ECOSYSTEM PROCESS AND FUNCTION OF TEMPORARY WETLANDS: BASELINE DATA FOR CLIMATE CHANGE PREDICTIONS

Report to the
WATER RESEARCH COMMISSION

by

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EXECUTIVE SUMMARY

BACKGROUND AND RATIONALE

According to Turok (2012) there will be an increase in the human population growth and urbanisation. In turn, there will be an increase in water supply demand consequently requiring high quality water supply. Parallel to the increasing demand of water supply, there is a predicted increase in temperature and variability of rainfall (e.g. geographically, periodicity and level of inundation), especially in South Africa (Schulze *et al.* 2011). Ephemeral wetlands are well adapted to cyclical drying and wetting conditions that include wide ranges in temperatures. The ephemeral wetlands in arid and semi-arid regions, or "drylands", are also adapted to the unpredictability and extremes changes in temperatures and inundation periodicity (Nouri *et al.* 2010). With the advent of global climate change, it is important to better understand the ecological function of these systems by drawing from this knowledge on how to better predict and manage permanent wetland systems with a view to understand how they cope and adapt to these climatic changes (e.g. shifts from permanent to seasonal or intermittent inundation). This project was designed to attempt to address gaps in knowledge such that it feeds into current water quality models that are designed to predict changes associated with global climate change by making them more robust. Especially as they pertain to temperature effects on lower trophic levels, specifically algal communities and its consequences for higher trophic levels. It also proposed to further address knowledge gaps in the ecological functioning of ephemeral wetlands in the Nelson Mandela Bay Municipality (NMBM) and responses to anthropogenic pressures.

AIMS OF THE PROJECT

There were five main aims of this project:

1. Determine rates of biogeochemical cycling generated by primary producers (micro- and macroflora) in temporary wetlands during different levels of inundation in order to better refine this process for use in global climate change models.

2. To examine lower trophic level relationships in temporary wetlands under different levels of inundation and link these to different climate change models.

3. To experimentally determine different temperature, water level and nutrient regimes that effect the growth and production of various algal taxa, for use and refinement in existing climate change and eutrophication models.

4. To model changes in ecosystem services derived from ephemeral wetlands in peri-urban and urban environments associated with changes in global climate.

5. To generate baseline information on selected urban wetland(s) in support of ICLEI-LAB (International Council for Local Environmental Initiatives – Local Action for Biodiversity) and Nelson Mandela Bay Metropolitan environmental services division to enable better management decisions to be made for rehabilitation, restoration and maintenance of wetlands in the face of climate change.

METHODS

In order to address the research aims above a range of desktop, field and laboratory methods was used. Aims 1, 2 and 3 were based on experiments and/or field data collection. Aim 4 was desktop and Aim 5 a combination of field data collection and desktop analyses.

Microcosm experiments using local sediments and either ambient or controlled temperatures were chosen for the laboratory component. These were set up in a series of experiments used to build up knowledge about conducting the experiments themselves as well as manipulating temperature to ascertain the response of primary and secondary producers to different conditions. The regional climate change projections of Schulze *et al.* (2011) were used to inform the experiments, using their intermediate projections of temperature based on the IPCC-TGICA (2007) A2 scenario. The field data collection was designed to obtain baseline data under current natural circumstances and track water chemistry, nutrients, algal biomass and community, as well as faunal abundance and community structure. Field data give a baseline under an ambient set of conditions with which to compare the microcosm experiments that focused on temperatures that may occur based on the current regional climate change projections. A desktop exercise was used to assess, in a generic way, what sort of ecosystem services ephemeral wetlands within the NMBM currently provide using WET-EcoServices (Kotze *et al.* 2009). The knowledge gained from the microcosm experiments, current field data collection and data from Schael *et al.* (2015), two scenarios were then assessed for these systems under extreme climatic shifts, one of a prolonged dry period or drought, and the other under extreme flooding.

As part of this project, an urban wetland of special concern was assessed using both WET-Health (Macfarlane *et al.* 2007) and WET-EcoServices (Kotze *et al.* 2009). It was initially designed as once-off data collection in 2015 for baseline data to aid a possible rehabilitation project on the wetland by Working for Wetlands (WfW), however follow-up data was collected in 2016 and parts slightly expanded from basic water quality to add heavy metals as well. Another urban site, one less heavily impacted until recent developments, Bridgemeade was also added in 2016. Although these urban sites are now permanent to semi-permanent wetlands hydrologically, they were historically ephemeral systems and are able to give an idea of the trajectory of wetland health and service provision in the face of urbanization.

RESULTS AND DISCUSSION

The results of the preliminary experiments using a range of sediments from different common hydrogeomorphological (HGM) units (Ollis *et al.* 2013) and average minimum (17°C) and moderate (24°C) summer temperatures to show that 1) successful hatching from the sediment egg bank was possible and 2) temperature had an effect on the numbers of hatchlings and species. These also demonstrated that sediments from depression and wetland flat HGM units were well suited to these experiments and had evolved an egg and seed bank whereas seeps did not have this same mechanism for repopulation after inundation/re-wetting and developed a different type of invertebrate community. It was then from there that sediments were then chosen for further experiments to more directly address the aims of the project.

Aim 1

The original approach to this aim was to conduct flux experiments in the field at different stages of the wetlands inundation cycle (newly inundated, middle and mature to drying). However, due to lack of rain within that time period, the sediment was then brought into the lab and a microcosm approach was adopted and flux experiments were conducted as if in the field, in three stages post inundation. Changes in water chemistry were evident across the experimental period, with electrical conductivity (EC)/total dissolved solids (TDS) steadily increasing along with dissolved oxygen, pH fluctuated, lowest during the initial inundation stage and highest during stage 2. Most of the nutrients measured were at their highest initially, and then by stage 2 were almost completely depleted (TOxN, SRP and TP). Net flux for each stage showed a continuous flux into the water column of TN, but a net influx back into the sediment of TOxN and Ammonia in stages 1 and 3, but efflux in stage 2, although the differences in the rates are subtle. TP and SRP both demonstrated an efflux during stages 1 and 2 (with SRP at a greater rate for up-take) and influx at stage 3. The rate of silica influx back into the sediment during stage 2 was the greatest change of any of the nutrients followed. Flux rates were variable, but also demonstrated the fairly fast release of nutrients into the water column upon inundation, rapid cycling from the sediment to the water column, and the rapid decline of nutrients over time in a closed system. The wetlands in this area are essential closed systems at a larger scale. This initial release of nutrients into the water column is essential for the development of phytoplankton which then serves as the first food of the newly hatched invertebrates. This cycling helps form both the algal community (benthic and water column) which in terms allows for the invertebrate community to develop and thrive. Shifts in temperature affect the water chemistry and rates of cycling and release of nutrients, therefore the communities that will colonise. More detailed methods and results are reported as part of a MSc thesis in Lategan (2016).

Aim 2

Field data collected over two separate inundation events in the winter and spring of 2015 demonstrated the importance of inundation duration and water depth on the development of both the algal and aquatic faunal communities. It also demonstrated the importance of temperature and timing of these events. The first inundation event lasted between 14-18 days and measured water temperatures ranged between 10-17°C, whereas the second event lasted between 24-34 days with temperatures ranging between 17-26°C. These differences had an effect on the timing of peak phytoplankton and MPB biomass and shift of peak invertebrate densities. In the cooler temperatures phytoplankton peaked early with a steady decline in biomass with MPBs with a slight lag behind phytoplankton, but with an early peak and slower decline. Both algal communities had < 20 species for the duration. The invertebrate densities had about an 8-10 day lag behind the algae, with an initial peak of mosquito larvae, which quickly declined and were replaced by branchiopods and zooplankton. In the longer, warmer inundation event, phytoplankton biomass had small “peaks” about every 8 days, with a dominance of chlorophytes and euglenoids. The MPB biomass increased asymptotically until peaking at day 20 and then again on day 28. There were more algal species found as compared to the first inundation event, with 35 phytoplankton species (chlorophyte dominated) and 32 MPBs (diatom dominated) Invertebrate densities peaked almost immediately with large numbers of mosquito larvae, but again, they were quickly replaced by branchiopods and zooplankton. Peak algal biomass as the wetlands were drying are most likely a reflection of the concentration of all the biomass into a smaller area.

Aim 3

The series of temperature experiments demonstrated what was seen in the field in terms of biomass patterns, but more clearly showed the shift in algal biomass and timing of hatching. Although chlorophytes were still dominant in the phytoplankton community at the lower temperatures, when presented with temperatures ≥ 34°C, cyanophytes and filamentous green algae dominated. When presented with very high temperatures and nutrient additions the community was reduced to one main species *Anabaena* sp. and little else. Species diversity sharply declined with temperature, and invertebrates perished at temperatures >40°C. Patterns of invertebrates in the more reasonable temperatures demonstrated an initial "bloom" of small zooplankters, rotifers, then replaced by cladocerans and ostracods. It is important to note that in the field, rotifers were seldom collected as the sampling mesh size was too course, therefore they were most likely missed. Higher temperatures are less successful in hatching large branchiopods that are indicative of ephemeral wetlands. These species are sensitive to temperature and tend to hatch in colder temperatures. This has implications on the long term viability of these species with both intermediate and distant future increased temperatures. However, their resilience lies in their ability of their egg bank to remain dormant for long periods of time under hostile conditions, the key would be to maintain the egg bank.

Laboratory experiments such as these enable us to manipulate key factors that then enable us to see the community shifts we would not otherwise given the unpredictability of rainfall and inundation in this region, as well as not having control as to what season and temperatures occur. Pairing field and experimental data gives us the ability to empirically predict shift that could occur with shifts in rainfall and temperature.

Aim 4

Combining experimental evidence, current field data collection and past data an assessment of the ecosystem services given by the small, ephemeral wetlands indicative of NMBM currently and project what could happen with different climate change scenarios to those services. Two extremes of the climate change spectrum were chosen to demonstrate this change, severe prolonged drought conditions and a period of prolonged wet or flood conditions. As temperature is generally more predictable and "stable", rainfall, especially in this region is not. The projections are for a 10-20% decrease in mean annual rainfall and a 2.2% increase in evapotranspiration levels (IPCC 2014a), however, one of the main difficulties is that day to day changes cannot be projected and it is widely thought that the variance or patchiness in rainfall in location, amount, duration and timing will increase (Schulze 2011, IPCC 2014c). In a region that already has high variance in rainfall on a monthly level, it is thought that there will be more extreme weather patterns with extended periods of drought and extended periods of flooding and less seasonal predictability (Schulze 2011, IPCC 2014c). Ephemeral wetlands, especially depression and wetland flat HGMs do not, in pristine, "normal" conditions give a high level of direct ecosystem services. They generally provide more indirect services such as biodiversity, tourism and research. As such, systems that have "low" direct human value, will show a decline in those services they do provide in prolonged dry phases. In prolonged wet phased and floods their direct and indirect services do generally increase. However there is a potential that the water quality, under increased urbanization (nutrient input) and higher temperatures could decline and produce large amounts of nuisance cyanophyte blooms and increased nuisance insects which could carry water borne diseases.

Aim 5

Urban wetlands are difficult to assess using current tools. In terms of WET-Health, almost all urban systems will have low present ecological state (PES) scores but virtue of their surrounding catchment. It is difficult to see health scores above a D, and most will fall to an F, as did both Pond 6 and Bridgemeade. These health score do not adequately take ecological functioning into account, and both these sites show some level of functioning, as well as ecosystem service provisioning. Both have an abundance of bird life with high diversity of birds, and a potential for recreational and tourism value.

Local and national government, NGOs and the community must come together to address this adequately. Until then, under increasing temperatures and more extreme weather events, this wetland will only continue to deteriorate and become more of a public health hazard than a service provider, if an effort to truly rehabilitate it is not made.

Bridgemeade, which also scored a PES of F, is not nearly as directly impacted as Pond 6, but the recent housing developments, lack of buffer and structural changes made to the wetland have affected its functioning. Great effort has been put in by community groups and WfW to clear the alien vegetation around the wetland, which has increased its aesthetic and recreational value. The nutrient levels and general water quality, however, is still not very good because of the run-off from the new housing developments. The constructed "retention pond" and drainage canal, built to protect the developments need to be rethought.

Urban wetlands have the potential to provide important ecosystem services, but they need to be managed properly and respected by the surrounding community. Education, as well as community and government partnerships must be fostered in order to aid in the management and protection of these wetlands and potential water resources.

GENERAL CONCLUSIONS

Although it was not possible to integrate out data into an existing model for water quality under changing climate regimes, we now have data on the response to both algae and invertebrates to different temperature regimes, which we have not had for our area to date. These data can be the beginning to developing some predictive capability around nutrient up-take and primary productivity and what will happen to the ecosystems within our water supply dams. How resilient will these ecosystems be and how can we put mitigation in place to prevent or control harmful algal blooms and decreasing water quality.

RECOMMENDATIONS FOR FUTURE RESEARCH

Experimental data is important to understand different components of ecosystems in a controlled way. They also can aid in developing and or feeing into models to predict the effect of change on ecosystems. However, along with needing more experimental data, we also need more long-term monitoring data. Models developed in the US, Europe and Australia are built around long-term data sets of climate and hydrological data, coupled with ecological data. South Africa has developed long term data on rivers, major dams and some major catchments, but there is very little long-term data on wetlands, big or small, permanent or ephemeral. We also lack consistent data collection across all regions of the country. We have made great strides in mapping and locating wetlands around the

country and that is now being iteratively refined and strengthened. Now that we have a much better idea of where our wetlands are and what type they are likely to be, we need to move towards understanding their interactions within the landscape, hydrological processes and ecological structure. In order to understand and make predictions about what can, and or will, happen to wetlands in a changing global climate, we need to get a better idea of how they are functioning now.

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WRC Reference Committee Members:

Dr Brilliant Petja	Water Research Commission (Chairman)
Dr Stanley Liphadzi	Water Research Commission (Interim Chairman)
Prof Guy Bate	Nelson Mandela Metropolitan University
Ms Nancy Job	University of the Free State
Dr Martin Ferreira	Jeffares & Green/University of Johannesburg
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Mr Geert Grobler	Department of Water and Sanitation
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Ms Christine Engelbrecht	Agricultural Research Commission

Research of this nature cannot be conducted without the contributions of many different people at different levels. Ms Jodi Lategan (2013-16, MSc Botany) contributed nutrient flux laboratory experiment data; Sixolile Mazwane, Riaan Weitz, and Emma Mangwiro contributed to temperature and sediment hatching experiments; Asandiswa Tuswa and Michael Larsen, urban wetland data. Dr Brigitte Melly (post-doc 2016), provided maps and GIS data for sites, modified from Schael *et al.* (2015). Their contributions were essential to the project, though portions of data collected were directly for fulfilment of their degree requirements, others were solely for the purpose of moving this project forward. Their time and dedication is acknowledged and greatly appreciated. Please see Appendix 9 for details of output from the project.

The project team would like to thank the following people, and institutions for their contribution:

NMMU Reserve Manager Craig Breedt for allowing us easy access to the NMMU Campus sites.

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Dr Nelson Miranda – gastropod type specimen identifications.

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TABLE OF CONTENTS

ABBREVIATIONS AND ACRONYMS

CBA	Critical Biodiversity Areas
Chl *a*	Chlorophyll *a*
CS	The Classification System by Ollis *et al.* (2013) revised from the NWCS (SANBI 2009)
DEA	Department of Environmental Affairs (National Government)
DO	Dissolved Oxygen measured in mg/L or %
DWS	Department of Water Affairs and Sanitation, also known as **DWAF** prior to 2009 and **DWA**, 2009-2014.
EC	Electrical Conductivity measured in µS/ cm or mS/m
ET	Evapotranspiration
GC	Global climate change
HGM	Hydrogeomorphic
ICLEI – LAB	International Council for Local Environmental Initiatives – Local Action for Biodiversity
IPCC	International Panel on Climate Change
MAP	Mean annual precipitation
MAR	Mean annual runoff
MDS	Multi-Dimensional Scaling
MOSS	Metropolitan Open Space System
MPB	Microphytobenthic algae
NFEPA	National Freshwater Ecosystem Priority Areas
NH_4^+	Ammonium
NMB	Nelson Mandela Bay (the Study Area)
NMBM	Nelson Mandela Bay Municipality
NMMU	Nelson Mandela Metropolitan University
NO_2^-	Nitrite
NWA	National Water Act of South Africa (Act 38 of 1998)
OM	Organic matter
PE	Port Elizabeth
PES	Present ecological state
RCM	Regional Climate Model

RCP	Representative Concentration Pathways
SA	South Africa
SANBI	South African National Biodiversity Institute
SAWS	South African Weather Service
Si	Silica
SIMPER	Similarity of Percentages
SRP	Soluble Reactive Phosphorus
TDS	Total Dissolved Solids measured in mg/L
TN	Total Nitrogen, organic and inorganic
TOxN	Total Oxidised Nitrogen
TP	Total Phosphorus
VB	Valley Bottom wetland
WfW	Working for Wetlands Programme

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

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1 GENERAL INTRODUCTION

1.1 Study Background

According to Turok (2012) there will be an increase in the human population growth and urbanisation. In turn, there will be an increase in demand for water supply, and for that water supply to be of a high quality. Parallel to this increasing water supply demand, is a predicted concomitant increase in air temperature associated with a variability of rainfall (i.e. across geographic areas, unpredictable periodicity and quantities), especially in South Africa (Schulze *et al.* 2011). Ephemeral wetlands are well adapted to the cyclical drying and wetting climatic events that exhibit ranges of temperatures and other environmental conditions. Ephemeral wetlands in arid and semi-arid regions, or "drylands", are also adapted to the unpredictability and extremes of temperatures and duration of inundation (Nouri *et al.* 2010). With the advent of global climate change, it is important to better understand the ecological function of these systems in order to draw knowledge so as to better predict and best manage how permanent wetland systems. Such information will be instrumental in understanding how these systems may adapt to changes in climatic such as possible shifts from permanent to seasonal or even intermittent inundation. This project was designed to try and address gaps in data and knowledge so that current water quality models, designed to predict changes with global climate change can be more robust. Especially as they pertain to temperature effects on lower trophic levels, specifically algal communities. It also proposes to further address our knowledge gaps of ecological functioning of ephemeral wetlands in NMB and their responses to anthropogenic pressures.

1.2 Project Aims and Objectives

The aims for project K5/2348 were as follows:

1. Determine rates of biogeochemical cycling generated by primary producers (micro- and macro-flora) in temporary wetlands during different levels of inundation in order to better refine this process for use in global climate change models.

2. To examine lower trophic level relationships in temporary wetlands under different levels of inundation and link these to different climate change models.

3. To experimentally determine different temperature, water level and nutrient regimes that effect the growth and production of various algal taxa, for use and refinement in existing climate change and eutrophication models.

4. To model changes in ecosystem services derived from ephemeral wetlands in peri-urban and urban environments associated with changes in global climate.

5. To generate baseline information on selected urban wetland(s) in support of ICLEI – LAB (International Council for Local Environmental Initiatives – Local Action for Biodiversity) and Nelson Mandela Bay Metropolitan environmental services division to enable better management decisions to be made for rehabilitation, restoration and maintenance of wetlands in the face of climate change.

1.3 General Approach

In order to address the research aims a range of desktop, field and laboratory methods were used. These have been broken up into two parts, with Aims 1 and 2, primarily experiment and/or field data collection driven and Aim 3, solely laboratory based, grouped into Part 1 (Chapters 3-6). Aim 4 was desktop based using available knowledge and WET-EcoServices (Kotze *et al.* 2009) and Aim 5, consisted of field data collection and desktop methods, grouped into Part 2 (Chapters 7-8). Chapters 1-2 consist of general introduction and background that is common to both sections.

This project has built upon base-line data collected by Schael *et al.* (2015) in WRC project K5/2181. Of the 1712 wetlands throughout the NMB that have been delineated (Figure 1.1) and typed through the Classification System (Ollis *et al.* 2013, Ollis *et al.* 2015), 45 sites were sampled once-off, with 6 sites monitored at different time intervals dependent on inundation periodicity and time of sampling (Schael *et al.* 2015).

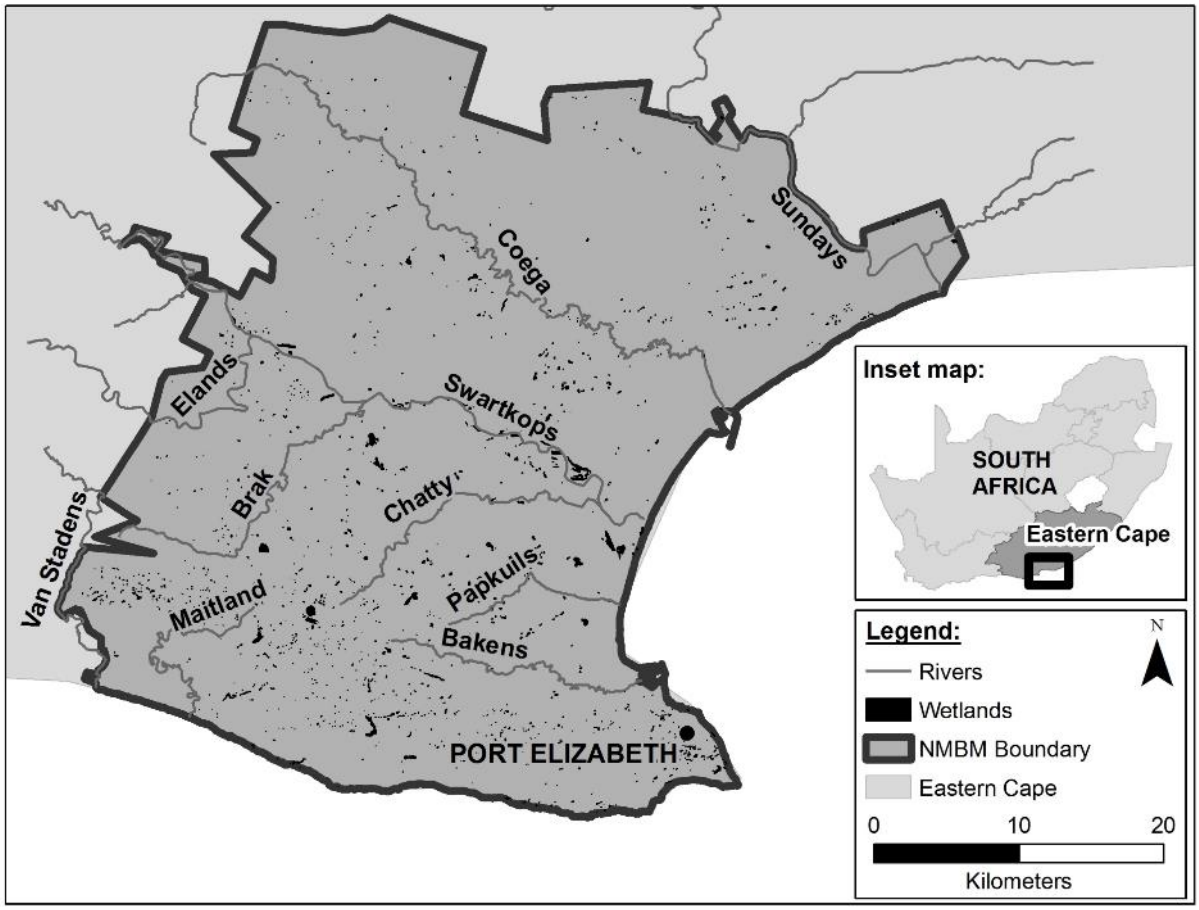

Figure 1.1 Distribution of wetlands in the NMB area from Schael *et al.* (2015).

The bulk of the wetlands identified in Schael *et al.* (2015) were small, < 1 ha in size, and of those the majority are ≤ 0.1 ha (Figure 1.2). Given the rainfall distribution and evaporation rates (Figure 1.3) within the NMBM, it has been observed that the majority of these wetlands are ephemeral, with a varied hydrological pattern of inundation periodicity (e.g. seasonal, intermittent and episodic) (Schael *et al.* 2015). The few larger wetlands, ≥ 5 ha are semi-permanent to permanent, and generally found in more urban areas or exist as commercial salt pans. These small systems, although

seeming to contribute little in overall water surface area to the municipality, do provide corridors and connectivity of water resources through the region. These small systems, which can be inundated at different times (dependent on where, when and how much rain falls) and varying duration, provide "island" refuges for aquatic vegetation and fauna (e.g. birds and breeding amphibians). They also provide for extra watering areas for livestock or game, and when the water subsides, the remaining vegetation becomes import for grazing mammals.

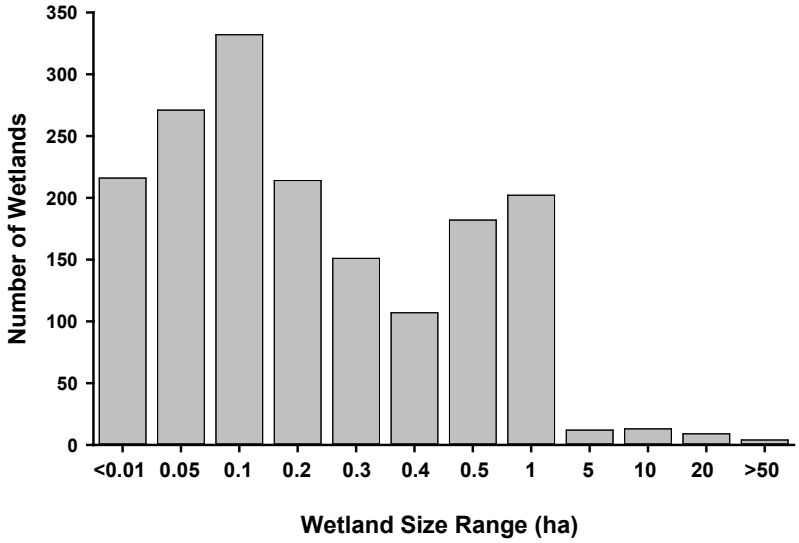

Figure 1.2 **Frequency distribution of numbers of wetlands in different size ranges from Schael *et al.* (2015). Note that the y-axis scale is not equally divided.**

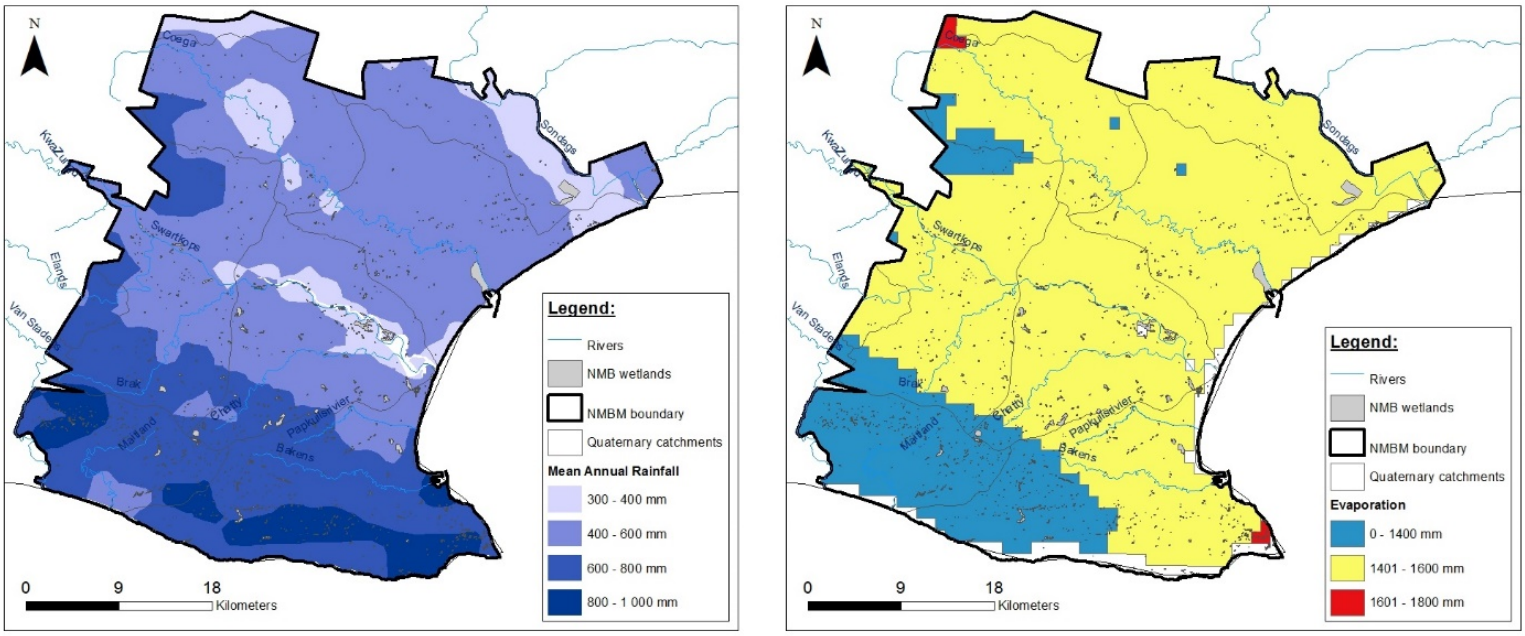

Figure 1.3 **Mean annual rainfall distribution (left) and evaporation rater per year (right) within the Nelson Mandela Bay Metropolitan area (NMBM). Data from South African Weather Service.**

This project was designed to provide data on lower trophic levels and their response to temperature and inundation duration in order for these data to better inform, and possibly be incorporated into, models around water quality/eutrophication and changes due to predicted climate change scenarios. It was also hoped that by examining the resilience and adaptability to extreme weather patterns of ephemeral wetland systems could aid in our understanding of what to expect if permanent wetlands/lakes/dams were to shift hydrological states to semi-permanent, seasonal or intermittent systems. What this project was not designed to do was to create a new model or models, but to be informed by current climate change model projections for our region. The predictions based on Schulze *et al.* (2011) were used as the context for the experiments conducted (further explained in Chapter 2). There is a paucity of baseline data for use in current water quality and eutrophication models, especially for data and information from within South Africa, and very little long term (>20 yrs.) monitoring datasets, especially for wetlands.

2 CLIMATE CHANGE, WATER RESOURCES AND WETLANDS

Evidence for anthropomorphic driven climate change has become the accepted reality within the scientific community (IPCC 2014c). Southern Africa is seen to be one of the most vulnerable regions (IPCC 2014b), and already scarce water resources will be hardest hit if the predicted reduction in rainfall, increased temperatures and evapotranspiration occur at predicted levels (Schulze 2011, IPCC 2014a, Kusangaya *et al.* 2014). An extensive review of the current state of climate trends and regional predictive models has been integrated by Kusangaya *et al.* (2014) and not necessarily reiterated here. The major conclusion drawn from Schulze (2007), Schulze (2011), Kusangaya *et al.* (2014) and others is that we are already experiencing a warming trend and greater variability (increasing extremes) in weather patterns. The predictions of regional climate models (RCMs) show that these increases will alter regional climatic patterns to the point where water resources in terms of availability, accessibility, quality and demand will be negatively altered (Kusangaya *et al.* 2014). The (IPCC 2014a) clearly outlined a key risk to the region's economy, government and societal structure is stress on water resources due to over exploitation, increasing degradation and increasing demand by a growing population. This risk is intensified in drought prone regions such as many parts of Southern Africa.

Schulze *et al.* (2011) RCMs were based on the (IPCC-TGICA 2007) AR4 (annual report 4) emissions scenarios. These scenarios integrated projections of increased CO_2 emissions integrated with different socioeconomic factors such as regionalization versus globalization, mitigation and emissions cutting, along with different rates of population and economic growth (IPCC-TGICA 2007). The A2, or "a very heterogeneous world with a continuously increasing global population and a regionally oriented economic growth that is more fragmented and slower than in other scenarios" (IPCC-TGICA 2007), is not the worst case scenario, but shows little effect of emission mitigation and regional expansion. This scenario assumes a 2.0-5.4°C increase in global temperatures. These scenarios have since been replaced in AR5 by representative concentration pathways (RCP) which have removed the socioeconomic aspect to the scenarios and focused on different emission rates and mitigations strategies (IPCC 2014a). These range from the "best case" scenario of high levels of global mitigation of emissions, RCP 2.6 to the "worst case" scenario of RCP 8.5, where nothing is done to alter the current trajectory of emissions.

The predictions for the "intermediate future", 2046-2065 and the "distant future", 2081-2100 of Schulze *et al.* (2011) were used for NMBM temperature regime changes. In the RCM, January maximum temperatures were predicted to increase between 2-2.5°C in the intermediate future and between 4-5°C in the more distance future. Schulze *et al.* (2011) used their "present" as 1971-1990 (January average maximum 28-30°C), for comparison to their RCM output. In Figure 2.1, we plot a smoothed long term monthly average temperatures in the NMBM area from 1950 to 2016 (data sourced from SA Weather Service) to demonstrate the monthly and seasonal temperature pattern currently experienced in the NMBM.

Overall averages over this period of average daily temperature, maximum and minimum temperatures are presented in Table 2.1. The average daily temperature in NMBM over the year is 17.7°C, with maximum of 22.6°C and minimum of 12.9°C, demonstrating the mild temperature and narrow temperature ranges and low variability (Figure 2.1 and Table 2.1).

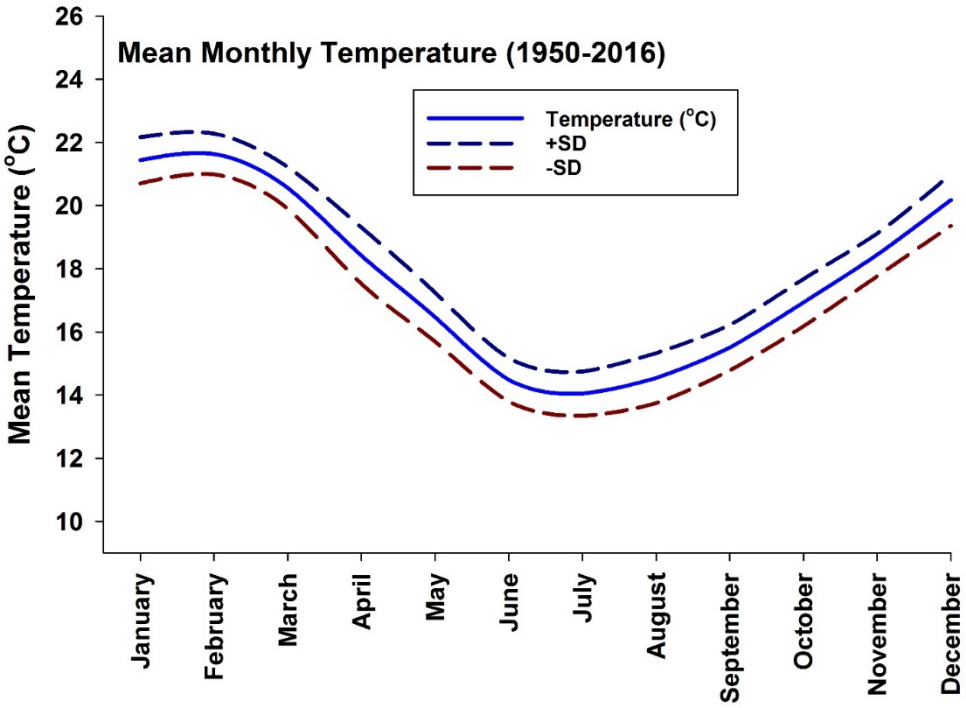

Figure 2.1 **Spline curve of average monthly temperatures and the standard deviation around the mean for long term data in the Nelson Mandela Bay Municipality (NMBM). Data from South African (SA) Weather Service.**

Table 2.1 **Daily average, maximum (max) and minimum (min) temperatures in Nelson Mandela Bay Municipality (NMBM) averaged monthly over a 66 year period, 1950-2016 (Data from SA Weather Service).**

	Daily Temperatures		
	Average	**Max**	**Min**
Year	17.7	22.6	12.9
January	21.4	25.7	17.4
July	14.1	19.9	7.9

Rainfall in the NMBM is highly variable and has been considered "bimodal" with peaks in precipitation in late spring and early autumn (Figure 2.2). This was true when examining total monthly rainfall data between 1960 and 1990, however when examining data between 1991 and 2016 the pattern is drastically altered (Figure 2.2). There are several "peaks" and valleys throughout the year of rainfall using this data set, with the main peak in August, or late winter (Figure 2.2). The other peaks are in mid-spring early autumn and winter (Figure 2.2). Combining the entire available data record, there is a flattening out of the rainfall pattern (Figures 2.2 and 2.3). The actual pattern lies with the variability around these monthly means, rather than the means themselves. Figure 2.3 demonstrates that standard deviation around the mean monthly totals, and shows that September and March are the least predictable and most variable rainfall months throughout the year.

