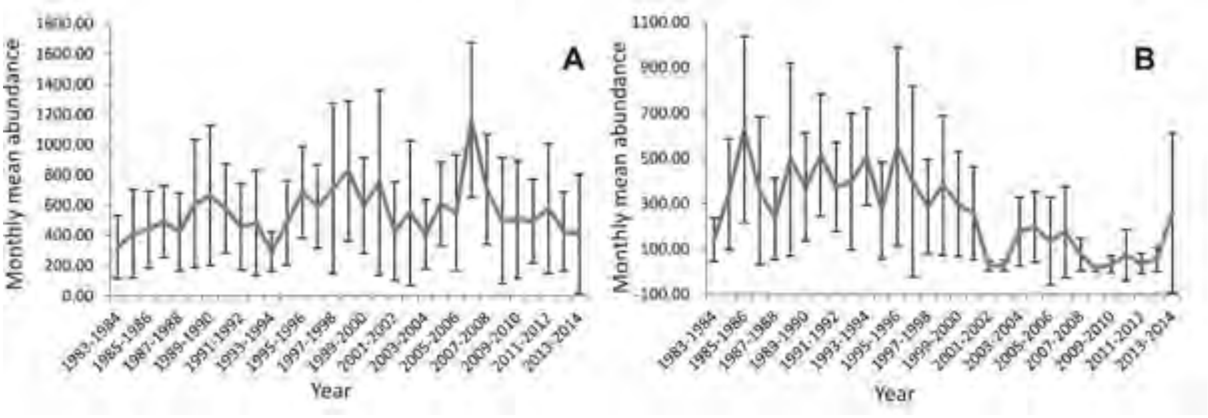

Fig. 3.2 Change in annual abundance of (A) resident waders for 12 months (July to June), and (B) migrant waders for 6 months (October to March) using CWAC count data from July 1983 - June 2014. Values represent means and standard deviations.

The monotonic regression showed a general increase in the abundance of resident waders in the last 31 years at Strandfontein, and a decrease in abundance of migrant waders.

For the migrants, total sums of squares was 926 676 with 30 degrees of freedom. The linear trend analysis estimated the decrease in abundance to be 13.35 birds per year (standard error 2.60, $t_{29} = 5.15$, $P<0.001$). This estimated decrease in migrant abundance over three decades was therefore estimated from this model to be ca. 400 birds. The residual sums of squared deviations was 484 457 with 29 degrees of freedom for this model. For the first 14 years, the monotonic regression estimated 398 birds (Fig. 3.3). The model then fitted decreases in five further steps, with the final step being 77 birds (Fig. 3.3). The model estimates that the decrease in abundance of migrants was 321 birds. This model has a total of six parameters (Fig. 3.3). The residual sum of squares for this model was 300 463 with 25 degrees of freedom, considerably smaller than the linear model. The approximate F-test based on the likelihood ratio principal has F distribution with $p2 - p1$ and $n - p2$ degrees of freedom, yields F = 3.83 with 4 and 25 degrees of freedom. This has a p-value of 0.0147, which suggests that the monotonic regression provides a better fit to the decrease of the migrants than the linear regression achieves.

For the residents, total sums of squares was 802 244 with 30 degrees of freedom. The linear trend analysis estimated the increase in abundance to be 3.52 birds per year (standard error 3.28, $t_{29}=1.07$, n.s.). The residual sums of squared deviations was 771 520 with 29 degrees of freedom for this model. In the monotonic regression, the first step was estimated to 322 birds (Fig 3.3). The model fitted increases in five further steps, with the final step being 592 birds for the last 19 years (Fig 3.3). The model estimates that the increase in abundance of residents was 270 birds. This model also has a total of six parameters (Fig 3.3). The residual sum of squares for this model was 658 165 with 25 degrees of freedom, a small decrease compared with the linear model. The approximate F-test yields F = 1.08 with 4 and 25 degrees of freedom which is not significant. Neither the linear regression nor the monotonic regression provides a significant fit to the pattern of increase of the residents. The inference is that the number of resident waders has remained stable during the study period.

The monotonic regression identified a few small step increases in abundance of residents in 1986/1987 and 1994/1995 periods, and a stable pattern thereafter. In contrast, the monotonic regression for migrant abundance showed stepped decreases starting at the 1997/1998 period, with another major dip in 2001/2002 and after 2008, low and stable numbers were evident.

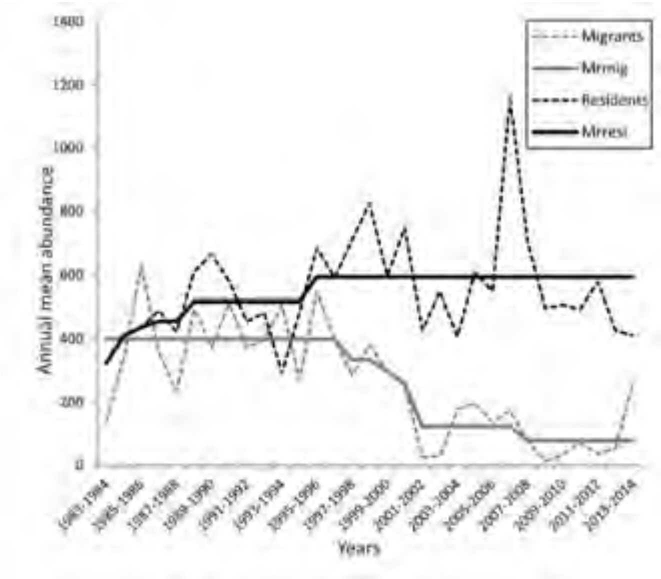

Fig. 3.3. Monotonic regression for monthly mean change in abundance of migrants (For October to March - non breeding months) and resident waders (June to July) at Strandfontein between 1983 and 2014. Mrmig - monotonic regression for migrants, Mrres - monotonic regression for residents.

On a species level, of the four most abundant species, only Curlew Sandpiper (*Calidris ferruginea*) (Fig. 3.4 B) numbers decreased drastically over the 31 year set data. Little Stint (*Calidris minuta*), Ringed Plover and Ruff show fluctuating trends over the time period, with Ringed plover numbers possibly declining. There appears to be decreases in all four species around 2001/2002 from previous years, followed by two peaked increases in mean numbers until 2008/2009. This was followed a drop and a recovery between 2012 and 2014. A similar trend was seen in the historic trend data (Fig 3.1) and change in abundance data (Figs. 3.2, 3.3) for migrant species.

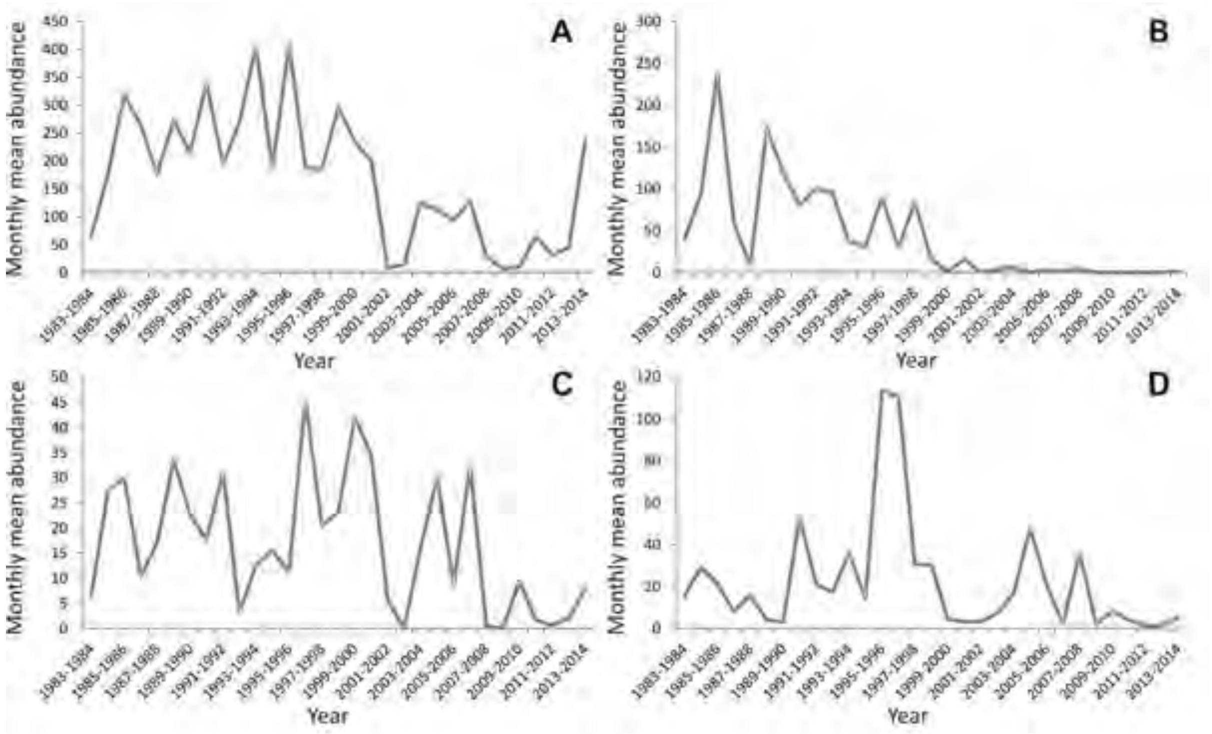

Fig. 3.4. Monthly non-breeding (October to March) mean trends for the most abundant migrant wader species at Strandfontein Little Stint (A), Curlew Sandpiper (B), Ringed Plover (C) and Ruff (D), data from CWAC counts, 1983 to 2014.

The historical seasonal trend results, analysed by monthly medians (Fig 3.1) and annual mean abundances (Fig 3.2, Fig 3.3, Fig 3.4 & Fig 3.5), showed similar trends overall, with migrant wader abundance decreasing, and resident wader abundance increasing between 1983 and 2014. The major difference in the two analyses was that of resident abundances in the last decade (2004-2014). The drop in monthly median abundance of residents from 2004-2014 is not reflected in annual mean data for the same period, which showed a marked increase in abundance.

The four most abundant migrant waders (Fig 3.4) made up a fair proportion of migrant, and to a lesser extent, all wader numbers. Little Stint made up 65.9% of migrant numbers, and 15% of all waders between 1983 and 2014, followed by Curlew Sandpiper (16.1%, 3.7%), Ruff (7.7%, 1.7%) and Ringed Plover (6.1%, 1.4%).

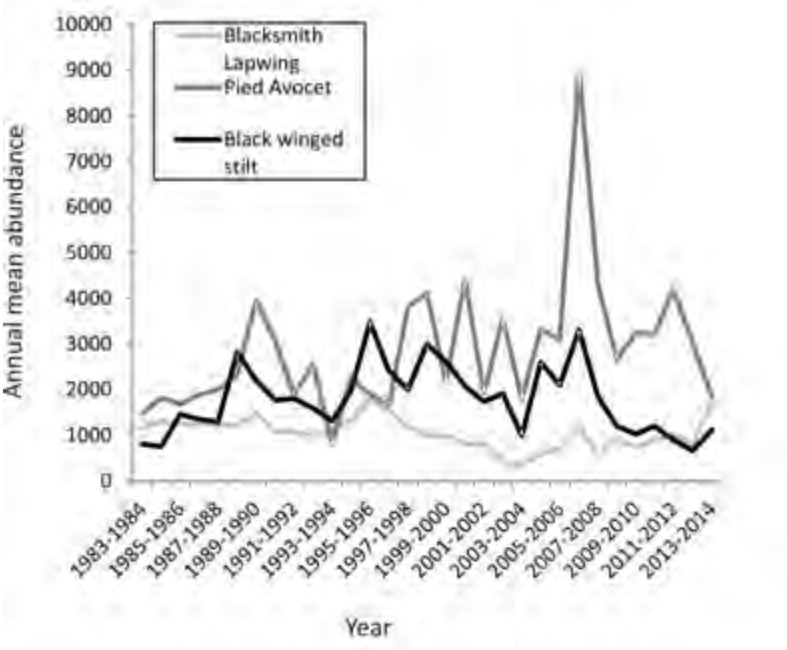

Fig. 3.5. Annual mean abundance of the three most common resident wader species (Pied Avocet, Blacksmith Plover and Black-winged Stilt) at Strandfontein between 1983 and 2014

The three most abundant resident wader species between 1983 and 2014 (Fig 3.5) were: Pied Avocet (48.1% of resident species, 37.2% of all wader species), followed by Blacksmith Lapwing 9 17.7%, 13.7%) and Black-winged Stilt (29.9%, 23.1%).

3.3.3 Species richness and diversity indices

A total of 33 species of waders belonging to six families were recorded during the 33 year CWAC census period. Of these, 20 species were migrants and 13 were residents. The migrant wader community typically had higher species richness than the resident wader community; however in some years (1987/1998, 1993 to 1996 and 2009/2010) the number of resident species exceeded that of migrants (Fig. 3.6). There was considerable fluctuation in species richness annually for waders; however an overall decline in richness at Strandfontein was evident, particularly after 2003/2004 when the recovery of both migrant and resident richness did not reach the upper peaks of the previous period. Species richness trends could be loosely associated with the historic seasonal variation abundance data for migrants, as a decline in wader abundance was noted in the most recent decade (2004-2014) in both datasets (however not evident for residents).

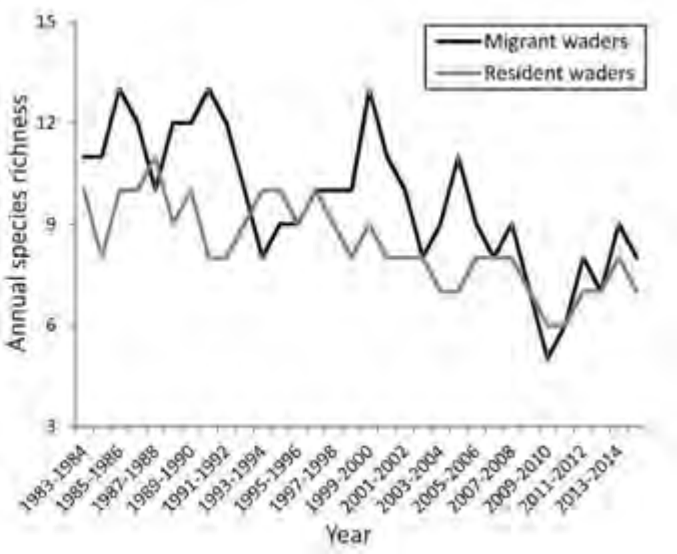

Fig 3.6. Species richness of resident and migrant waders at Strandfontein between July 1983 and June 2014.

