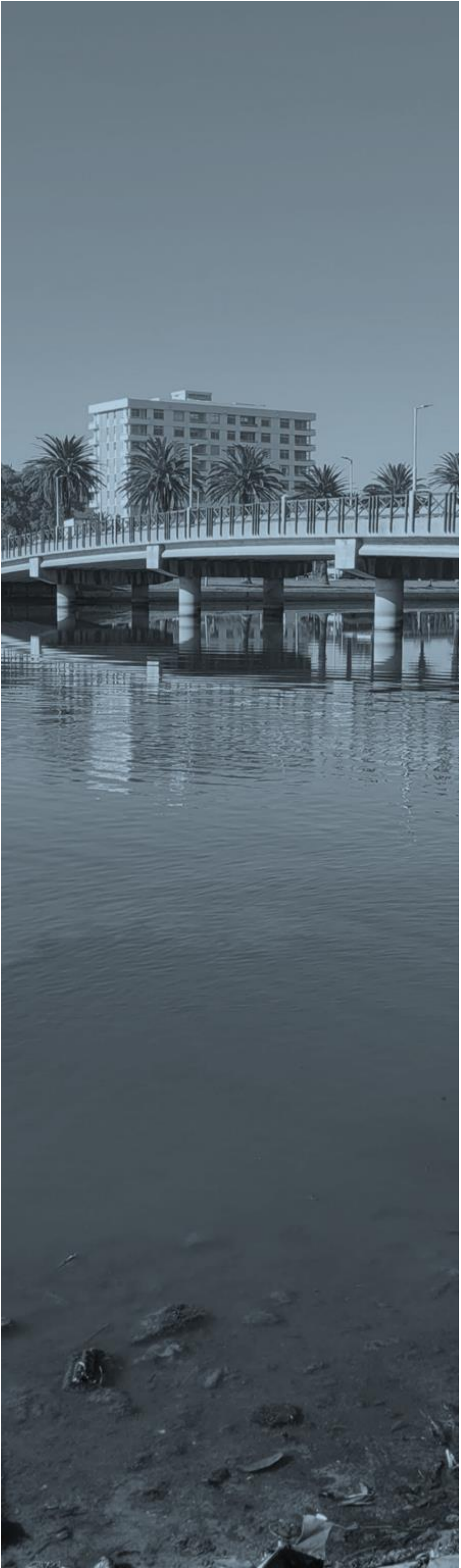

CITY OF CAPE TOWN
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Water Quality Remediation Plan for the Milnerton Lagoon

2023

VERSION: FINAL
DATE: 12 JUNE 2023

PREPARED FOR
City Of Cape Town
Biodiversity Management Branch
Environmental Management Department

PREPARED BY

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DOCUMENT DETAILS

Milnerton Lagoon Water Quality Remediation Plan

CLIENT

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Environmental Management Department
Directorate: Spatial Planning and Environment

PROFESSIONAL SERVICE PROVIDER

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EXECUTIVE SUMMARY

Background and introduction

The Milnerton Lagoon, located at the mouth of the Diep River in Cape Town, South Africa, is part of an urban estuary affected by high levels of pollution and other human impacts. Water quality in the lagoon has declined significantly in recent years, resulting in the closure of the lagoon and adjacent beach areas for recreational use due to the health risks posed to users.

The effects of poor water quality in the estuary have become tangible. The lagoon frequently produces a sulphurous odour and water is discoloured by high levels of suspended solids. Closures of the lagoon to aquatic sports and the adjacent beach to recreational users have impacted on both adjacent residents and on businesses relying on tourist visitors. Ecological impacts have also been observed in the lagoon, with a series of fish kills and a near-complete loss of biota in the lagoon downstream of the wooden bridge.

Recent and ongoing water quality problems in this system have been targeted for intervention in a Mayoral Priority Programme. Addressing the sources of pollution of the estuary is fundamental to the long-term success of any remedial actions and must include measures to re-establish effective treatment of sewage at the Potsdam wastewater treatment works as well as catchment interventions. The Potsdam Improvement Plan developed by the City is being implemented, and the City anticipates that barring any significant mechanical and electrical breakdowns, the treated effluent quality should improve significantly. Remediation of the accumulated contamination and water quality deterioration in Milnerton Lagoon is also prioritised and is the focus of this document.

Infinity Environmental (Pty) Ltd and PRDW South Africa were appointed by the City of Cape Town under a professional services framework contract to prepare a **Water Quality Remediation Plan** for the Milnerton Lagoon section of the Diep River Estuary, including short-term options to address the build-up of nutrient rich and contaminated sediments in the lagoon and the Diep River channel, and the associated water quality and odour issues.