This variability demonstrates the difficulties in predicting changes in daily or monthly rainfall patterns with RCM. The general estimates are on a yearly basis where Schulze *et al.* (2011) predict a 10% (intermediate) to 20% (distant) decrease in rainfall. The IPCC (2014a) for years 2081-2100 RCP 2.6 predicts a 10% decrease in precipitation and a 20% decrease at an RCP 8.5 level.

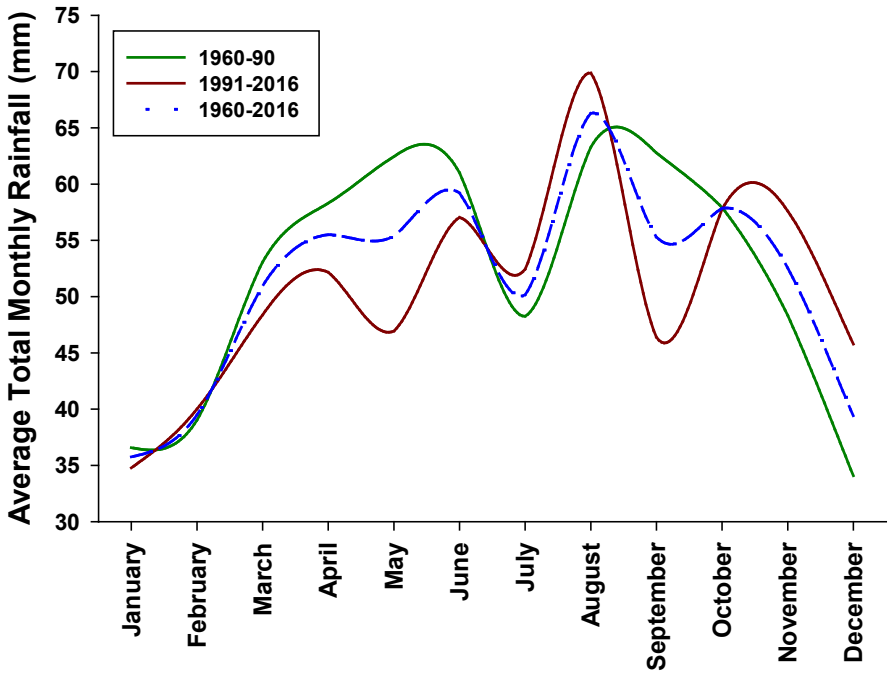

Figure 2.2 **Spline smoothed monthly average rainfall for different periods of the rainfall record for Nelson Mandela Bay Municipality (NMBM). Data from SA Weather Service.**

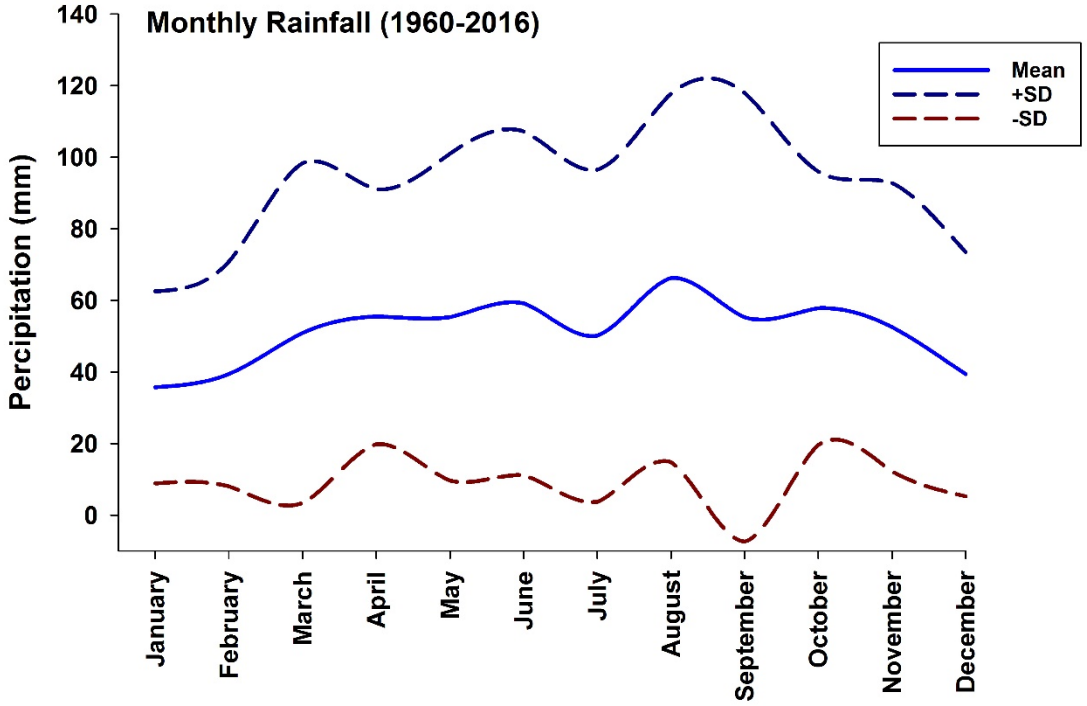

Figure 2.3 **Spline smoothed mean monthly rainfall between 1960 and 2016 with standard deviation plotted. Data from SA Weather Service.**

Given the large variance of precipitation within a month and between months, it is difficult to predict when these changes will take place. The coefficient of variation (CV) of monthly rainfall is extremely high for NMBM (Figure 2.4), with values ranging from 65% in April to > 100% in September. What this does demonstrate is that rainfall will present itself in "extreme" events of drought and floods, but have an average overall reduction in mean rainfall in the future.

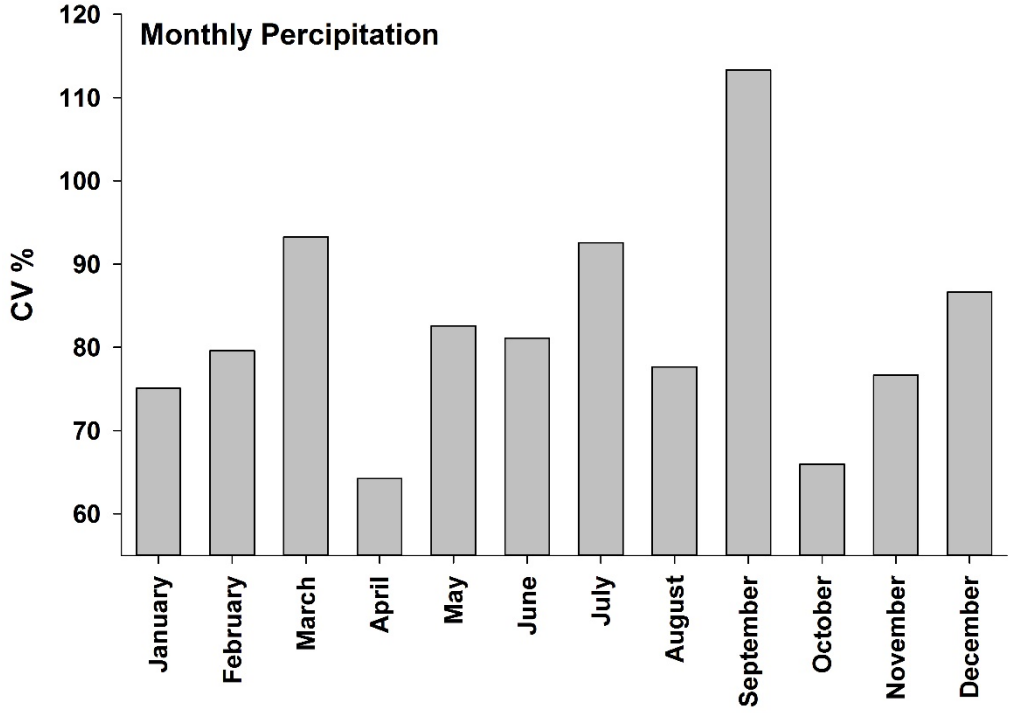

Figure 2.4 Coefficient of variation (CV expressed as percent) around the mean for average total monthly rainfall recorded by the SA Weather Service in the NMBM between 1960 and 2016.

What this temperature, rainfall and RCM prediction does is provide a background to the temperatures chosen for the experiments reported on and the context for what stresses wetlands and other water resources will be under in the future. Using Schulze *et al.* (2011) temperature scenarios, and applying Rivers-Moore *et al.* (2005) equation for predicting water temperature from maximum daily temperatures we plotted the different possible temperature scenarios for NMBM (Figure 2.5).

Several studies in North America (Poiani *et al.* 1996, Allan and Johnson 1997, Stanley *et al.* 2003, Pyke 2004, 2005a, b, Pyke *et al.* 2005, Voldseth *et al.* 2007, Johnson *et al.* 2010, Zhang *et al.* 2011), Australia (Brock *et al.* 2003, Nielsen *et al.* 2003, Brock *et al.* 2005, Roessink *et al.* 2008) and Europe (Folke *et al.* 2002, Mooij *et al.* 2007, Mooij *et al.* 2008, Jeppesen *et al.* 2010, Moss *et al.* 2010, Fragoso *et al.* 2011, Moss *et al.* 2011) have been evaluating the links between these changes in temperature, rainfall patterns and water quality/ecosystem response. They have much more hydrological long term data and have been able to develop models to aid in predictions of climate change.

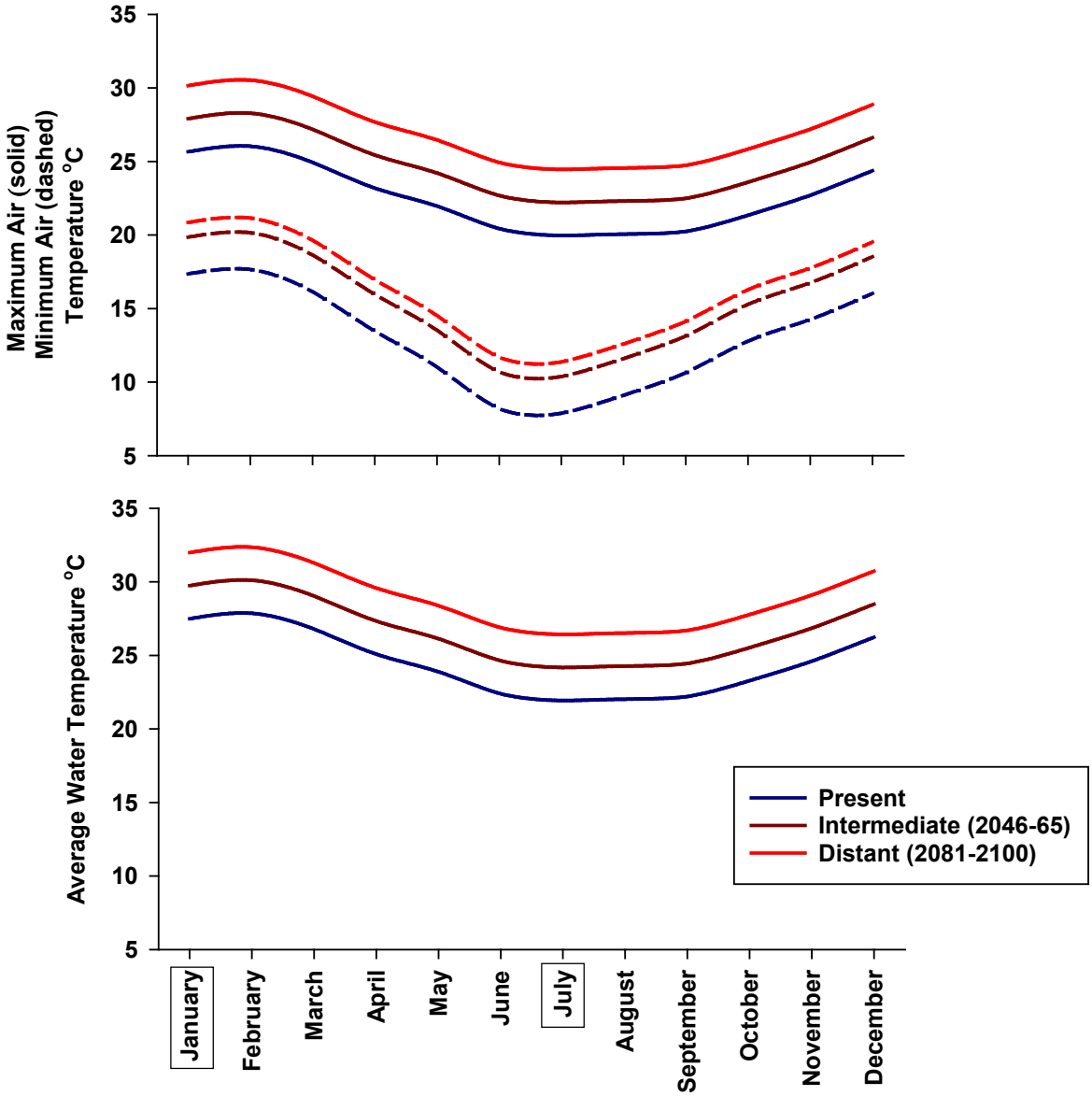

Figure 2.5 **Upper panel represents mean maximum (solid line) and minimum (dashed line) daily air temperatures (present: 1950-2016, data from SA Weather Service), and predicted intermediate and distant future temperatures. Lower panel represents calculated average water temperatures for the same three scenarios. Predictions from January maximum and July minimum, boxed (Schulze *et al.* 2011).**

PART 1

EXPERIMENTAL AND FIELD DATA

3 EXPERIMENTAL APPROACH AND METHODS

An important component of any type of model is to have primary data to guide, develop and enhance the applicability and accuracy of the output. In order to make predictions about how a certain parameter will respond under different conditions, such as increase in temperature, it is important to experimentally determine the response in a controlled environment. Although an experimental model can never completely replicate natural conditions and attendant responses, it does enable us to understand the general ecophysiological responses of plants and animals to different environmental parameters. Having a range of expected responses to a parameter, such as temperature, by a suite of taxa allows us to make better predictions on how an ecosystem or community could respond given those changes.

In order to address the different Aims, several different approaches were used. This chapter is broken up into the various components of each experiment or field data collection. General equipment used for recording physicochemical data collection and laboratory processing are discussed here. Unless specifically mentioned in relevant sections, equipment and methods were used in both field and laboratory data collection and processing.

All physicochemical measurements (temperature, pH, dissolved oxygen (DO), electrical conductivity (EC), salinity and total dissolved solids (TDS)) were recorded *in situ* with a YSI multiprobe, Crison multimeter, and/or Hanna multiprobe. Continuous water temperature measurements for microcosm experiments were logged and recorded using HOBO U20-001 water level and temperature data loggers, air temperature and relative humidity was logged with HOBO U10-003 loggers, all data were recorded at 15 min or 30 min intervals. Laboratory processing methods are discussed in Section 4.3.

3.1 Microcosm Experiments

Given ephemeral wetlands are dependent on enough sustained precipitation in order to become inundated, laboratory experiments were planned to address research aims as much as possible. Sediment was collected from a wide range of previously sampled sites (Schael *et al.* 2015) for use in a range of microcosm experiments, both small 2 L containers (2 L Exp. 1 and 2, Plates 1 and 2) and larger 30 L aquaria (FLUX, TExp1, TExp2 and TExp3, Sections 3.1.1 and 3.1.2). During 2013-14 and 2016 the NMBM was in a very dry phase, and as such monitoring sites were dry, therefore the approach to addressing the aims with regard to a field approach needed to be shifted toward laboratory experiments as outlined in this section. There was more rainfall in 2015, allowing for field data collection (Section 4.2), but not adequate enough for field experimentation.

3.1.1 Container Microcosms, HGM, and Temperatures

Sediments collected from 25 different wetlands and 3 different wetland types (depression (11), seep (8), wetland flat (7)) from around the NMBM region (Figure 3.1) were used in total (see Appendix Table 1 for location and site code details). Sediments were taken from the top 10 cm of soil cores collected as part of a previous project (Schael *et al.* 2015). Hatching experiments for these sediment samples were conducted at two different constant temperatures in 2 L containers (2L Exp1 = 17° and 2L Exp2 = 24°C respectively) under continuous light conditions in order to encourage optimal algal growth and invertebrate hatching (Brendonck 1996, Ketley 2007, Henri *et al.* 2014).

The 2L Exp1 experiment used 19 different sediments, 3 replicates, for a total of 57 experimental units and 2L Exp2 used 18, also with 3 replicates (54 units) (Plates 3.1 and 3.2). Sediments that did not produce hatchlings in the first experiment were not used in the second one, and other sites were added (Appendix Table1).

Plate 3.1 Dry sediment in experiment containers before inundation with distilled water.

Plate 3.2 Random placement of replicate sediment containers in temperature and light controlled cabinet, and inundated with distilled water.

The 2L Exp2 incorporated sampling for algal biomass (chl *a*), as well as invertebrates. Invertebrates were counted and removed daily. Every four days, along with the physicochemical data collection, 50 ml of water was collected from 2 randomly chosen replicates for chl *a* analysis. Distilled water at the same experimental temperature was added back into each container to replace what was removed. The invertebrates for each replicate were preserved in 70% ethanol after being allowed to mature for 1 to 2 weeks post removal.

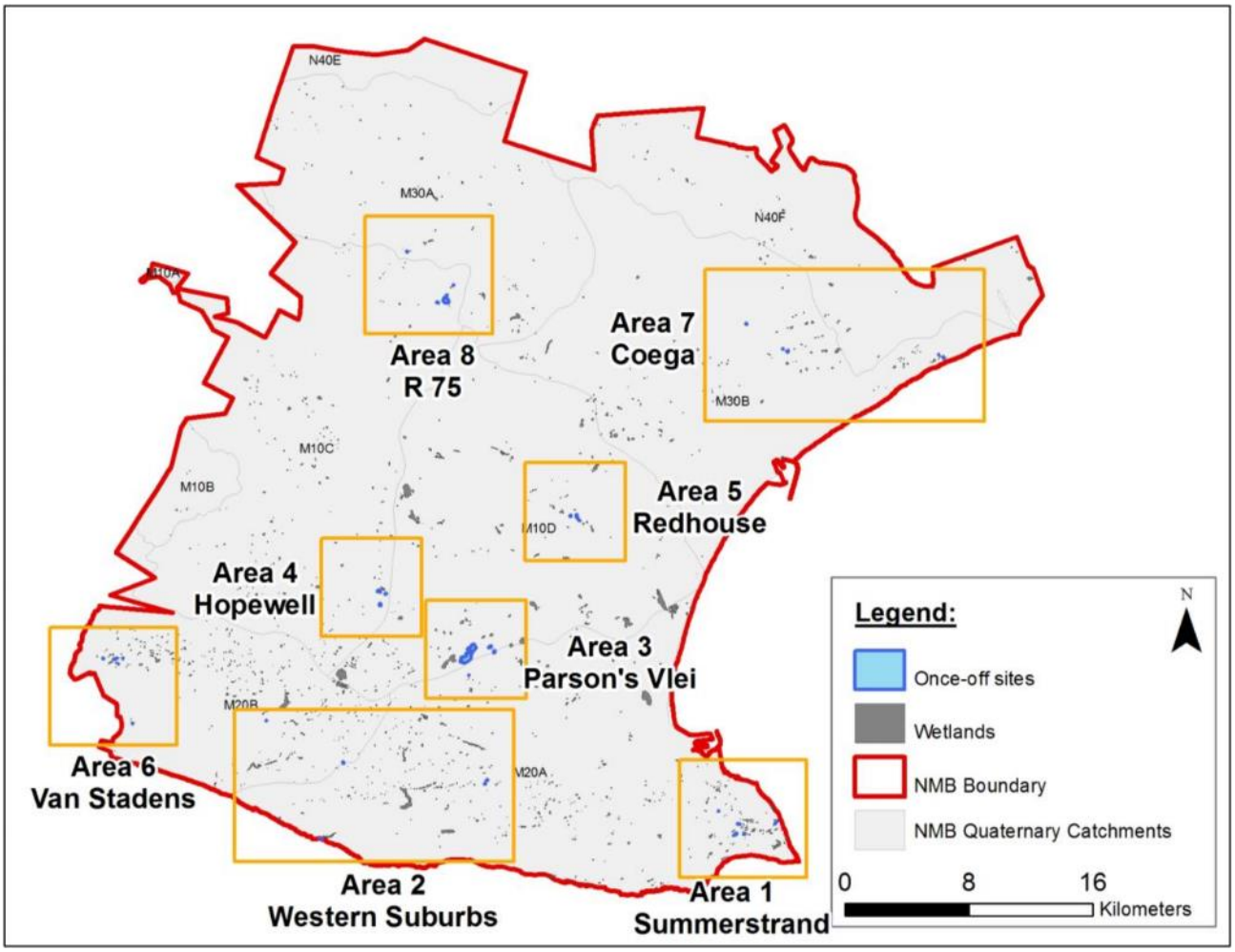

Figure 3.1 **Map of the different areas in NMB of sampled wetland sites in Schael *et al.* (2015), with sediment samples collected for use in current project as listed in Table 4.1.**

3.1.2 Geochemical Flux

The monitored ephemeral wetlands in the NMMU campus nature reserve had not received enough rainfall to sufficiently inundate them in order to conduct geochemical cycling experiments in the field as originally planned (e.g. site 1641b, Appendix Table 1). Therefore, we conducted the experiment using 30 L glass aquarium tanks as microcosms with sediment collected and dried from NMMU campus wetland 1641b, FLUX (Plate 3.3). This was done in different phases of the maturation of the sediment/water interface within each tank. The flux and cycling of nutrients (Nitrogen and Phosphorus) within a 24-h period was measured 72 h post tank sediment inundation. This was to see how quickly nutrients were released from the sediments into the overlying water column and made available to microalgae. The second 24-h experiment was conducted 13 days post inundation, providing a "mid-way" level maturity of the microcosm, and the last was conducted on day 28. These short-term experiments were designed to examine the fine-scale geochemical flux and address Aim 1.

In the FLUX studies , six tanks, 45 x 30 x 32 cm, were used and fitted with a glass cover that was sealed in place with marine type silicone during the geochemical flux experiments (Plates 3.3 and 3.4). Each cover had a 5 cm diameter central hole for ease of sampling, covered with a glass petri tray between sampling periods. There was also a small hole fitted with tubing and clamped for sampling water with a syringe will keeping the chamber sealed. The tanks were kept in ambient temperature and light conditions, outside under a shade covered area for the duration of the experiment (23 January-20 February 2015). Two tanks including their covers were painted black and used as dark control treatments. Approximately 6 kg of collected, sieved and dried sediment was used per tank and settled in each tank for a week before the addition of water. Each tank was filled with 30 L distilled water simulating its inundation and left for three days before the flux experiment began. Physicochemical parameters were recorded every 3 hours and two 50 ml replicate water samples were taken and filtered for nutrient analyses. All water was collected via a syringe through the tubing at a set distance from the sediment in each tank and replaced with an equal volume of distilled water kept at ambient temperature conditions. After the 24 h period was completed, samples for phytoplankton and MPB biomass were collected (TExp1).

Plate 3.3 **Outdoor tank set-up on day 3 for both geochemical cycling and germination / hatching experiments showing 3 of the "light" tanks and 2 of the "dark" tank controls.**

Plate 3.4 Outdoor tank set-up on during 24-h flux sampling for geochemical cycling experiments. Arrow shows dark treatment.

Benthic flux was calculated using the following equation adapted from Bartoli *et al.* (2003) and Pratihary *et al.* (2009)

$$F_x = \frac{(C_f - C_i) \times V}{A \times t}$$

Where:

F_x = Flux of the x species (mg.m^{-2}.h^{-1}). Positive values indicate flux out of the sediment and into the water column (efflux); and negative values the opposite (flux).

C_f = Final concentration of x (mg L^{-1})

C_i = Initial concentration of x (mg L^{-1})

V = Volume of water (l)

A = Surface area of sediment (m^2)

t = Incubation time/sampling interval (h)

The flux of the dark (control) tanks was then subtracted from the light tanks to determine up-take or release of nutrients by the autotroph community.

Plate 3.5 Outdoor tank experiment, dark tanks exposed to light post flux experiment, on day 16 post inundation.

3.1.3 Tank Microcosms with Temperature and Nutrient Addition

Upon the completion of the first 24-h flux monitoring, the experimental chambers were monitored every 4 days for 28 days for TExp1. Water for nutrients, microalgal chl *a*, microalgal community structure was collect and invertebrate hatchlings were sampled. The covers remained on, but the silicone was removed in order to sample the MPBs with a 10 mm diameter corer and dark control tanks were exposed to light (Plate 3.5). After the last sampling, water was drained, the macroalga *Chara* sp. and any other seedlings were collected and transplanted. All remaining sediment was mixed and dried for subsequent experiments.

Whilst the first germination/hatching tank experiment (TExp1) was done with ambient temperature and light conditions, the subsequent microcosm experiments were done under controlled light and temperature conditions. In TExp2, two temperatures were chosen, "normal" mean summer time temperature of 24°C and a prolonged higher temperature of 34°C representing extreme increase at an intermediate time period (2046-65) climate change models in IPCC-TGICA (2007) A2 scenario (Schulze *et al.* 2011), see discussion in Chapter 2. Two growth chambers were used, one set at 24°C and the other at 34°C. Due to experimental chamber malfunction 24°C was not achieved and rather a 26°C average was used that was still within the summer temperature parameters. The tanks as described in the FLUX experiments were used, but with an additional sediment from another depression wetland (RH1, Table 3.1). Each growth chamber was set up with two replicate tanks of each sediment type along with one control with no sediment added, only distilled water. Dried sediment from 1641b was used along with dried and sieved sediment from RH1, approximately 6 kg of each sediment was used per tank. As per previous experiments, 30 L of

distilled water was used to inundate the sediment in each tank. Sampling occurred every 4 days from inundation as per previous sections.

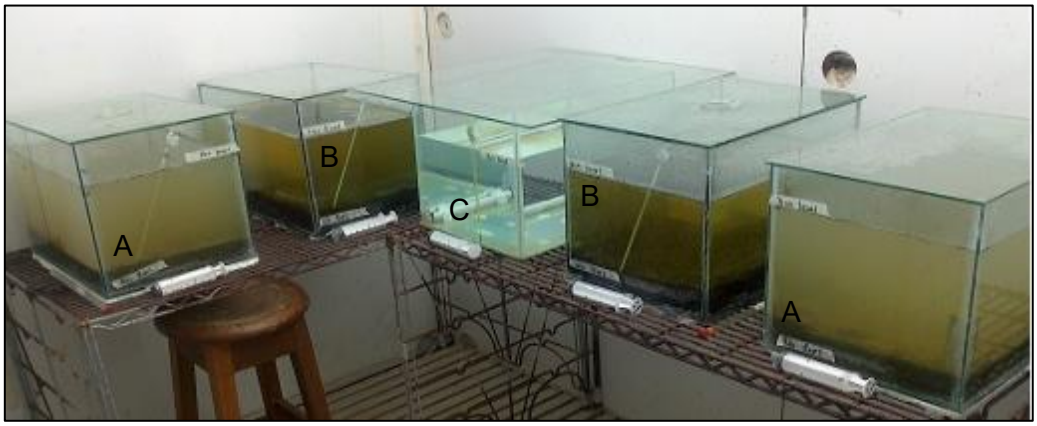

Plate 3.6 Experimental set-up for temperature experiments, showing the different sediments: A) RH1, B) 1641b and C) control tank.

The last experiment, TExp3, was done with a combination of nutrient addition and higher temperature in a temperature and light controlled chamber. It was determined from Schael *et al.* (2015) that the limiting nutrient in sediments and wetlands was phosphorus (P) with a background level of total phosphorus for this wetland sediment was determined to be ~25 µg L^{-1}, therefore this experiment tested the influence of increased P and the resulting influence on the algal and invertebrate communities. This was also coupled with predicted increase in summer temperatures of 34°C (see Chapter 2). Sediment from 1641b was used as per previous experiments with approximately 6 kg of dry sediment placed in each sediment treatment microcosm. The treatments were set up as a random block design of four replicates each of: Control tanks with distilled water only (C); sediment only (S); with 3 x phosphorus (~75 µg L^{-1}, 3P) addition; and 6 x phosphorus (~150 µg L^{-1}, 6P) addition to sediment treatments respectively. Sampling occurred as described for the previous experiments, however was discontinued on day 14 because of a malfunction in the growth chamber. Nutrients were added with inundation and then weekly until end of the experiment.

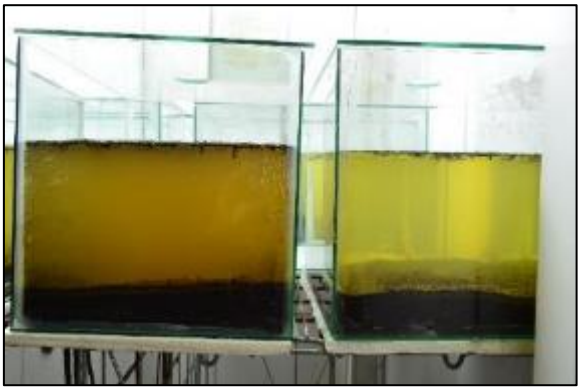

Plate 3.7 Portion of the experimental set-up in the growth chamber for the nutrient enrichment experiment.

3.2 Ephemeral Wetland Field Sampling

To address biological responses to inundation timing and periodicity, and support the laboratory experiments, data collection was done post inundation of small, ephemeral wetland sites on NMMU's campus reserve. One previously sampled site from Schael *et al.* (2015) was chosen (wetland flat 1593) as well two additional campus sites (wetlands flat, 1594 and dune depressions, 1765 [referred to as 1592] and 1641d) (see Appendix Table 1 for locations, Figure 3.2) that had not been sampled prior to this study. Physicochemical data were recorded, 2 replicate water samples of 500 ml was collected for nutrients and 1 L for biomass (Chl *a*) each were collected and processed in the lab as per Section 3.1, when enough water was present. Three to five replicate MPB cores (22 mm in diameter) for biomass determinate were collected dependent on substrate availability. One sample each was collected and preserved for phytoplankton and MPB community structure determination (Section 3.1). Invertebrates were collected using a 250 µm mesh kick-net for one 2 minute sweep throughout the wetland, when enough water present. In cases where the size and depth of inundation had decreased, qualitative samples of invertebrates were collected. Sites (except 1641d) were sampled at the end of July 2015 every 3-4 days post inundation until each site dried, 18 days later for a total of 5 sampling trips. The wetlands dried for a period of 2.5 weeks before the next significant rainfall event re-inundated the sites at the end of August 2015

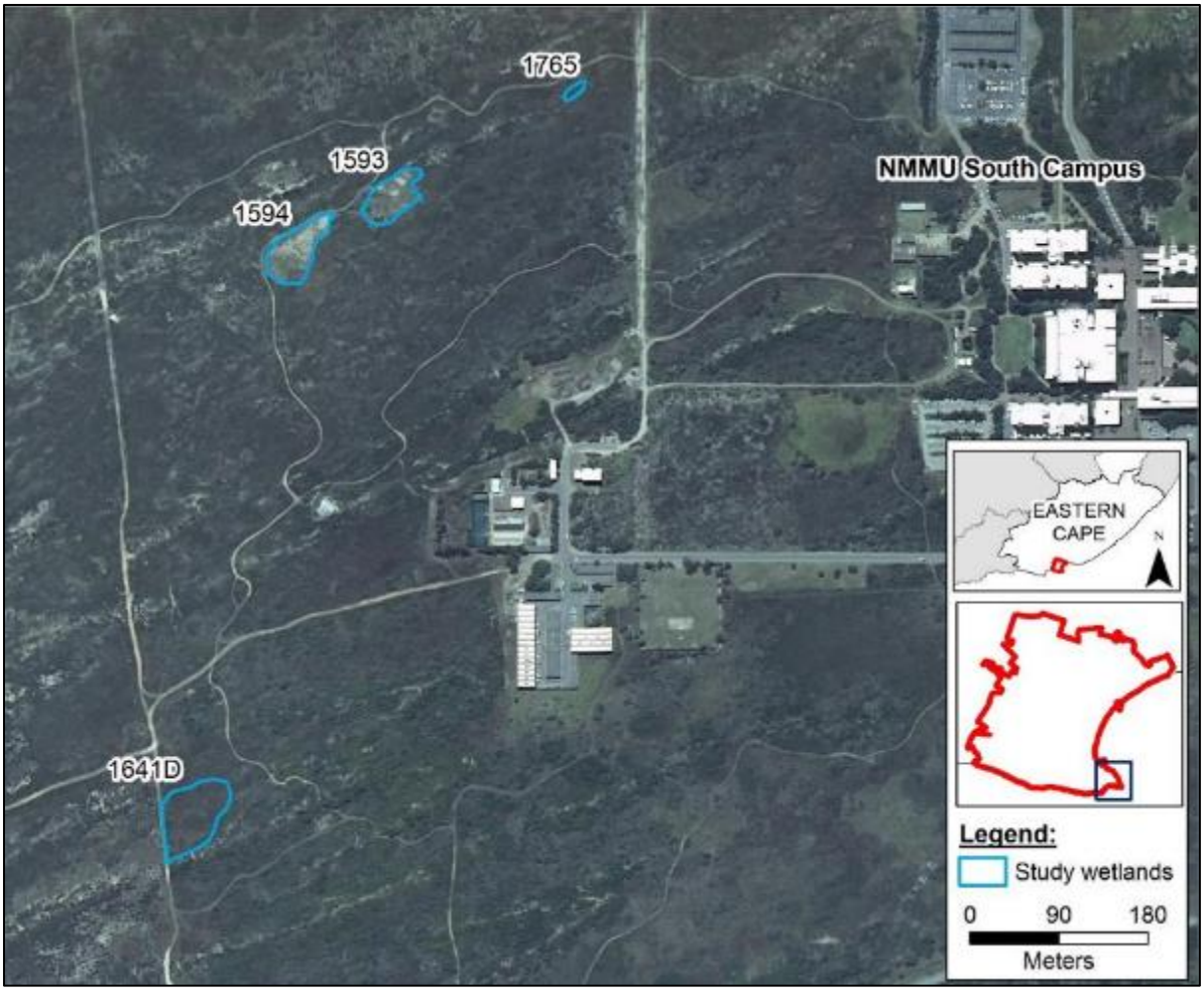

Figure 3.2 Study wetlands within the reserve, buildings as in A, wetlands 1765 (1592), 1593, 1594 and 1641d.

All four sites were sampled in the next inundation period at the end of August post inundation every 4 days until they dried, between 28-32 days later. Physical measurements were made at each sampling time at longest and widest points, GPS points were recorded along the wetted edge of each site when large enough. Depth was measured at the deepest point.