Diversity indices results show a similar, but more gradual negative trend over the 31 year period for residents and migrants, however both Shannon-Weiner and Simpson"s values for resident species were higher than those of migrants (Fig. 3.7). Shannon-Weiner diversity indices results for residents and migrants showed similar fluctuations to Simpson diversity indices, and the same was evident for migrants.

Simpson diversity indices fluctuated between 0.7 and 0.4 for residents, except for 1999-2000 where the value dropped to 0.36 (the lowest value). A similar trend was found for Shannon-Weiner diversity indices for residents, with small fluctuations varying between 1.4 and 0.6. Annual fluctuations in diversity were also apparent for Palearctic migrants, but with greater variation (Simpson"s index varied between 0.3 and 0.7, and Shannon-Weiner between 0.5 and 1.2).

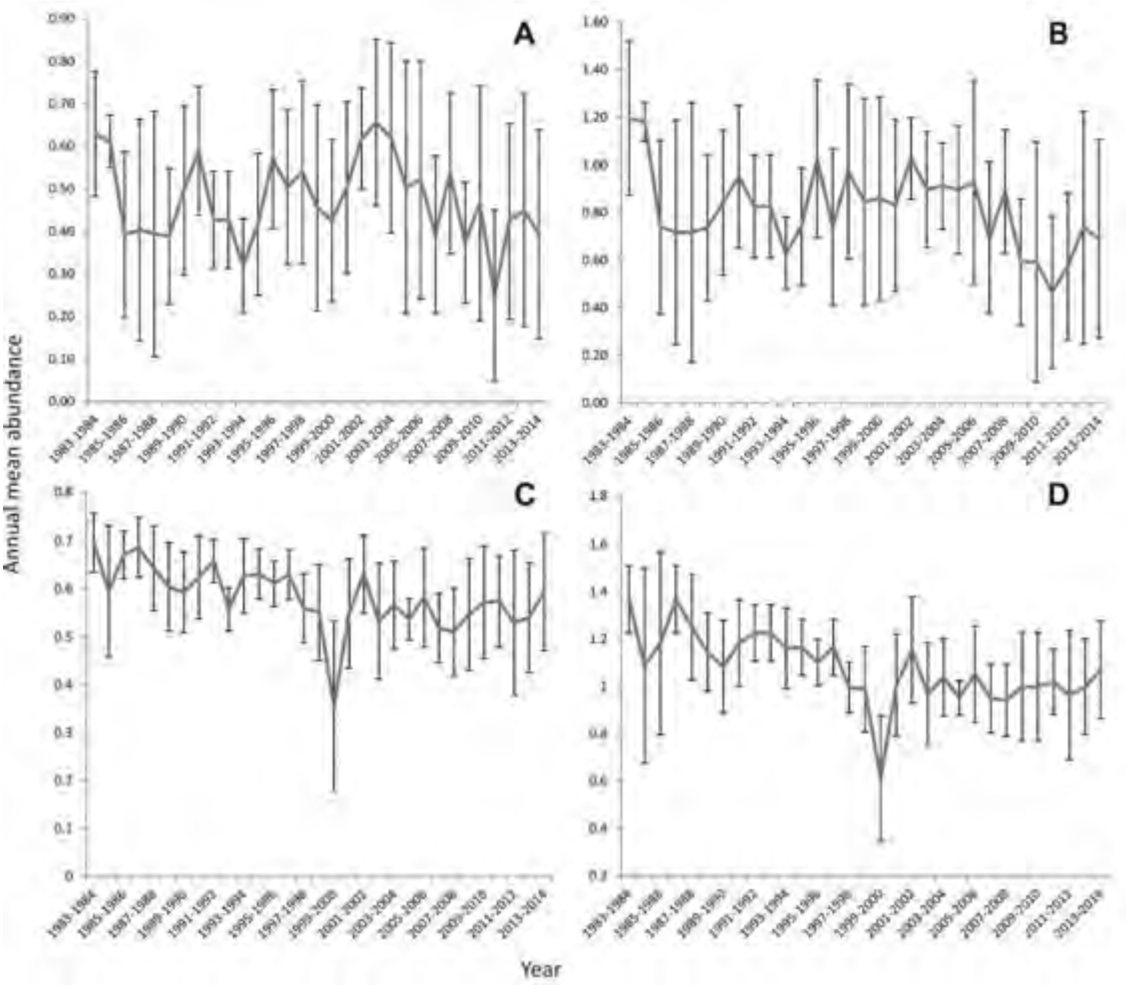

Fig. 3.7. Diversity Index scores for wader mean annual abundance at Strandfontein between July 1983 and June 2015. Migrant Simpson"s diversity indices (A), Shannon-Weiner diversity indices (B,), Resident wader Simpson diversity indices (C), and Shannon-Weiner diversity indices (D).

Findings suggest that the Shannon-Weiner diversity indices for migrant and resident waders were low overall for the 31 year period. The Simpsons score however, particularly in residents was quite high and relatively stable over the years compared to migrants which oscillate between years. The values were largely different, however overall trends showed similar results.

Of the 20 Palearctic migrants, 36% (n=12) had very low frequencies of occurrence (>1 individual), on average per annum (Table 3.2). Three of the species are listed as globally and regionally threatened and six are described as uncommon, rare or vagrant (Maclean, 2001). Chestnut-banded Plover and Greater Painted-snipe had only one individual recorded per species over the 31 year monitoring period (Table 3.2).

Table 3.2. Wader species with an annual mean of less than one individual present per year between 1983 and 2014 at Strandfontein.

Scientific Name	**Common Name**	**Total count 1983 - 2014**	**$\bar{x}$ birds/ yr**	**South African status (Maclean, 2001)**
Arenaria interpres	Ruddy Turnstone	23.00	0.70	Common migrant
Calidris canutus	Red Knot	5.00	0.15	Uncommon migrant
Calidris melanotos	Pectoral Sandpiper	2.00	0.06	Vagrant
Charadrius pallidus	Chestnut-banded Plover	1.00	0.03	Common resident
Limosa limosa	Black-tailed Godwit	3.00	0.09	Very rare vagrant
Numenius arquata	Eurasian Curlew	5.00	0.15	Uncommon migrant
Numenius phaeopus	Whimbrel	15.00	0.45	Common migrant
Phalaropus fulicaria	Grey Phalarope	12.00	0.36	Rare migrant
Phalaropus lobatus	Red-necked Phalarope	20.00	0.61	Rare migrant
Rostratula benghalensis	Greater Painted-snipe	1.00	0.03	Common migrant
Vanellus coronatus	Crowned Lapwing	13.00	0.39	Common resident
Xenus cinereus	Terek Sandpiper	2.00	0.06	Uncommon migrant

3.4 Discussion

3.4.1 Seasonal variation

At Strandfontein, waders showed similar seasonal variation patterns across all bracketed time periods with increased abundances in summer, where mean summer abundance exceeded mean winter abundance. The summer abundance peak was driven by the influx of migratory waders, which coincided with the arrival of Palearctic migrant waders in South Africa (Taylor *et al.*, 1999; Harebottle *et al.*, 2008). Arrival month of migrants varied slightly over all 10 year time intervals. In 1960s, 1983-1994 and 2004-2014, September was main arrival date. In the 1950s arrival time was late in November, and in 1994-2004 the migrants arrived earlier, in August. Similar variability was recorded in other local wetlands in South

Africa (Martin & Baird, 1987; Spearpoint *et al*., 1988; Harrison *et al*., 1997; Kalejta-Summers *et al*., 2001a; Kalejta-Summers *et al*., 2001b; Harebottle, 2012).

The majority of Western Cape wetlands showed increases in resident waders in winter months. CWAC between 1992 and 1997 at 34 Western Cape wetlands revealed that 67% (n = 23) have at least one species of resident wader which has a greater winter abundance compared to summer (viz. Berg River Estuary and De Mond) (Taylor *et al*., 1999). Between 1975 and 1986, median winter abundance at Langebaan Lagoon of resident waders exceeded the summer median in eight of the nine (88%) species recorded (Underhill, 1987). Numbers of resident waders wintering at the Berg River Estuary (1987 to 1989) were 23% higher than mean abundance during summer, with peak numbers in winter (except for Pied Avocets) (Velásquez *et al*., 1991). This could be due to juvenile recruitment and movement between favourable sites in winter (Harebottle *et al*., 2008). However, resident wader abundance at Strandfontein peaked in summer, and was lowest in winter for all species. Similar findings have been recorded at smaller satellite sites (Klienmond River Estuary, Rappenberg Bird Sanctuary, Rondevlei Section of the False Bay Nature Reserve, and Theewaterskloof Dam) (Taylor *et al*., 1999). In winter the pans at Strandfontein become inundated and there is an absence of exposed unvegetated littoral edge. Residents presumably have to move to other sites with suitable habitat and low winter counts are expected for the site (Kalejta-Summers *et al*., 2001b).

Peak abundance for migrant and resident waders varied between January and February in Strandfontein within the 62 year period, matching the pattern at Cape Recife, Algoa Bay, Eastern Cape, South Africa (Spearpoint *et al*., 1988). Generally wader abundance varies between months for different wetlands and peak abundance presumably coincides with the most favourable period (resources, habitat conditions) at sites. Resident shorebird numbers peak in June and migrants in February at Paarl Waste Water Treatment Plant (Harebottle *et al*., 2008). An analysis of seasonal trends in abundance of waders at Rietvlei Nature

Reserve, Western Cape, South Africa, showed that wader abundances peaked in November and December, and then declined rapidly. This decline coincides with the desiccation of the pans which results in minimal to no water for this period, generating a shrunken littoral feeding zone, forcing birds to find alternative sustainable habitats (Kalejta-Summers *et al*., 2001a). At the Berg River Estuary, Palearctic migrants peak in February and April which could suggest that migrants which have moved to the extreme southern part of their migratory range, revisit the site before departure to accumulate pre-migratory fat reserves (Velásquez *et al*., 1991). A decline in abundance of Palearctic migrants in January or December coincides with blooms of *Cladophora*, an algae which restricts access to food on intertidal mudflats (Kalejta & Hockey, 1991), and thus local movement is a response to temporarily unfavourable feeding conditions (Velásquez *et al*., 1991).

Peak abundance months are generally followed by departure of migrants and residents at Strandfontein. Departure months generally coincide with the peak mass values of migrant waders in late April, early May, which drives the start of migration parting of most wader species viz. Ruddy Turnstone (*Arenaria interpres*), Red Knot, Little Stint and Curlew Sandpiper (Summers & Waltner, 1979).

3.4.2 Historic trends in abundance

Overall, wader abundance increased somewhat between 1952 and 2004; and both migrants and residents exhibited declines in abundance from 2004 - 2014 for the long term median data analyses. Fluctuations in bird abundance at Strandfontein between the 1950s and 1980s could be linked to transformation of the WWTW. A substantial increase in wader abundance was evident between the 1950s and 1960s, and this can be explained by the increase in the water surface area. Between 1951 and 1969 the number of pans quadrupled from seven to 28, and the water surface area had an 18-fold increase (from 12.9 ha coverage to 223.1 ha) (Winterbottom 1960; Ashenkazi, 2001; Kalejta-Summers *et al*., 2001b). The increase in open water area is associated with a steep increase in wader species richness and abundance (Colwell & Taft, 2000; Whitfield *et al*., 2008). Additionally, in the 1960s the retention pans were drained and sludge was left to dry out in summer. Dried sludge pellets were either burned or removed as part of the treating process (Kalejta-Summers *et al*., 2001b). Draining the water would create suitable wader habit and this could explain the high abundance of migratory and residential waders at Strandfontein between 1952 and 1958, as a larger littoral edge was exposed. Kalejta-Summers (2001b) found similar increases in abundances at Strandfontein over the same period.

Large scale construction of the wastewater treatment works at Strandfontein was completed in 1979, resulting in 34 artificially created pans, with a water surface area of 306ha (Kalejta-Summers, 2001b). High resident abundance during this period suggests that the habitat at Strandfontein during this period was relatively stable, and conditions were favourable for waders in general.

The greatest change to the WWTW took place in 1980, when an activated sludge plant initiated operating (Ashenkazi, 2001). This potentially lowered the nutrient content of the ponds, as the plant dispatched „purified" water to the wetland after initially chemical treatment (Kalejta-Summers *et al*., 2001b). This process likely lowered the nutrient content of the water entering the pans. However, this did not appear to have a negative effect on wader abundance, as resident abundance climbed, and migratory wader abundance showed only slight decreases between 1962 and 2004. Migrant wader abundance in other wetlands however, appeared to have major declines during this period. Five of seven Palearctic species studied (Curlew Sandpiper, Grey Plover (Pluvialis squatarola), Sanderling (*Calidris alba*), Red Knot, and Ruddy Turnstone) showed declines in abundance at Langebaan Lagoon between 1975 and 2003 (Harebottle *et al*., 2006; Simmons *et al*., 2015). As resident waders were still plentiful during this period at Strandfontein, and migrant abundances fluctuating or declining in some instances, international factors could potentially be the cause of the reductions in abundances (Harebottle & Underhill, 2006).

The most significant decline at Strandfontein was between the periods 1993-2004 and 2004-2014. Migratory Palearctic and resident waders both expressed substantial declines in the last decade of the historic count data. Ryan (2013) also found that between 1980 and 2010, wader numbers along the False Bay coast, West Coast and South Coast were declining. The four most abundant waders in the 1980‟s declined by up to 90% (Ryan, 2013).