The terms of reference for the plan included a quantitative assessment of the potential effects of proposed interventions, accomplished using hydrodynamic and water quality modelling. Modelling was based on both existing (long-term) and new data gathered for this study, including bathymetric surveys, water quality and sediment analysis. Certain interventions could not be tested by modelling and were assessed qualitatively.

Remediation options assessed in this study included:

- Dredging of the lagoon to remove nutrient-rich sediment;
- Aeration of the lagoon to increase oxygen levels;
- Pumping of seawater into the Milnerton Lagoon to increase salinity, oxygen and flushing;
- Rerouting of Potsdam effluent to a marine outfall;
- Constructed reedbeds or off channel biofiltration for additional treatment of runoff; and
- Bioremediation of sediments through microbial inoculation

Current situation

The Diep River estuary is highly modified and located entirely within South Africa's oldest and most populous city. In this context, the management of water quality in the estuary is subject to several challenges. Water quality in the Milnerton Lagoon is impacted by multiple sources, the most significant of which are described below.

As for any estuary, the quality and quantity of inflows from the catchment is a major determinant of estuarine health and function. Flows in the Diep River are strongly seasonal, and during summer reduce to zero at the top of the estuary due to a combination of low summer rainfall and high abstraction in the catchment for agricultural purposes. For much of the year, therefore, the lagoon is fed only by treated sewage effluent from the City of Cape Town's Potsdam wastewater treatment works (WWTW), which treats sewage from 75 000 households as well as commercial and industrial areas, and by runoff from the urban catchment. The latter includes stormwater runoff with contaminant loads from roads, gardens, and industrial areas, but also raw sewage where failing or blocked sewer pipes or pump stations spill into the stormwater system. The natural hydrodynamic functioning of the system is altered by development in and around the estuary which constrains flows and alters the movement of sediment, reducing the flushing of accumulated sediments, nutrients, and contaminants entering the lagoon.

An analysis of water quality data suggests that the Potsdam WWTW was the most significant source of pollution loading in the lagoon during 2022 (Figure 1), discharging treated effluent that is high in solids and ammonia and has a high oxygen demand, well above compliance levels for these variables. Until 2019, the WWTW largely met its licence limits, but its performance deteriorated from 2019 and worsened significantly in 2022.

These changes could have significantly contributed to the recent observed deterioration in the ecological condition of Milnerton Lagoon and the associated odour problems resulting from the combination of a high and unmet oxygen demand and a high organic pollutant load. The data show an exponential increase in sediment and oxygen demand loading into the lagoon in 2021 and 2022, and in particular after May 2022. This appears to have led to a 'tipping point' in lagoon conditions, and ecologically toxic conditions now predominate in the lagoon with little to no dissolved oxygen, high suspended solids loading, and acutely toxic levels of un-ionised ammonia. The most recent and significant contributing factor appears to be the repeated contamination in 2022 of the Potsdam WWTW maturation ponds and the difficulties associated with procuring equipment to bypass and clean these ponds.

Other significant sources of pollution of the lagoon include sewage-contaminated runoff from informal areas in Du Noon and informally densified areas in Joe Slovo and Phoenix, as well as from periodic spills and overflows throughout the urban catchment when pump station failures or pipe blockages occur. These impacts are not quantified in available data, but contribute to the cumulative impacts on the lagoon.