3.3 Laboratory Processing

Although some parameters were collected *in situ*, many abiotic and biotic samples were brought back to the laboratory for processing. All water and soil samples were analyzed at NMMU by members of the project team, unless otherwise specified. Water samples collected for nutrients were not sent to nationally accredited laboratories for analysis due to the generally low nutrient levels in the samples that were mostly below the detection limits of external laboratories.

Phytoplankton and microphytobenthic (MBP) algal biomass was measured in terms of chlorophyll *a* (chl *a*). Collected water (volume determined by field or experiment collection) was filtered through GF/C filters unless otherwise stated. Chl *a* was extracted from the water column and sediment using 95% Ethanol and absorbance read at 665 nm using a spectrophotometer. Water samples collected for nutrient analysis were filtered through a 0.45 µm membrane filters then frozen until further analysis could be conducted. Total Nitrogen, Nitrate, Nitrite, Ammonium, Soluble Reactive Phosphorous, Total Phosphorous and Silica were analyzed according to the methods laid out for each parameter by Strickland and Parsons (1972), Bate and Heelas (1975)and Wetzel and Likens (1991), further details of these methods can be found in Schael *et al.* (2015).

Soil samples collected from cores in the field (Pond 6 and Bridgemeade) were brought to the lab for analysis of standard soil chemistry and composition, moisture content, organic content and electrical conductivity (EC) (Wentworth 1922, Strickland and Parsons 1972, Sparks *et al.* 1996). Three replicates of approximately 10 to 15 g of sediment per field sample were used to determine the soil moisture content (Black 1965). The samples were weighed and placed in an oven at 40°C to dry for 48 hours. The samples were then re-weighed to determine the percentage moisture content. The dried samples used to determine sediment moisture were then used for the percentage organic matter, which was calculated using the loss on ignition method (ashing) (Smith and Atkinson 1975, Briggs 1977). The soil samples were ashed in a muffle furnace at 550°C for 6 hours. The percentage organic matter was then calculated as the difference between the ashed weight and the dry weight.

Electrical Conductivity (EC) in the sediment was measured as an indicator of salinity as EC increases proportionally with salt concentration. EC was calculated using the methods described in The Non-Affiliated Soil Analysis Work Committee (1990). Soil samples were air dried and de-ionised water was added to 250 g of soil until a saturated paste was formed. The amount of de-ionised water added was noted and the paste was left to stand for at least one hour before filtering. The soil paste was then filtered through a Whatman no. 1 filter paper through a Buchner funnel, using suction. The filtrate was collected in a test tube and measured using a hand held Crison Conductivity Meter 524. The solution was also used to measure the pH of the extracted solution using a RE 357 Microprocessor pH meter calibrated to 4.7, 7 and 10.

4 MICROCOSOM EXPERIMENTAL RESULTS

4.1 Hatching Success by HGM and temperature

In the first experiment (2L-Exp1) there was an overall 63% success rate of invertebrate hatching from all of the sediments. All of the depressions (8), 3 of the 4 wetland flats, and only 2 of the 8 seep sediments had some hatching (Table 4.1). Overall success in the second experiment (2L-Exp2) was 77%. In 2L-Exp2, only 2 seep sediments were included, one from 2L-Exp1, both did have some success in hatching (Table 4.1). There were three new depression sediments and two more wetland flat sediments added to 2L-Exp2, of these two depression and one flat site did not have hatchlings (Details of sediment sites and hatching numbers in Appendix 1). In terms of the HGM types, Depressions were most successful overall with the highest number of hatchlings and best success rate (Figure 4.1). Wetland flats were as successful as Depressions, but with lower numbers of hatchlings (Figures 4.1 and 4.2). Seeps overall showed little to no hatching success with the exception of 2 sites, with one site contributing >1000 hatchlings in 2L-Exp2 to boost the overall mean (Figures 4.1 and 4.2) Numbers of hatchlings were much greater in the higher temperature 2L-Exp2 than in 2L-Exp1 for all but one of the repeated sediments (Figure 4.2). For instance, site 1310 on its own had 100 times more hatchlings at the higher temperature (Appendix 2). Hatching success was variable in between replicates within sites, as demonstrated by the standard deviations in Figures 4.1 and 4.2.

Table 4.1 The hatching success rate for each experimental temperature (17 and 24°C for 2LExp1 and 2LExp2 respectively) by HGM type and experiment number. The N = number of sediments used for each HGM type per experiment.

	Hatching Success (%)			
HGM Type	N	**2L-Exp1**	N	**2L-Exp2**
Depression	8	100	10	70
Wetland Flat	4	75	7	86
Seep	7	14	2	100

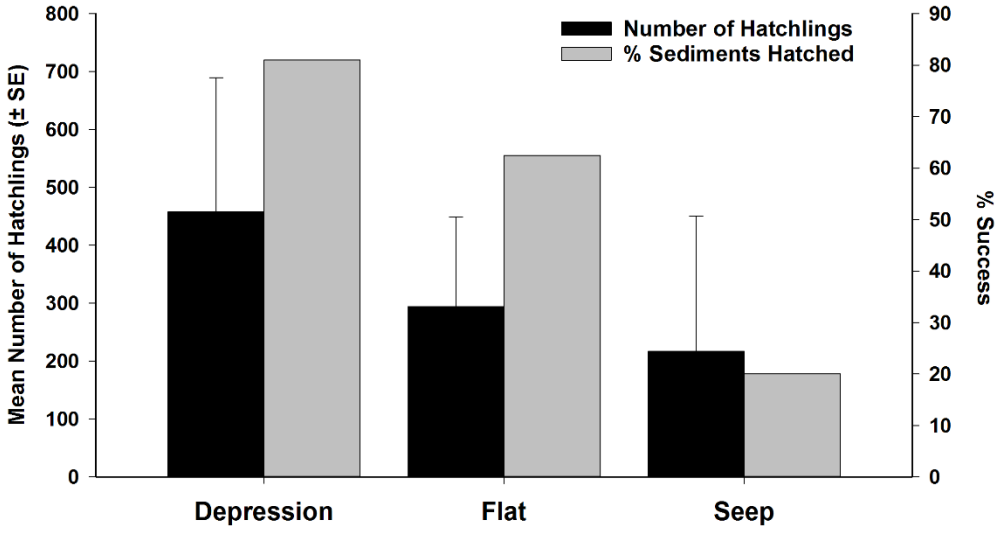

Figure 4.1 The overall mean number of hatchlings (±SE) and hatching success rate for both temperature experiments per HGM type.

Table 4.2 The range and mean ±SD for physicochemical parameters, temperature, pH, electrical conductivity (EC), dissolved oxygen (DO) and total dissolved solids (TDS) for the experiments in growth chambers set at 17° and 24°C (2L-Exp1 and 2L-Exp2 respectively).

	2L-Exp1				2L-Exp2			
Parameter	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**
Temperature (°C)	17.8	20.8	18.7	0.7	22.6	25.6	24.5	0.8
pH	4.4	8.0	6.9	0.6	5.7	8.5	7.4	0.3
Salinity (PSU)	0.0	0.3	0.06	0.07	0.0	0.2	0.04	0.04
EC (µS/cm)	7.1	499.0	106.0	125.5	5.3	437.0	77.2	67.6
DO (mg/L)	6.1	12.7	10.2	1.2	1.8	10.8	7.1	0.8
TDS (mg/L)	14.2	1004.0	211.2	251.6	10.6	878.0	155.7	135.7

The major branchiopod crustaceans (generally obligate ephemeral wetland taxa: *Branchiopodopsis* hodgsonii (Anostraca), *Leptestheria rubidgei* (Conchostraca) and *Triops granarius* (Notostraca), Cladocera (*Daphnia pulex, Simocephalus sp. Moina belli, D. dolichocephala, Alonella excusa*), copepods (*Lovenula falsifera, Metadiaptomus capensis*)and ostracods (Cyprididae spp.) were present in both experiments. The dominant algal groups present were cyanophytes (cyanobacteria), filamentous green algae (Chlorophyta, *Spirogyra* sp.) and the macroalga, *Chara* sp., was present within several of the replicates for various sites.

Algal biomass (Chl *a*) peaked within the first 12 days post inundation, depending from which wetland the sediment belonged and subsequently dropped to levels below 50 μgL^{-1} from day 16 onward (Figure 4.3). This initial algal biomass peak within the first four to 12 days fits the lag in hatching of invertebrates very well, in that most sediments with hatchlings only really start to produce large number of invertebrates after day 12. There was high variability between replicates in some cases, and only 2 of 3 were randomly selected for analysis. As no nutrients were added to the experiments over the 28 days, it would be expected that the early bloom of algae would deplete the nutrients inherent within the sediments under these experimental conditions, and furthermore for grazing pressure to also have an effect as the invertebrate population increased.

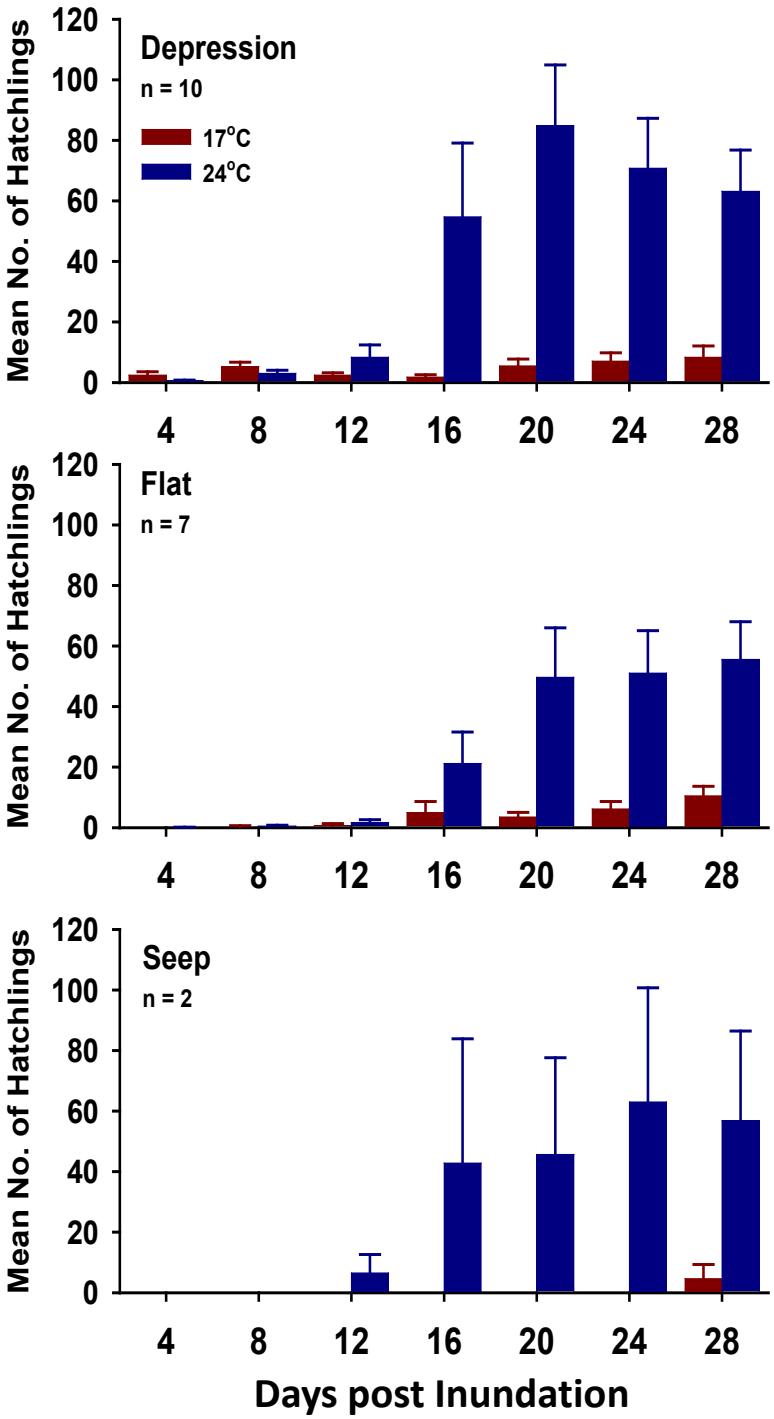

Figure 4.2 **The mean number of hatchlings (±SD) counted every four days post inundation of sediments collected from each of the HGM types from 2LExp1 in a 24°C temperature controlled environment and from 2LExp2, at 17°C.**

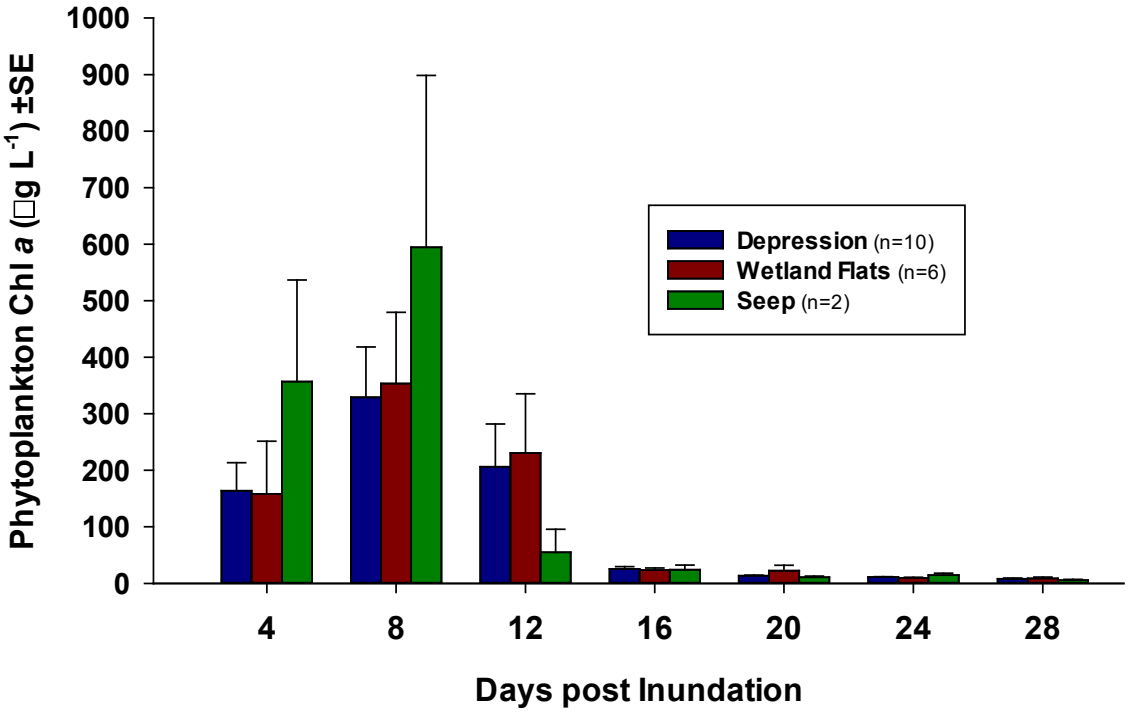

Figure 4.3 **Mean phytoplankton biomass (±SE) for each of HGM sediment types sampled every 4 days for 28-days of the 2LExp2. Note that the number of sediment replicates for each HGM type are in parenthesis.**

4.2 Nutrient Flux dynamics: Inundation and Sediment Development

As described in Section 3.1.2, physicochemical parameters were recorded and water collected for nutrient analysis every 3 hr over a 24 hr period, on day 3 post inundation (Stage 1), day 13 (Stage 2) and day 32 (Stage 3). As the experimental set up was carried out under shade cloth in an open air environment there was no temperature control, therefore water temperatures followed natural ambient conditions of the 32 day experimental period. FLUX experimental surveys are denoted with hatched green boxes (Figure 4.4). The overall mean water temperature was 24°C during the experimental period, whereas the air temperature average was 22°C. Water temperatures recorded during each flux experiment stage is shown in Figure 4.5. There was a marked increase in EC in both dark and light tanks from Stage 1 (early phase) to Stage 3 (late phase) as can be seen in Figure 4.6. Water temperature closely tracked changes in ambient temperatures. Figure 4.6 illustrates a similar pattern of increases between the stages for pH and TDS, and to some extent DO as the conditions within the microcosm matured.

Nutrient levels at each stage are illustrated in Figures 4.7 and 4.8. As a result of very low individual levels of nitrite and nitrate these were combined as TOxN (see Figure 4.7). When comparing stages, TP and SRP levels were generally much lower in Stages 2 and 3 as compared to the initial inundation Stage 1. In contrast, Silica had higher levels during Stage 2 than in both Stages 1 and 3 (Figure 4.8).

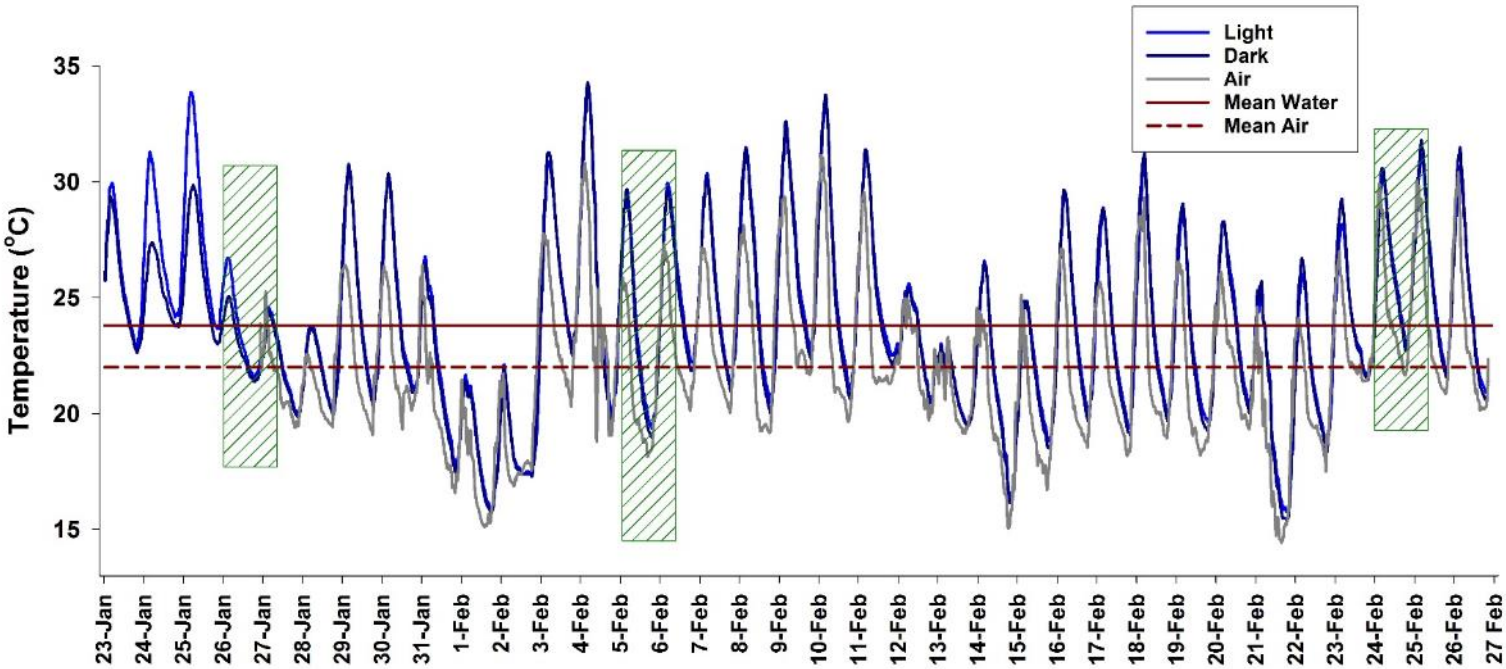

Figure 4.4 **Logged temperature data for ambient air (grey line) and water temperature (2 microcosms, 1 that was open to light continuously and 1 that was "dark" for 24 h during flux experiment). Data were logged every 30 min, overall air and water temperature mean values represented by red lines. Green hatched boxes represent flux experimental periods. These data relate to Flux and TExp1.**

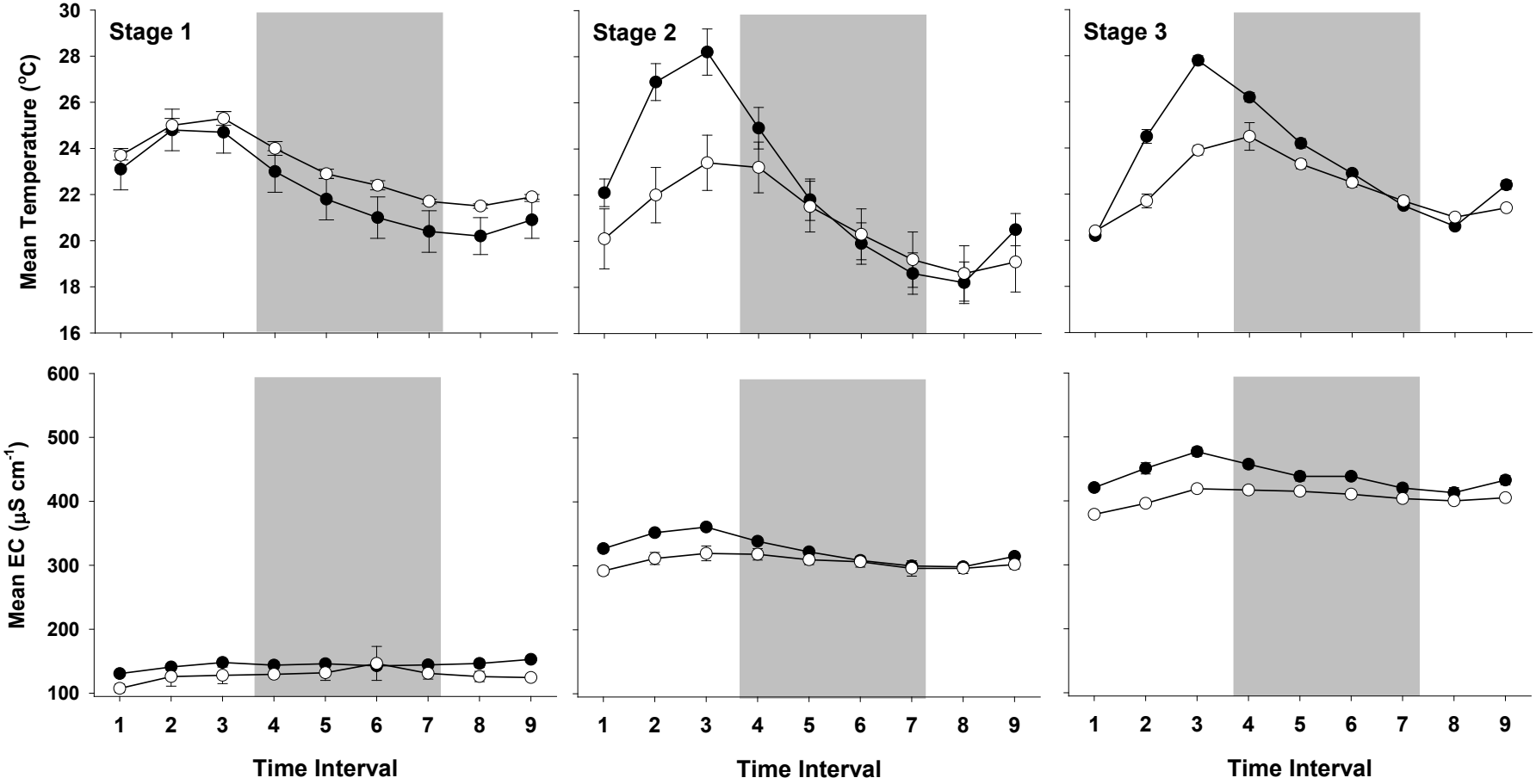

Figure 4.5 **Temperature (top row) and electrical conductivity (EC, bottom row) in light (open circles) and dark (closed circles) treatment tanks throughout the three flux experiments. The shaded area denotes samples collected during the night.**

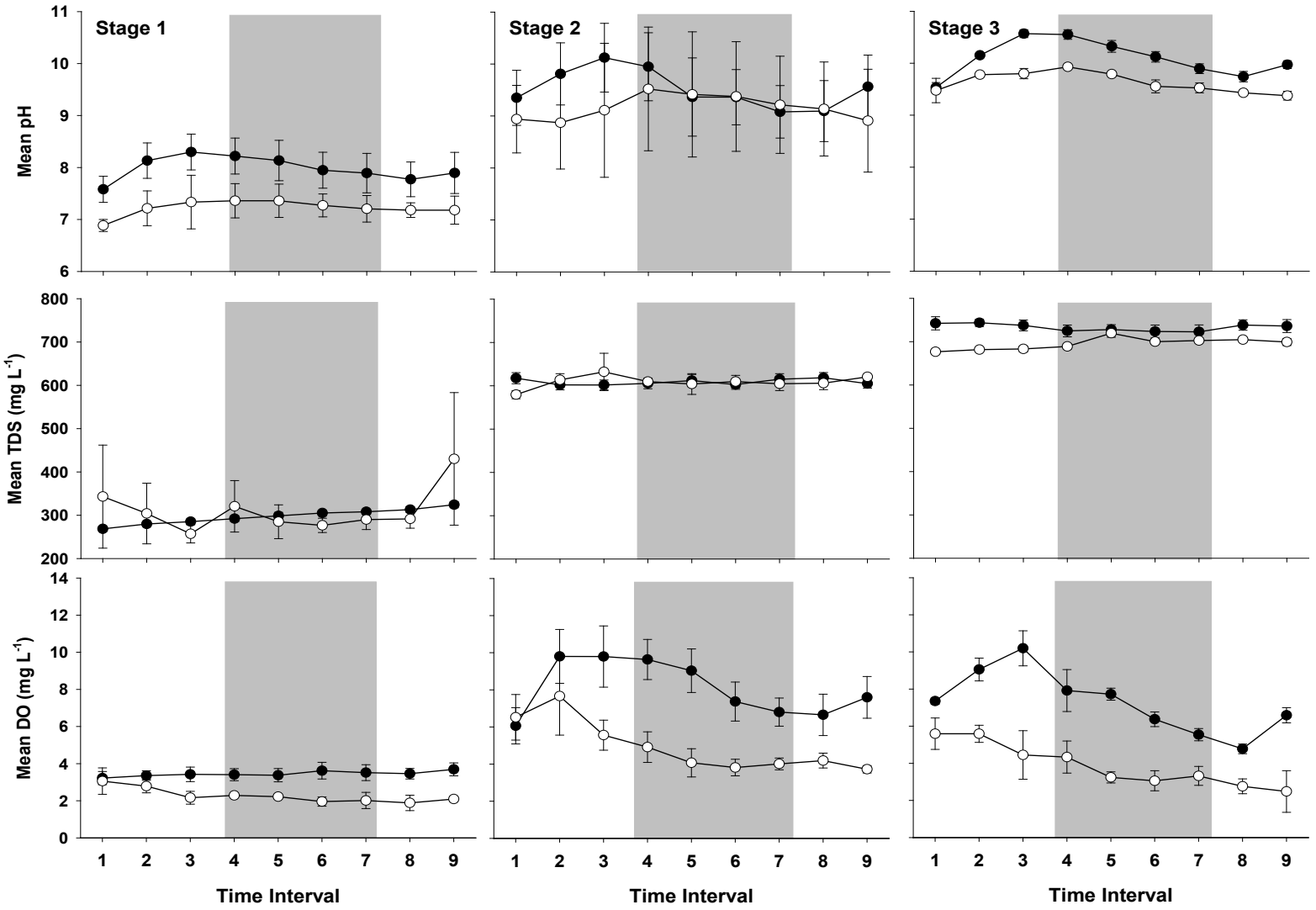

Figure 4.6 **The pH (top row), total dissolved solids (TDS, middle)) and dissolved oxygen (DO, bottom) in light (open circles) and dark (closed circles) treatment tanks throughout the three flux experiments. The shaded area denotes samples collected during the night.**

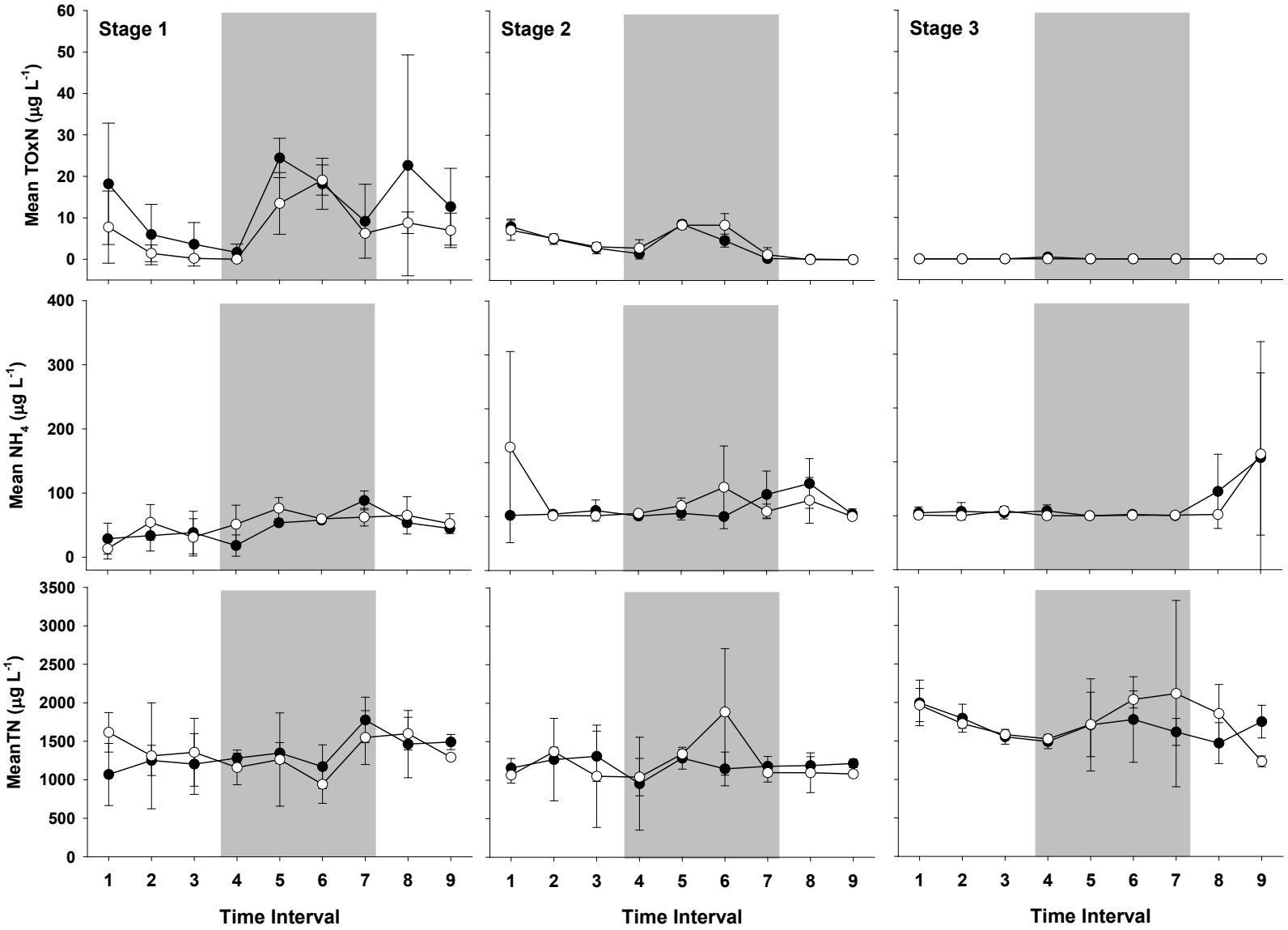

Figure 4.7 **The total oxidized nitrogen (TOxN), ammonia (NH_4^+) and total nitrogen (TN) levels in light (open circles) and dark (closed circles) treatment tanks throughout the three flux experiments. Shaded areas represent night time sampling period.**

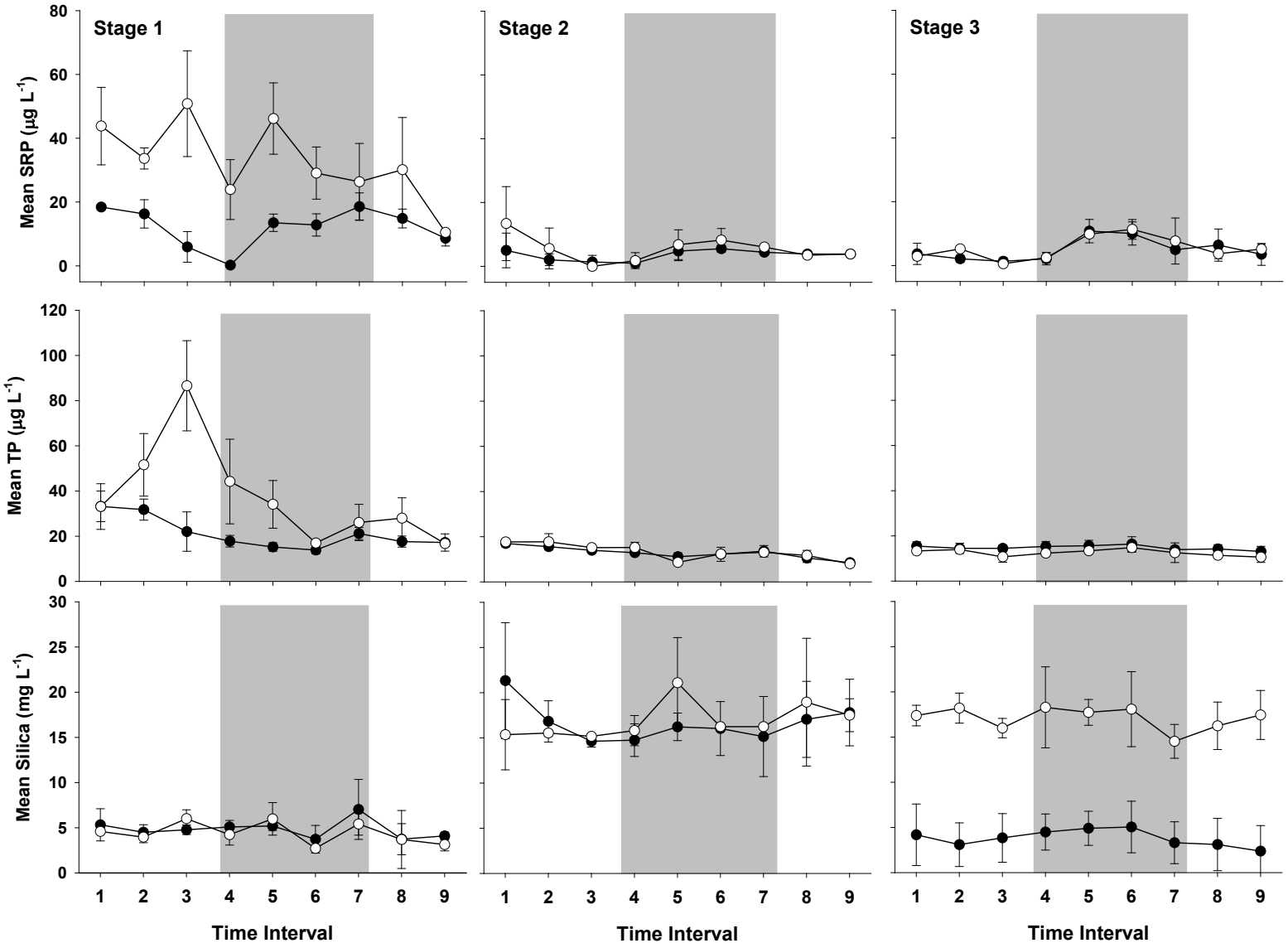

Figure 4.8 **The total phosphorus (TP) soluble reactive phosphorus (SRP) and silica (Si) levels in light (open circles) and dark (closed circles) treatment tanks throughout the three flux experiments. Shaded areas represent night time sampling period.**

Ammonium concentrations at Stage 1 averaged 49.1 ± 5.8 µg L^{-1} in all chambers (Figure 4.7). There was a significant decrease (H = 75.25; N = 162; $p < 0.001$) in overall NH_4^+ concentrations between Stages 1 and 2, however no significant difference between Stages 2 and 3 ($p > 0.05$). $NH4_4^+$ concentrations were highly variable at points during Stage 2 and Stage 3 flux experiments, but were generally low, below 10 µg L^{-1}.