As both wader groups exhibited population declines as per the historic analyses, localised environmental factors, particularly a decline in the habitat quality at Strandfontein could potentially play a role. The encroachment of reeds (*Typha capensis*) colonising the littoral edges may have resulted in minimal feeding zone for waders (Refer to Chapter 4). In addition, increased urbanisation would result in larger quantities of waste that needs to be treated. Thus larger volumes of water are needed to treat the effluent, and increased levels of water are pumped out from the plant. Therefore even in summer the water levels in the pans remain high. The systematic weir system at Strandfontein is highly volatile. Since the remodelling of the pans in the 1960s and 1970s, maintenance has been minimal to non-existent, and this probably resulted in damaged channels and pipes being damaged or blocked with sludge and silt, rendered the water system flow ineffective in part, this could possibly have altercations on the nutrient value, filtration process, quality of the water and water levels.

Seasonal variation patterns remained the same over bracketed time periods, wader arrival and departure remained constant over the months, however, the populations of migrants and resident‟s numbers (overall abundance) showed negative trends.

3.4.3 Identifying tipping points in changes in abundances

Fluctuations in annual wader abundance at Strandfontein could theoretically be linked to changes on a local (habitat change, environmental parameters) and global scale (demise of wader populations worldwide, as a direct result of breeding success in Tundra or habitat loss).

A steady decline in migrant wader abundance was noted over the 31 year period for which CWAC data was available (Fig 3.2 & Fig 3.3). This coincided with the historic trends when analysed for bracketed periods across seasons (Fig. 3.1) and with the specific species trends (Fig 3.4).

Reductions in abundance were noted in 1996/1997, and 2001/2002, and the highest mean abundance in 1985/1986. Similar increases (particularly of Curlew Sandpiper) were recorded at Langebaan Lagoon for this period, and this coincides with an increase in lemming abundance in the Arctic Tundra at the same time. Lemming (*Dicrostonyx torquutus* and *Lemmu sibiricus*) numbers were high in the Taimyr Peninsula in 1985; therefore Arctic Foxes (*Afopex fagopus*) and other higher trophic level predators are hypothesized to have avoided ground nesting birds and their eggs, and preyed on abundant lemming populations instead (Underhill, 1987). This could potentially have resulted in above average breeding success for Curlew Sandpipers, with increased numbers the following year - as detected at Strandfontein in 1986 where highest count of Curlew Sandpiper individuals were recorded (Fig 3.4 B).

The monotonic regression identified 1997/1998 as the major tipping point for the downward trend in abundance. The summer of 1997 marked the period when Palearctic migrant abundance declined at Strandfontein, and continued to do so throughout the 2000‟s and 2010‟s). Ruff (*Philomachus pugnax*) mean abundance declined at Strandfontein by 72% from 1996/1997 to 1997/1998. Little Stint mean abundance declined by 54% in 1996/1997, further supporting the downward trend.

The CWAC report for 1992-1997 indicates that Curlew Sandpiper and Little Stint both had declines in number (from previous years) at numerous wetlands, including Langebaan Lagoon and Lake St Lucia (Taylor *et al*., 1999). Both species are common, widespread and highly gregarious, usually foraging in large flocks, which bulk up numbers of migrants at Strandfontein (Maclean, 2001).

The summer of 2001/2002 marked the most extreme reduction in migrant mean abundance at Strandfontein, and the population did not recover after this (see tipping point in monotonic regression, Fig. 3.3). The overall negative trend in abundance of migrant waders for the 31 year study period is comparable to international declines of migratory waders using the EAF (Wetlands International, 2006). At a global level, the Fourth Edition of the Waterbird Population Estimates (WPE) for Wetlands International (WI) indicated that, where trends in waterbird have been studied from the six Ramsar regions – 41% of species were declining, 36% stable, and only 16% increasing in 2006 (Wetlands International, 2006). Similar trends are evident for waders. The Third edition of WPE shows a 51% decline for the family Charadriidae, and 49% decreasing trend for populations of Scolopacidae. These families are comprised of a high percentage of Palearctic migrants and show similar declining trends (Zöckler *et al.,* 2003). In 4th edition (3 years later) declines of 51 % (Charadriidae) and 54% Scolopacidae) indicate ongoing declines. Overall the number of decreasing populations exceeds increasing populations by 2.5 in Africa. Africa is the second highest region where the greatest declines have occurred (Wetlands International, 2001; Wetlands International 2006). At an international level, a 31 year analysis of the most abundant waders at the Tagus estuary, Portugal, indicated strong declines for Common Redshank (*Tringa totanus*), Dunlin (*Calidris alpina*), and Grey Plover (Catry et al., 2011).

However, the recovery in numbers of migrant waders numbers (an increase of 87%) in the last two years (2012 - 2014) at Strandfontein could be a result of increased management interventions directed towards waders. Water levels in three pans were manipulated during this period using the canal system. Water was stopped from entering the pan using wooden sluice boards at the main inlet in October, and evaporation reduced the water level to a very shallow layer (ca. 10 cm), maintained by occasional opening of the sluice gates. *Typha capensis* beds were mechanically removed from littoral zones of two pans which were water - controlled for waders (P2, 2013; P1, 2014). Increase in Palearctic migrant waders abundance may be related to this intervention for habitat quality at Strandfontein.

Resident wader numbers fluctuated between July 1983 and June 2014, but overall an increase in abundance was noted over the 31 year period. Tipping points where significant increases in abundance occurred were the periods 1987/1998 and 1995/1996. In contrast, when median seasonality data was used (Fig 3.1), the same period reflected a gradual decline in abundance. As the raw count data for this period was dominated by zeros, this influenced the value of the median.

Velásquez *et al* (1991) suggested that seasonal and annual fluctuations in resident abundance could be attributed to unsuccessful breeding years. This would result in localised movement between inland sites to the coast pre- or post-breeding (Winterbottom, 1960). This probably explains the influx of resident numbers to Strandfontein for 2006/2007. Habitat change or poor recruitment could potentially be the cause of the major declines in resident abundance in 1993/1994, however in the following years a stark increase point in the monotonic regression is noted for 1995/1996.

Many wetlands are interconnected, and birds will move seasonally within a network of wetlands to sites offering a high resource output. Pan profile, depth and water level fluctuations influence macrophyte abundance (hampering wader foraging) and invertebrate abundance. The interaction between these variables ultimately determines waterbird communities (see Chapter 4).

Such local movements are typical of most wader populations. European Oystercatcher (*Haematopus ostralegus*) abundance at the Banc d‟ Arguin (Mauritania), drastically decreased by 48% from 1980 to 1997. The decline in availability of the preferred food type (Giant cockle, *Anadara granosa*) explained the reduction in Oystercatcher numbers at this site and a compensatory increase in numbers at the neighbouring Bissa site (Zwarts *et al*., 1998). This emphasizes the need for multiple wetlands and integrated wetland networks for local seasonal movement. Post breeding movements of American Avocet (*Recurvirostra americana*) were monitored using radio telemetry for 185 breeding birds fitted with radio transmitters at five wetland sites in West America. Results from data analyses showed that 40 radio-marked Avocets moved 200 km between wetlands preceding migration, further highlighting the importance of local movement of individuals among habitat patches (Plissner *et al*., 2000).

A steady increase in resident wader abundance was apparent over the 31 year period for which CWAC data was available. Data presented for waders in The Third and Fourth edition WPE (2001 & 2006 respectively) showed that overall the Haematopodidae and Recurvirostridae (which include a high percentage of non-migratory species) are typically stable or increasing (Wetlands International, 2001; Wetlands International 2006; Zöckler *et al.,* 2003). In contrast, the trends for migrant species show declining numbers over the same period. Comparable patterns in African wetlands suggest that migrant wader abundance is decreasing in line with the global declines and detectable at their most southerly range, factors in their breeding range are probably underlying the declines (Simmons *et al*, 2015). However, migration back to the breeding grounds is an arduous and complex event, and loss of suitable habitats in the southern part of the range might exacerbate population declines of migrants.

3.4.4 Species richness and diversity indices

Species richness and diversity indices both show an overall negative trend between 1983 and 2014. Typically migrants have higher species richness than residents (20 of 33 wader species recorded are migrants), however these species are only present for a maximum of six months of the year, and thus an average per annum of migrants should be substantially less than the relative values. The diversity indices for Shannon-Weiner and Simpsons indicate higher index scores for the residents compared to migrants in most years. Shannon Weiner uses the number of species and evenness of species at a site to calculate the diversity index. The index values normally range from 0 - 5, and usually lie between 1.5 and 3.5. The higher the value, the higher the overall species richness and presence of rare species. Shannon-Weiner diversity index at Strandfontein fluctuates between 0.46 and 1.37 for all waders over the study period. Whilst residents show a more even spread than migrants, overall the low diversity score (evenness score) for residents and migrants suggests that there is relatively few successful species at Strandfontein.

The negative trend in species richness and diversity indices (although more subtle) can potentially be loosely associated with the negative trends in abundance of waders over the same period. However in most instances, years of high peaks in abundance do not coincide with high peaks in diversity. When comparing the migrant diversity indices and abundance over the 31 year review period, an agonistic result is evident between 2001 and 2003. During this period abundance is typically at an all-time low, whilst diversity indices (Simpsons and Shannon-Weiner) reveal their highest peak.

Diversity is reliant on numerous variables, as is abundance. Comparative studies of natural and artificial habitat types (mud, sand flats and aqua cultures) in Chongming Island, China, indicated that wader density, richness and diversity were reliant on the seasonal availability of habitats (Ma *et al*., 2004).

Similar patterns, albeit in widely different habitats and geographical areas are reported in the literature. Henningsson & Alerstam (2005) investigated and described the variables and processes determining the extensive distribution of species richness of waders in the Arctic tundra. Results show strong indications that species richness is reliant on; high primary productivity of resources, length of the summer, efficiency of connectivity through flyways, and historical tundra distributions.

Waders have varied feeding habits, and rely on certain variables for resources extraction (primary productivity, water level, vegetation cover etc.). High diversity and abundance of waders is generally associated with shallow water depths, where habitat requirement is maximised to suit most wader species (Colwell & Taft, 2010, Ma *et al*., 2004; Ma *et al*., 2010). Habitat quality could explain species richness and diversity indices. Varying ecological components (behaviour, population number) of specific species could also explain levels of richness and diversity. Some wader species (particularly migrants) are rare or vagrant, and these are often solitary (which will affect the Shannon-Weiner evenness). Parallel to this common species congregate in large flocks, are gregarious, viz. Little Stints, which can flock in thousands.

CHAPTER 4. IDENTIFYING OPTIMAL HABITAT REQUIREMENTS FOR WADERS

4.1 Introduction

Maximizing the littoral zone through regulation of water levels is a proven method of increasing wader abundance in wetlands. However, few studies have focused on analysing factors that determine the spatial and temporal distribution of waders in artificial water systems. Various interacting environmental variables influence the habitat preferences of waders, and in artificial wetlands these can be altered to cater for specific bird species (Powell, 1987; Boshoff *et al.,* 1991; Hockey &Turpie, 1999; Massero *et al.,* 2000; Lunardi *et al*., 2012; Manikannan *et al.*, 2012). Effective management of the wetlands requires prior identification of factors, which can then be integrated into a management plan that benefits not only resident but also migrant species whose conservation status necessitates interventions that provide suitable habitat in their southern migratory range.

Most ecological parameters that directly or indirectly influence broad and fine-scale distribution and abundance of waders cannot be controlled. However, variables which can be manipulated or transformed to positively influence the numbers of waders in Waste Water Treatment Works (WWTW) need to be identified for appropriate adaptive management and incorporation into annual operational plans. In this way -the value of WWTW for migrant waders can be enhanced, providing a reliable resource suited to the foraging needs of migrants replenishing fat reserves prior to their autumn departure to northern breeding grounds.

Waders respond to specific cues, such as water depth and prey availability, when selecting foraging habitat (Master *et al*., 2005).The ecology of a wetland, particularly the littoral zone, invertebrate biomass, vegetation cover and water levels all potentially influence habitat selection, which as a result affects the species richness and diversity of wader abundance at specific wetlands (Laubhanand & Gammonley, 2000).

4.1.1 Factors influencing habitat selection by waders

Lunardi *et al*. (2012) analysed the relationship between wader assemblages and environmental parameters on the intertidal flats of Baia de Todos os Santos, Brazil. The most important factors influencing the wader assemblages were invertebrate prey abundance, vegetation cover and amount of fine sediment. Both resident and migrant wader assemblages were dependent on the maintenance of heterogeneous intertidal habitats, each generated by a number of interacting factors (Lunard *et al*., 2012).

Factors influencing invertebrate abundance and community composition in wetlands are a reliable predictor of wader dispersion patterns. Some directly influence food availability, but others that have a more indirect influence on habitat quality are also of importance (Van Dusen *et al.,* 2012). Explanatory variables will have positive or negative association with wader abundance.

Water depth in a managed wetland context is generally the primary explanatory variable for waterbird diversity as it determines the habitat type available for foraging (Bolduc & Afton, 2008). In a managed wetland system the water depth should allow for continuous resource exploitation. This may be difficult to achieve throughout the year, as it is influenced by the interplay of rainfall and the ability to manipulate water levels. Water depth in wetlands fluctuate naturally and can change within days, rapidly modifying the suitability of the site as foraging habitat for waders. Eldridge (1992) describes optimal foraging habitat for waders as an extensive mudflat/littoral zone, with minimal vegetation and shallow water – 3 - 5 cm deep in a wetland with sloped banks and high density of invertebrate prey (at least 100 invertebrate individuals per square meter). An analysis of 25 managed, seasonal wetlands in the San Joaquin Valley, California, showed that water depth influenced density, abundance and species composition of wader communities (Colwell & Taft, 2000). Ideal habitat for the full range of wader species comprises shallow wetlands, with a mixture of exposed substrate (islands), as greatest diversity and abundance of waders is known to be associated with water depths of approximately 15 - 20 cm deep (Safran *et al.*, 1997).