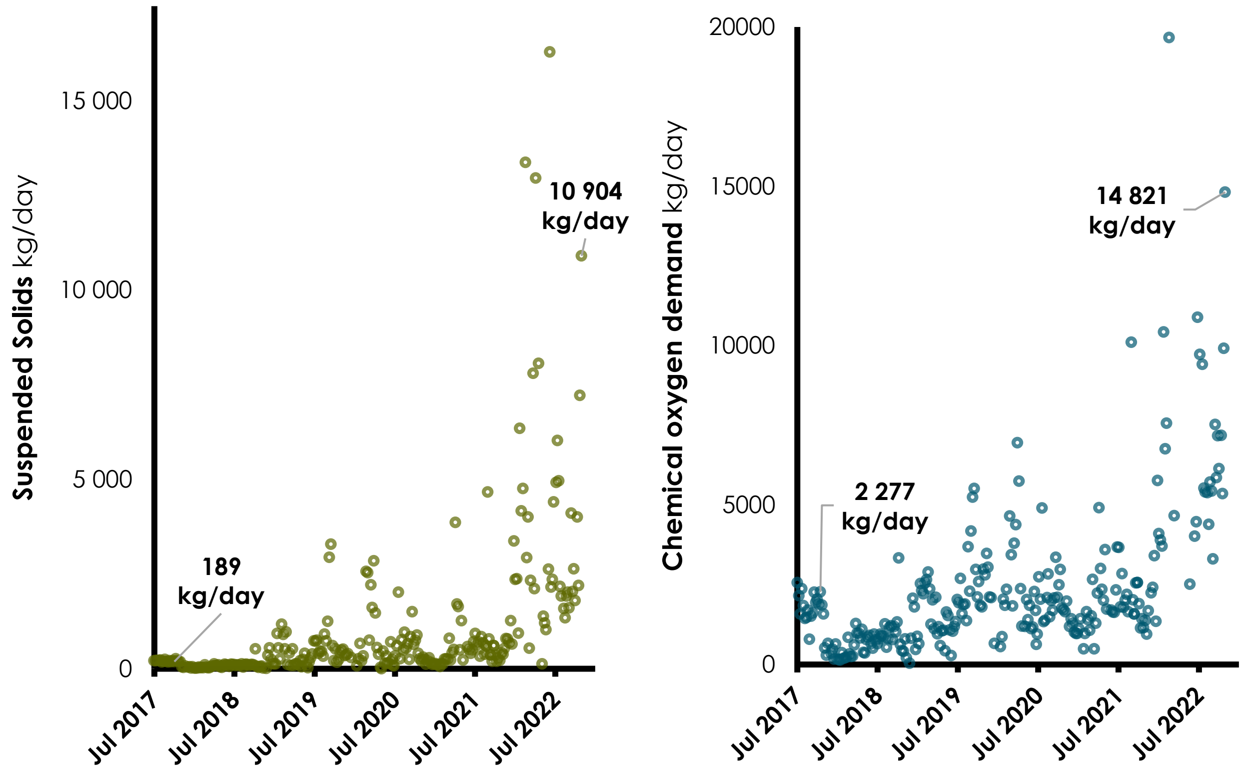


Figure 0-1. Potsdam WWTW final effluent loading for solids (left) and chemical oxygen demand [a measure of organic content] (right) between July 2017 and October 2022, showing a spike in 2019 and an exponential increase in the latter half of 2022

The Milnerton lagoon is characterised by a build-up of nutrient-rich, organic solids which have settled out on the bed of the waterbody. Conditions within these layers and the water above them are anoxic (i.e., without molecular oxygen) leading to a shift in bacterial populations and the production by bacteria of the sulphurous compounds responsible for the characteristic odours. Although the organic sediment in the lagoon is a historic problem that has built up over the past 20+ years, water quality in the lagoon deteriorated significantly during 2022, a period of poor and declining water quality associated with various challenges at the Potsdam WWTW.

Scope and focus of this Remediation Plan

Remediation as described in this report is envisaged as a set of short-term interventions that are aimed at improving management and repairing ecosystem function. Short-term remediation measures in this area are not expected to achieve the desired permanent ecological, human health and/or aesthetic outcomes unless there is a significant change at a catchment scale, reducing the routing of major pollutants into the Diep River and its associated watercourses.

Interventions to address the sources of pollution

Although beyond the scope of this remediation plan, other interventions to improve water quality by addressing the sources of the pollution form part of the 2021 Transversal Action Plan for the Lower Diep River and/or the Water Quality Improvement Plan for the Potsdam WWTW. Some of these interventions had already commenced at the time of finalisation of this report, while others were at various stages of planning.

Repairs, maintenance and reinstatement of critical processes and infrastructure at the Potsdam WWTW to recover levels of treatment to compliance with licensed standards is an urgent and critical measure, irrespective of the ongoing capital works projects to expand and upgrade the works. Specifically, bypassing of the Potsdam WWTW maturation ponds and their urgent remediation and reinstatement for settling and clarification is a critical intervention that could result in an improvement in Milnerton Lagoon, or contribute to conditions in which other remediation measures may achieve a long-term improvement. The City has since early 2022 been implementing a Potsdam Improvement Plan, culminating in the cleaning of the maturation ponds which started in March 2023.

Remediation of the reedbeds to the south of the WWTW to allow them to contain contaminated inflows is a necessary intervention, though not a rapid one and not one expected to on its own improve lagoon quality.

Separation of polluted inflows from clean runoff in the catchment, including construction of low-flow diversions of stormwater to sewer – specifically, the diversion of the Kleine Stink River around Du Noon, and the formalisation of a permanent low-flow diversion of the Erica Road outfall to the sewer east of Otto du Plessis and/or at the Milky Way sewer pump station.