Trends in TOxN in the dark and light chambers were fairly similar across all monitored stages. Both showed decreases over the first nine hours, followed by an increase and subsequent decrease in concentrations (Figure 4.7). Like with NH_4^+, the highest concentrations were recorded during Stage 1. By Stage 2, TOxN concentrations had dropped to barely detectible limits.

Silica concentrations in the first stage of the flux experiment were relatively low, with values ranging between 1.4 and 11.9 mg/L. There was no significant difference between mean concentrations light chambers ($p > 0.05$). By Stage 2, mean concentrations in all chambers had more than tripled, showing an overall significant increase from Stage 1 (H = 73.55; N = 162; $p < 0.001$). By Stage 3, silica concentrations had decreased to levels similar to Stage 1. By Stage 3 a weak diurnal trend was observed as silica concentrations decreased in the water column during the day and increased at night (Figure 4.8).

Unlike Si, SRP concentrations were highest in the dark chambers at Stage 1, averaging 32.7 ± 3.5 µg/L (Figure 4.8). This was significantly higher (H = 21.4; N = 54; $p < 0.001$) than the average of the light chambers. Stage 2 average SRP concentrations were significantly lower (H = 59.28; N =162; $p < 0.001$) than Stage 1. All three stages displayed diurnal trends with SRP concentrations decreasing during the day and increasing at night. The trends seen in TP were similar to those of SRP with concentrations decreasing significantly (H = 34.15; N = 162; $p <0.001$) from Stage 1 to 2; but not from Stage 2 to 3 ($p > 0.05$). TP concentrations during Stage 3 were more constant (Figure 4.8).

Flux rates were variable between sampling intervals for all nutrients (Lategan 2016). Overall net fluxes are illustrated in Figure 4.9. Positive values denote net flux into the water column from the sediment, whereas negative values show a net flux of nutrients back, or sinking, into the sediment. Within the first few days of inundation (Stage 1) there was a net positive flux of TP, SRP, Si and TN (Figure 4.8). There was a slight efflux of NH_4^+ and TOxN back into the sediment. After 13 days of inundation (Stage 2) the water column became a source of Si back into sediment whereas all other nutrients showed an efflux into the water column, although at generally low rates (Figure 4.9). By Stage 3, there was a net influx of all nutrients with the exception of TN. This is indicative of the sediments in this area, especially for ephemeral wetlands in relatively unmodified areas, nutrient supply in this way is low, but cycling of the nutrients there is immediate.

This cycling is important for the development of primary producers (emergent and submerged vegetation and macro- and microalgae) in the wetland. The nutrient levels and cycling rates help determine which algae, either from resting spores in the sediments or brought in aerially (birds, wind, etc.), will be able to develop, grow and compete to form the benthic (MPB) and pelagic (phytoplankton) communities.

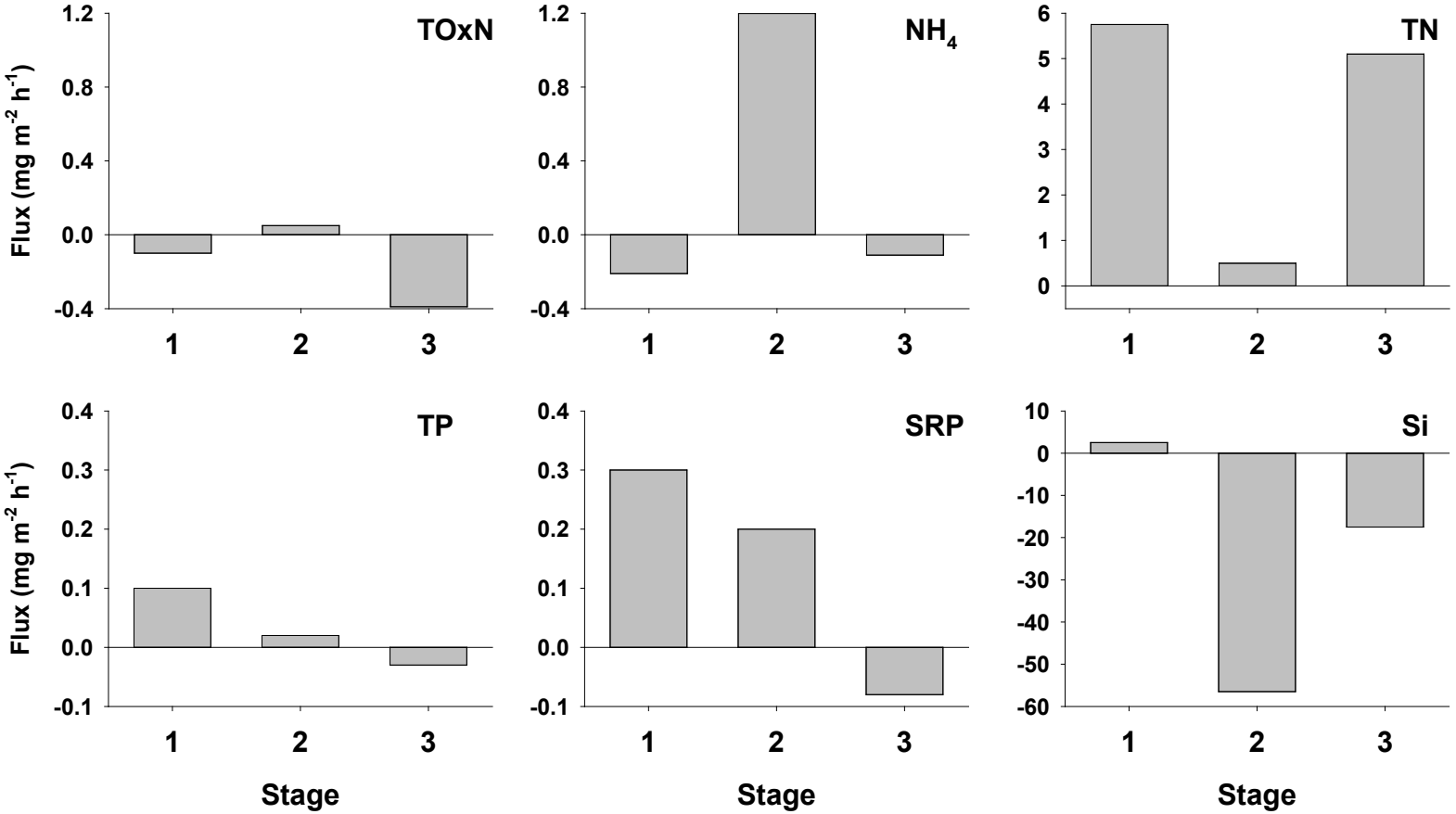

Figure 4.9 **Overall net nutrient flux attributed to autotrophic metabolism for each stage of sediment development for each nutrient measured. Negative values denote influx to the sediment; positive values denote efflux or nutrient release into the water column. Note y-axes are different for the different nutrients.**

4.3 Temperature Effects on Algae and Invertebrates

4.3.1 Ambient summer temperature and diel cycles

TExp1 was done conjunction with the FLUX experiment, under natural temperature conditions (Figure 4.4) where the overall mean was 24°C, but ranged from 15-35°C over the inundation period. Physicochemical parameters measured every 4 days is reported as averages across all tanks for each sampling period (Table 4.3). MPB biomass dominated across all sampling periods (Figure 4.10), with levels increasing steadily through the 28 day period. Phytoplankton biomass peaked at days 12 and 24 (Figure 4.10). Numbers of invertebrates were low for most of the experimental period until day 16 when they increased exponentially to peak on the last day (Figure 4.11). Numbers were highly variable per replicate (Figure 4.11).

Table 4.3 Physicochemical data, average (±SD) values for all tanks across the TExp1 experimental period. EC = electrical conductivity, TDS = total dissolved solids, DO = dissolved oxygen.

	Temperature (°C)		EC (µS/cm)		TDS (µg/L)		Salinity (PSU)		pH		DO (mg/L)	
Day	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
4	20.9	1.7	152.8	8.3	324.3	14.0	0.08	0.00	7.9	0.8	3.7	0.7
8	22.4	0.1	236.5	14.8	522.5	76.8	0.12	0.01	8.3	0.0	6.2	0.3
12	25.3	0.5	313.8	11.3	605.3	21.6	0.15	0.01	8.5	0.3	5.8	0.6
16	24.7	0.7	326.8	9.2	630.8	24.8	0.16	0.01	8.2	0.5	5.4	0.6
20	22.7	0.2	329.5	13.8	668.8	27.8	0.17	0.01	7.8	0.5	4.7	0.5
24	22.0	0.2	345.3	23.0	703.8	42.1	0.18	0.01	7.8	0.9	7.5	1.2
28	20.2	0.1	334.8	12.3	712.0	22.3	0.18	0.01	8.0	0.1	5.0	0.4

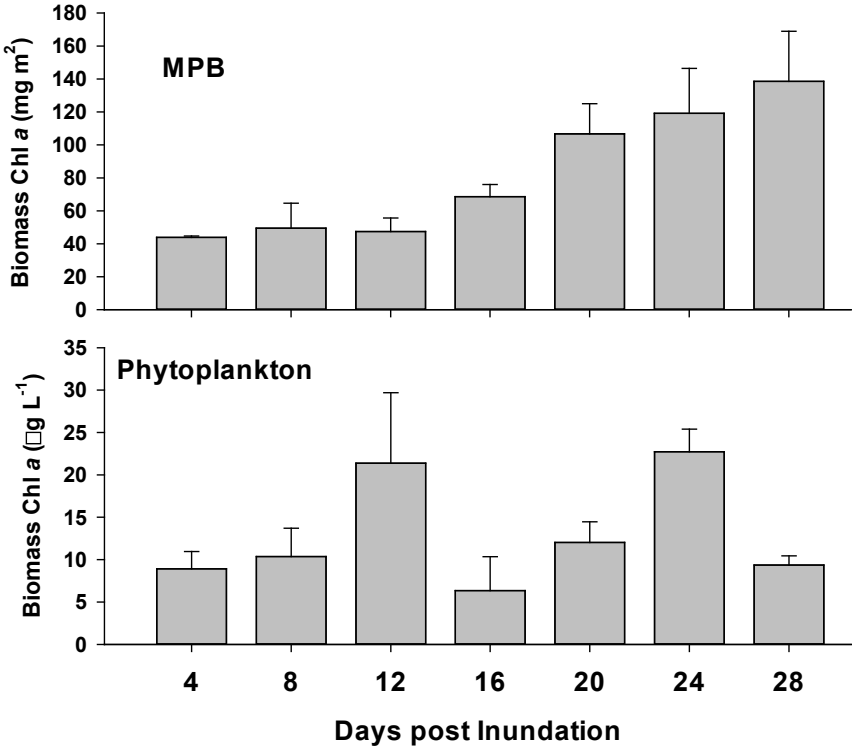

Figure 4.10 Mean algal biomass (±SD) over the TExp1 study period for sediment 1641b.

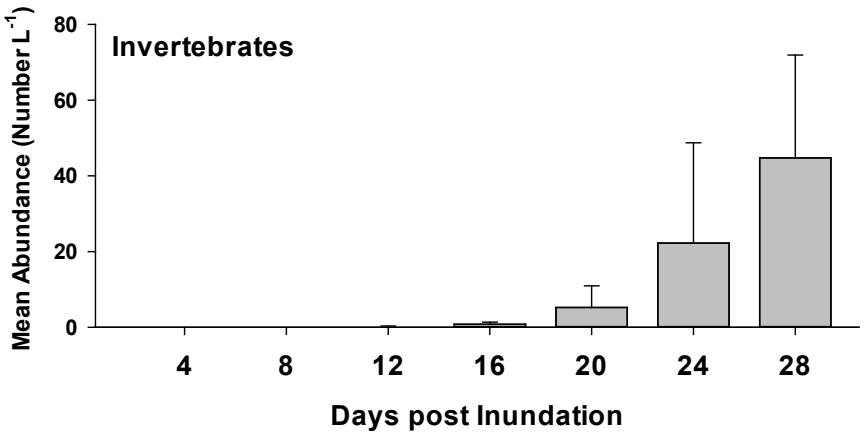

Figure 4.11 Mean number of invertebrates (±SD) over the TExp1 study period for sediment1 641b

The expectation is that diatoms (Bacillariophyceae) will dominate the sediment MPB community. In the case of this experiment, chlorophytes (green algae) were dominant in cell numbers (Figure 4.10), and generally made up of at least 8 different species within the genus of colonial *Scenedesmus* sp. (See Appendix 3 for full list). *Scenedesmus* spp. can limit their need for nutrient uptake by forming large colonies, this also serves as protection from grazers. There were 8 species of diatom present, 17 greens, 3 cyanophytes and 2 euglenoids. On day 24, an as yet to be identified euglanoid was the dominant species present (Figure 4.12). Euglenoids are both autotrophic and heterotrophic, feeding on microbes in the sediment and water column.

Chlorophytes were also the dominate group in the phytoplankton community for the first 20 days of the experiment, both by cell numbers (Figure 4.12) and number of species, with 13 species recorded. Diatoms were also present (4 species, dominated by *Amphora* sp.) but numbers were low until the last week of the experiment (Figure 4.12) where they steadily increased and were co-dominant with the blue-green *Anabaena* spp. There were two species each of Chysophyta, Cyanophyta and Euglenophyta throughout the experimental period, this were common throughout all of the experiments.

Invertebrates were depauperate by comparison to the algae. The community was comprised of *Moina* sp., a hardy, small cladoceran genus that is known to be tolerant of a wide range of water quality and ostracods (Cyprididae spp.) (Appendix 4). The *Moina* sp. were dominant (>60%) when invertebrates were present, with ostracods only gaining in terms of numbers by day 28, where they made up <40% of the total numbers of animals.

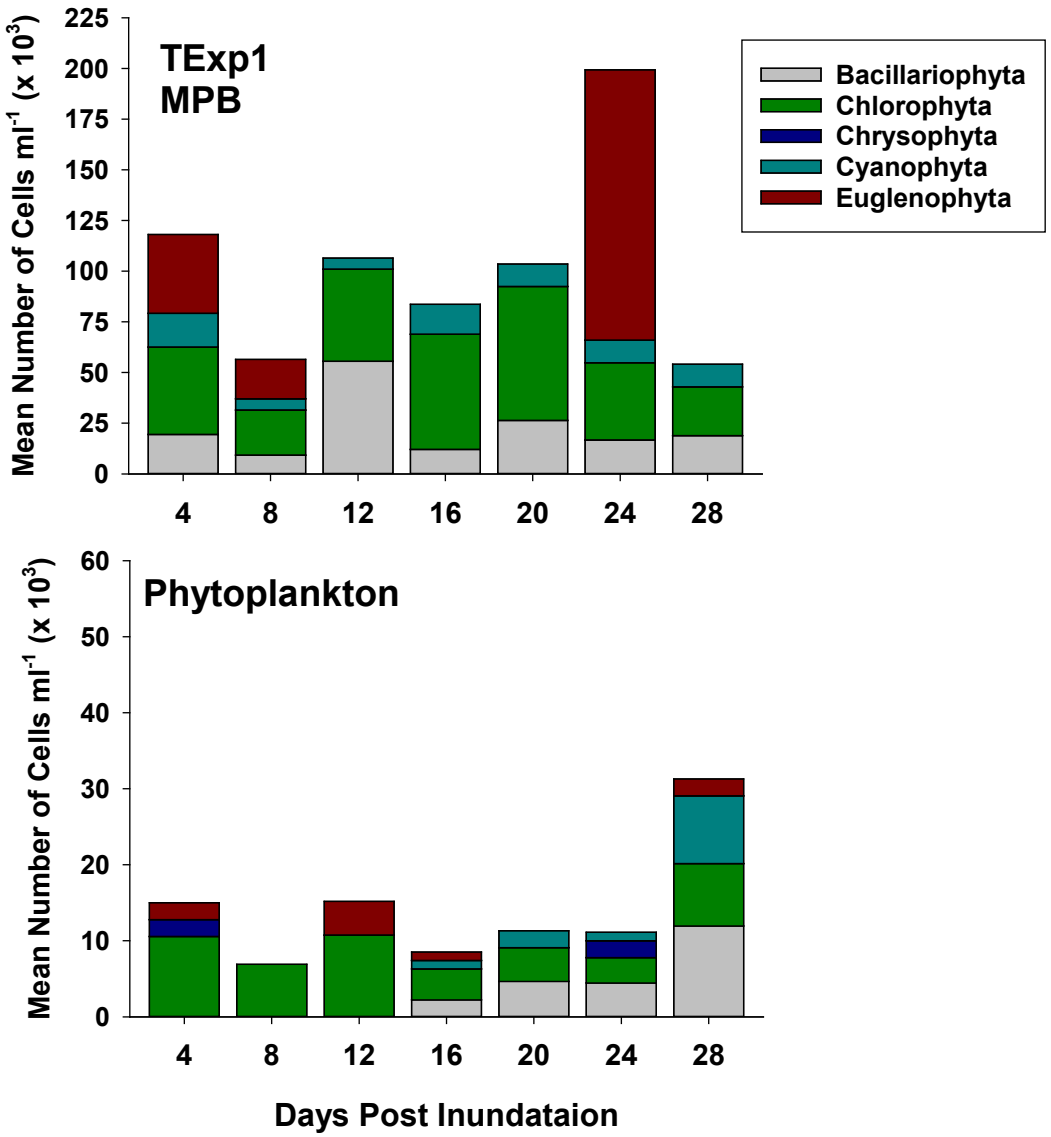

Figure 4.12 **Community composition of microphytobenthos (MPB, top) and phytoplankton (bottom) in microcosm experiment TExp1.**

4.3.2 Temperature controlled chamber experiments

TExp2 microcosms were used for sediment germination and hatching of invertebrates using two different sediments (one campus depression, 1641b and one depression from a drier region of NMB, Redhouse, RH1) at two different temperatures. One was deemed a "low" treatment and kept at 26°C and the other "high", at 34°C. The temperatures fluctuated within the microcosms as the experimental chambers experienced some malfunctioning, which introduced greater variability in temperature ranges at some points during the 28 day experiment (Figure 4.11).

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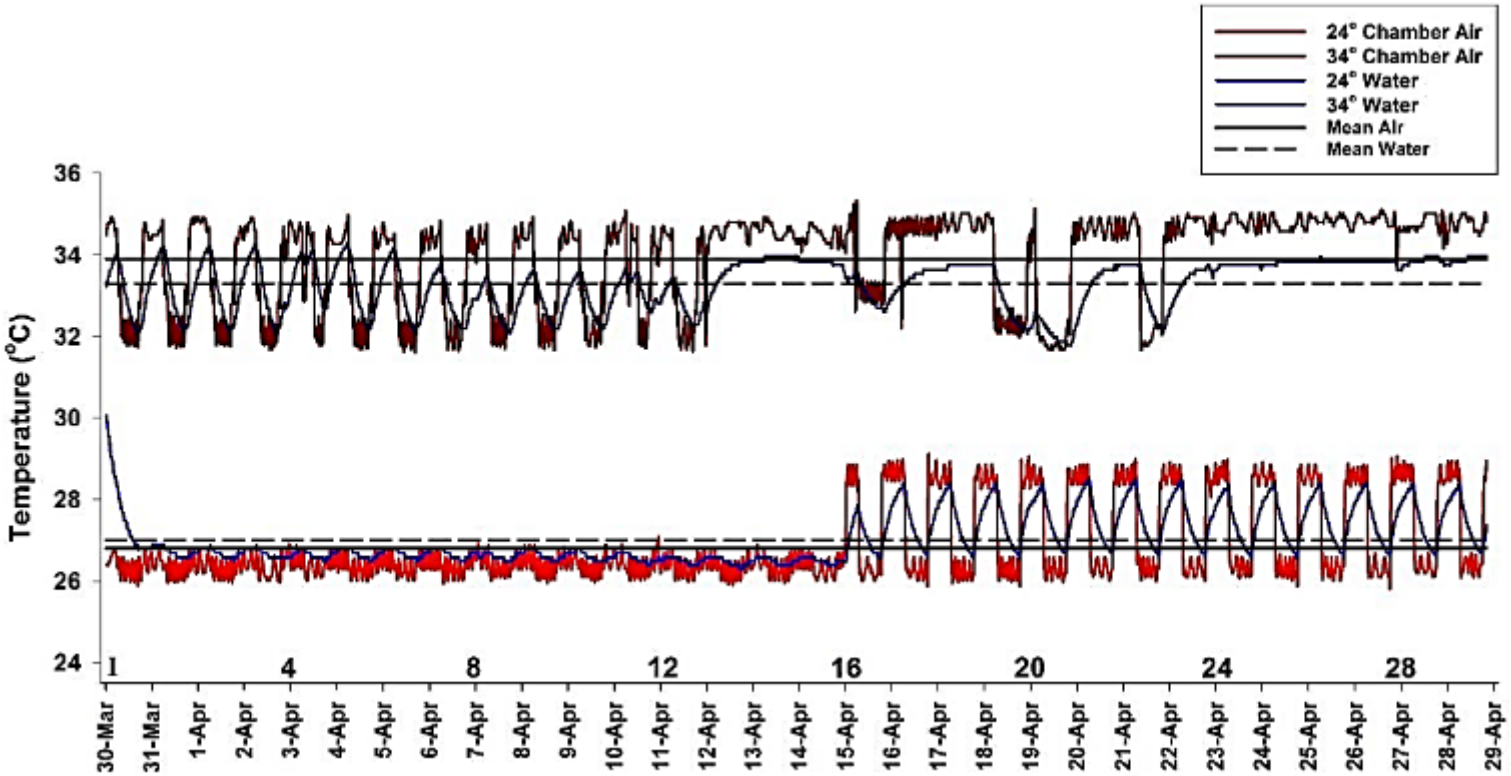

Figure 4.13 **Logged temperature data for air and water temperature in both growth chambers in TExp2, one set for target temperatures of 34°C (dark red and blue lines for air and water respectively) and the other at 26°C (light red and blue lines for air and water respectively). Data were logged every 30 min, overall air and water temperature mean values represented by black lines. Inundation represented by I, and each sampling time indicated by day number, 4-2.**

Despite the chamber fluctuations, the average temperatures across the sediments and were generally within a degree (Table 4.4). The EC for the RH1 sediments were higher than those from 1641b, and more variable between tanks, as can be seen by the high standard deviations in Table 4.4. The high temperature experimental sediments had much pH, tending toward alkaline, whereas the lower temperature pH was on the acidic side of neutral of 6.5 (Table 4.4). All other parameters were within the expected range.

Table 4.4 Physicochemical data, values for all tanks across the TExp2 average (±SD) over the 28-day experimental period. EC = electrical conductivity, DO = dissolved oxygen.

		Temperature (°C)		EC (µS cm^{-1})		Salinity (PSU)		pH		DO (mg L^{-1})	
Sediment/Temp		**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
1641b	**High**	33.5	1.1	296.4	68.7	0.12	0.03	8.5	0.8	7.1	2.1
RH1	**High**	33.1	1.0	515.0	157.9	0.21	0.06	8.3	1.3	8.2	4.3
1641b	**Low**	26.5	0.8	244.3	78.4	0.11	0.04	6.5	0.8	7.8	1.8
RH1	**Low**	26.3	1.0	385.0	142.7	0.18	0.07	6.4	0.9	8.5	2.5

The algal biomass results are shown in Figure 4.14 for phytoplankton and MPB biomass. There was variability within replicates, as can be seen in both figures by the high standard deviation. The phytoplankton biomass showed a pattern of early development, a dip by day 12 post inundation and then a resurgence by days 24 and 28. The patterns between the high and low temperatures were similar, but the biomass level generally higher for the low temperature treatment in sediment 1641b or the same. In sediment RH1 biomass was generally higher in the high temperature treatment (Figure 4.14).

Phytoplankton biomass was consistently under 50 µg L^{-1} with the exception of a few peaks in both sediments and temperature treatments. The peaks on days 24 (34°C) and 28 (26°C) in 1641b were due to the large amount of filamentous algae, *Spirogyra* sp. (Chlorophyta) and *Anabaena* sp. (Cyanophyta) present, and were in the large size fraction of algal cells (>2.5 µm) (Figure 4.15, see Appendix 3 for species list). The high biomass on day 8 for RH1 in the 34°C temperature treatment was most probably because of the formation of large colonies of the chlorophyte *Scenedesmus* sp. This shifted, however by the end of the experimental period to a dominance of *Chlosterium* sp., a large unicellular green and *Anabaena* sp. (Figure 4.15). Although the biomass was not as great within the high temperature treatment in RH1, there was a clear dominance of *Anabaena* sp. when examining the community structure of the samples.

MPB biomass was less variable between temperature treatments and sediments throughout the experiment, with 1641b sediment with higher biomass, especially in the first 12 days (Figure 4.14). The only peaks above the average level during the experimental period were for day 28, but there was no pattern with sediment or temperature levels.

The MPB community for sediment 1641b was dominated by diatoms (*Craticula* spp. and *Rhopalodia* sp., Appendix 3), whereas the RH1 sediment was dominated by *Scenedesmus* sp., green algae (Figure 4.16, Appendix 3.)

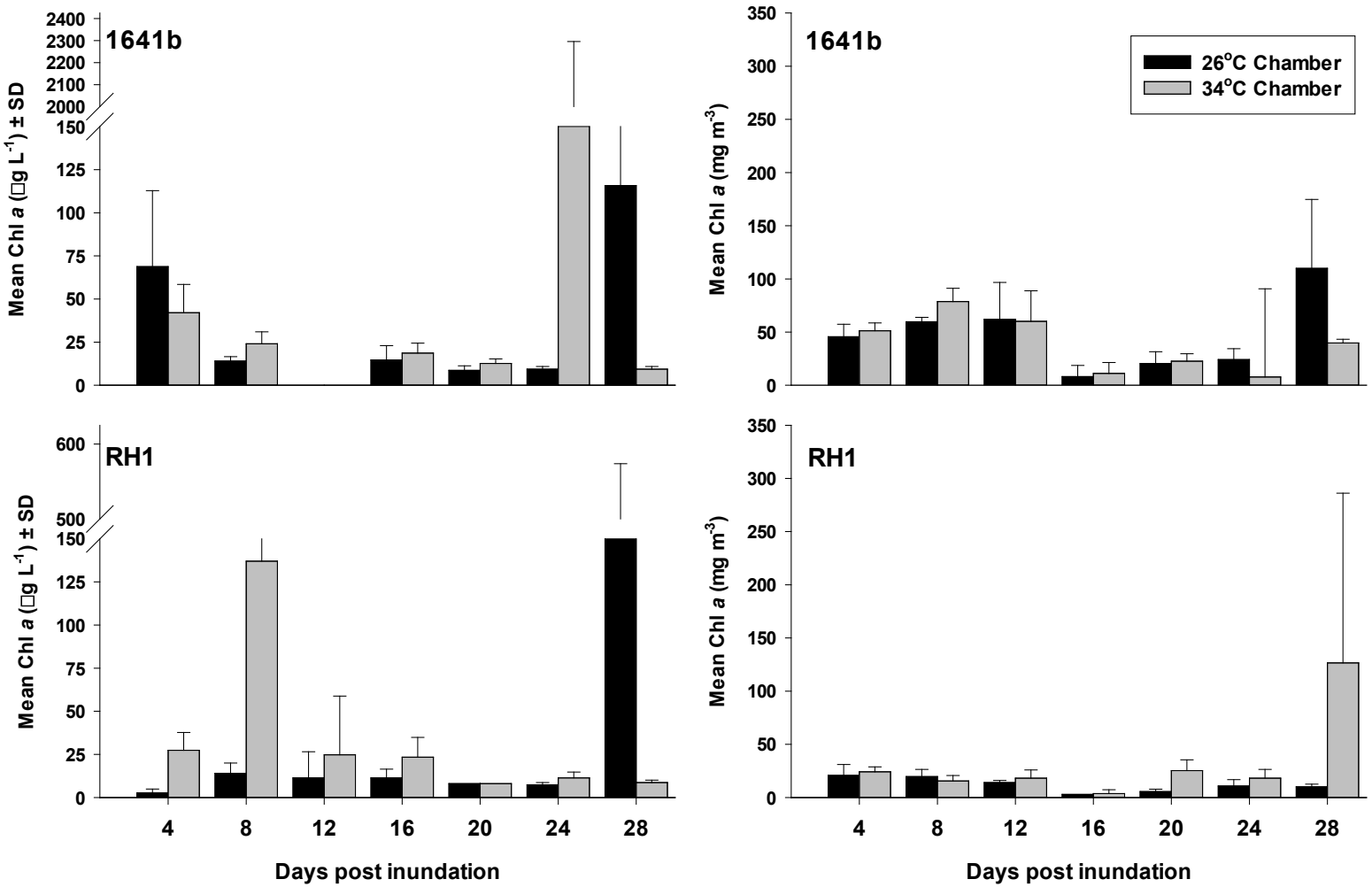

Figure 4.14 **Mean (±SD) phytoplankton (left) and microphytobenthos (MPB, right) biomass over the TExp2 study period for sediments 1641b (top) and RH1 (bottom) and both experimental temperatures. Note differences in the x-axis ranges in each figure.**

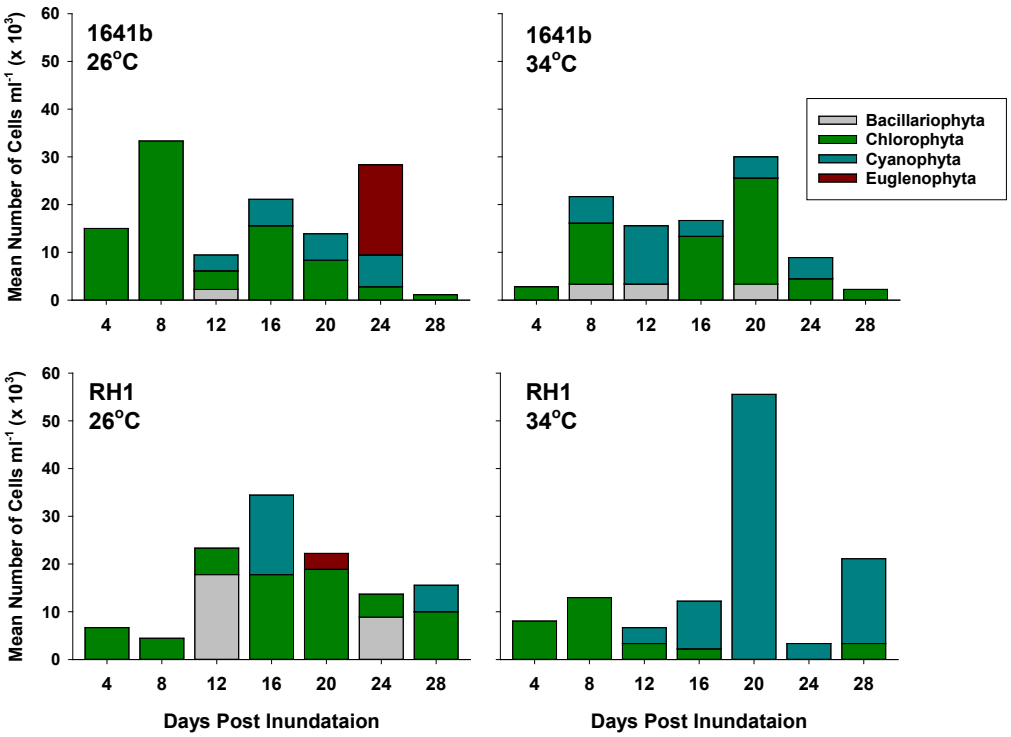

Figure 4.15 **Phytoplankton community composition over the TExp2 study period for sediments 1641b (top) and RH1 (bottom) and both experimental temperatures.**

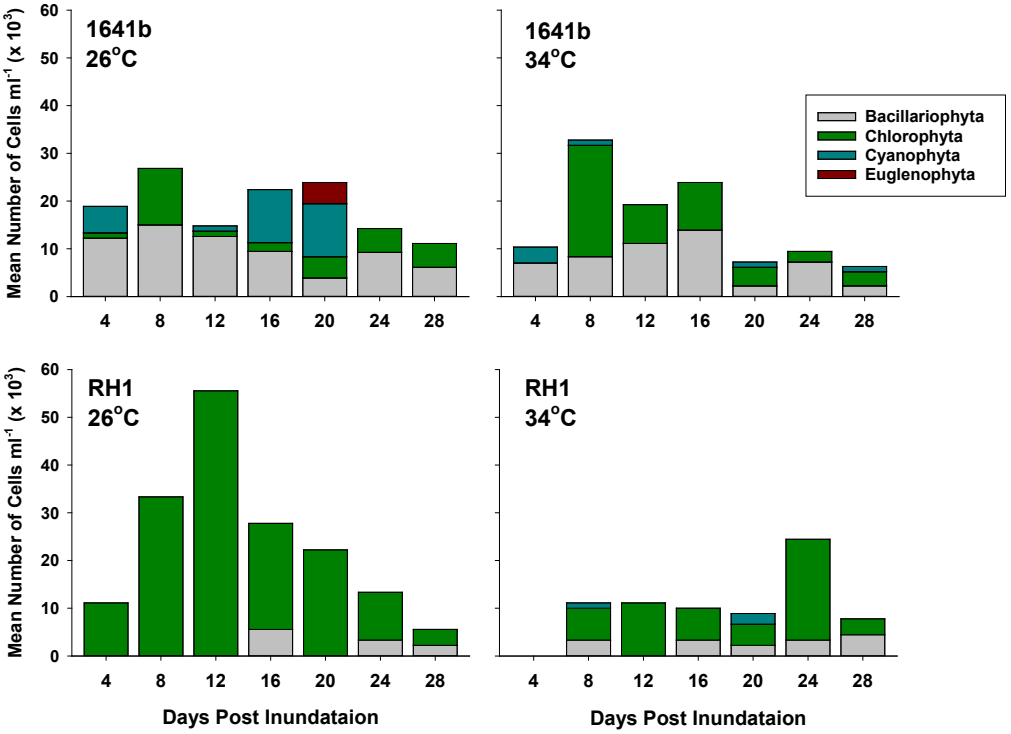

Figure 4.16 **Microphytobenthic (MPB) community composition over the TExp2 study period for sediments 1641b (top) and RH1 (bottom) and both experimental temperatures.**

There was a delay of invertebrate hatching until day 8 post inundation (Figure 4.17). Overall numbers were higher in the RH1 sediments than the 1641b sediments with a significant peak in numbers for both sediments and in the lower temperature treatment occurring at the end of the experiment (Figure 4.17). The response to temperature varied between the sediments, with more hatchlings per litre in the high temperature on days 4-16 in 1641b and the hatching response to the lower temperature in RH1sediments was consistently better throughout the experiment. In fact, with the exception of the end of the experiment, RH1 sediments produced more hatchlings throughout the experimental period than 1641b (Figure 4.17).