In natural systems, food availability for waders is often limited by (unpredictable) high water levels. However in a disturbed water system, (much like that of a WWTW) where the conditions are optimal for maximising invertebrate biomass, (eutrophic system with excess sludge formation) resource availability can be maximised by manipulating water levels. Lowering water levels gradually over a period of time will maximize the foraging habitat as it increases substrate surface area, and thus extractable benthic invertebrate community biomass (Rehfisch, 1994; Sanders, 1999).

Lake Benmore, a managed water system in New Zealand supported increased numbers of Black Stilts (*Himantopus novaezelandiae*), when the water level of the delta was lowered during draw down periods. The exposed wet substrata augmented the exposed benthic invertebrate biomass, and resource availability showed a positive correlation with decreased lake water level (Sanders, 1999). Rehfisch (1994) demonstrated that waders consumed a moderately small proportion of the total mean annual biomass in artificial experiment stations at Humber Estuary, England. Most of the primary production was unavailable due to high water levels restricting the bird"s access to the resource. In summer and autumn when the mean water depth was manipulated to be at its lowest, the entire recorded biomass was utilised. This shows that the limiting factor for waders is primarily resource access rather than abundance, particularly for substrate-probing waders using tactile methods in prey location (Finn *et al*., 2008). Artificial wetland-based reserves, where certain variables can be manipulated, can therefore rapidly become as productive as natural systems by providing optimal food resources for birds, with manipulation of water level being the primary habitat management tool.

A well-established technique to provide new foraging habitat for waders is the revitalisation of the invertebrate community resource, accomplished by seasonally lowering the water levels to provide freshly exposed littoral zones (Velasquez, 1992; Rehisch, 1994; Sanders, 2000; Taft *et al*., 2002; Sanchez *et al*., 2006; Ma *et al*., 2010). This maintains an early successional stage in the invertebrate community, and optimizes biomass availability while ensuring continued colonization (Rehisch, 1994; Sanders, 2000). Invertebrate communities are quickly established after flooding (depending on duration and frequency) and disturbance, especially given the ubiquitous nature and rapid life cycles of primary food items such as dipteran larvae. Plant cover can also influence invertebrate species assemblage, as communities are associated with open water, mud or vegetated areas (Rehfisch, 1994; De Szalay & Resh, 2000). However waders prefer to forage in unvegetated areas (Taft *et al*., 2008; Russel *et al*., 2014). Peak abundance of migrant waders frequenting South Africa in the austral summer overlaps with the period of high abundance and productivity of invertebrate community at South African wetlands.

Prey density (invertebrate biomass) is reliant on numerous variables. Terrestrial vegetation cover and aquatic macrophytes can influence invertebrate community structure and micro distribution. Increased species richness and community abundance are associated with the presence of macrophytes, as they create additional living spaces in the water column (Gregg & Rose, 1985). De Szalay & Resh (2000) showed that invertebrate community distribution could be explained by the distribution of macrophytes at a wetland. Wetlands with low macrophyte cover had greater species richness, and areas with open water had less diversity, but a higher abundance of benthic invertebrates. This is reflected in the local foraging preferences of waders, which may congregate and feed in close proximity to vegetated zones of wetlands (Handel & Gill, 1992; Kalejta & Hockey, 1994). However, very dense stands of macrophytes (e.g. *Typha* beds) may inhibit access to resources (Russel *et al.*, 2014).

Management practices can regulate aquatic invertebrate recruitment by manipulating aquatic macrophyte growth (De Szalay & Resh, 2000). Wetland vegetation provides waders with wind shelter and concealment from (aerial) predators. Dunlins and Grey Plovers have been recorded to actively exploit aquatic and terrestrial vegetation for shelter during high wind conditions, and those that maintain territories in areas that are sheltered from the wind were found to feed for longer periods (Hammond & Pearson, 1994). Weather conditions can suppress foraging activity and affect the local distribution of waders. During high wind conditions Dunlins are more likely to form large, compact roosting flocks in a sheltered environment (behind foliage), than to forage (Handel & Gill, 1992). Under ideal conditions, most waders feed at uncovered tidal flats, and littoral edges of open water systems with minimal vegetation (Eldridge, 1992; Hammond & Pearson, 1994; Rehfisch, 1994; Safran *et al.*, 1997; Sanders, 1999).

Soil substrate properties are another variable known to influence habitat selection in waders as they stimulate the regeneration of the invertebrate community living in the sediment (Lourenço *et al.*, 2005). Both waders and their prey are most abundant on soft substrates (sand, mud), which provides a foraging advantage for waders and an ideal burrowing habitat for their prey (Finn *et al.*, 2008).

At a site that has readily available resources waders will utilise different components of the invertebrate resource through different trophic specialisations (resource partitioning). Variation in beak and other body dimensions allows waders to forage in varying water levels, mud depths and in varied habitats. Therefore access to different food resources differs across the species. The competitive exclusion principle may not apply to waders, as several species typically coexist, presumably since diversity in bill and foraging behaviour allow resource partitioning. Considerable morphological variation and microevolution is even evident within a species (Cowell, 2010), e.g. European Oystercatchers are suggested to have three bill types; pointed, chisel-shaped, and blunt as well as their intermediates for efficient prey handling (Hulscher, 1984). Bills of individual Oystercatchers changed their morphology when food type was changed and the bird was forced to adapt its feeding method (Swennen *et al*., 1983). Feeding behaviours of waders is probably constrained primarily by individual morphology, particularly of the bill. Different waders utilise different foraging water depths, which can be associated with tarsus and culmen length (Baker, 1979; Safran *et al*., 1997). Large waders are able to handle heavier invertebrates and waders with longer bills have access to prey which burrow deeper into the substrate (Durell, 2000). While waders use a multitude of feeding techniques, tactile and visual feeding cues are considered the most relevant during foraging (Barbosa, 1995; Turpie & Hockey, 1997; Barbosa & Moreno, 1999). Tactical foragers are non-selective feeders when compared to visual foragers, resulting in differences in resource exploitation. These differences in foraging technique might explain fine scale dispersal as they influence initial habitat selection (Kalejta, 1993).

Waders migrating south for the austral summer feed in moderately dense, diverse species assemblages and inter- and intra-specific antagonism is likely to occur (Kalejta, 1993). Birds often cope with these increased pressures by diversifying their feeding range and using more than one feeding behaviour. Along with resource partitioning related to anatomy, habitat preferences for feeding sites also influence dispersion patterns. Blacksmith Lapwing and Kittlitz"s Plover prefer to feed on dry land, Three-banded Plover and Little Stint in the littoral edge or fringe, whilst Black-winged Stilt (*Himantopus himantopus*) and Ruff are more commonly observed feeding in open water (Maclean, 2001). The differences in feeding habitat coincide with different feeding techniques, and can vary in time and space. This highlights the importance of habitat heterogeneity that would offer the diversity of habitat required by the diverse feeding habitats of different waders (Van Dusen *et al*., 2012).

Dispersion of the most abundant migrant and resident waders in the Berg River Estuary, South Africa, was related to differences in prey abundance (Kalejta, 1993). Curlew Sandpiper distribution could be explained by density of nereid worms (*Ceratonereis erythraeenis*) and Grey Plover dispersion on the estuary was related to biomass of nereid worms (Kalejta, 1993; Hockey & Turpie, 1999). There was considerable overlap in the diets of all species observed and a difference in feeding techniques was reflected in variance in prey size and number consumed. Wader diet is thus flexible and generalised (Santos *et al*., 2005; Castro *et al*., 2009; Bowgen *et al*., 2015).

Various biological and ecological factors have been proposed to directly and/or indirectly influence migrant and residential wader distribution at various scales, including wetland size, substrate type, area of terrestrial or aquatic vegetation cover, water transparency, wind effects, predator-prey relationships, and extent of littoral edge zone (Powell, 1987; Boshoff *et al.,* 1991; Hockey &Turpie, 1999; Massero *et al.,* 2000; Lunardi *et al*., 2012; Manikannan *et al*., 2012).

The aims of this chapter are to identify habitat variables that accurately predict fine scale wader abundance at the Strandfontein WWTW, with the objective of using this information to optimally manage habitat for both resident and migratory waders. Climate change is predicted to reduce rainfall and increase summer temperatures in the Western Cape Province (Magadza, 1994; Hulme *et al*., 2001; Pio *et al*., 2014), which will likely accelerate the drop in water levels of inland wetlands in early and mid-summer. This chapter will discuss if the mitigation provided by the managed pans at Strandfontein have the potential to increase the importance of this site for waders in their southern migratory range.

4.2 Methods

4.2.1 Study site and data collection

Observational bird counts, invertebrate benthos sampling, and measurement of environmental variables (extent of littoral zone, % vegetation cover, water level fluctuations, and wind speed) were carried out at primary pan one (P1) of the Strandfontein Section of the False Bay Nature Reserve, over a one year period between July 2014 and June 2015 (). Refer to Chapter 2 (Figs. 2.1, 2.2, p. 27) for details of study site and collection methods.

4.2.2 Tracking seasonal variation in invertebrate abundance and biomass

Monthly mean values for invertebrate biomass and abundance over the 11 plots were obtained by first using means of the 10 substations at each plot, and then obtaining means across the 11 plots providing a single monthly mean value for the pan. The relationship between invertebrate biomass and abundance was tested with a linear regression.

For comparison of seasonal variation in biomass and abundance invertebrate data were further categorised into four main groups (worms, snails, aquatic insects and their larvae, and terrestrial insects). Chironomids were the most abundant invertebrate taxon, and were isolated from the aquatic insects group and used as an additional variable for the GLM. Seasonal variation across the major invertebrate categories were examined for the study period using monthly mean values per site.

4.2.3 Factors influencing wader abundance

To identify explanatory biotic and abiotic variables associated with fluctuations in wader abundances; six environmental and biotic variables (rainfall, change in water level, invertebrate biomass and abundance for the major groups (see 4.2.2 above), littoral edge size and vegetation cover) and count data for the response variable (wader abundance) were subject to a Generalised Linear Model (GLM), with a Poisson distribution and logarithmic link function (McCullagh & Nelder, 1989).

Environmental parameters were sampled once monthly over 12 months at 11 plots along the perimeter of pan one (n=132). Bird counts were done three times monthly (one week before environmental sampling, during environmental sampling and one week after environmental sampling) over 12 months at the same 11 plots (n = 330) (Appendix 1) .The monthly environmental variable data per plot were used for the three monthly bird counts (in total 30 monthly bird counts were undertaken at 11 plots).

Model selection was guided by backwards and forward selection techniques and the Akaike Information Criterion (Linhart & Zucchini 1986). Analyses were performed using Genstat (Release 17)(VSN International 2014).

Linear regressions were used to test the relationship between monthly mean resident wader abundance and mean chironomid abundance, and fluctuating water levels and chironomid abundance (monthly means) in the pan over the sampling period.

4.2.4 Community composition data at different sites

To determine if the 11 plots had distinctive communities, and whether the effects of initial vegetation clearing was reflected in community composition of invertebrates and waders, total species abundance (July 2014 – June 2015) were analysed with PRIMER 6, (version 6.1.15) with the PERMANOVA+ add-in (version 1.0.5)(Clarke & Warwick, 1994; Anderson *et al*., 2008). Values were first 4th root transformed, and a similarity matrix constructed using the Bray-Curtis similarity coefficients. Multi-Dimensional Scaling (MDS) plots and cluster analyses were performed, and the significance of the separation of groups tested with an

ANOSIM. Stress factors on MDS plots were all <0.15, indicative of an acceptable 2D representation of the data (Clarke & Warwick, 1994).

4.3 Results

Over the study period, 11 species and 1 698 individual waders were recorded in the observational counts at the entire pan. These comprised seven resident and only four migrant species with total abundances of 1 111 and 587 respectively for the entire sampling period. The 5 280 cored mud samples (four samples per subplot x 10 subplots, x 11 plots, x 12 months) contained 5372 individuals and over 61 invertebrate taxa (identified to morphospecies).

4.3.1 Seasonal variation trends in invertebrate abundance and biomass

There was no significant relationship between mean invertebrate abundance and biomass ($P = 0.97$, $r = 0.01$, $F = 0.001$, $n = 11$) for the pan. Invertebrate biomass remained relatively stable during the course of the year, with the exception of small peaks in October and February. Invertebrate abundance increased in early and mid-summer (November to February) with a peak in January ($n = 5.68$), that matched a biomass peak over roughly the same period. Peaks in abundance and biomass had the largest standard deviations, showing the greatest variation from the mean in these months (Fig. 4.1).

Terrestrial invertebrates had the greatest biomass of all categories and water snails the lowest (Fig 4.2). For the major four groups of invertebrates, biomass peaked in October (albeit only slightly for aquatic invertebrates), and again in late summer (apart from aquatic snails). As with the mean invertebrate biomass, biomass by taxon had only minor seasonal fluctuation. Terrestrial invertebrates had the greatest biomass of all categories and water snails the lowest.

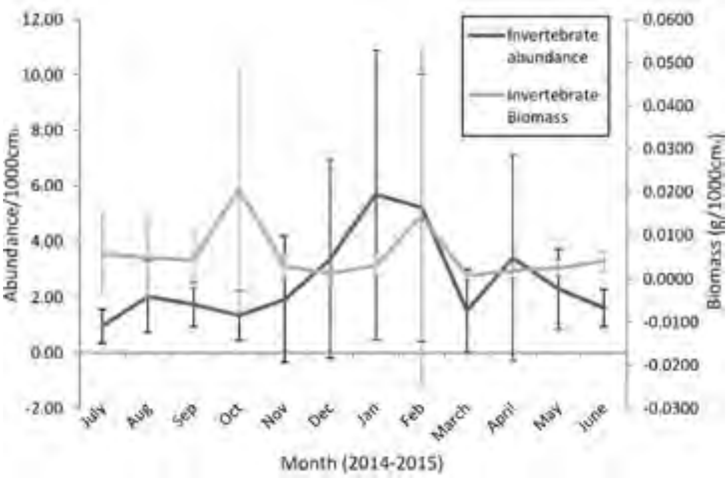

Fig. 4.1. Mean (±standard deviation) invertebrate abundance and biomass over the period (July 2014 – June 2015) at 11 sites at P1, Strandfontein.

Invertebrate abundance varied seasonally in all groups save water snails (Fig 4.2). Nearly all groups showed a decline in abundance over summer, apart from chironomids whose numbers increased substantially in summer. Chironomids and aquatic invertebrate abundance increased over summer (December to May), peaking in January and April respectively and a decrease in abundance was apparent outside these months. The inverse was evident for terrestrial invertebrates and worms, which showed increased abundances during winter months (May to October), peaking in September and October respectively. Terrestrial invertebrates thus had the greatest biomass, and chironomids the greatest abundance. Water snails had the lowest biomass and abundances overall.

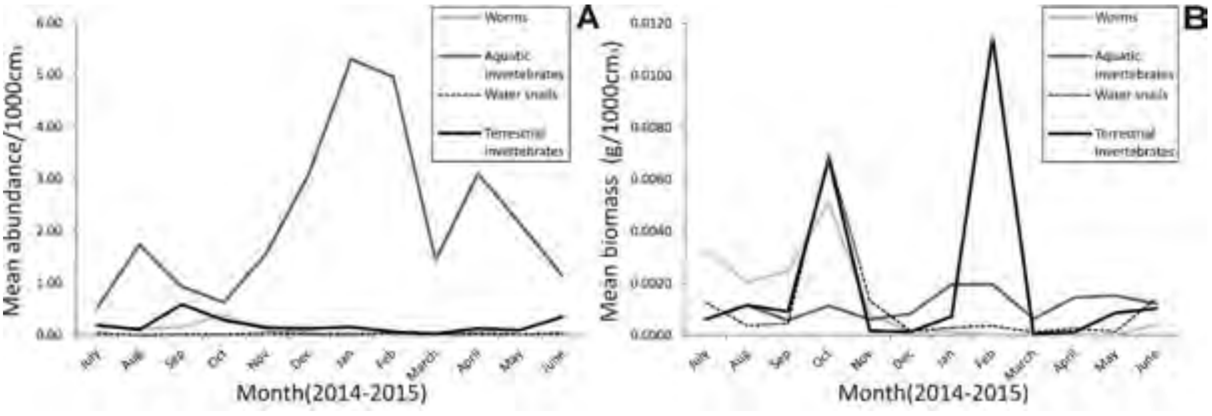

Fig. 4.2. Monthly mean invertebrate abundance (A) and biomass (B), for four major invertebrate categories over the study period July 2014 – June 2015 at P1, Strandfontein.

When abundance and biomass of the same invertebrate groups were reported as annual medians (Fig. 4.3), aquatic invertebrates had the highest abundance and biomass as reflected in the mean data (Fig. 4.1), although there was considerable variability in numbers for all groups.

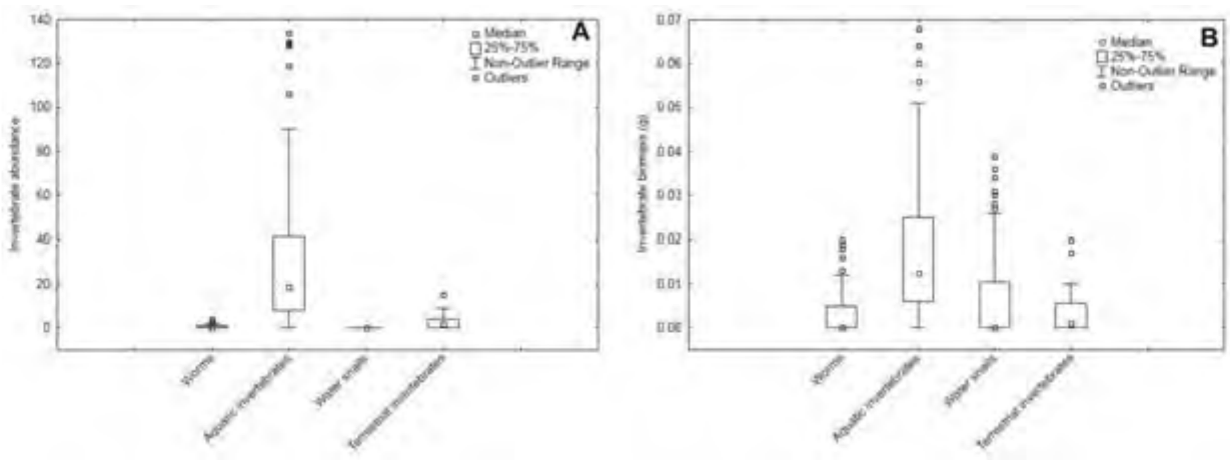

Fig. 4.3. Median invertebrate abundance (A) and biomass for (B), for four major invertebrate categories for the study period (July 2014 – June 2015) at P1, Strandfontein. Extreme outliers were removed.

4.3.2 Factors influencing wader abundance

The GLM examined the association between wader abundance and various explanatory variables (wind speed, vegetation cover %, invertebrate biomass and abundance -per group, littoral zone size and fluctuating water level). Results accounted for 27% variability in total wader number, explained by only three of the variables. Vegetation cover had a significant negative relationship with wader abundance, while increased chironomid abundance and retreating water levels were positively associated with increased numbers of waders (Table 4.1).

Table 4.1 The selected GLM (Poisson distribution, logarithmic link function) identified three explanatory variables (F 3,325 = 39.7, P<0.0001). The data consisted of results of 30 monthly bird surveys, and 12 monthly environmental surveys at 11 sites in pan P1, Strandfontein Sewage Works. This model accounted for 26.8% of the deviance.

Parameter	Estimate	Standard Error	t 325	p values
Constant	1.69590	0.04160	40.80	<0.0001
Fluctuating water level	-0.00025	0.00002	-16.40	<0.0001
Vegetation Cover	-0.02710	0.00161	-16.87	<0.0001
Chironomid abundance	0.00575	0.00037	15.59	<0.0001

Chironomid abundance was negatively associated with water level (P = 0.85, r = 0.05, F = 0.03, n = 12). Resident wader abundance showed a significant positive relationship with chironomid abundance (P = 0.012, r = 0.69, F = 9.37, n = 12) (Fig. 4.4).

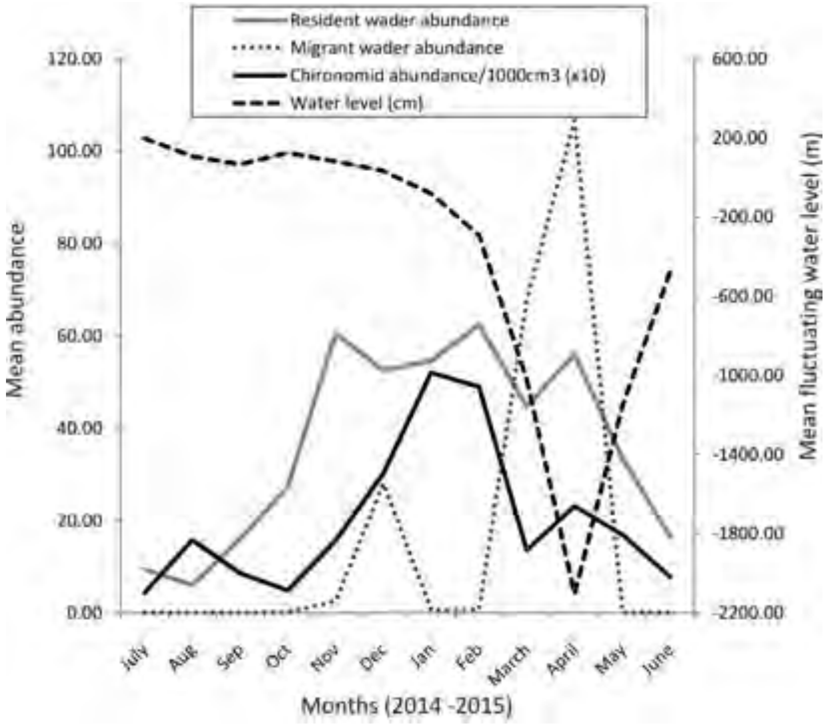

Fig. 4.4. Mean abundance of resident and migrant waders, chironomids (X 10) and fluctuating water level for all 11 plots on Pan 1, Strandfontein.

4.3.3 Wader community composition at plots

The ordination of the 11 plots based on invertebrate community composition showed no difference between community composition at the control (C) and bulldozed (B) plots.

Instead plots appeared to cluster in a spatial pattern with site, not treatment, defining position of plots on the MDS (refer to map of pan – Chapter 2, Fig. 2.2). Treatment (bulldozed, control) did not appear to influence wader communities, with similarity between communities related to plot position (ANOSIM for factors C & B; Global R = -0.139, P = 0.89, Fig. 4.5B). The same was true for invertebrate communities, with treatment not explaining cluster position (ANOSIM for factors C & B; Global R = 0.001, P = 0.39, Fig. 4.5A). For invertebrate community composition, plots C4 and B5 which had the highest similarity score were adjacent but differed in topography and flora cover. B4 was cleared of reeds and all surface vegetation prior to sampling start in July 2014. C4 remained heavily vegetated throughout the study period, with a dense reed bed fringing the bank.

Plot B1 was an outlier with particularly high abundance of species, whereas most other plots were generally similar in terms of wader composition. Plots B3, C3, B4 and C4 clustered together by their wader community composition; however they comprised both cleared and vegetated plots. Similarity of community composition of clusters appears to be related to position of plots. The same principle applies to the invertebrate ordination; with clusters comprising nearby plots rather than treatment plots e.g. cluster (B3, C3 and B4).

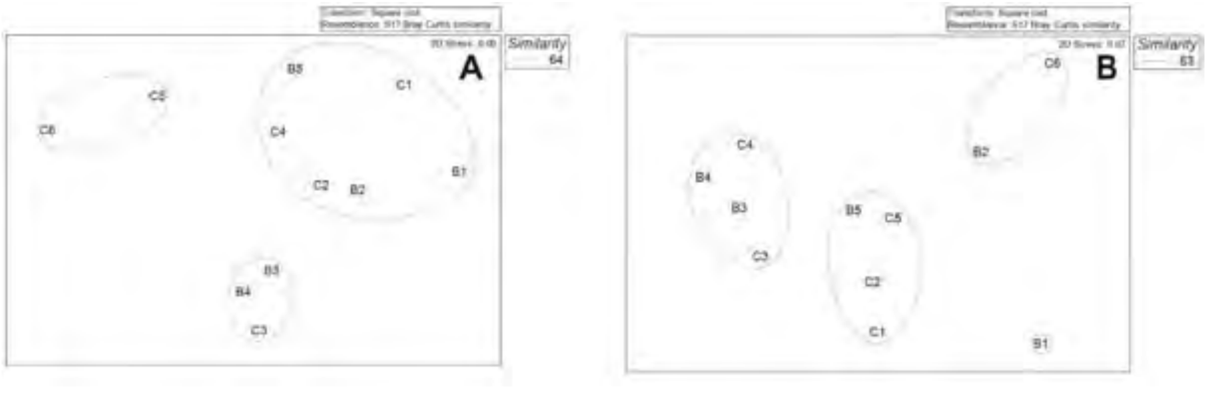

Fig. 4.5. Ordination of sampling plots based on invertebrate abundance (A) and wader abundance (B) from June 2014 to July 2015 at P1, Strandfontein. C – control plots, B – bulldozed plots.

4.4 Discussion

4.4.1 Seasonal variation trends in invertebrate abundance and biomass

Aquatic invertebrates had the greatest annual abundance and biomass when compared to worms, aquatic snails and terrestrial invertebrates occurring in the littoral zone. Invertebrate biomass peaked in spring, while invertebrate abundance remained fairly constant over the year, apart from a dramatic increase in numbers of chironomids over the summer months. This increase in chironomid populations coincided with retreating water levels (Fig. 4.3), providing ideal conditions for feeding waders. This pattern is similar to that seen in hyper-eutrophic wetlands in Florida, where chironomid abundance peaked in summer. Larval cycle of chironomids there ranged between 14 and 22 days, allowing rapid recruitment as water temperatures rise (Cowell *et al*., 1981). In contrast, biomass of terrestrial invertebrates and worms at Strandfontein peaked in September and October, similar to trends reported by King (1983), for macroinvertebrates in the Eerste River, Western Cape, South Africa. Thus invertebrate prey is available year round, but with increased productivity in summer when reflected by abundance, and peaks in spring when biomass is considered. The latter measure is likely influenced by dry weight measurements, especially of snail shells.

Wader diet is influenced by both the abundance and availability of preferred invertebrate prey, and seasonal switching between prey types is a common practice due to seasonal variation in invertebrate resources (Puttick, 1978; Cowell *et al*., 1981; Worrall, 1984; Kalejta, 1993; Kalejta & Hockey, 1994). The diet of Red Knots inhabiting the intertidal flats of the Wadden Sea, Western Europe substituted bivalves in summer to mud snails in autumn according to analyses of faecal samples (Dekinga & Piersma, 1993). Kalejta (1993) also highlighted seasonal prey switching (Puttick, 1978; Worrall, 1984).

At the Berg River Estuary and Langebaan Lagoon, nereid worms were found to be one of the essential prey items for Curlew Sandpipers (Puttick, 1978; Kalejta, 1993; Kalejta & Hockey, 1994). However at salt marshes at Odiel marshes, south-west Spain, Chironomid larvae abundance was positively associated with counts of foraging waders (Sanchez *et al*., 2006). A similar association was detected for Strandfontein, where the summer peak in chironomid abundance coincided with the period over which migrant species occur.

Water snails at Strandfontein probably do not make up a large part of wader diet due to their low density. *Assiminea* snails were reportedly one of the fundamental components of the Curlew Sandpipers diet at the Langebaan lagoon between 1974 and 1975, however when the snails were most abundant (spring) they were often avoided, presumably due to their small size and putative smaller energy intake compared to other more abundant prey items (Puttick, 1978).