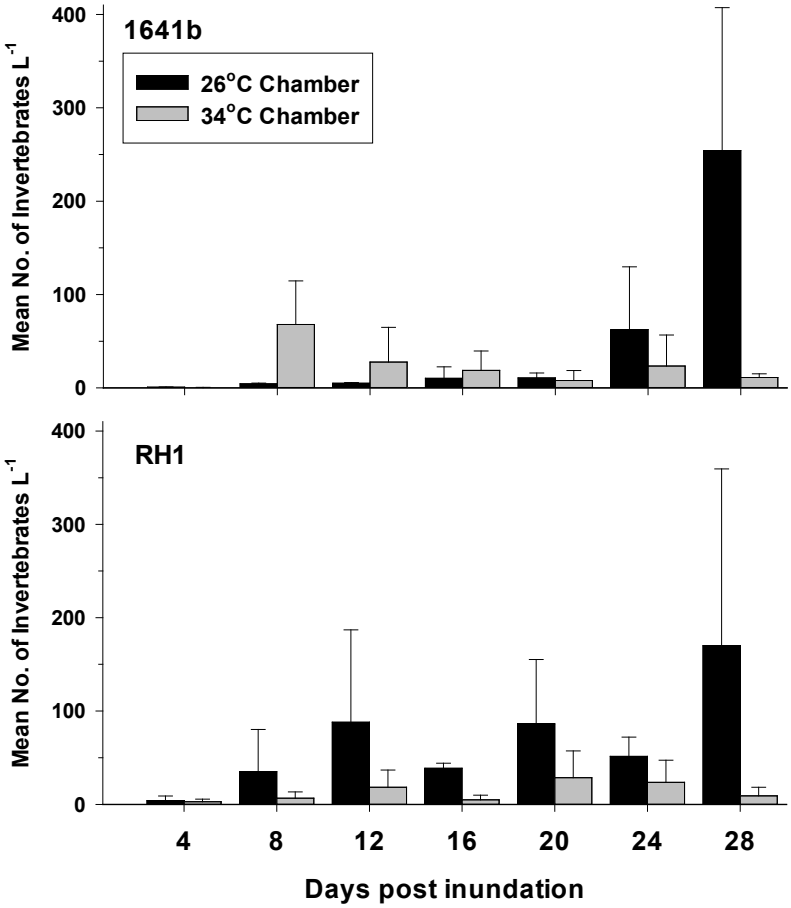

Figure 4.17 Mean invertebrate density (±SD) over the TExp2 study period for sediments 1641b (top) and RH1 (bottom) and both experimental temperatures.

Unlike TExp1, which only had the successful hatching of two taxonomic groups of invertebrates, Cladocera (*Moina* sp.) and Ostracoda, both sediments and temperature regimes produced a greater diversity of groups (Figure 4.18) and number of species (Appendix 4). Although a few of the obligate ephemeral large branchiopods (Conchostraca and Notostraca) were present, they were in low numbers and could not be shown on Figure 4.18, but are listed in Appendix 4. Overall there were 13 species, 10 of which were in RH1 sediment at 26°C, and 7 species at 34°C. There were fewer species present in the 1641b sediments, with 6 and 7 in the low and high temperatures respectively. However, the numbers of taxa were greater in the lower temperature treatments, by approximately 2.5-3 times the total numbers of the 34°C treatments. In both sediments and temperatures there was a clear succession pattern of rotifers emerging first followed by the cladocerans and ostracods. Temperature had more of an effect on which taxonomic groups were dominant, however, with the

ostracods dominant in terms of numbers in the lower temperature and the cladocerans dominant in the high temperature. When the experiment was terminated and filtered on day 32, additional invertebrate samples were collected, the numbers of animals had risen 10 x than that of day 28 in the lower temperature treatment, with a switch to a dominance of ostracods, followed by cladocerans, where the dominant cladoceran was a very small Chydoridae, *Alonella* sp. which is reflected in the final numbers of Appendix 4. There was only a slight increase in animal numbers on the same date for the higher temperature treatments, but with the cladocerans (*Moina* spp.) still the dominant group.

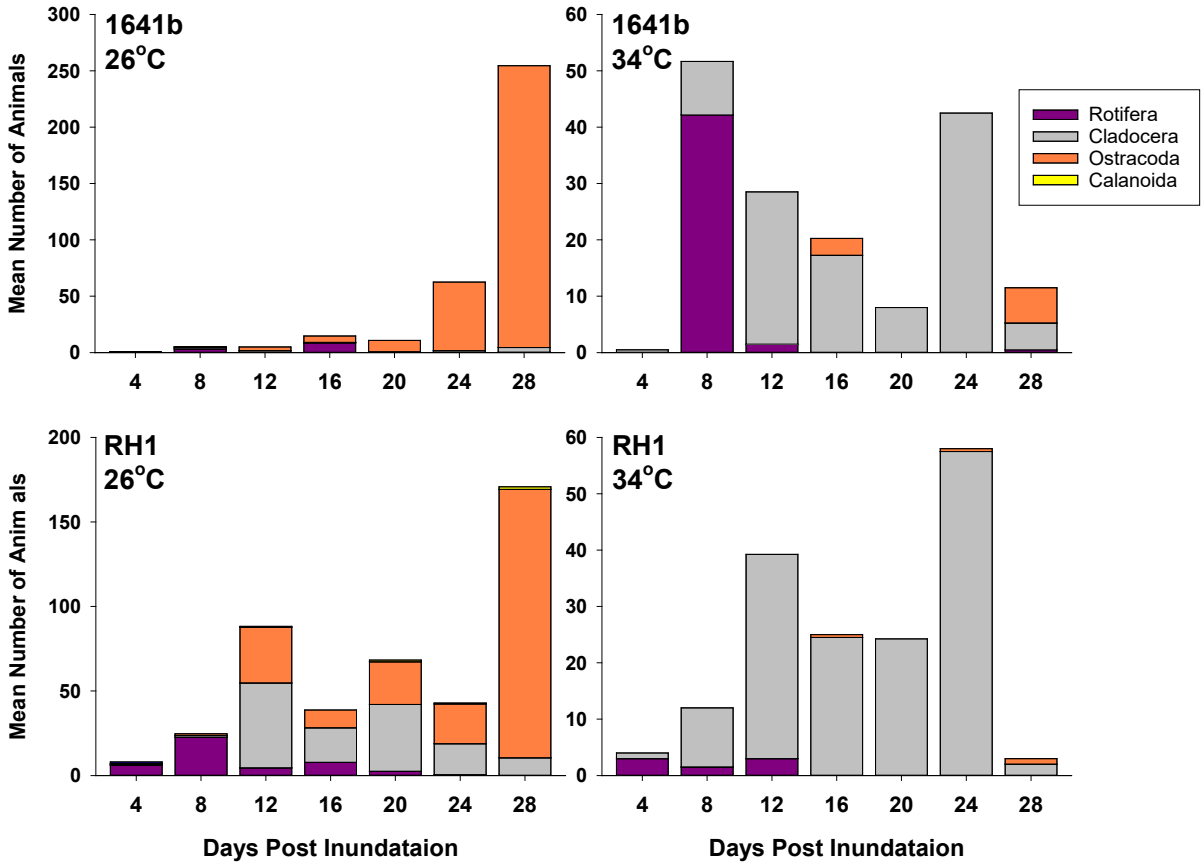

Figure 4.18 Invertebrate community composition (mean number of animals per L within each taxonomic order) for each sampling day, sediment and temperature treatment. Note that y-axes are not equivalent between panels.

4.3.3 Nutrient addition and high temperature

In the last experiment, TExp3, the temperature was set at the high level of 34°C and nutrients were added as per Section 3.1.3. The experiment was to last the standard 28 days, however due to a major malfunction with the growth chamber, where temperature exceeded 45°C at times (Figure 4.19), the experiment was terminated. Physicochemical variables are given in Table 4.5 and a similar to other values from experiments with this sediment (1641b) with the exception of extremely low DO levels.

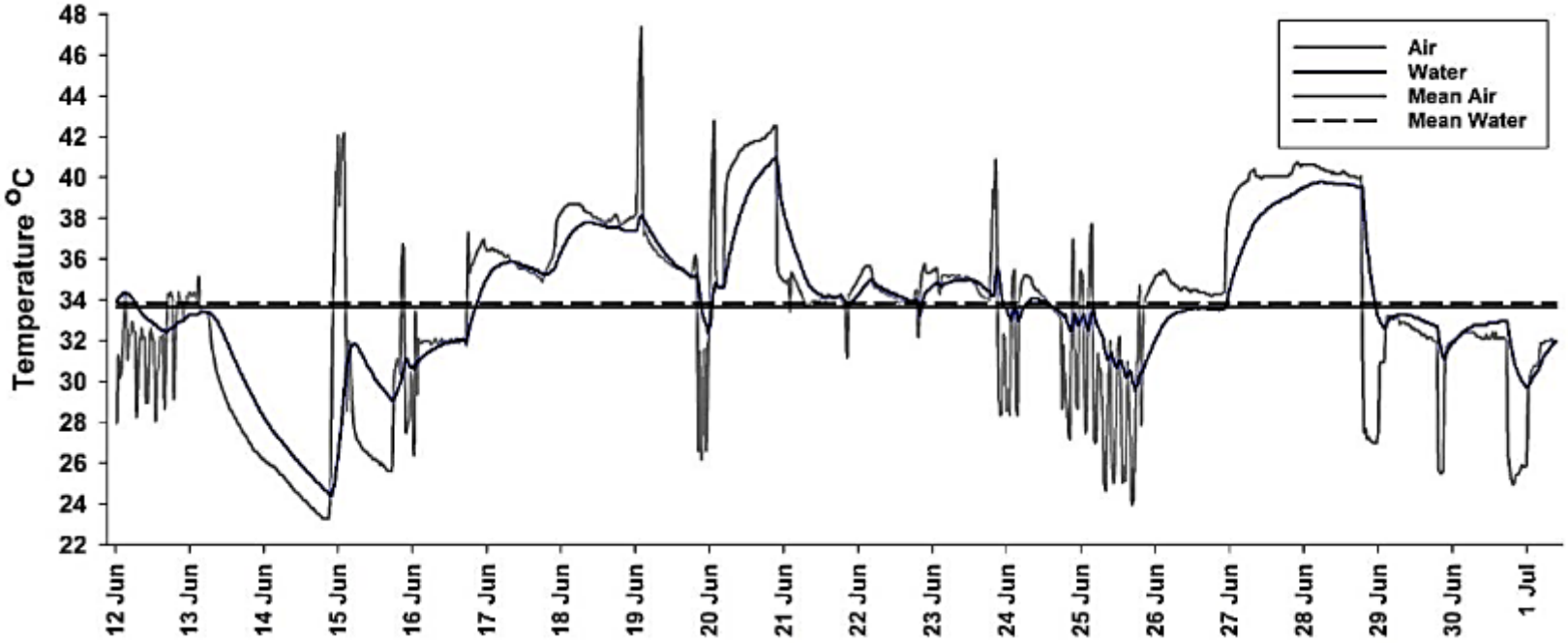

Figure 4.19 **Logged temperature data for ambient air (grey line) and water temperature (2 microcosms, blue line) TExp2. Data were logged every 30 min, overall air and water temperature mean values represented by black lines.**

Table 4.5 **Physicochemical data from TExp3, average (±SD) values for each treatment per sampling day until termination of the experiment. Sediment (1641b) only, 3P = 3P:N ratio, 6P = 6P:N ratio addition. EC = electrical conductivity, TDS = total dissolved solids, DO = dissolved oxygen.**

		EC (µS/cm)		TDS (µg/L)		Salinity (PSU)		pH		DO (mg/L)	
Day	**Treatment**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
4	**Sediment**	351.0	39.4	706.0	78.7	0.08	0.01	6.7	0.7	0.3	0.2
	3P	314.0	7.1	631.3	14.1	0.07	0.01	7.2	0.4	0.3	0.4
	6P	324.7	9.5	651.7	18.3	0.07	0.00	6.8	0.4	0.5	0.6
8	**Sediment**	718.3	45.1	1443.3	90.7	0.16	0.01	6.9	0.3	1.7	1.3
	3P	679.5	22.9	1365.5	45.9	0.15	0.01	7.0	0.2	2.9	2.0
	6P	685.3	24.3	1376.8	48.9	0.15	0.01	7.0	0.3	4.2	1.3
12	**Sediment**	846.8	79.6	1702.3	159.7	0.19	0.02	7.8	0.6	5.6	2.6
	3P	853.0	80.8	1714.3	162.4	0.19	0.02	7.7	0.8	5.0	4.0
	6P	808.3	77.4	1624.5	155.8	0.18	0.01	7.9	0.5	6.9	3.7
16	**Sediment**	487.7	276.4	980.1	555.5	0.13	0.01	8.5	0.6	9.3	2.8
	3P	727.3	78.6	1461.3	156.3	0.16	0.02	8.8	0.5	8.6	1.9
	6P	624.3	76.9	1255.8	157.2	0.14	0.01	8.8	0.5	11.5	2.1

The EC levels were high as per previous higher temperature experiments, and pH was initially in the 6 range, but by day 16 post inundation increased to 8. The DO levels were low for all treatments at the start of the experiment but slowly increased as the algal biomass (primarily MPB, Figure 4.20) increased. The spikes in temperature within the chamber, cooling and then heating up to extremes seemed to have an effect on the physicochemical and biological parameters. Phytoplankton biomass was almost non-existent, with MPB levels greater (Figure 4.20), however still lower than the previous experiments (Figures 4.10 and 4.14). There was not a significant difference in measured biomass between treatments ($p > 0.05$). It is not certain if this was because of the erratic temperature conditions or a flaw in nutrient addition.

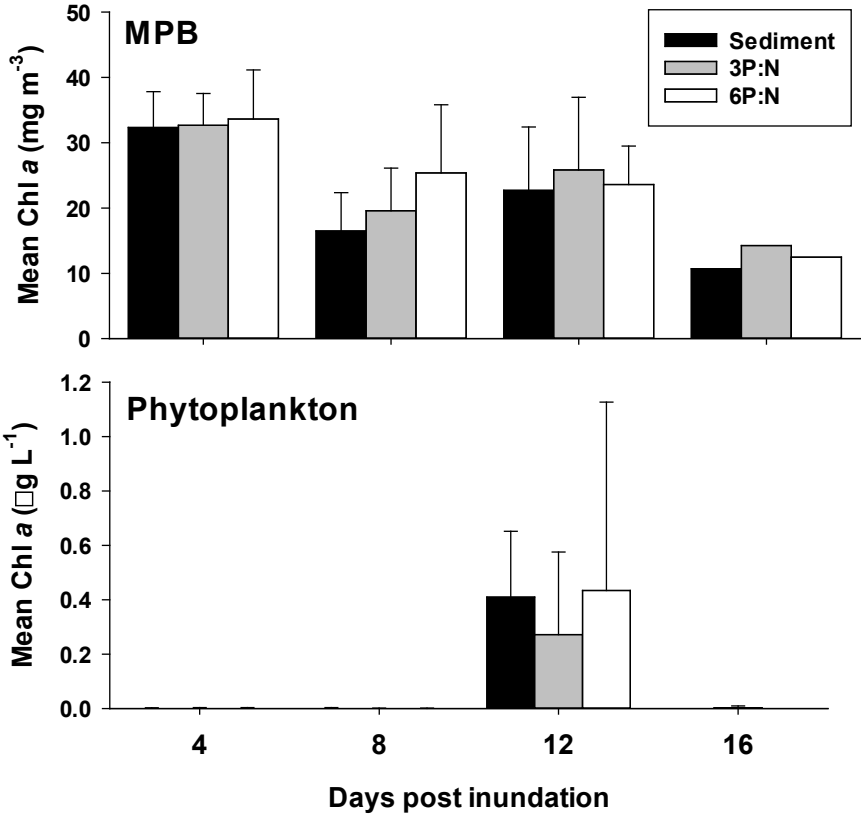

Figure 4.20 Mean (±SD) microphytobenthic (MPB, top) and phytoplankton (bottom) biomass over the TExp3 study period for sediment 1641b nutrient addition. 3P = 3P:N, 6P = 6P:N. Note phytoplankton values for days 4, 8 and 16 were below 0.1 and cannot be seen on scale.

Although the measured algal biomass was very low in this experiment, there was evidence of large amounts of epiphytic algal growth on the sides of the tanks and floating as "scum" on top of the water. This was only noted and had not been setup for quantification. An example of this filamentous "bloom" of *Spirogyra* sp. and *Anabaena* sp. is shown in Plate 4.1 below.

Invertebrate hatching failed almost completely. Within the first 8 days there were fewer than 20 invertebrates (ostracods and cladocerans) collected from all of the experimental treatment tanks, thereafter all invertebrates that had emerged died off and there were no further hatchlings seen or collected. Even the *Moina* spp. and ostracods, which tend to dominate in warmer water situations and show a wide tolerance of water quality conditions were unable to develop and grow. There were

two factors that most probably influenced this lack of development, the first being the extreme temperatures, but the second the lack of “edible” algae for the newly hatched and growing zooplankton to consume. Filamentous greens and blue-greens are not easy for these small zooplankters to handle and eat. Therefore, if any invertebrates did hatch in the high temperatures, they most likely starved soon afterwards for the lack of small phytoplankton available (Figure 4.20).

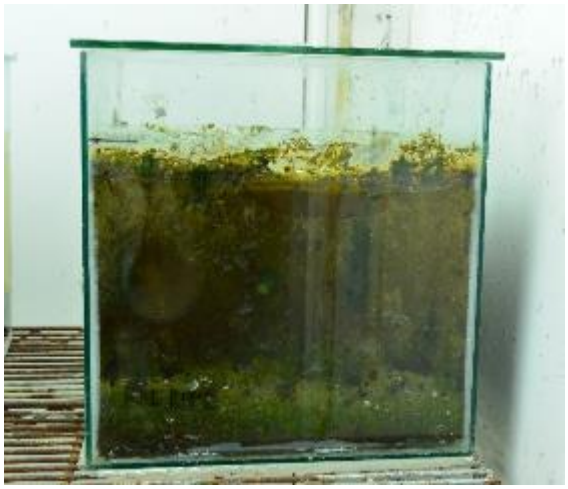

Plate 4.1 Example of epiphytic mat development in TExp3.

4.4 Algal Biomass and Invertebrate Density with Temperature

To synthesize the experiments into general relationships between temperature, algal emergence and development and invertebrate biomass over an inundation period we distilled our experimental data down into their principal components. We took relative biomass and abundance plotted as a spline curve over time and found that there is a very different pattern emerging from experiments done at temperatures less than 25°C and those greater (4.21).

What is noticeable is the immediate buildup of algal biomass prior to high density invertebrate emergence 8 to 10 days later in the lower temperature experiments. This is then followed by an asymptotic increase of invertebrates and decline in algal biomass. In the higher temperature experiments, we see almost a concurrent buildup of algal biomass and invertebrate development with the algal biomass somewhat more stunted and declining mid-inundation and then a rapid increase toward the later stages, probably due to nutrient recycling. The invertebrate response is almost a sine wave, with a gradual increase, decline and then parallel increase toward the end of our experimental period. These relationships have baring on how these communities will respond with changing climatic patterns and how they will shift with high temperatures under short inundation durations and suggest that their response will be altered under prolonged inundation coupled with cooler temperatures and decreased evaporation during extreme rain events.

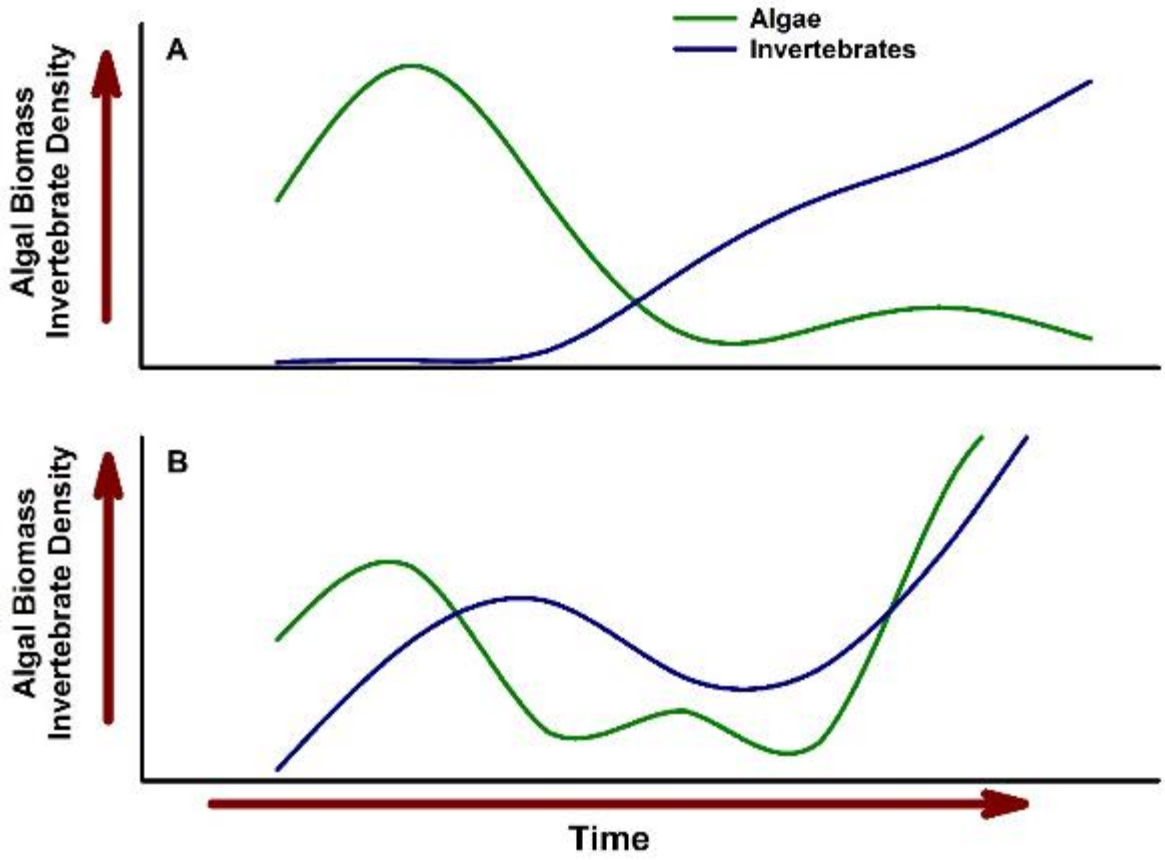

Figure 4.21 **The relationship between algal biomass accumulation and timing to that of invertebrate hatching and density at temperatures A) < 25°C and B) > 25°C.**

5 CHANGES IN EPHEMERAL WETLANDS WITH INUNDATION

Field data were collected to link changes of inundation level with physicochemical parameters and biological response. Hydrological periodicity of sampled sites range from seasonal to intermittent. Rains in July and August 2015 on the NMMU campus allowed for sampling to occur at the same level as the experiments for two inundation periods from initial fill to drying.

The three campus sites, 1765 (=1592), 1593 and 1594 were initial chosen and then 1641d was added in the second inundation event (Figure 3.2) and were sampled until dry. Sites 1592 and 1593 were at their maximum depth, 21 cm, on 27 July and slowly decreased until 1593 dried completely on 7 August (Figure 5.1). Even though 1594 was deepest, 23 cm, on 30 July, it was also dry by 7 August. The depression wetland of 1592 took the longest to completely dry, lasting until 11 August (Figure 5.1). During the second inundation event, depths were >25 cm for all sites, and after inundation on 24 August did not dry until 23-27 September.

Temperatures were recorded at each site and varied between 17 and 10 °C throughout the first inundation event, and 17 to 26 °C during the second inundation event (Figure 5.2).The pH at all sites and between 6.5 and 8.5 with an overall mean across sampling dates of 7.5, but was variable from site to site and over both inundation periods (Figure 5.3). The electrical conductivity (EC) began high in 1593, decreased markedly and then steadily increased over time. Sites 1592 and 1594 showed a pattern of increasing EC over time as well (Figure 5.3).

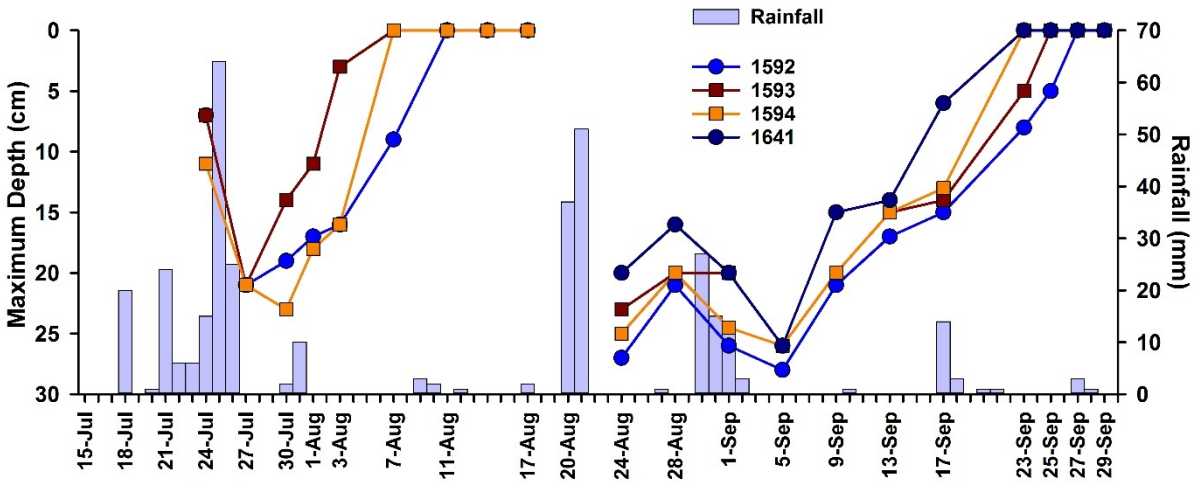

Figure 5.1 **Measured maximum depth estimate for NMMU Campus sites in relation to daily total rainfall. Round symbols are depression HGMs and squares represent wetland flats. 1592 = 1765 on Figure 3.2**

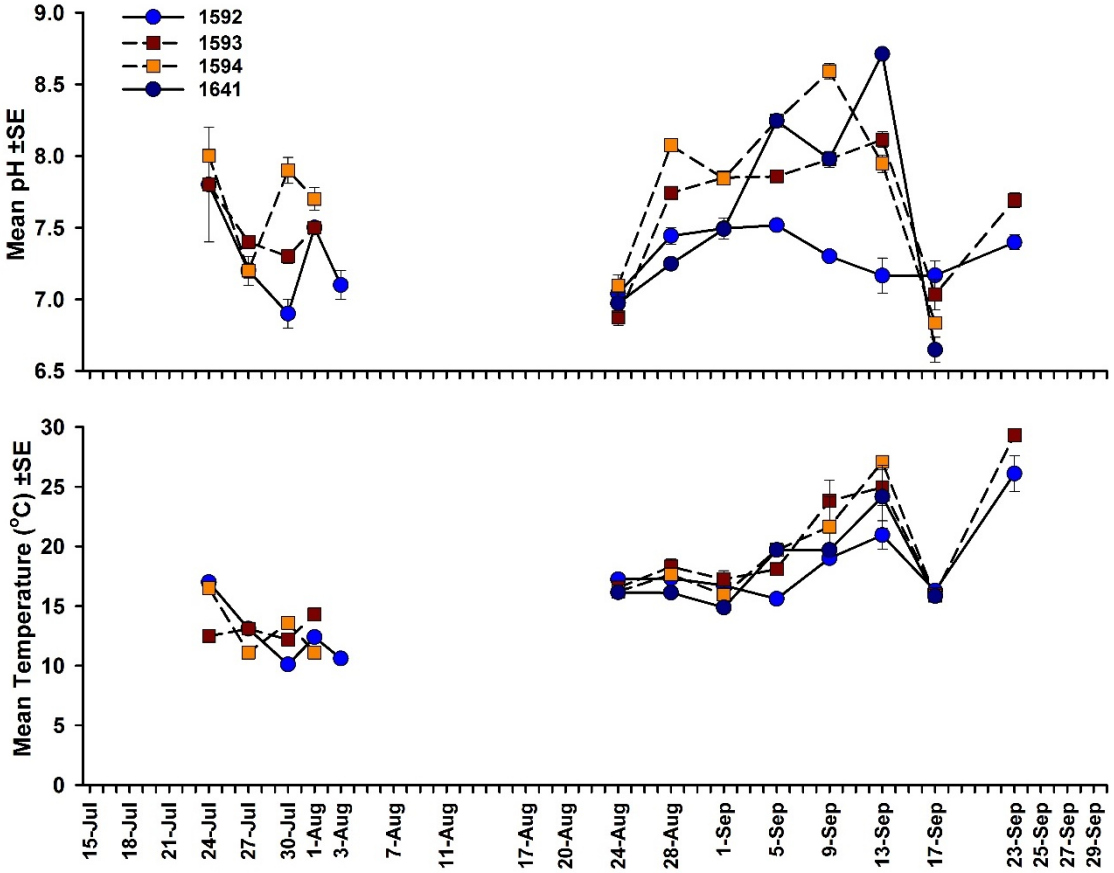

Figure 5.2 **Temperature (bottom) and pH (top) at each site during each inundation period, symbols as in Figure 5.1.**

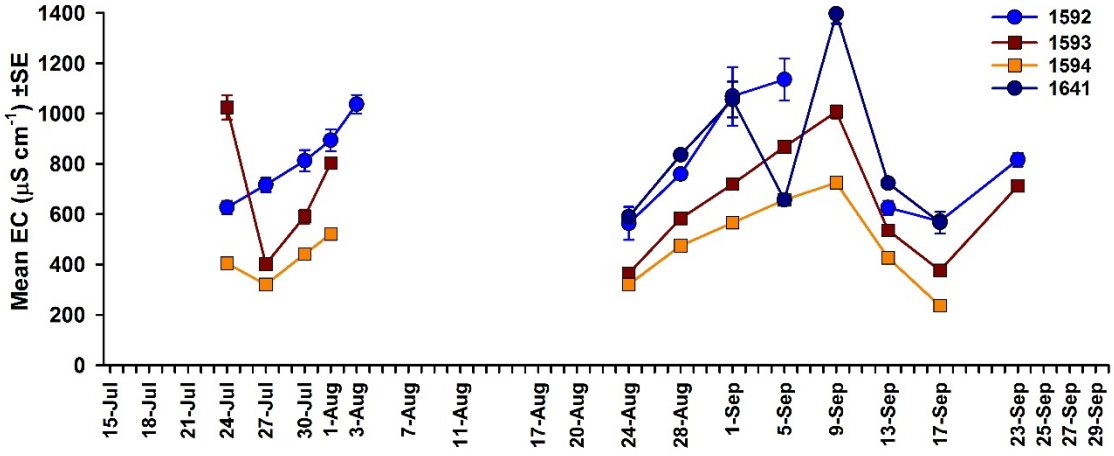

Figure 5.3 **Electrical conductivity (EC) at each site during each inundation period, symbols as in Figure 5.1.**

Phytoplankton biomass peaked approximately 4 days post inundation and steadily decreased over time (Figure 5.4). MPB biomass peaked approximately 8 days post inundation and maintained a lower level of biomass for the duration of the sampling period (Figure 5.4).

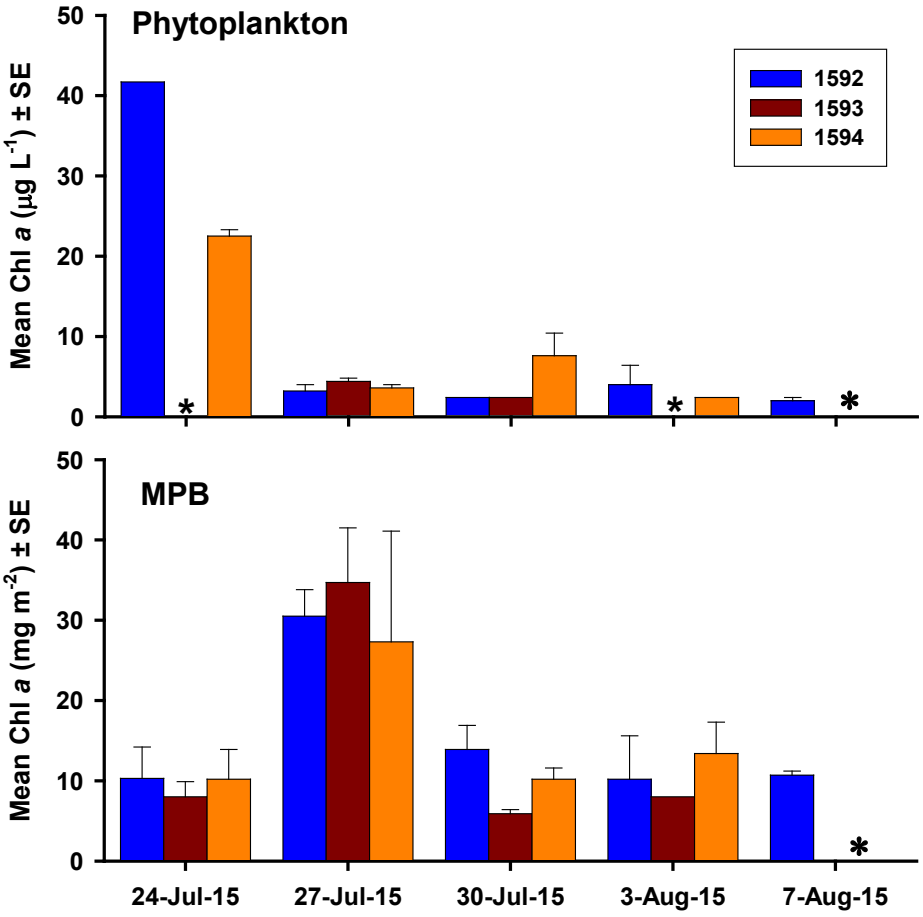

Figure 5.4 **Algal biomass, phytoplankton (top) from the water column samples and microphytobenthos (MPB, bottom) from sediment cores during the mid-July to beginning of August inundation period. Asterisks denote wetlands with too little water to sample or time when the wetland had dried so there was no sample.**

The pattern during the second event, a warmer and longer inundation period, was different form the first, in that the greatest peak for both phytoplankton and MPB biomass occurred at the end of the inundation cycle (Figure 5.5). The end spike in biomass could be due to concentration effects as the wetlands were drying (maximum depth was 8 cm and 5 cm for 1592 and 1593 respectively). Focusing in on the first 28 days, there was small peak of phytoplankton 4 and 8 days post inundation, but the main peak was about day 24 (Figure 5.5). The MPBs peaked around day 20 for most of the sites sampled.