4.4.2 Factors influencing wader abundance

The GLM identified retreating water level and increasing chironomid abundance as good indicators of wader abundance. The temporal matching between these variables and the summer feeding period of waders suggests that chironomids are most likely the major dietary item during this period (all other dietary categories fed into the model did not contribute in explaining the variance in wader numbers). There was a positive relationship between resident wader abundance and chironomid abundance through the year, suggesting that residents may alter local feeding sites in relation to food resources.

For the Pied Avocet (48.1% of resident abundance) fluctuations may relate to local dispersion in breeding sites as well as links to food resources and water levels. However, Blacksmith Lapwing (17.7% of resident waders) and Black-winged Stilt (29.9% of resident abundance) do not display local movement and breed at Strandfontein changes in their numbers are therefore most likely associated with fluctuating food resources with birds, moving between different pans at Strandfontein. Vegetation cover was negatively associated with wader abundance, although there was no shortage of prey items in this habitat. The presence of vegetation could potentially hinder resource exploitation, as compact foliage and root systems may prevent efficient foraging compared to unvegetated areas. The dominant emergent macrophyte at the study site was *Typha capensis*. The density of the vegetation cover could presumably have an effect on the accessibility of prey resources for waders. Wader foraging efficiency may be negatively impacted where increased vegetation cover occurs, as waders are typically small and cannot access the invertebrates that inhabit the area beneath the dense strands of macrophytes. Waders prefer open unvegetated habitat, and generally (with some exceptions) avoid densely vegetated areas when foraging (Davis & Smith, 1998; Twedt *et al*., 1998; Darnell & Smith, 2004; Rogers & Hulzebosch, 2014, Rogers *et al*., 2015). However vegetated zones at a waterbody may act as indicators of high resource areas to waders. Grey plover in the Berg River Estuary favoured high vegetation cover areas whilst foraging possibly for concealment (Kalejta & Hockey, 1994). The density and nature of the emergent vegetation would determine its access to waders.

Exposed sandbanks on shores of Eilandvlei (one of the lakes which comprise the Wilderness Lakes System) Western Cape, South Africa, were encroached by *Typha capensis* in the last two decades. This negatively impacted the abundance of foraging waders (Russel *et al*., 2014).The opposite also occurred with clearing of *Typha* beds, with the exposed banks showing an increased number of waders until such time that the area was recolonized by *Typha* (Russel *et al*., 2014).

Water depth influences accessibility to invertebrates in the mud zone or shallow water beyond that, and affects wader abundance and richness (Russel *et al*., 2014). Continually receding water level in summer maintains mud and shallow water, providing a constant food resource. However, seasonal influences likely resulted in their being no clear negative relationship between chironomid abundance and water level.

Chironomids (*Chironomus formosipennis*) were the most abundant invertebrate species at Strandfontein. These worms have haemoglobin allowing them to survive in organically enriched environments (like a WWTW) which is subject to relatively low oxygen concentrations, particularly in summer months, when other aquatic invertebrates might be limited by oxygen availability (Picker *et al*., 2004; Griffiths *et al*., 2015). Water level manipulation in artificial pans would maximize prey availability. Hockey *et al*. (1998) reported similar results during an experimental trial at the Berg River Estuary, Western Cape, South Africa. Prey availability was artificially increased by dropping the water level in the salt pans, permitting wader access to previously restricted concentrations of chironomids, resulting in increased numbers of Curlew Sandpipers (Hockey *et al*., 1998).

The distribution of both invertebrate and wader communities was not associated spatially with either cleared or control plots (some of which had dense *Typha* beds), but instead MDS clusters for both groups (Fig. 4.4) were based on proximity of sites to one another. This suggests that factors other than the presence of emergent macrophytes determined community composition of sites. For invertebrate community composition plots C4 and B5 had the highest similarity score, but differed in topography and flora cover. B4 was cleared of reeds and all surface vegetation prior to sampling start in July 2014. C4 remained heavily vegetated throughout the study period, with a dense reed bed fringing the bank. The similarity between the two sites suggests that the treatment of the floral community (particularly *Typha*) does not significantly affect the invertebrate community composition and the location of the plots (as they lie next to each other in the pan) appears to have a greater effect on the invertebrate community structure than the treatment of the plots. It is possible that the foraging area of a flock of waders was greater than the plot dimensions, or that a flock foraged at the boundary between two adjacent plots.

Wader abundance at Strandfontein during the sampling period July 2014 to June 2015 highly increased in summer with the seasonal influx of both migrant and residential species, matching resource abundance and availability. As water levels retreated, the reedbeds which had previously excluded waders no longer hampered foraging behaviour, and waders were able to access the invertebrate resources in these plots (Rehfisch, 1995; Russel *et al*., 2014; Kalejta-Summers *et al*., 2001).

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CHAPTER 5. CONCLUSION AND MANAGEMENT RECOMMENDATIONS

5.1 Problems facing waders

Waders and other waterbirds appear to be particularly hard-hit by ongoing habitat transformation and climate change. Approximately 38% of waterbird populations are declining, and only 20% increasing (Wetlands International, 2012). Although resident wader populations appear to be stable or even increasing, most of the Palearctic waders with viable census data have decreasing populations - largely attributable to habitat degradation or loss (Wetlands International, 2002; Delany *et al.*, 2003; Zöckler *et al.*, 2003; Wetlands international, 2006). Similar patterns were found at Strandfontein for trends over the past 31 years. Resident wader numbers showed minor fluctuations over the past 31 years, however migrant numbers showed ongoing and consistent declines. Tipping points from the monotonic regression show that migrant populations first started to decline in 1997/1998, with the greatest tipping point in the summer of 2001/2002. Between July 1983 and June 2014 migrant abundance declined by 82% at Strandfontein. Migrant abundance, and migrant and resident species richness and diversity all exhibited a decline at Strandfontein between 1983 and 2014. In line with the findings for Strandfontein, similar negative trends for migrants and increasing trends for residents are evident at other nearby waterbodies such as Langebaan Lagoon. This waterbody supports wader communities with similar species composition to those at Strandfontein (Harebottle *et al.*, 2006; Harebottle & Underhill 2006; Simmons *et al.*, 2015). This suggests that local habitat transformation is not the primary factor behind the decline in migrant waders, as resident wader populations appear to be largely stable, as indicated by global reports and data from Strandfontein. Palearctic migrants would be far more susceptible to habitat transformation and other potentially limiting factors, as they would be subject to multiple impacts across large spatial scales. Wader migration exploits the natural cycle of seasonal change across the globe (Robinson *et al*., 2009). It is an adaptation which bypasses seasonal fluctuations in food resources, and is measured by breeding success in the Tundra. Successful breeding is dependent on successful migration to southern sites where fuel stores are replenished adequately in anticipation of the return flight to breeding grounds (Piersma & Lindström, 2004). Refuelling at reliable sites and avoiding predation is imperative for successful breeding, as the generation of male breeding plumage is positively associated with adequate fat stores (body mass). Full male breeding plumage in Bar-tailed Godwits is an honest indicator of sufficient fat reserves and can be related to flight performances during the return migration, typically a bird"s ability to cope with the stress of the journey (Piersma *et al*., 2001). If migration is compromised, recruitment is negatively affected. Threats to stop-over sites and winter ranges (sites in South Africa) would ultimately impact clutch size and recruitment. However, Ryan (2013) suggests that the negative trend in migrant abundance in South Africa could be a direct result of global population declines. Threats at both summer and winter ranges which impact the suitability of wetlands to provide suitable habitat for waders include; exploitation of biological resources by agriculture and aquaculture activities, urbanisation, and human disturbance and natural system modifications (e.g. impoundments), at stop over sites (Kirby *et al.,* 2008). Rakhimberdiev *et al.* (2011) found that migrating Ruff (*Philomachus pugnax*) numbers have been declining at wintering ranges in Netherlands and Sweden as a response to habitat loss. Distribution of Ruffs has shifted in an eastward direction, with increased populations now breeding in Western Siberia.

In addition to the impacts of urbanisation, climate change is considered to be a threat to migratory waders, as it affects timing of ecological events and processes. In addition to its impact on phenological events, climate change could also reduce the extent of available foraging area on a global scale, as rising sea levels might reduce the width of mudflat and sandy shores which are bounded at the high water mark by rocky headlands (Galbraith *et al.*, 2002).

Natural fluctuations in population numbers are the norm, and for Palearctic migrants these fluctuations appear to be linked with complex predator cycles in the Tundra (Underhill, 1987; Blomqvist *et al.*, 2002). A switch in prey type taken by Arctic Foxes and other arctic predators affected the abundance of Curlew Sandpiper (Underhill, 1987), Knot (Underhill *et al.*, 1989), and Sanderling (Summers *et al.*, 1987) at Langebaan Lagoon (one of the southernmost ranges of the East Atlantic Flyway). Numbers of migrants in the southern part of their range declined in the following year, when lemming population crashed in the Taimyr Peninsula breeding grounds (Summers *et al.*, 1987; Harebottle & Underhill, 2006). Presumably, predators for whom lemmings represented a major dietary resource then switched to wader eggs and chicks (Piersma & Lindström, 2004; Schmidt *et al.*, 2012). Curlew Sandpipers have departed earlier from breeding grounds due to decreased breeding success, as a result of increased predation (median departure days have advanced by 23 days between 1946 and 2005) (Barshep *et al.*, 2010). These natural fluctuations could be impacted negatively by climate as shorter, warmer winters may impact lemming populations over the long term (Schmidt *et al.*, 2012).

Shorter, warmer winters in the Tundra resulting from climate change could potentially alter prey resources at the breeding site. The energy used for egg formation after mating is derived from biomass gains obtained from food resources at the breeding grounds after migration viz. income breeding (Meltofte *et al.*, 2007). Meltofte *et al* (2007) showed that in three species of wader; seasonal variation, availability and abundance of prey were potentially affected by climate change. This had a substantial influence on the timing of egg laying in Greenland between 1995 and 2005. Migration of Whimbrel (*Numenius phaeopus*) to their northern breeding grounds from the wintering range at Banc d'Arguin, relies both on the annual cycle of prey species at the breeding grounds and at stop overs en route to the Tundra (Zwarts, 1990).Phenological mismatches between the timing of wader migration and breeding cycles of invertebrate prey could therefore impact migrants in all parts of their range. Several studies have shown that waders adjust the date of departure from arctic breeding grounds according to local climate conditions, and in many cases long term data shows no trend matching to climate at all (Rehfisch & Crick, 2003).

The arrival month of migrants at Strandfontein (wader wintering range) was the same from 1983 to 2014. When analysed by decade, starting from the 60‟s to the 1980‟s -September was the month when migrants started appearing. In the 1990‟s waders first appeared in August, and then from 2004-2014 they appeared again in September. Thus the patterns at Strandfontein do not support a general trend for earlier arrival date with time. It is possible that the birds have in fact arrived earlier in the region, but take some time to sort locally and eventually arrive at Strandfontein. During the study period (July 2014 to June 2015) at P1, wader abundance started increasing in September; however the first migrants were only recorded in November. The biggest increase in wader abundance between October and November 2015 (68%) coincided with the greatest increase in chironomid abundance (56%). April showed the greatest rate of water recession (51% change between monthly records), which coincided with the highest wader abundance at P1. This was followed by a substantial decrease in wader abundance and sudden increase in water level in May.

Chironomid abundance and receding water levels were strong explanatory variables for the dispersion of waders across the 11 sites at Pan 1. Currently migrant arrival is timed optimally to resource availability, but climate change predictions for the Western Cape Province (lower rainfall and increased temperatures) might alter this. In Glensaugh, Scotland, global warming is predicted to impact the abundance of crane flies (Tipulidae), the favoured invertebrate prey of Eurasian golden Plovers (*Pluvialis apricaria)*. In response to the decrease in food resources, Plover range is predicted to contract, as species distributions are predicted to contract and shift northward and as an outcome of climate change (Pearce-Higgins *et al*., 2010). Changes in distribution ranges are not restricted to the northern hemisphere, as 13%

of the 408 of South African terrestrial bird distributions analysed between 1970s and 1990s had range changes, possibly a result of climate change (Hockey *et al*., 2011).

South Africa is predicted to experience reduced rainfall and warmer temperatures as a result of climate change (Hulme *et al*., 2001; Pio *et al*., 2014). For inland wetlands, this could have serious implications for waders, as it may lead to water bodies drying out before summer ends and migrants have replenished fuel stores for the migration back to the Tundra. The Western Cape wetlands do not experience summer rainfall, thus increased summer temperatures are likely to accelerate drying out of the shallow water pans favoured by feeding waders.

Managed wetlands become more important to wader conservation in these instances, especially since they typically have some legal protection and are not prone to habitat loss via urbanisation. Key variables that have been identified as being linked to increased wader abundance can be manipulated to some degree in artificial habitats. Wader abundance is limited by environmental variables which inhibit foraging (viz. high water levels and narrow, littoral edges, dense vegetation cover, steeply-sloped waterbody profile, and prey avaliabilty).

This study identified chironomid abundance and receding water level as strong predictors of fine-scale wader abundance, and emergent macrophyte stands as having a negative impact on wader abundance. Invertebrate abundance, dominated by chironomids, increased in summer, coinciding with the drop in water level. Receding water levels expose littoral habitat and chironomid resources, but also create additional feeding habitat below dense stands of macrophytes that were situated along the highest water level mark.

Historical and medium-term trends in wader abundances at Strandfontein reveal that the site has been used by waders since at least 1952 when the WWTW enlarged the original small waterbody. Although Strandfontein currently only supports a small wader population when compared to other Western Cape wetlands, it is nevertheless regarded as a reliable site for waders, especially residents (Kalejta-Summers *et al*, 2001b; Marnewick *et al*., 2015).In the last decade (June 2004 to July 2014) median CWAC data showed that resident wader abundance increased by 75% between August and February at Strandfontein, suggesting that residents had moved from other local wetlands to Strandfontein when conditions became favourable. Strandfontein supports >1% of the global population of gregarious waterbird species; notably populations of Pied Avocet and Black-winged Stilt (Marnewick *et al*., 2015). However, it does not appear to offer ideal year-round foraging habitat. Resident wader abundance increased during summer, and only a small population stay in winter when the migrants have left When analysed separately, migrants were present when chironomid levels were fairly high, but clearer patterns were seen in the tight relationship between residents and chironomid abundance, suggestive of local dispersion relative to prey abundances.