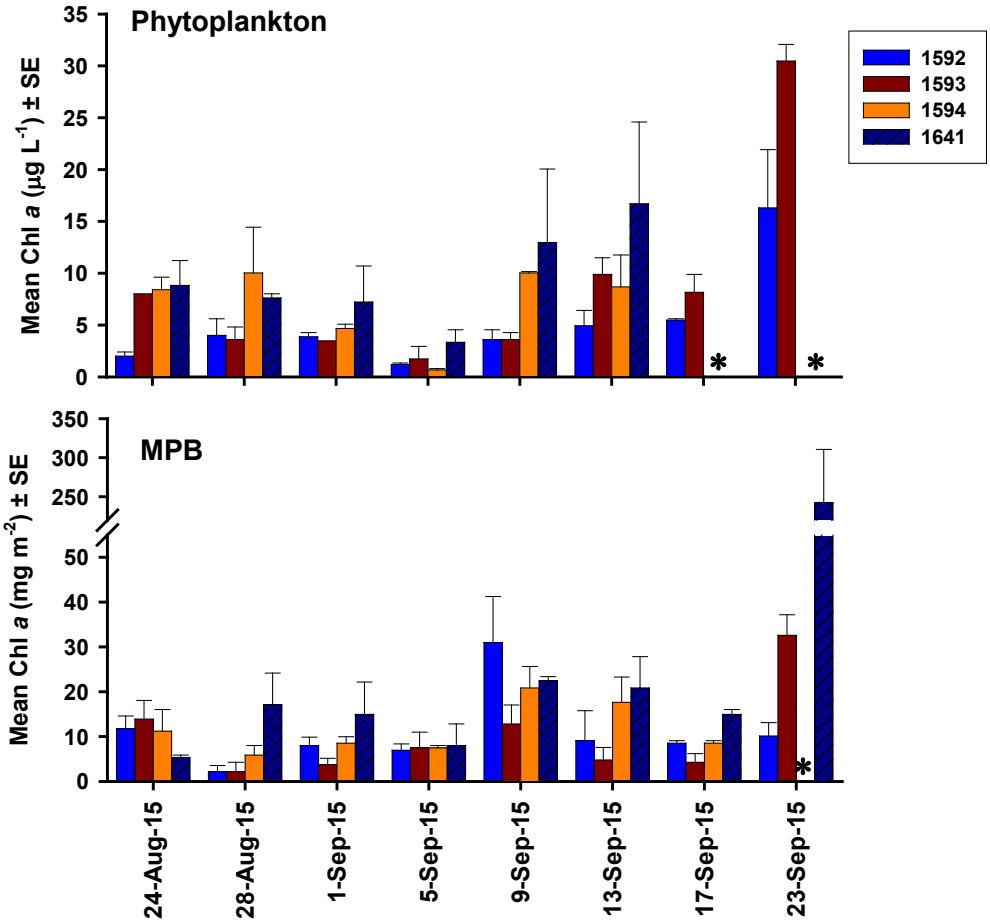

Figure 5.5 **Algal biomass, phytoplankton (top) from the water column samples and microphytobenthos (MPB, bottom) from sediment cores during the mid-August through September inundation period. Asterisks denote wetlands with too little water to sample or time when the wetland had dried so there was no sample.**

Phytoplankton samples were irregularly sampled during first inundation event in July (event 1) due to low water levels, the sites and dates where phytoplankton were collected showed that the Chlorophyta (greens) were the dominant group in sites 1592 and 1593, but they were co-dominant with cyanophytes (cyanobacteria) in site 1594 (Figure 5.6). The chlorophytes were the most specious, with 8 species, followed by diatoms with 4 (Appendix 5). Cyanophytes and euglenoids were represented by *Anabaena* spp. (2) and *Trachelomonas* sp. respectively (Appendix 5). Of the chlorophytes present, *Chlamydomonas* sp. was present in all samples across sites and dates and was the dominant green. Diatoms did not feature in sites 1593 and 1594, however were present in 1592 and by day 20, just prior to drying, *Gomphonema* sp. was co-dominant with *Chlamydomonas* sp. and *Scendesmus maximus* (greens).

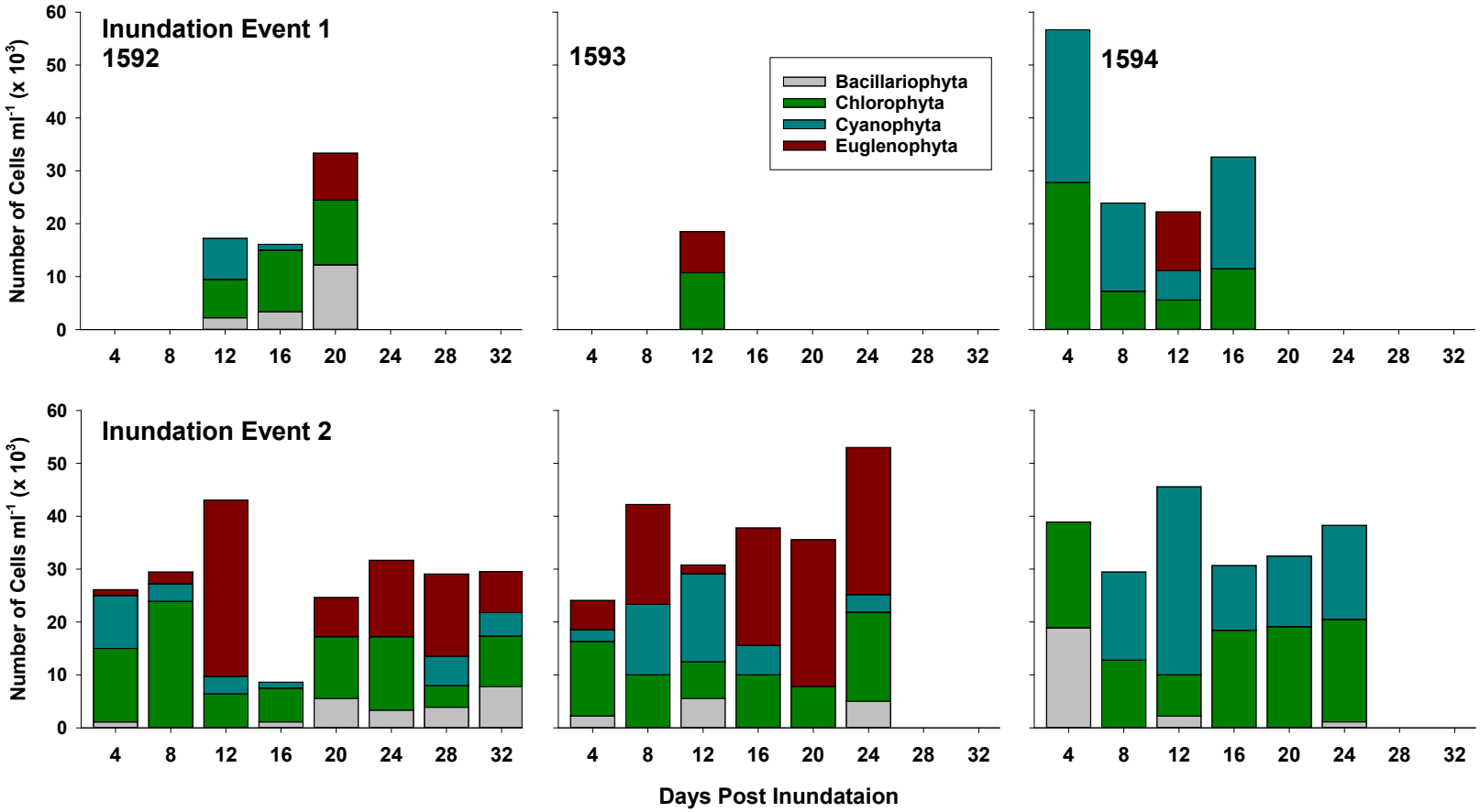

Figure 5.6 **Phytoplankton community (mean cells per ml) over the inundation periods between July and August 2015, top row is inundation event 1 and the bottom row represents event 2. Days with no bars were not sampled due to water levels too low for adequate sampling during event 1 and or complete drying of the site. Site 1592 is represented in the left panels, 1593 in the middle and 1594 on the left.**

Overall, there were more algal species in the second inundation event, due to longer duration generally and a longer period of depths >15 mm (~25 days vs < 10 days in event 1), before evaporation and drying occurred. There were 35 species identified, 23 chlorophytes, 11 diatoms, one cyanophyte and five euglenoids (Appendix 5). The phytoplankton community pattern was different during the second inundation event. In three sites, 1592, -93 and -94, greens were dominate for the first 4-8 days, with a community shift occurring by day 12 in all sites, but not all with the same pattern of dominance. The euglenoids (*Euglena* spp., *Phacus* spp. and *Trachelomonas* sp.) were dominant from day 12 (with some exceptions) until the end of the inundation period. I sites 1592 and 1593 (Figure 5.6) and *Trachelomonas* sp. was dominant in site 1641d during the entire period (Figure 5.7). Site 1594 was characterized by greens (*Chlamydomonas* sp., *Coccomonas* sp. and *Scenedesmus* spp.) and *Anabaena* sp. (cyanophyte).

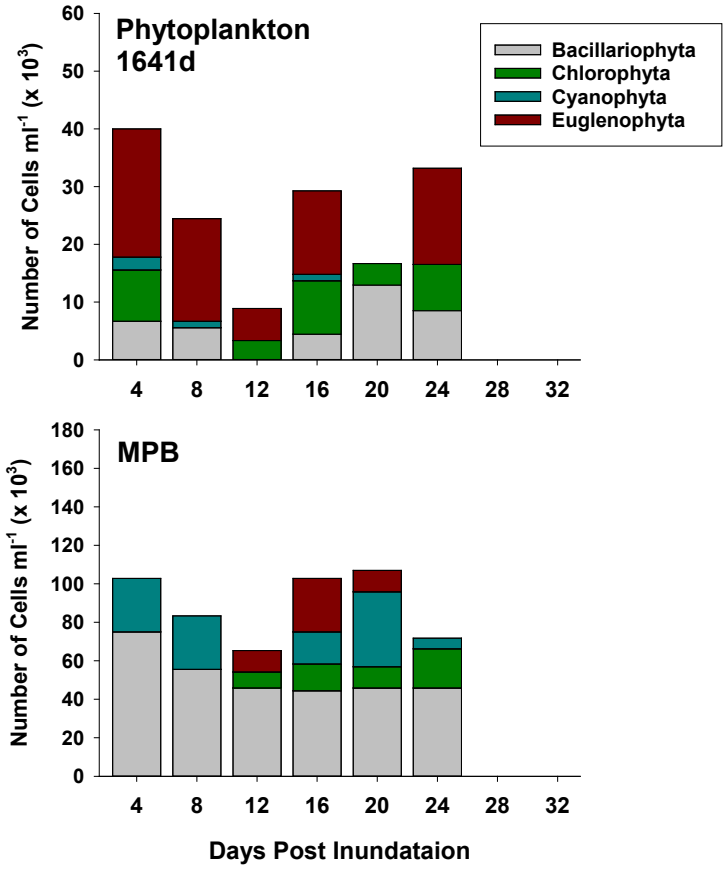

Figure 5.7 Phytoplankton community (top) and microphytobenthos (MPB) by (mean cells per ml) for inundation event 2 between August and September 2015 for site 1641d. Days with no bars were due to the complete drying of the site.

The MPB community during the second inundation event, like the phytoplankton, was more diverse with 32 species recorded versus the 19 during the first event (Appendix 5). This was also partly due to the additional samples and site. However, there was probably a temperature and inundation periodicity affect as well. Unlike the microcosm experiments, the MPB community was not dominated by chlorophytes during either inundation event (Figures 5.6 and 5.8), but generally by diatoms, and on some dates, cyanophytes. Diatoms across all sites and dates were represented by 16 species, chlorophytes had 11, cyanophytes 3 and euglenoids by 2 (Appendix 5).

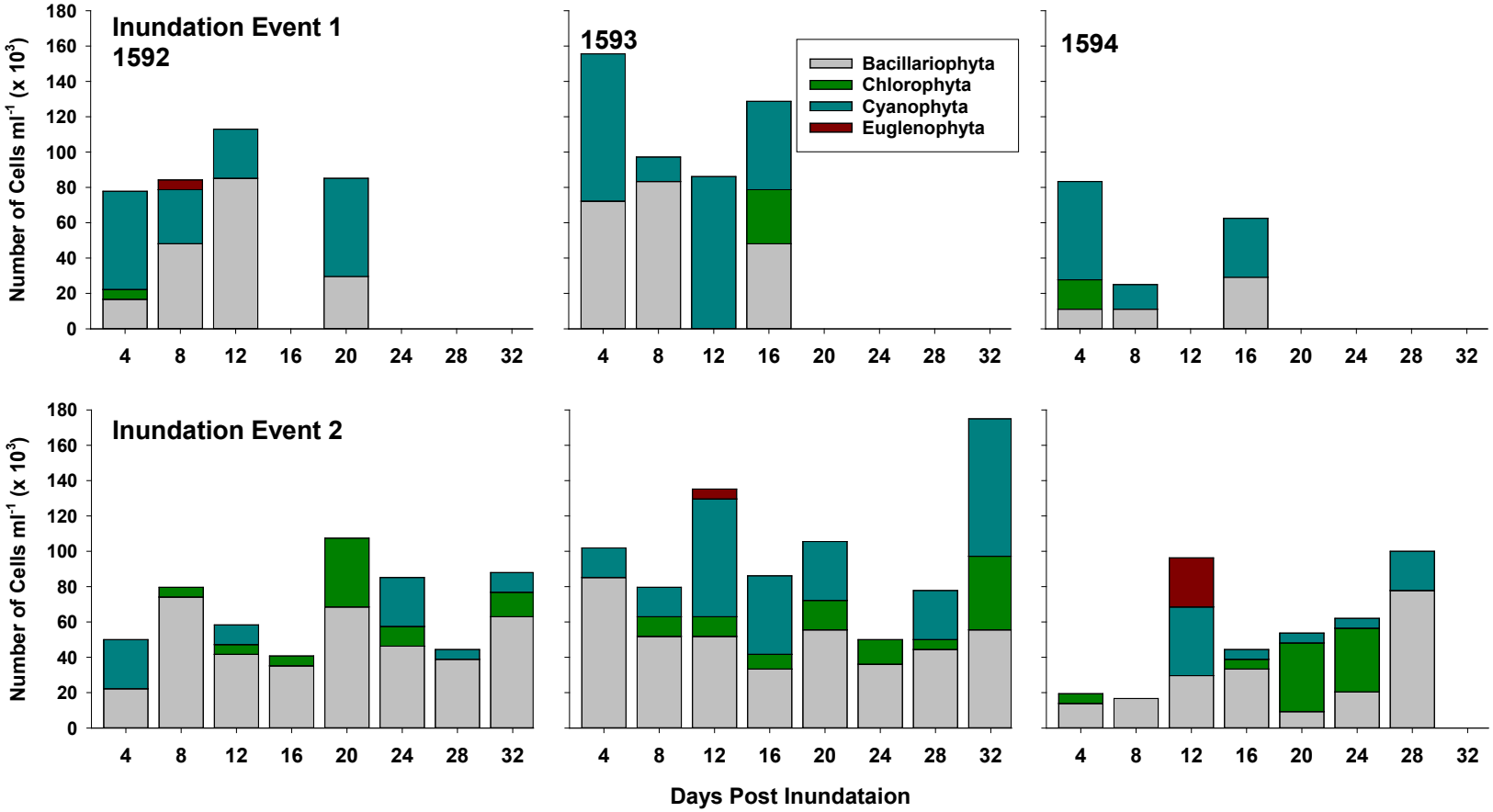

Figure 5.8 **Microphytobenthic community (MPB, mean cells per ml) over the inundation events between July and August 2015, top row is inundation event 1 and the bottom row represents event 2. Days with no bars were not sampled due to water levels too low for adequate sampling during event 1 and or complete drying of the site. Site 1592 is represented in the left panels, 1593 in the middle and 1594 on the left.**

There were four dominant genera of diatoms that were represented across sites and over the inundation periods, *Nitzschia* spp., *Navicula* spp., *Craticula* spp. and *Amphora* sp. Once again, the filamentous cyanophyte present at all sites was *Anabaena* spp., with site 1592 having additional species in lower densities (see Appendix 5 for taxa). Euglenoids were present, however not in the numbers seen within the phytoplankton samples (Figures 5.5-5.8). The chlorophytes, when present were dominated by *Scendesmus* spp., similar to the microcosm experiments.

The pattern of invertebrate numbers during inundation period one was very similar to those seen in the experiments, with the peak numbers on days 12 and 16 before declining. During the August/September inundation (event 1) invertebrate numbers peaked for 1592, 1593 and 1641 on days 8 and 12 (Figure 5.9). The pattern for 1594 was different, delayed to days 20 and 24 and generally lower than the other sites (Figure 5.10), because of influx of insects migrating into the area and colonizing.

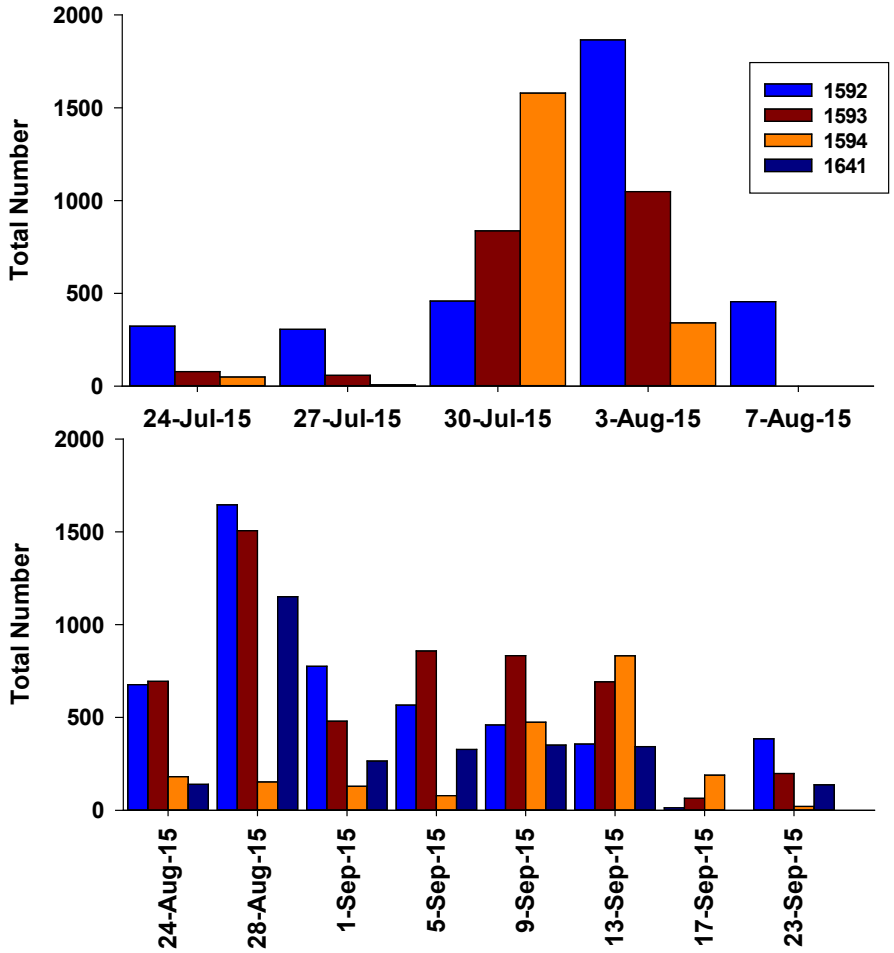

Figure 5.9 Total number of invertebrates collected during the first inundation event in July (top) and then during the mid-August through September inundation event (bottom).

For ease of presentation and interpretation, invertebrates were split into functional groups rather than taxonomic orders or families (Figure 5.10, Appendix 6). Obligate ephemeral wetland taxa of An ostraca, Notostraca and Conchostraca were grouped as “branchiopods”.

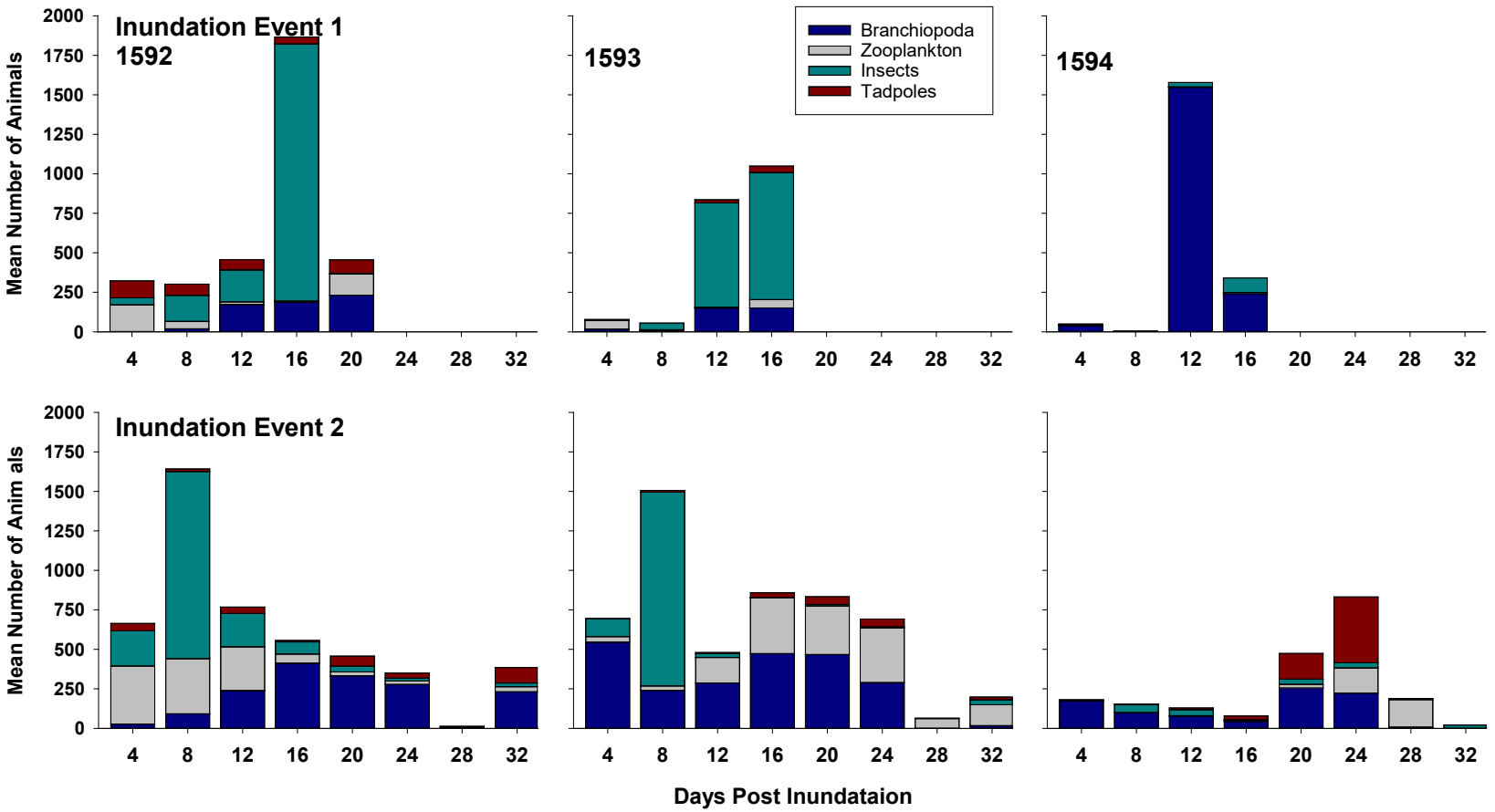

Figure 5.10 **Invertebrate community composition by functional group for sites 1592 (left), 1593 (middle) and 1594 (right) for inundation event 1 (July-early August 2015, top row) and event 2 (mid-August – mid-September 2015, bottom row). "Branchiopoda" includes Notostraca, Conchostraca and Anostraca; "Zooplankton" includes Cladocera, Copepoda, and Ostracoda, "Insects" includes all fully aquatic and aquatic stages of insects. A full 32-day series is shown here for comparison between events and algal figures, days without bars denote times when depth was too shallow to sample or wetland was dry (see Figure 5.1).**

Although Cladocera are technically also part of the Branchiopoda, they were grouped in with general "zooplankton" as those identified at these sites are small and not confined solely to ephemeral systems, but occur and I wide range of water bodies, just as the Copepoda and Ostracoda. All aquatic and semi-aquatic stages of insects were group together. Although tadpoles are vertebrates, they are an important part of the aquatic fauna and food web of the wetlands, and as can be seen in Figures 5.10 and 5.11, and substantial in terms of numbers in some sites. Although biomass was not determined, tadpoles contribute the greatest proportion of animal biomass, when they are present.

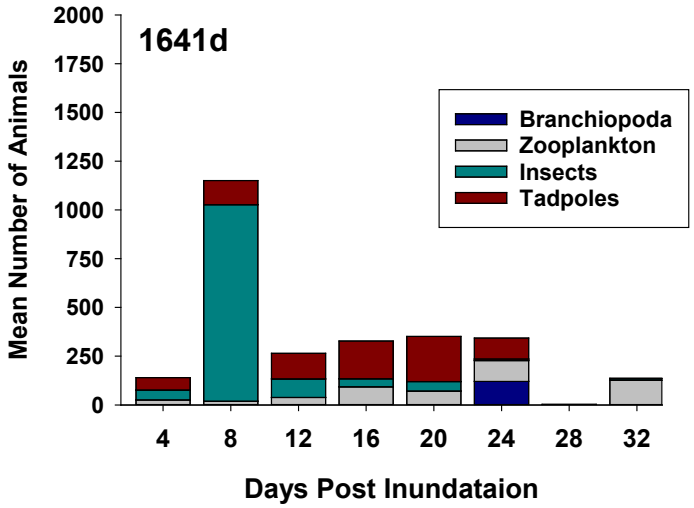

Figure 5.11 Invertebrate community composition by functional group for site 1641d for inundation event 2 (mid-August – mid-September 2015). Functional groups as described in Figure 5.10.

Zooplankton and branchiopods were the dominant groups immediately after inundation in most sites during both inundation events. The exception was site 1641d, where insects and tadpoles were immediately dominant (Figure 5.11). Insects moved in quickly and were dominant in sites 1592 and 1593, primarily made up of Chironomidae (semi-aquatic, mud dwelling taxa) and Culicidae (mosquitoes) by day 8 (Appendix 6 for species list). Inundation event 1 was characterized by the successful hatching and development of *Triops granularis* (Notostraca) and *Branchiopodopsis hodgsonii* (Anostraca), both obligate ephemeral wetland taxa. Inundation event 2 not only had *T. granularis* and *B. hodgsonii* but also the conchostracan, *Leptestheria rubidgei*.

In terms of trophic status, grazers or the grazing stages (i.e. nauplii of *T. granularis*) of invertebrates and tadpoles were present, days 4-12, thereafter there was a mix of planktivorous grazers (*B. hodgsonii*, *Daphnia* sp., culicids and some Chironomidae), benthic grazers (*L. rubidgei, Moina* spp., tadpoles, ostracods, and tadpoles) and predators (cyclopoids, dytiscids and *T. granularis*). This shift from algivore dominated community to a predator one is most probably why there was rebound in algal biomass toward the end of the inundation period, along with the concentration affect due to the evaporation drying up process of the wetlands.

6 IMPLICATIONS FOR A WARMER MORE UNCERTAIN TIME

Combining experimental evidence, current field data collection and past data an assessment of the ecosystem services given by the small, ephemeral wetlands indicative of NMBM currently and project what could happen with different climate change scenarios to those services. Two extremes of the climate change spectrum were chosen to demonstrate this change, severe prolonged drought conditions and a period of prolonged wet or flood conditions. As temperature is generally more predictable and "stable", rainfall, especially in this region is not. The projections are for a 10-20% decrease in mean annual rainfall and a 2.2% increase in evapotranspiration levels (IPCC 2014a), however, one of the main difficulties is that day to day changes cannot be projected and it is widely thought that the variance or patchiness in rainfall in location, amount, duration and timing will increase (Schulze 2011, IPCC 2014c). In a region that already has high variance in rainfall on a monthly level, it is thought that there will be more extreme weather patterns with extended periods of drought and extended periods of flooding and less seasonal predictability (Schulze 2011, IPCC 2014c). Ephemeral wetlands, especially depression and wetland flat HGMs do not, in pristine, "normal" conditions give a high level of direct ecosystem services. They generally provide more indirect services such as biodiversity, tourism and research. As such, systems that have "low" direct human value, will show a decline in those services they do provide in prolonged dry phases. In prolonged wet phased and floods their direct and indirect services do generally increase. However there is a potential that the water quality, under increased urbanization (nutrient input) and higher temperatures could decline and produce large amounts of nuisance cyanophyte blooms and increased nuisance insects which could carry water borne diseases.

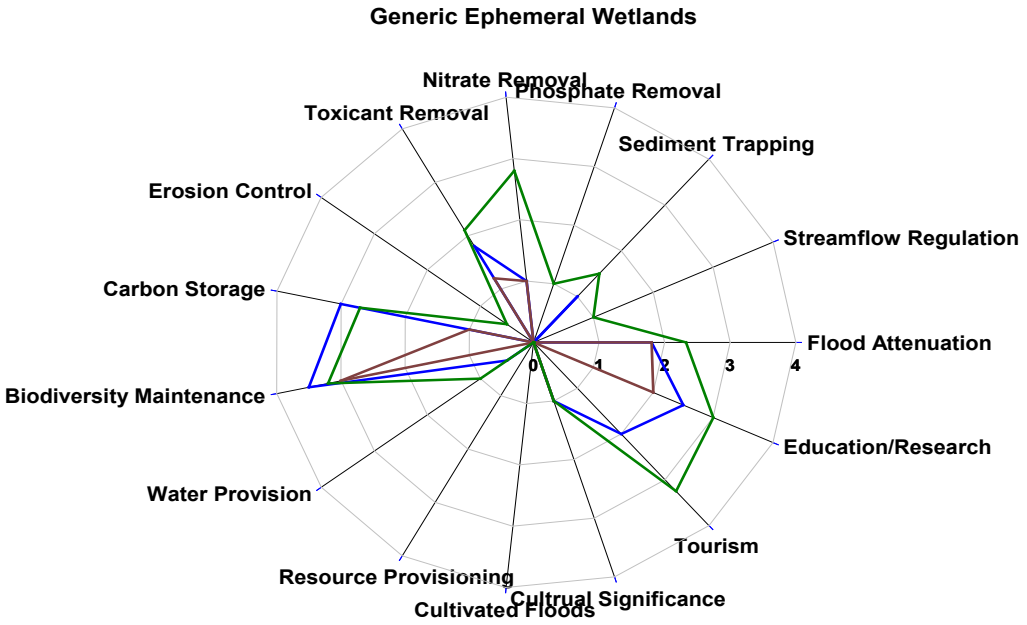

Figure 6.1 Ecosystem service provision for representative ephemeral wetlands in the NMBM area as evaluated by WET-EcoServices (Kotze *et al.* 2009). Scale ranges from 0 (no service) to 4 (highly effective given the opportunity to provide the service). Blue line represents present service provision, brown line represents provision under prolonged dry conditions, green line represents provision under prolonged wet conditions.

PART 2

URBAN WETLAND ASSESSMENT

7 URBAN WETLAND FIELD SAMPLING

Unlike the other parts of the study, the urban wetlands are permanently inundated, although historically these sites were most likely seasonal wetlands that have changed with the urbanization of their landscape.

Pond 6 is located within the Nelson Mandela Bay Metropolitan area (NMBM) along the South Western floodplain of the lower Swartkops Estuary (Figure 7.1). Baseline data for Pond 6, a largely degraded urban wetland was done towards providing information to NMBM and Working for Wetlands for a planned rehabilitation project. The wetland was sampled prior to a planned clean-up as a once-off exercise in March 2015. The first phase of clean-up was scheduled to begin 30 March. Follow-up sampling, with a reduced number of parameters was then completed in April, June and July 2016.

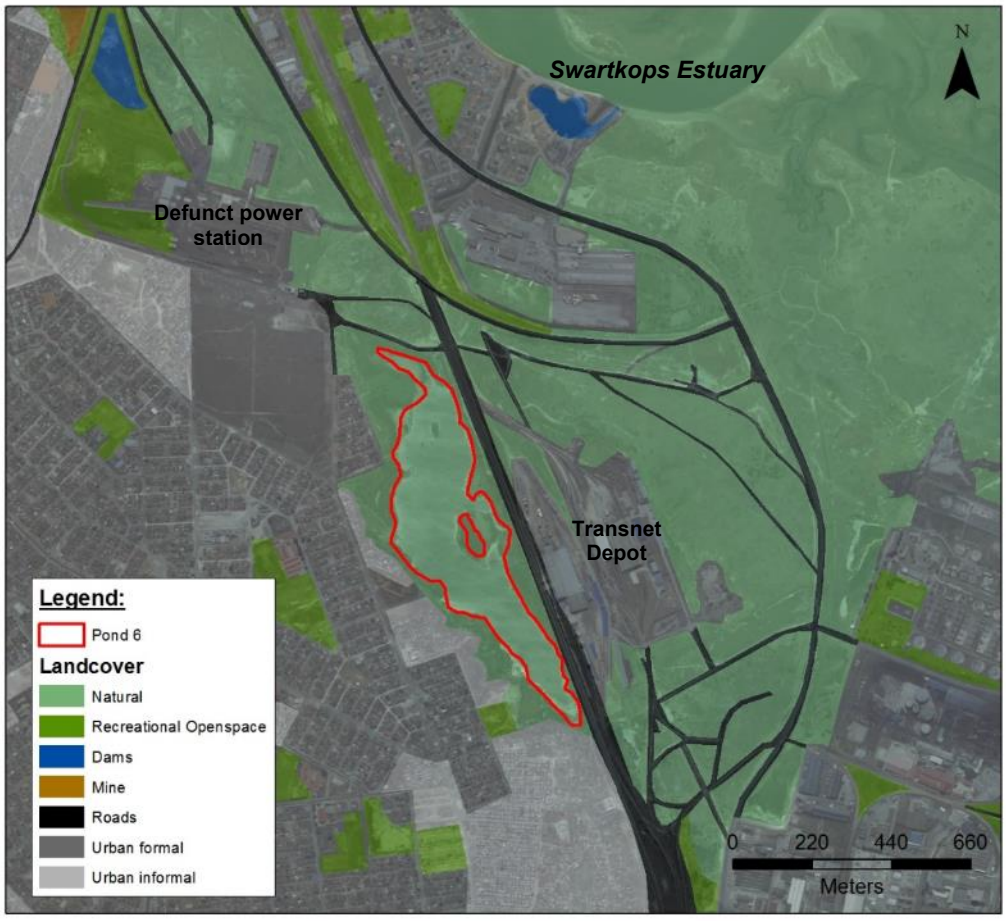

Figure 7.1 Land cover and setting for the urban wetland, Pond 6.

Stewart (2009)classified the NMBM into 12 broad habitat units consisting of 52 unique vegetation types with the dominant vegetation biomes being Fynbos and Subtropical Thicket biomes. The wetland is located within the Sundays Valley Thicket and Wetland type habitats both occurring within the Subtropical Thicket biome. The dominant vegetation type found within the catchment is the Motherwell Karroid Thicket (Stewart 2009).

Three sites were chosen within the wetland for the 24-25 March 2015 sampling, one at the northern most section near one of the storm water inflows, a second site closer to the central portion of the

wetland, also near a storm water inflow and the last at the southernmost end and below the last of the inflow areas (Sites 1-3, Figure 7.2). Basic physicochemical parameters were recorded, two replicate water samples collected for nutrient analysis and phytoplankton biomass as well as additional samples for determination of total and faecal coliforms as well as *E. coli* (the later were taken to accredited national laboratory, PathCare Lab code 8160EF) at each site. Two to three replicate samples per site were collected for MPB biomass determination, dependent on available substrata. One sample from each site was collected for phytoplankton and MPB community analysis and fixed with Lugol's solution. Macroinvertebrate samples were collected with a kick-net with 1 mm mesh size, each sample was timed for 1.5 min. Invertebrates were then fixed in 70% EtOH in the field. Sediments were collected at each site from 1 to 3 points dependent on the topography of each site. Cores were augured and observational data (colour and texture, presence/absence of wetland soil characteristics such as mottles) recorded every 10 cm from the surface to a maximum of 60 cm. Vegetation was surveyed from 10 m^2 plots from a gridded map at stratified random points of vegetation change in order to get a representation of vegetation present. The Braun-Blanquet system of plant cover determination was used for each species type present. Unknown taxa were collected and tagged in the field for further identification in the laboratory. Soil moisture was recorded using a soil moisture probe, at three different points within each plot. The height of the dominant plant species was measured and recorded. Water fowl were not quantitatively sampled, but the species and estimated numbers of each species were recorded. A list of sited birds from the area was accessed from the UCT Avian Demography Unit database, from the Coordinated Waterbird Counts Project (CWAC, http://cwac.adu.org.za/sites.php?sitecode=33522536) between 1998 and 2009, combining summer and winter surveys in all years, to augment data for health assessment. Methods of data collection can be found in Taylor (1999) and on the ADU website, http://cwac.adu.org.za/instructionsprotocol.php.