Current habitat at Strandfontein could be manipulated further to create a larger network of wetland areas with added heterogeneity, suited for wader foraging. Winter rain and reduced evaporation result in high water levels and a decreased littoral edge which are difficult to manipulate with the existing channel system. In addition, steep bank profiles fringed with *Typha* beds dominate parts of the littoral zone in winter and autumn months. This is likely responsible for the local movement of residents to other wetlands. Exposing the shore line and increasing the littoral zone by removing *Typha* beds, and at the same time periodically dropping water levels could increase resident abundance in winter months, and potentially make the site more attractive to migrants in summer. Water levels can be manipulated throughout the year ensuring a constant abundance of chironomids and other invertebrates (Taylor *et al*., 1999; Ashenkazi, 2000). Managing wetland vegetation would increase littoral edge maximising feeding area. This would be advantageous to those Palearctic migrants arriving earlier to the southern edge of their non-breeding range. *Typha capensis*, a common aquatic macrophyte at Strandfontein, Rondevlei and Zeekoevlei sections of the False Bay

Nature Reserve, were shown to have a negative association with wader abundances; however they do offer ideal habitat for a range of other birds, including herons, bitterns, gallinules and a wide variety of passerines (Maclean, 2001). A management plan directed at conserving waders would lower the value of the WWTW to these species. Given the large number of pans at Strandfontein, it might be possible to identify and manage a proportion for waders, and leave other well-vegetated pans with steep profiles for conservation of other birds. Dense strands of reeds have been successfully removed from pan edges at Strandfontein in 2012, 2014 and 2015, with marked increases in the littoral zone.

5.2 Management interventions for wader conservation at Strandfontein WWTW

The Strandfontein section of the False Bay Nature Reserve has been managed to improve conservation of its biota since 2006. A plan of operation is updated annually, and an Integrated Reserve Management Plan (IRMP) for the broader False Bay Nature Reserve is reviewed and updated every five years, and needs to take the specific conservation priorities of the different components of the reserve into consideration. A number of interventions could be included into the management plans for the area, utilising the existing infrastructure at the WWTW to improve the value of the site for both resident and migrant waders. These include:

- Creating mudflats by upgrading the current channel and flow system
- Evaluating the effect of sludge on waders abundance and feeding efficiency
- Manipulating pan profiles
- Drafting Environmental Management Plans (EMP) for local City of Cape Town Municipality wastewater treatment works

The baseline data collected here for historical and seasonal variability of wader populations at the WWTW would serve as a reference against which to evaluate any management intervention that is implemented (adaptive management). In addition, the fine-scale abundance and diversity data collected for P1 Strandfontein could be used as a baseline for documenting future trends.

Retreating water level was identified as the primary variable positively associated with wader abundance that could be manipulated with the existing infrastructure. Ashkenazi (2001) suggested creating mudflats, by draining the water from the wagon wheel primary ponds of WWTW"s. This process has in fact been an active part of the management of the Strandfontein WWTW area since 2006, however in current conditions, the capacity to drain or increase water in a pan has become a lengthy process, as many of the weirs and channels are either inaccessible or blocked. The connectivity of the 30 interconnected pans and canals has been compromised by neglecting to maintain the linking channels and outflow/inflow pipes. Water flow into the primary pans can be regulated at the wagon wheel cement sluice system by using wooden boards to stop inflow. Canals around the edges of the pans can be used to divert water to another pan, however, mostly due to blocked outlet channels, water cannot be drained from a pan. Upgrades of this channel system are essential for rapid regulation of water levels. Retaining a constantly retreating water level and shallow depths of 10 - 20 cm in more than one pond over the course of the year could maximize wader numbers. This process will involve continuously topping up the water in the summer months, as the pans quickly dry out due to the combined actions of wind and sun. Additional measure would be decreasing the water level in the winter months, when rainfall causes rising water levels and pans often become inundated. This would increase the value of the wetland to both resident and migrant wader communities by maximising the exposure of productive littoral zones (Sanders, 2000). Water levels in winter are generally very high, and littoral zone is compromised for resident waders. An investigation into the effectiveness of creating optimal habitat (fluctuating levels) for resident waders during winter months would be beneficial to the management of the site.

Sludge build up in the Strandfontein pans is another result of a poorly-maintained system. The build-up of sludge in sand-lined pans could potentially affect the abundance and richness of invertebrate species. Foraging in Whimbrels has been found to be more successful in sandy areas than in muddy areas, possibly due to increased visibility (Velásquez & Navarro, 1993). Since the pans at Strandfontein WWTW have a thick layer of sludge, surveys evaluating the effect of sludge on waders abundance and feeding efficiency need to be conducted before any management measures can be implemented. The pans can be completely isolated once the water system flow is correctly managed, and a single pan can be left to dry out over the course of a year. The dried up sludge can be removed mechanically by a bulldozer, leaving a sandy substrate. Although sludge may provide ideal habitat for aquatic oligochaetes and chironomid larvae, it may also reduce feeding efficiency of waders that probe for food.

Manipulating pan profile has considerable potential for maximizing foraging area, by providing a complex and heterogeneous littoral zone. The basins of the Strandfontein pans could be graded to create an uneven and undulating profile, generating areas of varying water depth over a longer period (Rehisch, 1994; Sanders, 1999; Cowell &Taft, 2000; Sanders, 2000, Bolduc & Afton, 2008; Eglington *et al*., 2010). The changing littoral zone may also benefit invertebrates that prefer shallow water. The general profile of the pan banks should be graded to create a gentle sloping edge, as steep, rigid bank profiles have very narrow littoral zones.

Another minor threat for waders at Strandfontein specifically is water quality, as Strandfontein is largely eutrophic, and blooms of cyanobacteria and green algae occur regularly. *Microcystis* blooms reduce oxygen levels and are associated with death of waterbirds at sewage works or in sewage-polluted water bodies (Murray & Hamilton, 2010). In 2012 - 2015 samples of dying waterbirds viz. Kelp Gulls (*Larus dominicanus*) and Haurtlaubs Gulls (*Chroicocephalus hartlaubii*) found at Strandfontein were examined by veterinarians and it was concluded that *Clostridium botulinum* blooms had produced a neurotoxin which caused muscular paralysis. Outbreaks were typically associated with rising temperatures in summer, and high invertebrate abundance. These toxins typically affect gulls and ducks (Murray & Hamilton, 2010). However the possibility cannot be excluded that this may extend to waders, as aquatic invertebrates found near dead birds which had died of botulism, carried the toxin (Duncan & Jensen, 1976).

The Strandfontein Section of the False Bay Nature Reserve has Ramsar status and is a formal Important Birding Area (IBA) from Birdlife South Africa (Marenewick *et al*., 2015). Currently, unproclaimed City of Cape Town nature reserves operate under the Nature and Environmental Conservation Ordinance No 19 of 1972. This ordinance pertains to any wild fauna or flora and is applicable to any local, private or provincial established nature in the Western Cape. Strandfontein is yet to be proclaimed as a Protected Area (PA) by the Department of Environmental Affairs (DEA), and PA status would enable enforcement of stricter environmental legislation, particularly the National Environmental Protected Areas Act (NEMPAA). Protected area proclamation would protect the site in the long term against habitat transformation.

The integration of the recommended management interventions into the annual plan of operations for Strandfontein could enhance the value of the reserve as a resource for both resident and migrant waders. Subject to adaptive management it may offer a reliable feeding site in the event that climate change impacts the quality of satellite inland waterbodies in terms of the needs of migrants. Further investigation into the effect of *Typha capensis* beds on the invertebrate resources and their availability to waders needs to be undertaken. However, negative association of these beds with wader numbers suggests that current management actions of eliminating these reedbeds would benefit waders. Such management recommendations should be drafted and included into the IRMP for the broader False Bay Nature Reserve in order for connecting wetlands (Zeekoevlei and

Rondevlei) to optimise wader habitat (Russel *et al*., 2014; Taft *et al*., 2008). Incorporating most of the suggested managing manipulations into an Environmental Management Plan (EMP) for local City of Cape Town Municipality wastewater treatment works facilities (Athlone, Mitchells Plain, and Macassar) in order to potentially mitigate habitat loss and possible impacts of local climate change on feeding habitat for migrant waders.

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APPENDICES

Appendix1. Bird counts from the 11 plots at Pan P1 at Strandfontein Section of the False Bay Nature Reserve from July 2014 to June 2015.

		Wader species										
Plot	**Date**	*Burhinus vermiculatus*	*Calidris minuta*	*Charadrius hiaticula*	*Charadrius pecuarius*	*Charadrius tricollaris*	*Gallinago nigripennis*	*Himantopus Himantopus*	*Philomachus pugnax*	*Recurvirostra avosetta*	*Tringa nebularia*	*Vanellus armatus*
B1	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
B1	15.07.2014	2	0	0	0	0	0	0	0	0	0	15
B1	21.07.2014	0	0	0	0	0	0	1	0	0	0	2
B1	06.08.2014	1	0	0	0	0	0	0	0	0	0	0
B1	18.08.2014	0	0	0	0	0	0	0	0	0	0	8
B1	10.09.2014	0	0	0	0	0	0	0	0	0	0	8
B1	15.09.2014	0	0	0	0	0	0	1	0	1	0	18
B1	14.10.2014	0	0	0	0	0	0	4	0	0	0	24
B1	20.10.2014	0	0	0	0	0	0	4	0	0	0	20
B1	30.10.2014	0	0	0	0	0	0	3	0	0	0	5
B1	10.11.2014	0	0	0	0	0	0	27	0	0	0	39
B1	18.11.2014	0	0	0	0	0	0	1	4	0	0	30
B1	10.12.2014	0	0	0	0	0	0	3	11	2	0	26
B1	17.12.2014	0	0	0	0	0	0	8	1	3	1	18
B1	13.01.2015	0	0	0	0	0	0	4	0	10	0	2
B1	19.01.2015	0	0	0	0	0	0	10	0	5	1	13
B1	10.02.2015	0	1	0	0	0	0	14	0	4	0	0
B1	16.02.2015	0	1	0	0	0	0	3	0	1	0	1
B1	24.02.2015	0	0	0	0	0	0	0	0	0	0	1
B1	11.03.2015	0	0	0	0	0	0	0	0	0	0	12
B1	16.03.2015	0	7	7	0	0	0	0	0	0	0	7
B1	27.03.2015	0	26	0	0	0	0	2	0	6	0	6
B1	08.04.2015	0	50	0	0	3	0	4	0	1	0	22
B1	13.04.2015	0	50	0	0	0	0	1	0	0	0	1
B1	22.04.2015	0	61	0	1	0	0	0	0	3	0	3
B1	13.05.2015	0	0	0	0	0	0	2	0	0	0	0
B1	21.05.2015	0	0	0	0	0	0	0	0	3	0	0
B1	25.05.2015	0	0	0	0	0	0	2	0	0	0	0
B1	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
B1	22.06.2015	0	0	0	0	0	0	0	0	0	0	0
C1	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C1	15.07.2014	0	0	0	0	0	0	1	0	0	0	0
C1	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
C1	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C1	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C1	10.09.2014	0	0	0	0	0	0	0	0	0	0	2
C1	15.09.2014	0	0	0	0	0	0	0	0	0	0	6

C1	14.10.2014	0	0	0	0	0	0	1	0	0	0	0
C1	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C1	30.10.2014	0	0	0	0	0	0	2	0	0	0	0
C1	10.11.2014	0	0	0	0	0	0	0	0	0	0	5
C1	18.11.2014	0	0	0	0	0	0	0	0	0	0	4
C1	10.12.2014	0	0	0	0	0	0	0	0	0	0	6
C1	17.12.2014	0	0	0	0	0	0	11	15	4	11	3
C1	13.01.2015	0	0	0	0	0	0	4	0	4	0	0
C1	19.01.2015	0	0	0	0	0	0	2	0	7	0	4
C1	10.02.2015	0	0	0	0	0	0	2	0	3	0	5
C1	16.02.2015	0	0	0	0	0	0	2	0	6	0	11
C1	24.02.2015	0	0	0	0	0	0	2	0	4	0	3
C1	11.03.2015	0	0	0	0	0	0	0	0	2	0	0
C1	16.03.2015	0	0	0	0	0	0	0	0	0	0	0
C1	27.03.2015	0	0	0	0	0	0	0	0	0	0	2
C1	08.04.2015	0	2	0	0	0	0	0	0	0	0	1
C1	13.04.2015	0	1	0	0	0	0	0	0	2	0	0
C1	22.04.2015	0	21	0	0	0	0	2	0	6	0	0
C1	13.05.2015	0	0	0	0	0	0	0	0	1	0	0
C1	21.05.2015	0	0	0	0	0	0	0	0	4	0	2
C1	25.05.2015	0	0	0	0	0	0	1	0	3	0	2
C1	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
C1	22.06.2015	0	0	0	0	0	0	3	0	4	0	0
B2	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
B2	15.07.2014	0	0	0	0	0	0	1	0	0	0	0
B2	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
B2	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
B2	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
B2	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
B2	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
B2	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
B2	20.10.2014	1	0	0	0	0	0	0	0	0	0	0
B2	30.10.2014	0	0	0	0	0	0	2	0	1	0	0
B2	10.11.2014	0	0	0	0	0	0	0	0	0	0	3
B2	18.11.2014	0	0	0	0	0	0	0	0	1	0	1
B2	10.12.2014	0	0	0	0	0	0	0	0	5	0	4
B2	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
B2	13.01.2015	0	0	0	0	0	0	0	0	1	0	0
B2	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
B2	10.02.2015	0	0	0	0	0	0	0	0	2	0	10
B2	16.02.2015	0	0	0	0	0	0	0	0	0	0	0
B2	24.02.2015	0	0	0	0	0	0	0	0	0	0	0
B2	11.03.2015	0	0	0	0	0	0	0	0	0	0	0
B2	16.03.2015	0	0	0	0	0	0	0	0	0	0	0
B2	27.03.2015	0	77	8	0	0	0	0	0	0	0	0
B2	08.04.2015	0	0	0	0	0	0	0	0	0	0	0