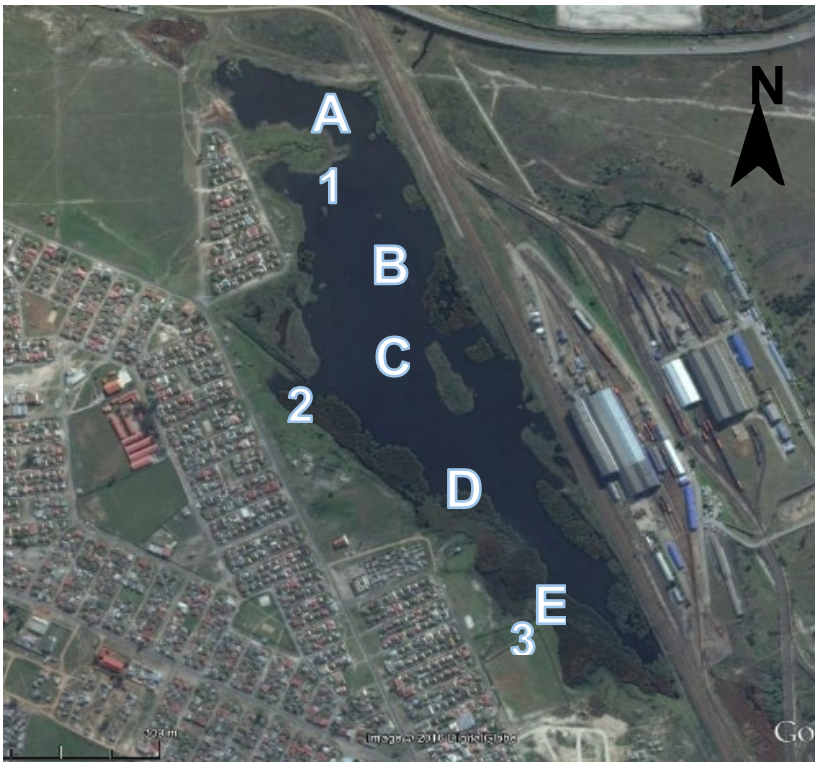

Figure 7.2 Sampling sites for Pond 6, 2015 sites numbered 1-3; 2016 sites letters A-E.

The 2016 sampling (26 April, 21 June and 28 July) concentrated on recording physicochemical parameters, the collection of water samples for nutrient analysis, chl *a* biomass of phytoplankton and MPB, as well as algal community. Vegetation cover of the dominant wetland was mapped for the entire wetland. A health assessment was completed using WET-Health tool of Macfarlane *et al.* (2007). Possible ecosystem services were determined using WET-EcoServices (Kotze *et al.* 2009) assessment tool.

A second urban wetland, Bridgemeade, was added in 2016 (Figure 7.3). This wetland is under anthropogenic pressures, but to a lesser extent than Pond 6 and with a more recent transformation from a peri-urban to fully urban landscape setting. There has been an extensive and intensive build-up of urban formal housing in the area, alien tree infestation (*Acacia longifolia*) and drainage engineered by NMBM to protect the new housing developments that were not well placed given their proximity to the wetland.

Figure 7.3 **Bridgemeade wetland showing sampling five sampling sites around the wetland. The "main wetland" sampling points designated A, C and E. Two of the three engineered retention ponds were sampled and are designated as R1 and R2.**

The sampling regime for this wetland was similar to that of Pond 6 in 2016 occurring on 27 April, 22 June and 25 July, focusing on physicochemical parameters, water for nutrient level determination and algal samples. Vegetation was done at a broad scale, concentrating on mapping the distribution of the major plant species. Sediment cores were collected at 3 sites for particle size, organic matter, soil moisture and physicochemical (pH, EC, REDOX) measurements in the lab.

Additionally, heavy metal content of Cadmium (Cd), Chromium (Cr), Manganese (Mn) and Aluminum (Al) was determined for sediments and *Typha capensis* in Pond 6 and Bridgemeade in 2016. Sediment cores were collected at Pond 6 in 2015 and concentrations of Cd, Cooper (Cu), Mn, Nickle (Ni) and Zinc (Zn) were determined.

7.1 Laboratory Processing

Laboratory processing follows the same methods as discussed in Chapter 4, Section 4.3 with the exception of heavy metals analysis discussed below.

The *T. capensis* plants collected at Pond 6 and Bridgemeade were washed thoroughly with deionised water to remove periphyton and sediment (Phillips *et al.* 2015). Each wetland plant was fractionated into roots, stems and leaves. 50g of each fresh sample will be dried at 70°C for 48 hours and ground using an analytical mill and homogenised to ensure even element distribution and then kept for heavy metal analysis (Phillips *et al.* 2015).

Sediment and plant samples for heavy metal analysis were digested using an overnight wet digestion with HNO_3 method from Du Laing *et al.* (2003). All samples were chemically analysed for presence of heavy metals using Flame Atomic Absorption Spectrometer (AAS). Standard series for all metals will be prepared from stocks of known concentrations and will be represented in ppm (parts per million) equivalent to mg/L.

8 URBAN WETLAND ASSESSMENTS

8.1 Pond 6 Survey

Pond 6 is a highly disturbed permanent wetland in the middle of a formal and informal urban development and industrial area (Figure 7.1). It is a hydrologically permanent system currently, maintained by a system of storm and waste water discharge channels. It was most likely an ephemeral system in the past and part of the relic floodplain system of the Swartkops River. Using the wetlands classification system of Ollis *et al.* (2013), it is classified as an unchannelled valley bottom wetland HGM. The wetland is approximately 1.6 km in length and 0.55 km in width and covers an area of about 0.175 km^2. It is generally a large, shallow water, ecosystem with a depth range between 0.75 m to 1.2 m. As a result of the anthropogenic pressures taking place within the immediate catchment the wetland conservation and rehabilitation attention is required (NMBM SOER 2011). The wetland has become a dumping ground for the local community as evidenced by the large amounts of litter in and around the wetland, from small plastics from household refuse, to large building rubble and large household item disposal (furniture, tyres etc.). Despite interventions by different organizations to initiate clean-ups over the last several years, and the latest occurring in April 2015 by Working for Wetlands, this solid waste pollution persists. The area is also heavily grazed by cattle kept in the area (Plate 8.1). Vegetation could be seen to be grazed to the base, especially some of the tenderer wetland plants such as *Schoenoplectus decipiens*. Other plants, such as the dense stands of reeds of *Typha capensis* and *Phragmites australis*, however were not grazed. There were also newly erected informal shacks and long drops near the margins at the southern end of the wetland in March 2015 (Plate 8.2) which persisted in 2016.

Plate 8.1 Cattle grazing on the edge of Pond 6 during a site visit 29 March 2015.

On the March 2015 sampling date, the wetland was estimated to be about half full, the water was very cloudy and green with a distinct odor of hydrogen sulfide and algae. Maximum depth of 70 cm was recorded at site 3 (Table 8.1), but it is estimated that the maximum depth was probably a little over 1 m at central parts of the wetland. EC and Salinity levels show the water to be brackish with slightly alkaline pH levels. The TDS levels were very high, as levels should be <1 000 mg L^{-1} (DWAF 1996b), and the levels measured were >2 300 mg L^{-1} (Table 8.1).

Plate 8.2 **General state of the outer edge of the wetland taken at the southern end (Figure 6.2), arrows show the *Phragmites australis* (and *Typha capensis*, not shown) fringe at the edge of this portion of the wetland, as well as the different types of rubbish and building rubble evident from the north to the south of the wetland.**

Table 8.1 **Physicochemical data for the three sites sampled at Pond 6 in March 2015.**

	Site		
Parameter	**1**	**2**	**3**
Depth (cm)	45	45	70
Temperature (°C)	22.4	22.7	26.4
DO (mg/L)	4.2	4.1	20.0
EC (mS/cm)	5.39	5.46	5.83
pH	8.34	8.35	8.63
Salinity (PSU)	3.1	3.1	3.1
TDS (g/L)	23.0	23.3	23.2

Bacterial levels exceed target recreational water quality levels (DWAF 1996c), and most definitely are well above any domestic water use levels according to DWAF (1996b) (Table 8.2). Risk of gastrointestinal infection is high for full contact with water with >2000 per 100 ml of faecal coliforms, and >400 per 100 ml of *E. coli* (DWAF 1996b). The risk of gastrointestinal illness decreases with intermediate contact with water with 1000-4000 faecal coliforms per 100 ml (DWAF 1996b). Faecal coliforms also exceed levels for young and mature cattle, target levels are <200 per 100 ml, with levels greater than >1000 per 100 ml putting young cattle with significant risk of infection and mature cattle at risk for infection (DWAF 1996d). All water quality guideline levels were exceeded at 2 of 3 sites, with only mid-level risk at site 3.

Table 8.2 Bacterial counts from two replicate samples at each Pond 6 site, the values for most of the bacterial types exceeded laboratory limits, where an average of the two samples was possible the standard deviation is given in parentheses.

Site	Total Coliforms (#/100 ml)	Faecal Coliforms (#/100 ml)	*E. coli* (#/100 ml)
1	>2420	>2420	>2420
2	>2420	>2420	>2420
3	>2420	216.5 (± 3.5)	129.5 (± 2.1)

Algal biomass was high for both phytoplankton and MPB levels (Table 8.3). Water quality targets >30 µg L^{-1} are considered to be at nuisance levels and aesthetically unacceptable (DWAF 1996c). Nutrient levels were not particularly high given its urban context (Table 8.4), but can be considered P limited, as TN levels were greater than TP, with an average N:P ratio of 41:1. The relatively high TN levels across the wetland denote a mesotrophic state (500-2500 µg L^{-1}) according to DWAF (1996a), bordering on an eutrophic state. The major difference is the possibility of nuisance algal blooms of cyanobacteria in particular, if the input rates are increased. The TP levels do exceed levels considered by DWAF (1996a) as being in an eutrophic state, which ranges between 25-250 µg L^{-1}, the average over the three sites was 61 µg L^{-1}, although not without some variability. Although, both algal biomass primary nutrient levels denote the wetland would be considered eutrophic, none of the nutrients when combined with the pH, DO and temperature levels at the time of sampling could be considered to be at toxic levels. Cyanobacteria were present, but not to the levels that could be considered to be toxic to animals or humans.

Table 8.3 Algal biomass at each of the Pond 6 sites in March 2015 (n = 2), see Figure 7.2 for site locations.

	Phytoplankton Chl *a* (µg/L)		MPB Chl *a* (mg/m^2)	
Site	**Mean**	**SD**	**Mean**	**SD**
1	305.8	6.0	105.5	65.5
2	273.7	1.5	52.7	20.9
3	560.3	72.6	61.2	41.8

Table 8.4 Mean nutrient (±SD) levels from 2 replicated sampled collected at each of the three sites at Pond 6 (no replicate for TN). TN = total nitrogen; TP = total phosphorus; SRP = soluble reactive phosphorus.

	Site 1		Site 2		Site 3	
Nutrient	**Mean (µg L^{-1})**	**SD**	**Mean (µg L^{-1})**	**SD**	**Mean (µg L^{-1})**	**SD**
Ammonia	34.7	37.8	23.7	17.4	28.7	32.8
Nitrate	79.5	13.5	1224.5	419.1	111.5	0.2
Nitrite	45.5	22.0	121.4	85.3	42.2	26.7
Silica	101.2	68.9	202.5	53.0	63.7	12.4
TN	2125.9	-	2961.9	-	1671.9	-
TP	61.4	65.9	44.6	32.4	81.3	71.9
SRP	25.6	12.8	23.7	10.7	20.4	2.1

Sediment was characterized from core samples collected at each sampling site. All sites were gravel and sand dominated (Figure 8.1). Site 1 was characterized by gravel and medium sand, whereas Site 2 was dominated by fine and medium sand. Sediment at Site 3 was found to be more evenly distributed amongst the size classes, but with medium sand and gravel dominating. Soil pH was circum neutral and conductivity was relatively high (Table 8.5), and much higher than was measured in the surface water (Table 8.1). The percentage of soil moisture and organic matter (OM) content are also shown in Table 8.5 for each site sampled. Approximately 96% of soils in South Africa have less than 2% OM (Du Preez *et al.* 2011). Organic matter in NMB ephemeral wetlands averaged approximately 3.4% (Schael *et al.* 2015), which demonstrates that the averge of >11% in Pond 6 is high for this area by comparison and expectation. This suggests an organic layer build-up that might be considered "peat" in the context of semi-arid regions in South African (Soil Classification Working Group 1991, Job 2009).

Table 8.5 Sediment electrical conductivity (EC), pH, soil moisture (SM) and organic matter (OM) for replicate sediment core samples collected at three sites at Pond 6.

	EC (mS cm^{-1})		pH		SM %		OM %	
Site	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
1	25.5	3.8	7.5	0.2	13.0	1.9	12.3	0.6
2	20.9	9.0	7.6	0.2	28.0	3.0	9.4	3.8
3	9.8	1.0	7.4	0.1	35.5	10.7	12.8	4.5

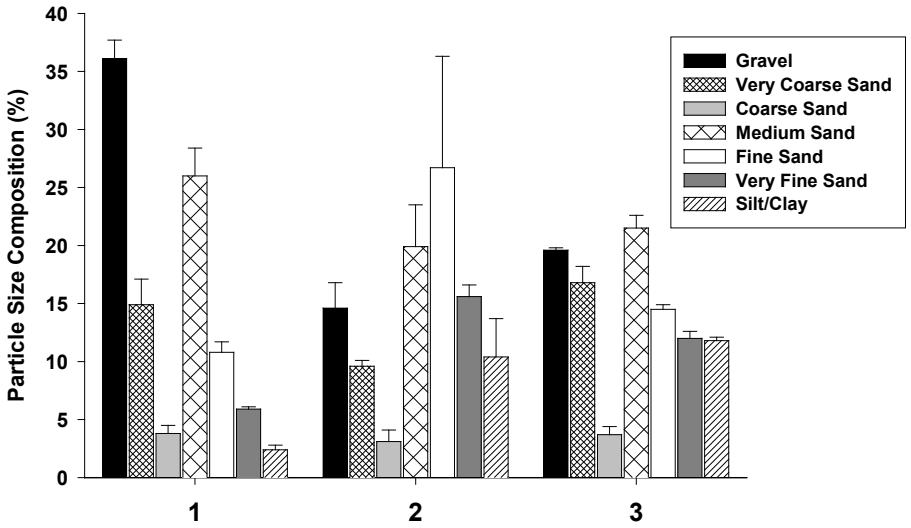

Figure 8.1 Sediment particle composition from samples collected at three sites at Pond 6 (Figure 2.5). Categories are based on the Wentworth scale (Wentworth 1922).

Sediment collected at each of the 3 sites in 2015 were analysed for heavy metal content for five elements (Section 7.1). Overall the values for all metals averaged over the three sites are shown in Table 8.6. Cadmium values were higher than what would be considered acceptable (DWAF 1996b), the other elements have high levels, however are within acceptable limits. Levels did not differ significantly between the three sites, however there was a pattern of a steady decrease in Ni from site 1 to 3, and variability between replicates of Zn.

Table 8.6 Average concentrations (mg kg^{-1}) and standard deviation (SD) of heavy metals measured in sediment samples from three sites within Pond 6 (n = 6 for each element), March 2015. Cd = Cadmium; Cu = Copper; Mn = Manganese; Ni = Nickel; Zn = Zinc.

	Concentration (mg kg^{-1})	
Element	**Mean**	**SD**
Cd	87.6	4.4
Cu	111.8	9.6
Mn	288.3	76.1
Ni	144.4	112.6
Zn	120.3	56.4

Invertebrate samples were dominated by the hemipterans Notonectidae (*Anisops* spp.) and Corixidae (*Sigara* spp.) with a few *Chironomus* sp. (Chironomidae: Diptera) and very little else. There was evidence of fish present, but these were not sampled. There was a very low biodiversity of invertebrates but high numbers of those taxa present. The taxa identified are all very tolerant of a wide range of water quality and adapted to low oxygen levels.

Vegetation plots showed large areas of dominant plant species with very little diversity. There was a mix of alien invasive weeds as well as typical wetland sedges and reeds (Table 8.7). Lower floodplain/valley floor wetlands along the coast are not necessarily highly diverse by nature.

Table 8.7 Presence/Absence of dominant plant species found within the plots examined at the Pond 6 wetland.

Family	Taxon	1	2	3
Aizoaceae	*Delosperma patersoniae*		+	
Amaranthaceae	*Sacrocornia pillansii*	+	+	
Anacardiaceae	*Rhus* sp.		+	
	Searsia laevigata		+	
Asteraceae	*Felicia* sp.	+		
	Scenacia sp.		+	
	Scenacia sp./*Microglossa* sp.	+		
	Xanthium stumarium	+		
Cyperaceae	*Bolboschoenus maritimus*	+		
	Schoenoplectus decipiens	+	+	+
Juncaceae	*Juncus kraussii*	+		
Mesembryanthemaceae	*Carpobrotus deliciosus*		+	
Poaceae	*Cynodon dactylon*		+	+
	Paspalum disticum	+		+
	Pennisetum distichum	+		+
	Pennisetumlandestinum c	+	+	
	Phragmites australis			+
	Setaria sp.	+	+	+
Solanaceae	*Lycium cinereum*	+	+	
Typhaceae	*Typha capensis*	+		+

Despite the highly polluted nature of the wetland, the bird life was seen to be thriving. Numbers were observational estimates and only birds that were readily recognisable were recorded (Table 8.8) although only 5 species were seen on the day of sampling as per casual observation, 56 species have been recorded in the area through the CWAC (Appendix 3).

Table 8.8 List of water birds recorded at Pond 6 on 29-30 March 2015 with estimates of numbers.

Species	Common Name	Relative abundance
Phoenicopterus ruber	Greater Flamingos	>100
Fulica cristata	Red-knobbed Coot	20-50
Vanellus armatus	Blacksmith Lapwings	10-20
Bubulcus ibis	Cattle Egrets	20-50
Platalea alba	African Spoonbills	5-10

To strengthen our baseline data and bring in elements of seasonality, several follow-up sampling events were done in 2016. Two sampling sites were added (Figure 7.2). The physicochemical parameters measured at all sites in April, June and July 2016 are given in Table 8.9. The majority of the parameters recorded were similar from month to month and between years. However, there was a notable decrease in TDS values between end of March 2015 and April 2016 with over a 50% reduction. The values in April still exceeded DWAF (1996b) levels, whereas in June they were either slightly less than, or at, acceptable levels (Table 8.9) and after a rainfall event of 49 mm over 3 days in July the levels were below the limits.

Table 8.9 Range (min/max), average and standard deviation (SD) of each physicochemical data for five sites sampled at Pond 6 in 2016.

	April				June				July			
Parameter	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**
Depth (cm)	45	95	**72**	19	43	86	**66**	16	52	96	**79**	17
Secchi Depth (cm)	5	10	**7.8**	1.8	10	12	**10.8**	0.8	12	17	**13.6**	1.9
Temperature (°C)	15.8	17.6	**17.0**	0.7	11.1	11.5	**11.4**	0.2	13.3	14.0	**13.5**	0.3
EC (mS cm^{-1})	5.45	5.59	**5.50**	0.05	4.59	5.05	**4.90**	0.18	3.05	3.73	**3.48**	0.26
Salinity (PSU)	3.51	3.57	**3.55**	0.03	3.02	3.31	**3.13**	0.13	1.91	2.37	**2.21**	0.17
TDS (g l^{-1})	12.64	13.44	**12.84**	0.34	9.17	11.21	**10.03**	0.74	6.10	7.46	**6.95**	0.52
pH	9.14	9.21	**9.18**	0.03	7.98	9.02	**8.64**	0.39	8.24	8.76	**8.50**	0.21
DO (mg l^{-1})	5.97	9.65	**8.00**	1.35	0.13	10.41	**5.70**	4.31	5.74	10.23	**8.04**	1.74

Algal biomass was high for both phytoplankton (all of 2016) and MPB (sampled in June and July only) levels (Table 8.10). These levels remain at what is considered to be, nuisance levels and aesthetically unacceptable (DWAF 1996c). Biomass was much greater in April 2016 as compared to the March 2015 and the other 2016 sampling dates (Tables 8.3 and 8.10). Variability within a site was high for MPB samples, showing a level of patchiness in algal development and habitat exploitation. Despite the lack of clarity of the water, with Secchi depth never greater than 17 cm throughout the 2016 sampling period, algal biomass was high. Although biomass itself was more than likely the cause of the "murky" water.

Table 8.10 Algal biomass at each of the Pond 6 sites in autumn (April) and winter (June and July) of 2016, see Figure 6.2 for site locations. Only one phytoplankton sample per site was collected in April and no microphytobenthic (MPB) samples were collected.

	Phytoplankton Chl *a* (µg L^{-1})					MPB Chl *a* (mg m^{-2})			
	April	June		July		June		July	
Site	Mean	Mean	SD	Mean	SD	Mean	SD	Mean	SD
A/1	769.8	139.0	12.1	140.3	9.5	265.8	20.9	132.9	29.8
B	684.3	167.9	16.6	147.0	3.8	378.7	43.3	103.4	62.6
C/2	764.2	179.6	9.1	137.7	28.4	393.4	40.3	335.4	196.9
D	1101.3	160.4	3.0	98.9	3.8	388.2	378.9	88.6	23.9
E/3	558.7	81.3	9.1	101.6	7.6	372.3	177.5	169.8	46.2

As has been demonstrated by this brief sampling period, Pond 6 has issues in terms of water quality and should not be used for human consumption, cattle/livestock watering or recreation (submersion), regardless of the season. It is not evident that it is ever used for the local population in terms of drinking or recreation, however it is heavily relied upon for livestock. Cattle and goats frequently are taken there to both drink and graze. This leads to unhealthy animals and a real possibility for some heavy metal loading into the meat of the animals through bioaccumulation by grazing on the plants in the area. This would have to be determined through further research. There was no evidence of fishing around the wetland, but there are fish present. These fish, if caught and eaten, could also be contaminated, and pose a public health risk. This wetland, in its current state, is a risk to the surrounding population in terms of water quality, potential for harmful cyanobacterial blooms (which have the potential to give off airborne toxins that can affect breathing, especially to those with upper respiratory problems). The amount of solid waste dumped in and around the wetland could also pose a risk to the community. Despite this, however, there is a great potential for this wetland to provide recreational value for bird watchers given the high diversity of water birds that utilize the wetland. Overall wetland health and ecosystem service provision will be discussed further in Section 8.3, with implications for the wetland with climate change.

Figure 8.2 **Pond 6 and immediate urbanized area from 2004-2016. A) 1 January 2004; B) 26 October 2006; C) 18 December 2011; D) 20 March 2016. Images from Google Earth.**

8.2 Bridgemeade Survey

Unlike Pond 6, which has been urbanized since at least the 1960s and has changed little over the last 15 years (Figure 8.2), Bridgemeade has transformed rapidly since 2004 (Figure 8.3). In 2004, the wetland was near-natural in a peri-urban area. The main disturbance in 2004 is the road leading to the suburban housing development (Figure 8.3a). It is clear from this image that the wetland was a seasonal endorheic depression with no natural channel inflow or outflow. Surrounding vegetation appears to have been fynbos typical of that area, with sedges dominating the wetland. There was a stark change by 2006, when the first of the planned housing developments was built (Figure 8.3b). The development, coupled with a high rainfall year (822.5 mm, SA Weather Service), saw subsequent modifications made to the wetland in order to protect the housing infrastructure (Figure 8.3c). These modifications took the form of an outlet channel running north along the new housing development, parallel to the road, three retention ponds and storm water drainage points into the wetland form the two other housing development areas currently being built (Figure 8.3d). These developments have changed the surrounding vegetation and allowed for the invasive Port Jackson (*Acacia saligna* to infest the area. These stands have been cleared by community and Working for Wetlands initiatives. The wetland vegetation structure has also most likely changed from a sedge dominated system (*Eleocharis* sp. and *Schoenoplectus* spp.) to one dominated by reeds (*Typha capensis*). The overall hydrology has also shifted as well as creating a more permanent to intermittently dry system, as opposed to a seasonal wetland.

The physicochemical properties of the wetland are generally within standard acceptable levels. It is a fresh, slightly alkaline system with normal TDS levels (Table 8.11). Unlike in Pond 6 where TDS was measured in g L^{-1}, these levels are with water quality standards and measured in mg L^{-1}. There is evidence of some litter in around the wetland, but very little as compared to Pond 6. The clarity of the water, as measured with the Secchi disc depth, was not very good on the three sampling dates reported, demonstrating a fair amount of turbidly, especially in site R1, in June and July with values of 7 and 9 cm respectively. The main wetland (sites A, C and E) tended to be clearer on average with the exception of April where all sites had low Secchi disc values.

Table 8.11 Range (min/max), average and standard deviation (SD) of each physicochemical data for five sites sampled at Bridgemeade in 2016.

	April				June				July			
Parameter	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**
Depth (cm)	33	>100	**37.8**	5.6	52	126	**77.4**	33.2	47	52	**49.7**	2.5
Secchi Depth (cm)	6	8	**6.6**	0.9	7	36	**25.6**	11.3	9	50	**28.2**	14.5
Temperature	14.8	14.9	**14.9**	0.04	11.7	13.8	**12.4**	0.9	11	12.3	**11.38**	0.50
EC (mS/cm)	1.79	1.79	**1.79**	0.02	0.52	1.10	**0.92**	0.26	0.67	1.31	**0.98**	0.31
Salinity (PSU)	0.86	0.89	**0.88**	0.01	0.32	0.75	**0.61**	0.19	0.36	0.75	**0.61**	0.18
TDS (mg L^{-1})	3.58	3.59	**3.58**	0.01	0.43	0.97	**0.80**	0.24	1.34	2.61	**1.95**	0.62
pH	9.19	9.36	**9.28**	0.07	7.10	8.17	**7.80**	0.48	7.87	8.15	**8.01**	0.13
DO (mg L^{-1})	10.01	11.12	**10.53**	0.44	0.62	16.65	**12.69**	6.77	5.95	9.41	**8.19**	1.44

Similar to Pond 6, the algal biomass for both phytoplankton and MPBs was high and exceed DWAF water quality standards during all sampling periods (Table 8.12). The retention pond sites, R1 and 2

were consistently higher than the main wetland in terms of phytoplankton biomass, accounting for some of the turbidity issues noted, especially in R2.

Table 8.12 Algal biomass at each of the Bridgemeade sites in autumn (April) and winter (June and July) of 2016, see Figure 6.3 for site locations. Only one phytoplankton sample per site was collected in April and no microphytobenthic (MPB) samples were collected. Sites R1 and R2 data collection began in June.

	Phytoplankton Chl *a* (µg L^{-1})					MPB Chl *a* (mg m^{-2})			
	April	June		July		June		July	
Site	Mean	Mean	SD	Mean	SD	Mean	SD	Mean	SD
R1		661.8	31.8	112.3	3.8	36.9	16.4	147.7	119.3
R2		475.8	22.7	422.3	34.0	347.0	94.0	123.4	25.4
A	220.5	109.1	3.0	86.9	5.7	107.6	8.9	115.0	10.4
C	108.7	77.0	3.0	78.9	5.7	67.5	23.9	89.7	4.5
E	139.0	113.3	12.1	61.5	3.8	248.9	68.6	124.5	8.9

The clearing of alien vegetation has aided greatly in the general improvement of this wetland and it is known to local bird watchers as a good spot to see birds like the African Marsh Harrier (*Circus ranivorus*) an important bird of prey found in wetlands. Unfortunately there are no CWAC records for this area to draw on exact numbers and species of birds present. Overall health and ecosystem service provision is discussed in the following section.

8.3 WET-Health and Ecosystem Services Assessment

Using the WET-Health (Macfarlane *et al.* 2007) and WET-EcoServices (Kotze *et al.* 2009) tools both Pond 6 and Bridgemeade were evaluated. The assessment system involves several steps and preliminary tables before reaching final scores. Examples of these tables can be found in Appendix 7.

Both sites were assess and found to be in a present ecological state (PES) of class F for all three of the indicator parameters: hydrology, geomorphology and vegetation. All parameters are scored separately and each taken into consideration. The hydrological assessment for Pond 6 is summarized in Tables 8.13 and 8.14. The dominant impacts governing the overall score is the surface roughness and infilling at the site and the alteration of water inputs (increase from storm water drainage).

The hydrological assessment for Bridgemeade is summarized in Tables 8.15 and 8.16. Although the impacts were not of the same magnitude as in Pond 6, the main influences of the overall score were surface roughness and, in this case, the excavation of retention ponds as well as discharge canals, both adding and removing storm water from the surrounding area.

Figure 8.3 **Bridgemeade wetland and immediate area from 2004-2016. A) 13 March 2004; B) 8 November 2006; C) 18 December 2011; D) 20 March 2016. Images from Google Earth.**

Table 8.13 Overall magnitude of impacts of on-site activities on water distribution and retention patterns at Pond 6

Activity	Magnitude of impact
1. Canalization & stream modification	0.03
2. Impeding features	0.33
3. Surface roughness	4.2
4. Direct water losses	0.24
5. Infilling excavation	1.6
Total score of magnitude of on-site activities in the HGM unit (sum of the above scores)	6.4

Table 8.14 The present hydrological state of Pond 6 derived from the integration of the Water Inputs and Water distribution and Retention assessments

Water inputs		6	Water distribution & retention					6.4
			Water Inputs					
			None	Small	Moderate	Large	Serious	Critical
			0-0.9	1-1.9	2-3.9	4-5.9	6-7.9	8-10
Water distribution & retention patterns	None	0-0.9						
	Small	1-1.9						
	Moderate	2-3.9						
	Large	4-5.9						
	Serious	6-7.9					9	
	Critical	8-10						
Present hydrological state of the wetland HGM unit:							9	
Present hydrological state category							F	

Table 8.15 Overall magnitude of impacts of on-site activities on water distribution and retention patterns on Bridgemeade

Activity	Magnitude of impact
1. Canalization & stream modification	0.1
2. Impeding features	0.28
3. Surface roughness	3.0
4. Direct water losses	0.8
5. Infilling excavation	1.6
Total score of magnitude of on-site activities in the HGM unit (sum of the above scores)	5.78

The geomorphological assessment for both sites was the same, both with similar constraints and modifications made due to roads and housing development (Table 8.17). Likewise, the vegetation assessment for both sites were the same (Table 8.18). This is due to the level of invasive weeds and trees, not only around each wetland but in both catchments, as this is a catchment level assessment. Due to the large urban extent of the catchments, this was expected.

Table 8.16 The present hydrological state of the wetland derived from the integration of the Water Inputs (3.2.1) and Water distribution and Retention (3.2.3) assessments

Water inputs		6	Water distribution & retention					6.4
			Water Inputs					
			None	Small	Moderate	Large	Serious	Critical
			0-0.9	1-1.9	2-3.9	4-5.9	6-7.9	8-10
Water distribution & retention patterns	None	0-0.9						
	Small	1-1.9						
	Moderate	2-3.9						
	Large	4-5.9						
	Serious	6-7.9				8		
	Critical	8-10						
Present hydrological state of the wetland HGM unit:							9	
Present hydrological state category							F	

Table 8.17 Overall magnitude of impact scores and the present wetland geomorphic state category for both Pond 6 and Bridgemeade sites.

Impact category	Score
The effect of altered water inputs on the geomorphology of the HGM unit	10
The intensity and magnitude of impact of erosional features	0.06
Intensity and magnitude of impact of depositional features	1.63
Overall Present Geomorphic state = Sum of the above three scores	11.69

Impact score	Description	State category
10**	Modification have reached a critical level as geomorphic processes have been modified completely.	F

Table 8.18 Overall magnitude of scores and present wetland state of vegetation category for both Pond 6 and Bridgemeade.

Disturbance class	Description of disturbance class	Extent (%) of catchment	Intensity of impact score	Magnitude of impact score	Present vegetation state category
1	Infrastructure	>75	10	7.5	
2	Bare ground & sports fields	>10	8	0.8	
HGM Magnitude of impact score (Sum of magnitude of impact scores)				8.3	F

Under different climate change scenarios these assessments would not be altered from their current state and would probably only deteriorate further, mainly to do with altered hydrological state and possible increase in alien vegetation and vegetation clearing for development. Both sites, however could improve given some rehabilitation and subsequent management of the systems. Admittedly, Pond 6 would need a great deal of intervention and community by-in and cooperation. Bridgemeade would have been better served if wetland buffers had been put in place before the housing developments were planned and approved.

Even though the overall wetland health of both sites is classified as an F, it doesn't mean that they don't still provide for some important ecosystem services. Not all types of wetlands are effective at providing all types of ecosystem services, and not all have are presented with the opportunity to provide a service. Both effectiveness and opportunity are taken into account in scoring. In part, because of their hydrological alterations, both wetlands provide for flood attenuation with a score of 3. In terms of nutrient and toxicant removal, both do provide this service to some extent and there is a need for this service in order to deal with inputs from the run-off from the developments surrounding each wetland. However, Bridgemeade is slightly more effective (3) in providing this service than Pond 6. Perhaps because of the size of the development surrounding Pond 6 and population density as compared to Bridgemeade at the moment. There is real potential and ability for phytoremediation of nutrients and toxicants (i.e. heavy metals) through uptake by reeds present in abundance at both wetlands, which provides an important service.

Both sites provide for maintenance of biodiversity to a larger extent than would have been expected given the water quality and overall health of the systems. Aquatic flora and fauna completely dependent on the water are more limited to hardy, broadly tolerant species and do not have a high diversity at present. The bird community, however is very diverse and thriving, especially at Pond 6. If managed correctly, and Pond 6 is able to be cleaned up and rehabilitated, this site could provide for some important eco-tourism to bring in bird watchers to the area. It could also provide for recreation or green open spaces for local residents. This recreational provision is much stronger for Bridgemeade, as the site itself is less damaged and views as safer for people to come to.

Under different climate change scenarios the different services will expand and contract, as both opportunities as well as effectiveness in provision changes. For instance in time of drought, flood attenuation will not be provided and these is a large potential for the channels that did provide the service could deteriorate, become overgrown with vegetation and therefore not be as effective. Likewise, as changes in weather patterns become less predictable and more severe, times of deluge could expose that the current geomorphology and hydrological functioning will be unable to function as well as a flood attenuation service. Global climate change, and the local effects on climatic patterns will change the overall functioning and service provision of our wetlands, and in urban settings this will have huge effects on the local population.

8.4 General Conclusions

Urban wetlands are difficult to assess using current tools. In terms of WET-Health, almost all urban systems will have low present ecological state (PES) scores but virtue of their surrounding catchment. It is difficult to see health scores above a D, and most will fall to an F, as did both Pond 6 and Bridgemeade. These health score do not adequately take ecological functioning into account, and both these sites show some level of functioning, as well as ecosystem service provisioning. Both have an abundance of bird life with high diversity of birds, and a potential for recreational and tourism value (Figure 8.4).