B2	13.04.2015	0	0	0	0	0	0	1	0	1	0	2
B2	22.04.2015	0	40	0	0	0	0	0	0	0	0	0
B2	13.05.2015	0	0	0	0	0	0	3	0	1	0	2
B2	21.05.2015	0	0	0	0	0	0	0	0	0	0	0
B2	25.05.2015	0	0	0	0	0	0	0	0	0	0	2
B2	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
B2	22.06.2015	0	0	0	0	0	0	0	0	1	0	0
C2	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C2	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
C2	21.07.2014	0	0	0	0	0	0	5	0	0	0	0
C2	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C2	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C2	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
C2	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
C2	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
C2	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C2	30.10.2014	0	0	0	0	0	0	0	0	2	0	0
C2	10.11.2014	0	0	0	0	0	0	0	0	0	0	0
C2	18.11.2014	0	0	0	0	0	0	0	0	0	0	4
C2	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
C2	17.12.2014	0	2	0	0	0	0	3	12	0	0	0
C2	13.01.2015	0	0	0	0	0	0	0	0	0	0	0
C2	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
C2	10.02.2015	0	0	0	0	0	0	0	0	4	0	4
C2	16.02.2015	0	0	0	0	0	0	0	0	0	0	0
C2	24.02.2015	0	0	0	0	0	0	2	0	5	0	1
C2	11.03.2015	0	1	0	0	0	0	2	0	2	0	2
C2	16.03.2015	0	0	0	0	0	0	1	0	2	0	0
C2	27.03.2015	0	19	0	0	0	0	1	0	2	0	2
C2	08.04.2015	2	0	0	0	0	0	0	0	0	0	2
C2	13.04.2015	0	0	0	0	0	0	0	0	0	0	0
C2	22.04.2015	0	0	0	0	0	0	0	0	0	0	2
C2	13.05.2015	0	0	0	0	0	0	2	0	11	0	0
C2	21.05.2015	0	0	0	0	0	0	0	0	2	0	0
C2	25.05.2015	0	0	0	0	0	0	0	0	0	0	0
C2	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
C2	22.06.2015	0	0	0	0	0	0	0	0	3	0	0
B3	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
B3	15.07.2014	0	0	0	0	0	0	1	0	0	0	0
B3	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
B3	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
B3	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
B3	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
B3	15.09.2014	0	0	0	0	0	0	0	0	0	0	2
B3	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
B3	20.10.2014	0	0	0	0	0	0	0	0	0	0	0

B3	30.10.2014	0	0	0	0	0	0	0	0	4	0	1
B3	10.11.2014	0	0	0	0	0	0	0	0	0	1	2
B3	18.11.2014	0	0	0	0	0	0	0	0	4	0	0
B3	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
B3	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
B3	13.01.2015	0	0	0	0	0	0	3	0	2	0	0
B3	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
B3	10.02.2015	0	0	0	0	0	0	7	0	2	0	0
B3	16.02.2015	0	0	0	0	0	0	9	0	2	0	2
B3	24.02.2015	0	0	0	0	0	0	2	0	0	0	0
B3	11.03.2015	0	0	0	0	0	0	3	0	2	0	0
B3	16.03.2015	0	0	0	0	0	0	4	0	2	0	0
B3	27.03.2015	0	0	0	0	0	0	3	0	1	0	2
B3	08.04.2015	0	0	0	0	0	0	0	0	0	0	2
B3	13.04.2015	0	0	0	0	0	0	3	0	0	0	0
B3	22.04.2015	0	0	0	0	0	0	0	0	0	0	3
B3	13.05.2015	0	0	0	0	0	0	0	0	0	0	0
B3	21.05.2015	0	0	0	0	0	0	0	0	0	0	0
B3	25.05.2015	0	0	0	0	0	0	3	0	1	0	2
B3	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
B3	22.06.2015	0	0	0	0	0	0	0	0	0	0	0
C3	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C3	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
C3	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
C3	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C3	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C3	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
C3	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
C3	14.10.2014	0	0	0	0	0	0	2	0	0	0	0
C3	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C3	30.10.2014	0	0	0	0	0	0	0	0	4	0	2
C3	10.11.2014	0	0	0	0	0	0	0	0	0	0	0
C3	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
C3	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
C3	17.12.2014	0	3	0	0	0	0	2	0	2	0	1
C3	13.01.2015	0	0	0	0	0	0	10	0	3	0	4
C3	19.01.2015	0	0	0	0	0	0	11	0	0	0	3
C3	10.02.2015	0	0	0	0	0	0	9	0	3	0	0
C3	16.02.2015	0	0	0	0	0	0	11	0	0	0	8
C3	24.02.2015	0	0	0	0	0	0	7	0	2	0	4
C3	11.03.2015	0	0	0	0	0	0	3	0	0	0	2
C3	16.03.2015	0	0	0	0	0	0	2	0	2	0	1
C3	27.03.2015	0	0	0	0	0	0	1	0	0	0	0
C3	08.04.2015	0	0	0	0	0	0	0	0	0	0	0
C3	13.04.2015	0	0	0	0	0	0	10	0	0	0	2
C3	22.04.2015	0	0	0	0	0	0	0	0	0	0	0

C3	13.05.2015	0	0	0	0	0	0	0	0	0	0	0
C3	21.05.2015	0	0	0	0	0	0	0	0	0	0	0
C3	25.05.2015	0	0	0	0	0	0	0	0	0	0	0
C3	19.06.2015	0	0	0	0	0	0	1	0	0	0	0
C3	22.06.2015	0	0	0	0	0	0	0	0	0	0	1
B4	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
B4	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
B4	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
B4	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
B4	18.08.2014	0	0	0	0	0	0	3	0	0	0	0
B4	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
B4	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
B4	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
B4	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
B4	30.10.2014	0	0	0	0	0	0	0	0	0	0	0
B4	10.11.2014	0	0	0	0	0	0	2	0	0	0	0
B4	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
B4	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
B4	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
B4	13.01.2015	0	0	0	0	0	0	1	0	0	0	1
B4	19.01.2015	0	0	0	0	0	0	4	0	0	0	0
B4	10.02.2015	0	0	0	0	0	0	7	0	2	0	3
B4	16.02.2015	0	0	0	0	0	0	0	0	0	0	0
B4	24.02.2015	0	0	0	0	0	0	2	0	1	1	2
B4	11.03.2015	0	0	0	0	0	0	3	0	1	0	0
B4	16.03.2015	0	0	0	0	0	0	7	0	0	0	1
B4	27.03.2015	0	0	0	0	0	0	3	0	0	0	0
B4	08.04.2015	0	0	0	0	0	0	0	0	0	0	2
B4	13.04.2015	0	0	0	0	0	0	8	0	0	0	0
B4	22.04.2015	0	0	0	0	0	0	0	0	0	0	0
B4	13.05.2015	0	0	0	0	0	0	0	0	1	0	0
B4	21.05.2015	0	0	0	0	0	0	0	0	1	0	1
B4	25.05.2015	0	0	0	0	0	0	0	0	0	0	0
B4	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
B4	22.06.2015	0	0	0	0	0	0	0	0	0	0	2
C4	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C4	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
C4	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
C4	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C4	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C4	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
C4	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
C4	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
C4	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C4	30.10.2014	0	0	0	0	0	0	0	0	0	0	0
C4	10.11.2014	0	0	0	0	0	0	0	0	0	0	0

C4	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
C4	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
C4	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
C4	13.01.2015	0	0	0	0	0	0	0	0	0	0	0
C4	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
C4	10.02.2015	0	0	0	0	0	0	2	0	3	0	0
C4	16.02.2015	0	0	0	0	0	0	1	0	4	0	0
C4	24.02.2015	0	0	0	0	0	0	2	0	0	0	2
C4	11.03.2015	0	0	0	0	0	0	3	0	1	0	0
C4	16.03.2015	0	0	0	0	0	0	0	0	0	0	2
C4	27.03.2015	0	0	0	0	0	0	4	0	2	0	1
C4	08.04.2015	0	0	0	0	0	0	5	0	1	0	0
C4	13.04.2015	0	0	1	0	0	0	0	0	0	0	0
C4	22.04.2015	0	0	0	0	0	0	3	0	1	0	0
C4	13.05.2015	0	0	0	0	0	0	0	0	0	0	2
C4	21.05.2015	0	0	0	0	0	0	2	0	2	0	2
C4	25.05.2015	0	0	0	0	0	0	0	0	0	0	0
C4	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
C4	22.06.2015	0	0	0	0	0	0	0	0	0	0	1
B5	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
B5	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
B5	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
B5	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
B5	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
B5	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
B5	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
B5	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
B5	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
B5	30.10.2014	0	0	0	0	0	0	0	0	0	0	0
B5	10.11.2014	0	0	0	0	0	0	0	0	0	0	0
B5	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
B5	10.12.2014	0	0	0	0	0	0	0	0	0	0	2
B5	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
B5	13.01.2015	0	0	0	0	0	0	0	0	0	0	0
B5	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
B5	10.02.2015	0	0	0	0	0	0	3	0	0	0	0
B5	16.02.2015	0	0	0	0	0	0	7	0	0	0	2
B5	24.02.2015	0	0	0	0	0	0	1	0	28	0	0
B5	11.03.2015	0	0	0	0	0	0	1	0	1	0	0
B5	16.03.2015	0	0	0	0	0	0	0	0	0	0	2
B5	27.03.2015	0	0	0	0	0	0	0	0	0	0	0
B5	08.04.2015	0	0	0	0	0	0	0	0	2	0	1
B5	13.04.2015	0	33	0	0	0	0	5	0	1	0	2
B5	22.04.2015	0	0	0	0	0	0	3	0	0	0	2
B5	13.05.2015	0	0	0	0	0	0	3	0	9	0	0
B5	21.05.2015	0	0	0	0	0	0	0	0	0	0	0

B5	25.05.2015	0	0	0	0	0	0	3	0	0	0	1
B5	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
B5	22.06.2015	0	0	0	0	0	0	0	0	3	0	0
C5	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C5	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
C5	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
C5	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C5	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C5	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
C5	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
C5	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
C5	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C5	30.10.2014	0	0	0	0	0	0	0	0	0	0	0
C5	10.11.2014	0	0	0	0	0	0	0	0	0	0	0
C5	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
C5	10.12.2014	0	0	0	0	0	0	0	0	0	0	0
C5	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
C5	13.01.2015	0	0	0	0	0	0	0	0	0	0	0
C5	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
C5	10.02.2015	0	0	0	0	0	0	0	0	0	0	0
C5	16.02.2015	0	0	0	0	0	0	8	0	3	0	0
C5	24.02.2015	0	0	0	0	0	0	0	0	0	0	0
C5	11.03.2015	0	0	0	0	0	0	2	0	3	0	0
C5	16.03.2015	0	30	0	0	0	0	0	0	9	0	0
C5	27.03.2015	0	13	0	0	0	0	3	0	2	0	0
C5	08.04.2015	0	0	0	0	0	0	4	0	7	0	3
C5	13.04.2015	0	10	0	0	0	0	7	0	6	0	1
C5	22.04.2015	0	3	0	0	0	0	3	0	9	0	3
C5	13.05.2015	0	0	0	0	0	0	4	0	0	0	0
C5	21.05.2015	0	0	0	0	0	2	2	0	7	0	0
C5	25.05.2015	0	0	0	0	3	0	0	0	6	0	4
C5	19.06.2015	0	0	0	0	0	0	9	0	0	0	0
C5	22.06.2015	0	0	0	0	0	0	1	0	3	0	0
C6	01.07.2014	0	0	0	0	0	0	0	0	0	0	0
C6	15.07.2014	0	0	0	0	0	0	0	0	0	0	0
C6	21.07.2014	0	0	0	0	0	0	0	0	0	0	0
C6	06.08.2014	0	0	0	0	0	0	0	0	0	0	0
C6	18.08.2014	0	0	0	0	0	0	0	0	0	0	0
C6	10.09.2014	0	0	0	0	0	0	0	0	0	0	0
C6	15.09.2014	0	0	0	0	0	0	0	0	0	0	0
C6	14.10.2014	0	0	0	0	0	0	0	0	0	0	0
C6	20.10.2014	0	0	0	0	0	0	0	0	0	0	0
C6	30.10.2014	0	0	0	0	0	0	0	0	0	0	0
C6	10.11.2014	0	0	0	0	0	0	0	0	0	0	0
C6	18.11.2014	0	0	0	0	0	0	0	0	0	0	0
C6	10.12.2014	0	0	0	0	0	0	0	0	0	0	0

C6	17.12.2014	0	0	0	0	0	0	0	0	0	0	0
C6	13.01.2015	0	0	0	0	0	0	0	0	0	0	0
C6	19.01.2015	0	0	0	0	0	0	0	0	0	0	0
C6	10.02.2015	0	0	0	0	0	0	0	0	0	0	0
C6	16.02.2015	0	0	0	0	0	0	0	0	0	0	0
C6	24.02.2015	0	0	0	0	0	0	0	0	0	0	0
C6	11.03.2015	0	0	0	0	0	0	0	0	0	0	2
C6	16.03.2015	0	20	0	0	0	0	0	0	0	0	2
C6	27.03.2015	0	10	0	0	0	0	0	0	0	0	2
C6	08.04.2015	0	0	0	0	0	0	2	0	0	0	2
C6	13.04.2015	0	40	8	0	0	0	0	0	0	0	0
C6	22.04.2015	0	0	0	0	0	0	0	0	0	0	1
C6	13.05.2015	0	0	0	0	0	0	0	0	0	0	0
C6	21.05.2015	0	0	0	0	0	0	0	0	0	0	0
C6	25.05.2015	0	0	0	0	0	0	0	0	0	0	2
C6	19.06.2015	0	0	0	0	0	0	0	0	0	0	0
C6	22.06.2015	0	0	0	0	0	0	0	0	0	0	0