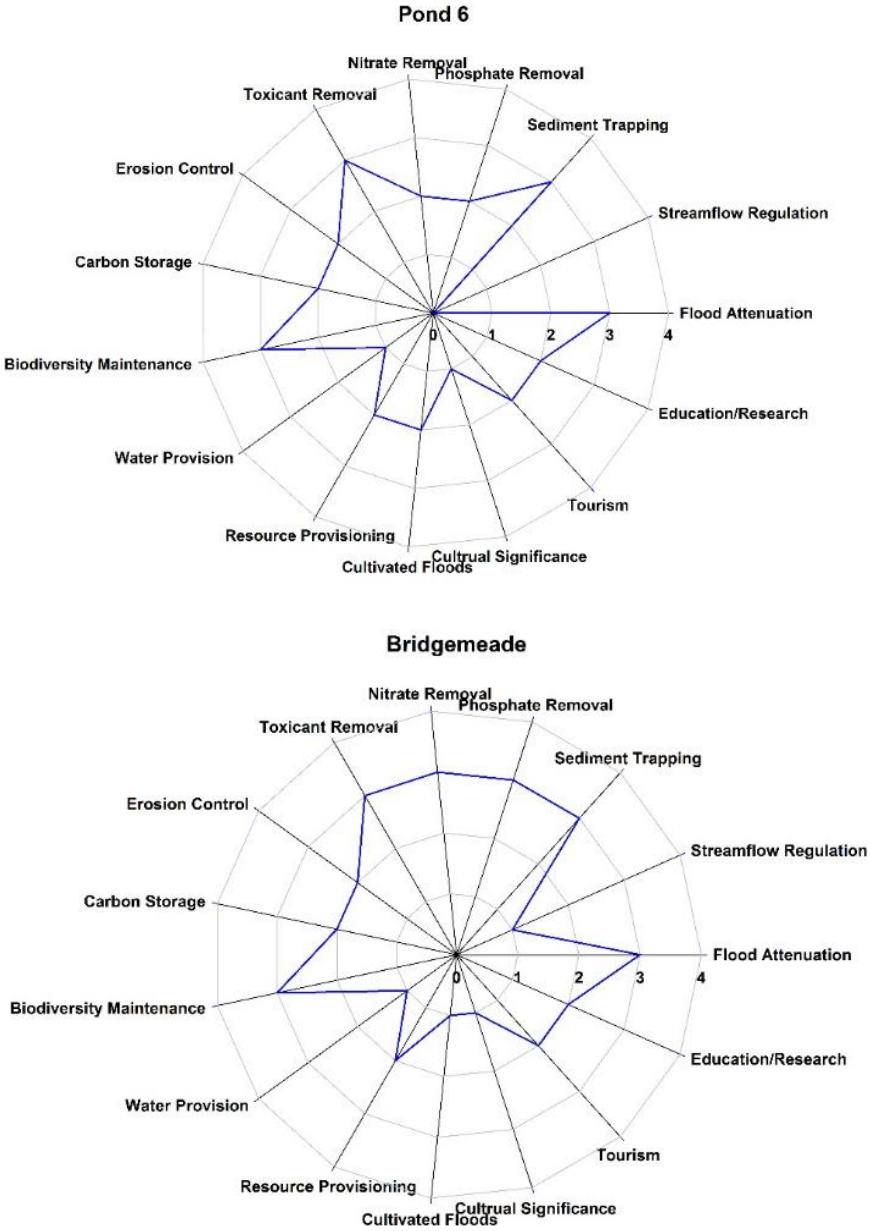

Figure 8.4 Ecosystem services presently provided by Pond 6 (top) and Bridgemeade (bottom) as evaluated by WET-EcoServices (Kotze *et al.* 2009). Scale ranges from 0 (no service) to 4 (highly effective given the opportunity to provide the service).

There is no doubt that Pond 6 has earned its PES score of F, the bacterial, nutrient, algal and solid waste levels are all above DWS standards. Attempts for WfW and the NMBM to clean up the site, in terms of solid waste and waste water inputs, has not seen significant changes made to this system. Liter traps are currently being placed in key run-off areas and hopefully will start to reduce the amount of solid waste entering the wetland. The issues around this wetland are broad and societal. Local and national government, NGO's and the community must come together to address this adequately. Until then, under increasing temperatures and more extreme weather events, this

wetland will only continue to deteriorate and become more of a public health hazard than a service provider, if an effort to truly rehabilitate it is not made.

Bridgemeade, which also scored a PES of F, is not nearly as directly impacted as Pond 6, but the recent housing developments, lack of buffer and structural changes made to the wetland have affected its functioning. Great effort has been put in by community groups and WfW to clear the alien vegetation around the wetland, which has increased its aesthetic and recreational value. The nutrient levels and general water quality, however, is still not very good because of the run-off from the new housing developments. The constructed "retention pond" and drainage canal, built to protect the developments need to be rethought.

Urban wetlands have the potential to provide important ecosystem services, but they need to be managed properly and respected by the surrounding community. Education, as well as community and government partnerships must be fostered in order to aid in the management and protection of these wetlands and potential water resources.

9 CONCLUDING REMARKS AND RECOMMENDATIONS

Climate change in aquatic ecosystems that could permanently alter and or reduce functioning and ecosystem service provision through creating too much ecosystem disturbance that may then exceed its inherent resilience (Schulze *et al.* 2011, Junk *et al.* 2012, IPCC 2014c, a). Under conditions predicted for our area as discussed in both sections, there could be widespread decrease in water quality in remnant pools as seasonal and permanent wetlands dry. This would then lead to decreases in egg/seed banks (Brendonck and De Meester 2003, De Meester *et al.* 2005, De Roeck *et al.* 2007, Waterkeyn *et al.* 2008), decrease in habitat and loss of refugia for migrant species and species that have an aquatic stage in their life-cycle(Brooks and Hayashi 2002, Brooks *et al.* 2002, Brooks 2009), as well as an overall loss of wetlands (Pyke 2004, 2005b, Sánchez-Andrés *et al.* 2010, Davidson 2014, Schook and Cooper 2014). Shifting temperatures could shift species ranges and conceivably allow for more successful invasive species to increase their ranges and outcompete less tolerant native species (Beisner *et al.* 1997, Dukes and Mooney 1999, Ehrenfeld 2008, Rahel and Olden 2008, Dallas and Ketley 2011, Dallas and Rivers-Moore 2014). Shifts in algal taxa, as we have shown can lead to greater eutrophication and nuisance algal blooms (Feuchtmayr *et al.* 2009, Mooij *et al.* 2009, Jeppesen *et al.* 2010, Moss *et al.* 2011, Elliot 2012, Paerl and Paul 2012). It is important to know how regional wetlands respond to current climate in order to predict how they will shift to changes.

In order to full be able to utilize existing models and integrate our shift of algal biomass and community changes we need:

- We need better long term hydrological data for non-fluvial wetland systems;
- Understand interflow and/or subsurface connectivity during inundation periods, especially for systems that lack direct groundwater or fluvial connections;
- Access to quinary hydrological data to add to more data collection;
- Further explore ecophysiological responses of key algal and invertebrate taxa to different thermal regimes.
- Evaluation of habitat and wetland loss in response to climate changes, which systems are most vulnerable to permanent loss.

We also need better focus on the functioning and evaluation of our urban and urbanized wetlands. Wetlands can be resilient to changes in the landscape and climate, but we need to understand where the “tipping points” occur and how effective are different rehabilitation efforts in reversing anthropogenically induced disturbance, what altered stable states can be acceptable for both the ecosystem functioning and human health, and how can we maximize ecosystem service provision while protecting the resource.

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11 APPENDICIES

Appendix 1 Wetlands whose sediments were used for hatching experiments, their wetland identification code (WET ID), field code, geographical zone (area, Figure 1), brief descriptor, coordinates, and hydrogeomorphological unit (HGM). An "X" designates if sediment used in a microcosm experiment

WET ID	Code	Area	Descriptor	X-coord	Y-coord	HGM	2LExp1	2LExp2	FLUX	Tank Exps	Field Sites
1593	CR1	1	NMMU Campus	25.65959	-34.00777	Flat		X			X
1594	CR5	1	NMMU Campus								X
1765	CR92	1	NMMU Campus			Depression					X
1641a	RESA	1	NMMU Campus	25.65568	-34.01411	Seep		X			
1641c	RESB	1	NMMU Campus			Depression			X	X	
1641d	RESD	1	NMMU Campus			Depression					X
1626	SBG1	1	NMMU Campus	25.66291	-34.01336	Flat		X			
1647	DFTN	2	Draaifontein Road	25.32904	-33.94909	Depression	X	X			
329	TC1	2	Theescombe	25.48184	-33.98550	Depression		X			
326	TC2	2	Theescombe	25.48374	-33.98322	Flat		X			
1655	SV1	2	Seaview seep	25.36819	-34.01784	Seep	X				
1654	SV2	2	Seaview seep	25.36622	-34.01732	Seep	X				
790a	PV1a	3	Parsonsvlei	25.48581	-33.90509	Seep	X				
789b	PV3b	3	Parsonsvlei	25.48801	-33.90770	Seep	X				
1699	PV4	3	Parsonsvlei	25.47032	-33.92215	Flat	X				
944	HW1	4	Hopewell	25.40724	-33.87354	Depression		X			
947	HW2	4	Hopewell	25.41190	-33.87525	Depression	X	X			
1019	RH1	5	Redhouse	25.54057	-33.82971	Depression	X	X		X	
1648	RH2	5	Redhouse	25.54439	-33.82872	Depression	X	X			
1017	RH3	5	Redhouse	25.54470	-33.82998	Flat	X	X			
1016	RH4	5	Redhouse	25.54663	-33.83190	Flat	X	X			
1679	VSM1	6	Yellowoods Farm	25.22572	-33.91622	Seep	X	X			
1668	VSM2	6	Yellowoods Farm	25.23575	-33.95090	Flat	X	X			
1651	CDD3	7	Coega Dunes	25.79658	-33.73422	Depression	X	X			
1311	CZ6-1	7	Coega Zone 6	25.39075	-33.73305	Depression	X	X			
1310	EW1	7	Coega Elephant wallow	25.68759	-33.73170	Depression	X	X			
1359	PL1	7	Coega Powerlines	25.66212	-33.71678	Depression	X	X			
1691	BED1	8	Berg 'n Dal Farm	25.42619	-33.67663	Seep	X				
1382	R75-4a	8	Hillslope Seep	25.44693	-33.70510	Seep	X				

Appendix 2 **Site codes, HGM types and zones (see Appendix 1 and Figure 4.1 for details) for sediments used in the 2L experiments, with total number of hatchlings counted over each 28 day experiment. 2LExp1 = 17°C and 2LExp2 = 24°C**

WET ID	CODE	HGM	Zone	Number of Hatchlings	
				2LExp1	2LExp2
1647	DFTN	Depression	2	4	538
329	TC1	Depression	2		0
944	HW1	Depression	4		0
947	HW2	Depression	4	133	53
1019	RH1	Depression	5	211	2328
1648	RH2	Depression	5	48	618
1651	CDD3	Depression	7	2	0
1311	CZ6-1	Depression	7	17	792
1310	EW1	Depression	7	26	2601
1359	PL1	Depression	7	347	979
1593	CR1	Flat	1		337
1626	SBG1	Flat	1		261
326	TC2	Flat	2		0
1699	PV4	Flat	3	0	
1017	RH3	Flat	5	75	1224
1016	RH4	Flat	5	82	1066
1668	VSM2	Flat	6	0	192
1641a	RESA	Seep	1		4
1655	SV1	Seep	2	0	
1654	SV2	Seep	2	0	
790a	PV1a	Seep	3	0	
789b	PV3b	Seep	3	0	
1679	VSM1	Seep	6	14	1285
1691	BED1	Seep	8	0	
1382	R75-4a	Seep	8	0	
			Grand Total	**959**	**12278**

Appendix 3 List of algal taxa identified in experiments TExp1 and TExp2 for microphytobenthos (MPB) and phytoplankton samples, listed by sediment and temperature treatment. Total cell number (x 10^3) over the experimental period are given. As TExp1 was based outside and was subject to natural environmental conditions it is listed as "ambient", for comparison to the temperature controlled experiments.

		1641b						RH1			
		MPB			Phytoplankton			MPB		Phytoplankton	
Division/Class	**Taxon**	***Ambient***	***26°C***	***34°C***	***Ambient***	***26°C***	***34°C***	***26°C***	***34°C***	***26°C***	***34°C***
Bacillariophyta	*Amphora* sp.	27.8			37.8						
	Craticula ambigua	177.8	16.7	15.6			6.7				
	Craticula sp.		85.6	47.8	28.9	2.2	3.3		3.3	17.8	
	Cyclotella sp.	5.6									
	Cymbella sp.	111.1	4.4	15.6	23.3			3.3	1.0	8.9	
	Diadesmis sp.	5.6									
	Navicula sp.	38.9	7.8					7.8			
	Pinnularia sp.	5.6									
	Rhopalodia sp.	15.6	44.4	21.1	1.1						
	Unknown diatoms								3.3		
Chlorophyta	*Characium angustum*	11.1									
	Chlamydomonas sp.				4.4						
	Chlamydomonus globosa				5.6	14.4					
	Chlorella sp.				53.3						
	Chlorogonium elongatum	116.7			8.9	11.1					23.3
	Chlorogonium sp.		1.1								
	Chlosteriopsis sp.									2.2	
	Closterium sp.				1.1					11.1	
	Coelastrum sp.	16.7			1.1						
	Diacanthos belenophorus			4.4							
	Franceia sp.				6.7						
	Monoraphidium sp.	1.0			1.0						
	Oedogonium sp.	11.1									

		1641b						RH1			
		MPB			Phytoplankton			MPB		Phytoplankton	
Division/Class	**Taxon**	***Ambient***	***26°C***	***34°C***	***Ambient***	***26°C***	***34°C***	***26°C***	***34°C***	***26°C***	***34°C***
Chlorophyta	*Oocystis* sp.	155.6	24.4	3.3			5.6		3.3	5.6	8.9
	Pediastrum boryanum var. cornutum	66.7									
	Pediastrum sp.	5.6									
	Scenedesmus acutus				5.6						
	Scenedesmus arcuatus	44.4									
	Scenedesmus bernardii	261.1	11.1			27.8	2.2	8.9	6.7	5.6	1.1
	Scenedesmus bicaudatus	44.4									
	Scenedesmus caudatoaculeolatus			12.2							
	Scenedesmus dimorphus	727.8	25.6		74.4	77.8	36.7	1.0	1.0	26.7	
	Scenedesmus dispar		1.1								
	Scenedesmus ellipticus	27.8									
	Scenedesmus intermedius			56.7							
	Scenedesmus microspina	11.1									
	Scenedesmus opoliensis var. mononensis	1.0									
	Scenedesmus sp.	377.8	14.4	32.2	27.8	56.7	31.1	111.1	38.9	28.9	18.9
	Sphaerolopsis sp.			1.1	7.8						
	Spyrogira sp.		8.9		28.9	4.4		4.4	12.2		2.2
	Unknown chlorophytes	11.1					2.2			18.9	25.6
Chrysophyta	*Ochromonas* sp.				2.2						
	Unknown chrysophyte				2.2						
Cyanophyta	*Anabeana* sp.	83.3	28.9	5.6	21.1	21.1	3.0		3.3	22.2	9.0
	Anabeana viguieri	27.8			4.4						
	Cylindrospermum stagnale	16.7									
	Oscillatoria sp.			1.1							
Euglenophyta	*Strombomonas* sp.		4.4								
	Trachelomonas sp.	44.4			2.2						
	Unknown euglenophyte	166.7			7.8	18.9				3.3	

Appendix 4 **Invertebrate taxa identified in experiments TExp1 and TExp2 samples, listed by sediment and temperature treatment. Total number of animals per L over the experimental period, including day 32 when experiments terminated. As TExp1 was based outside and was subject to natural environmental conditions it is listed as "ambient", for comparison to the temperature controlled experiments. * denotes species that appeared post 28 days.**

			1641b			RH1	
Order	**Family**	**Taxon**	***Ambient***	***26°C***	***34°C***	***26°C***	***34°C***
Calanoida	Diaptomidae	*Lovenula falcifera*				2.0	
		Metadiaptomus capensis				7.5	
Cladocera		Cladocera sp.			10.0		22.0
	Chydoridae	*Alonella excusa**		464.0			
	Daphniidae	*Daphnia dolichocephala*		1.5		49.5	
	Moinidae	*Moina belli*			40.0	0.5	44.0
		Moina sp.	401.0	259.0	169.5	2489.0	383.0
Conchostraca	Leptestheriidae	*Lepthestheriella* sp.				4.5	
Notostraca	Triopsidae	*Triops granarius*				0.5	
Ostracoda	Cyprididae	Cyprididae sp.	126.5	7633.5	19.0	2260.5	10.5
Rotifera	Asplanchnidae	Asplanchnidae sp.		8.5	2.0	99.5	6.0
	Philodinidae	Philodinidae sp.		8.5	25.5	2.0	3.0
		Rotifera sp.			101.0		4.5

Appendix 5 Microphytobethos (MPB) and Phytoplankton species list for NMMU Campus sites between July and September of 2015, total number of cells (x 10^3) over the entire sampling period for inundation events 1 and 2.

		MPB		Phytoplankton	
Division/Class	**Taxon**	**1**	**2**	**1**	**2**
Bacillariophyta	*Achnanthes sp.*		5.0		
	Amphora sp.	116.7	3.0	5.6	12.2
	Craticula ambigua	55.6	166.7		27.8
	Craticula cuspidata	83.3	483.3		1.1
	Craticula sp.		194.4		2.2
	Cymbella sp.		2.0		
	Fragilaria sp.	55.6			
	Gomphonema sp.	111.1	27.8	12.2	2.2
	Hantzschia sp.	61.1			
	Navicula sp.	155.6	927.8	2.2	28.9
	Navicula sp. 1		166.7		
	Nitzschia borealis				12.2
	Nitzschia recta		638.9		8.9
	Nitzschia umbonata	35.6	933.3	1.1	3.0
	Nitzschia sp.	333.3			
	Pinnularia borealis		27.8		
	Pinnularia viridiformis		88.9		12.2
	Pinnularia sp.		38.9		
	Rhopalodia sp.		122.2		
	Tryblionella sp.		38.9		
	Unknown diatoms				38.9
Chlorophyta	*Chlamydomonas sp.*	5.6	11.1	124.4	253.3
	Coccomonas sp.			11.1	
	Cosmarium sp.				27.8
	Eudorina sp.				153.3
	Gloeomonas sp.				22.2
	Lombomonas sp.				33.3
	Oocystis sp.		5.6	25.6	3.0
	Pediastrum duplex				2.2
	Pediastrum duplex var typicum		38.9	18.9	12.2
	Pediastrum simplex				4.0
	Pediastrum sp.				26.7
	Pedinomonas sp.		11.1		23.3
	Pedinopera sp.			16.7	
	Polytoma sp.				3.0
	Scenedesmus acuminatus				5.6
	Scenedesmus acutus				7.8
	Scenedesmus bernadii	16.7	188.9		16.0
	Scenedesmus dimorphus		116.7		81.1
	Scenedesmus maximus		22.2		3.0
	Scenedesmus quadricauda	5.6	44.4	2.2	167.8
	Scenedesmus sp.		33.3		17.8
	Sphaerellopsis sp.		22.2	6.7	12.2
	Spyrogira sp.				8.9
	Unknown chlorophyte	72.2	77.8	4.4	15.6
Cyanophyta	*Anabeana oblonga*				28.9
	Anabeana sp.	244.4	55.6	13.3	
	Cylindrospermum sp.	377.8	527.8	96.7	168.9
	Homoeothrix sp.		16.7		
	Komvophoron sp.	27.8			

		MPB		Phytoplankton	
Division/Class	**Taxon**	**1**	**2**	**1**	**2**
Cyanophyta	*Planktothrinx sp.*	72.2			
Euglenophyta	*Euglena clavata*	38.9			
	Euglena sp.				2.2
	Phacus orbicularis				165.6
	Phacus sp.		27.8		
	Trachelomonas sp.				2.2
	Unknown euglenophyte				47.8

Appendix 6 Macroinvertebrate and frog (tadpole) species list for NMMU Campus sites between July and September of 2015. Taxon arranged by functional group, obligate ephemeral wetland branchiopods from egg banks, followed by zooplankton from egg banks primarily, migrant taxa such as insects and tadpoles, semi-aquatic invertebrates and then terrestrial invertebrates that live in marginal vegetation.

Functional Group	Order	Family	Taxon	Inundation Event 1	Inundation Event 2
Branchiopoda	Anostraca	Branchipodidae	*Branchiopodopsis hodgsonii*	902	3238
	Conchostraca	Leptestheriidae	*Leptestheria rubidgei*	17	800
	Notostraca	Triopsidae	*Triops granarius*	1850	900
Zooplankton	Cladocera	Daphniidae	*Daphnia barbata*		56
			Daphnia longispina	1	57
			Ceriodaphnia reticulata		35
		Moinidae	*Moina micrura*	1	116
	Copepoda	Cyclopoida	*Ectocyclops phaleratus*	41	409
			Cyclopoid copepodites		47
		Unspecified	copepodites	35	44
	Ostracoda	Cyprididae	Cyprididae spp.	435	2648
Insects	Coleoptera	Dytiscidae: Agabinae: Agabini	*Agabetes* sp.		1
		Dytiscidae: Colymbetinae: Colymbetini	*Rhantus* sp.	1	4
		Dytiscidae: Dytiscinae: Hydaticini	*Hydaticus* sp.		2
		Dytiscidae: Hydroporinae: Bidessini	*Hydroglyphus* sp.		2
		Dytiscidae: Hydroporinae: Hydrovatini	*Hydrovatus* sp.		1
		Dytiscidae: Hydroporinae: Hyphydrini	*Hydropeplus* sp.		5
			Hyphydrini sp.	1	58
		Helophoridae	*Helophorus sp*	5	
		Hydrophilidae: Hydrophilinae: Hydrophilini	Hydrophilini sp.		1
	Diptera	Chironomidae: Chironominae	Chironominae sp.		1
		Chironomidae: Orthocladinae	*Cricotopus* sp.		108
			Pseudosmittia sp.	249	142
			Bryophenocladius sp.	39	14
		Chironomidae: Tanypodinae	Tanypodinae sp.		46
		Culicidae	Culicidae spp.	13	
		Culicidae: Culicinae	*Aedes* sp.	3354	3953
			Culex sp.		215
	Ephemeroptera	Baetidae	*Cloeon* sp.		22
	Hemiptera	Corixidae	*Sigara* sp.	3	

Functional Group	Order	Family	Taxon	Inundation Event 1	Inundation Event 2
	Hemiptera	Notonectidae	*Anisops* sp.	2	20
Tadpole	Anura	Bufonidae	*Amietophrynus rangeri*	38	118
		Phrynobatrachidae	*Phrynobatrachus natalensis*		1
		Pipidae	*Xenopus leavis*		265
		Pyxicephalidae	*C. boettgeri/S. grayii*		499
			Cacosternum boettgeri	391	842
			Cacosternum nanum		20
			Strongylopus grayii	8	156
		Unspecified	Anura		62
	Colembolla:				
Non-insects	Entomobryomorpha	Sminthuridae	Sminthuridae	4	2
		Isotomidae	Isotomidae	1	0
	Gastropoda	Unspecified	Gastropoda	1	8
Terrestrial	Araneae	Unspecified	Fall-in spiders		5
	Gastropoda		Fall-in snails	3	0
	Isopoda		Fall-in isopods	5	4
	Coleoptera		Fall-in beetles		33
	Hemiptera		Fall-in bugs		23
	Lepidoptera		Fall-in caterpillars	8	

Appendix 7 **Birds (species and common names given) recorded between 1998 and 2009, average number (±SD) seen over a 12 year period of summer and winter water bird counts in the Pond 6 region (PE Power Station Pans), data from the ADU CWAC database http://cwac.adu.org.za/sites.php?sitecode=33522536, last accessed 12 July 2016.**

Species	Common Name	Average	SD
Actophilornis africanus	African Jacana	1.6	1.9
Alopochen aegyptiacus	Egyptian Goose	14.2	14.5
Anas capensis	Cape Teal	92.0	62.2
Anas erythrorhyncha	Red-billed Teal	14.3	9.2
Anas hottentota	Hottentot Teal	0.2	0.6
Anas hybrid	Hybrid Mallard Duck	0.2	0.4
Anas platyrhynchos	Mallard Duck	0.2	0.4
Anas smithii	Cape Shoveler	105.1	67.2
Anas undulata	Yellow-billed Duck	60.8	40.7
Dendrocygna bicolor	Fulvous Duck	5.7	9.5
Dendrocygna viduata	White-faced Duck	2.2	3.2
Oxyura maccoa	Maccoa Duck	6.2	8.2
Fulica cristata	Red-knobbed Coot	289.4	213.2
Gallinula chloropus	Common Moorhen	5.3	4.7
Porphyrio madagascariensis	African Purple Swamphen	0.2	0.4
Tadorna cana	South African Shelduck	51.0	49.5
Netta erythrophthalma	Southern Pochard	2.1	3.5
Anhinga rufa	African Darter	0.2	0.4
Ardea cinerea	Grey Heron	6.7	3.7
Ardea goliath	Goliath Heron	0.3	0.6
Ardea melanocephala	Black-headed Heron	0.2	0.6
Ardeola ralloides	Squacco Heron	0.3	0.6
Nycticorax	Black-crowned Night-Heron	0.3	0.6
Bubulcus ibis	Cattle Egret	7.5	18.7
Egretta alba	Great Egret	0.2	0.4
Egretta garzetta	Little Egret	8.5	8.9
Calidris minuta	Little Stint	47.6	68.2
Alcedo cristata	Malachite Kingfisher	0.1	0.3
Ceryle rudis	Pied Kingfisher	0.3	0.5
Charadrius hiaticula	Common Ringed Plover	1.2	1.7
Charadrius pecuarius	Kittlitz's Plover	7.2	6.4
Charadrius tricollaris	Three-banded Plover	9.6	5.0
Vanellus armatus	Blacksmith Lapwing	45.0	24.5
Chlidonias hybrida	Whiskered Tern	0.1	0.3
Chlidonias leucopterus	White-winged Tern	56.6	142.2
Sterna caspia	Caspian Tern	0.7	0.9
Sterna hirundo	Common Tern	0.1	0.3
Himantopus	Black-winged Stilt	106.6	65.4
Larus cirrocephalus	Grey-headed Gull	194.0	128.7
Larus dominicanus	Kelp Gull	4.6	4.6
Larus hartlaubii	Hartlaub's Gull	0.3	1.2

Species	Common Name	Average	SD
Motacilla capensis	Cape Wagtail	13.3	8.1
Phalacrocorax africanus	Reed Cormorant	9.1	14.8
Phalacrocorax carbo	White-breasted Cormorant	6.8	5.7
Philomachus pugnax	Ruff	91.0	73.1
Phoenicopterus ruber	Greater Flamingo	132.3	80.5
Platalea alba	African spoonbill	6.3	4.9
Plegadis falcinellus	Glossy Ibis	0.1	0.3
Threskiornis aethiopicus	African Sacred Ibis	20.1	15.1
Recurvirostra avosetta	Pied Avocet	21.3	29.5
Podiceps nigricollis	Black-necked Grebe	8.6	17.7
Tachybaptus ruficollis	Little Grebe	97.9	69.6
Calidris ferruginea	Curlew Sandpiper	18.7	22.3
Tringa glareola	Wood sandpiper	1.2	1.5
Tringa nebularia	Common Greenshank	2.3	2.2
Tringa stagnatilis	Marsh Sandpiper	13.4	17.1

Appendix 8 **The following tables represent the input tables for determining the health status of both Pond 6 and Bridgemeade wetlands using the WET-Health tool (Macfarlane *et al.* 2007). Final scores in the main text.**

Table A8.1 **Different types of land use type activities within the Pond 6 catchment potentially altering the quantity of water entering the HGM unit and the overall magnitude of their effects (Adapted from: Macfarlane *et al.* 2007)**

Reduced flows:			
Land-use activity descriptors	**Extent (%)**	**Intensity of water loss**	**Magnitude**
Irrigation	<5	0	0
Alien plants	<5	0	0
Plantations	0	0	0
Sugar	0	0	0
Overall magnitude of reduced water inputs to the HGM unit from the immediate catchment:			**0**
Increased flows:			
Description of level of increase:			**Magnitude**
Additional flows are more than equal to the natural situation			7
Overall magnitude of increased and decreased flows to the HGM unit:			
Overall magnitude: Increased flows + Decreased flows			**7**

Table A8.2 **Factors that potentially alter the flood-peak magnitude and frequency received by the Pond 6 HGM**

Land-use activity descriptor:	**Score**
Level of reduction:	
Level of reduction in relation to dams within the wetlands immediate catchment	NA
Level of increase:	
Extent of hardened surfaces within the immediate catchment	8
Extent of areas with bare soil within the immediate catchment	2
Overall magnitude score:	
Combined score: level of reduction + level of increase	**10**

Table A8.3 **Combined magnitude of the impact of altered quantity and pattern of inputs into Pond 6, taking into account the wetland unit's vulnerability**

Descriptor:	**Magnitude score**
Reduction in water quantity inputs (Table 3.1)	7
Alteration to flood-peaks (Table 3.2)	10
Magnitude of impact based on the HGM type, altered quantity of water inputs and the altered pattern of water inputs:	**6 (Serious – 51% to 79% of hydrological integrity has been lost)**

Appendix 9 Capacity building and knowledge dissemination

Student Theses

Larsen, M.R. estimated graduation 2018. An integrated conceptual health model and predicted climate change trajectories for two urban wetlands in Port Elizabeth, South Africa.

Lategan, J. 2016. The dynamics fo microalgal communities in response to environmental variables and nutrient fluxes in ephemeral wetlands in the Nelson Mandela Bay Metropole. Nelson Mandela Metropolitan University, Port Elizabeth. MSc Thesis. 214 p.

Mangwiro, E.R. 2016. Sediment-water flux responses of microalgae following nutrient enrichment. Nelson Mandela Metropolitan University, Port Elizabeth. Honours Thesis. 54 p.

Mazwane, S.L. 2016. The role and growth response of germinated microalgae to environmental conditions. Nelson Mandela Metropolitan University, Port Elizabeth. Honours Thesis. 48 p.

Mazwane, S.L. estimated graduation 2018. Assessing the response patterns of microalgae to varying environmental conditions in ephemeral wetland sediments. Nelson Mandela Metropolitan University, Port Elizabeth. MSc Thesis.

Tuswa, A. 2017. Assessing heavy metal content in urban wetland macrophytes and sediments in NMBM. Nelson Mandela Metropolitan University, Port Elizabeth. Honours Thesis. 49 p.

Weitz, R. 2016. The ecology and response patterns of micro-invertebrates and microalgae to environmental conditions following inundation of ephemeral wetland sediments. Nelson Mandela Metropolitan University, Port Elizabeth. Honours Thesis. 52 p.

Weitz, R. estimated graduation 2018. Food-web structure and lower trophic level dynamics in small, ephemeral wetlands: Algal and invertebrate emergence from dry wetland sediments under ambient and experimental conditions. Nelson Mandela Metropolitan University, Port Elizabeth. MSc Thesis.

Conference/Symposium Presentations

Gama, P.T. and D.M. Schael. 2017. Vulnerability of coastal wetlands in Nelson Mandela Bay Metro. Coastal and Marine Research symposium, Nelson Mandela Metropolitan University, 20 April. (Invited presentation)

Schael, D.M., B.L. Melly and P.T. Gama. 2017. Conservation and Management of Wetlands with Growth and Development: The Case of the Nelson Mandela Bay Metropolitan Area. Local Climate Solutions for Africa 2017: Water and Climate, 22-24 March 2017. (Invited presentation).

Melly, B.L., D.M. Schael and P.T. Gama. 2016. Can we identify vulnerable wetland systems without site-specific data? National Wetlands Indaba, 25-28 October, Swadini ForeverResort, South Africa.

Schael, D.M. and P.T. Gama. 2015. Biomonitoring and rapid health assessments for ephemeral wetlands in South Africa: Can invertebrates play an integral role? National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Gama, P.T., D.M. Schael and A. Tuswa. 2015 Phantom wetlands: ephemeral wetlands of the NMMU nature reserve. National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Mazwane, S.L., R. Weitz, D.M. Schael and P.T. Gama. 2015. The growth response patterns of ephemeral wetland microalgae to environmental conditions. National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Weitz, R., D.M. Schael and P.T. Gama. 2015. The ecology and response patterns of microinvertebrates and microalgae to environmental conditions following inundation of ephemeral wetland sediments. National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Lategan, J., P.T. Gama and D.M. Schael. 2015. What the flux? The *ins* and *outs* of ephemeral wetland development. National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Schael, D.M. and P.T. Gama. 2015. The influence of wetland type and inundation duration on aquatic invertebrate species composition in small, ephemeral wetlands. Southern African Society of Aquatic Scientists Conference, 28 June-3 July, Drakensburg, South Africa.

Conference Posters

Larsen, M.R., P.T. Gama and D.M. Schael. 2016. Can a degraded urban wetland still provide ecosystem services? National Wetlands Indaba, 25-28 October, Swadini Forever Resort, South Africa.

Mazwane, S.L., P.T. Gama and D.M. Schael. 2016. Comparing biomass and community composition of ephemeral microalgae in the field versus microcosm studies. National Wetlands Indaba, 25-28 October, Swadini Forever Resort, South Africa.

Mazwane, S.L., P.T. Gama and D.M. Schael. 2016. Comparing biomass and community composition of ephemeral microalgae, how climate change may affect primary producers: field versus microcosm studies. 3rd National Conference on Global Change, 5-8 December, Durban, South Africa.

Tuswa, A., P.T. Gama and D.M. Schael. 2016. Pre-rehabilitation bacterial and heavy metal assessment of an urban wetland: The story of Pond 6, Port Elizabeth. NRF Internship programme presentations, February 2016, CPUT, Cape Town, South Africa

Tuswa, A., P.T. Gama and D.M. Schael. 2015. Pre-rehabilitation assessment of an urban wetland: The story of Pond 6, Port Elizabeth. National Wetlands Indaba, 20-23 October, Rawsonville, South Africa.

Weitz, R., D.M. Schael and P.T. Gama. 2016. The influence of changing hydroperiods on the succession of invertebrate communities of an ephemeral wetland. National Wetlands Indaba, 25-28 October, Swadini Forever Resort, South Africa.

Workshop/Symposium participation and presentations

Wetlands in Drylands: Past, Present & Future Trends in Ecosystem Service Provision, Stonehenge Conference Venue, Parys, Free State, South Africa. 9-12 November 2014. This event is being organised through the Royal Society's South Africa-UK Scientific Seminar Scheme. Funded in the UK by the Department for Business Innovation & Skills (BIS) & in South Africa by the National Research Foundation (NRF). D.M. Schael session co-chair; P.T. Gama group leader.

Wetland Prioritisation Workshop, Nelson Mandela Bay Municipality, Local Action for Biodiversity (LAB): Wetlands and Communities, Pine Lodge, Port Elizabeth: 5-6 May 2015. B.L. Melly presented NMMU data, D.M. Schael, P.T. Gama and B.L. Melly participated in development of prioritisation tool.

WIDS2017. D.M. Schael and P.T. Gama invited to give a key note address for session 'Ecological dynamics of Wetlands in Drylands' meeting of the Wetlands in Drylands (WIDS) Research Network, to be held at Macquarie University, Australia from 24-26 July 2017.