Implementation of the capital upgrades and expansion of the Potsdam WWTW is strongly supported. It is not anticipated that these works will be concluded before the year 2025 and the remediation actions considered in this report are therefore assessed both in terms of the need to reverse the impacts on the lagoon, and their ability to ameliorate water quality in the short term before the Potsdam upgrades occur.

DATA AND MODELLING

This Remediation Plan is based on a combination of long-term, existing data and new data collected for the purposes of the assessment and modelling.

A **hydrodynamic model** of the estuary was constructed using the DHI Group Mike21C two-dimensional (horizontal) model, and used to simulate sediment transport and scour in the lagoon.

A one-dimensional HEC-RAS **water quality model** was developed and used to test the impacts of proposed remediation measures against a baseline of current conditions, as well as the future Potsdam upgrades.

New **data collected** for the assessments included:

- A bathymetric survey and sediment depth quantification in the lagoon and river channel
- Water quality sampling at catchment inflows and in-lagoon quality sampling of a full daily tidal cycle at spring tide
- Sediment granulometric grading and quality classification for disposal purposes

The assessments also draw on existing and long-term data collected by the City and others.

Recommended remediation measures

Water quality modelling was undertaken to assess the performance of proposed interventions in relation to the current water quality of the Milnerton Lagoon. The proposed interventions were modelled, with a number of variations, and the simulated water quality was compared with the baseline situation for parameters including dissolved oxygen, salinity, nutrients, and algae. A relative improvement or deterioration in these parameters is the basis for the assessment of expected effects of the intervention. The expected positive outcomes of the intervention, including ecological benefits in addition to the water quality effects, were also qualitatively considered. Financial costs of interventions are estimated, and indicative timeframes for implementation also determined as the time required to conclude design, approvals and procurement processes is a key driver of whether an intervention is a viable short-term solution to the Milnerton Lagoon's water quality issues.

Dredging of the Milnerton Lagoon

A total of 136 550 cubic metres of organic sediment has built up in the lagoon over the past few decades, with thicknesses ranging from one to two metres. Dredging would remove this material from the system for dewatering and offsite disposal. Dredging of the lagoon in up to three phases, depending on budget availability, was investigated.

Red = organic sediment accumulation

Modelled effects of dredging

DISSOLVED OXYGEN	**Significant increases in dissolved oxygen levels in the lagoon**
NUTRIENTS & ALGAE	**No significant effect on nutrients unless inflow quality is improved**

Costs are estimated at **R 133 million** to dredge and dispose of organic sediments from the entire lagoon. Dredging only the area between Woodbridge and the mouth is estimated at **R 77 million.** If only the area downstream of Woodbridge is dredged and inflows of organic solids continue at 2022 levels, it is expected to take up to ten years for organic sediments to accumulate to today's levels in the lower lagoon. Up to 20 years will be required for sediments to accumulate to current levels if the whole lagoon is dredged and inflows of solids continue at 2022 rates. **Repeated or maintenance dredging will be needed unless the inflows of organic sediment are reduced at the same time that dredging is implemented.** Capital upgrades at the Potsdam WWTW are expected to reduce its contribution to suspended solids loading to near zero in the medium term, while cleaning of the maturation ponds currently underway is also anticipated to improve the situation relative to 2022. Dredging will require a large area on the shoreline for dewatering and handling of the dredged material before disposal – sites investigated for this purpose include the public open space along the R27 near Loxton Road. Environmental authorisations will be required, and at least **24 months** will be required for these processes and the procurement of contractors. **The proposed dredging is predicted to have a substantial ecological benefit for the health of the lagoon and is recommended for implementation.** Dredging of the lagoon will, however, provide only short-lived improvements at high cost if the discharge into the lagoon of effluent with significantly elevated nutrients and solids is not mitigated. Other sources of pollution contributing to the organic sediment build up will also need to be addressed and minimised to avoid the need for repeated dredging.

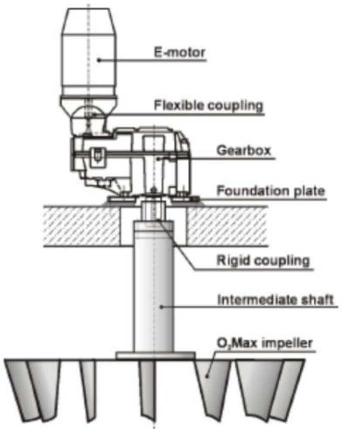
Aeration of the Milnerton Lagoon

Oxygen levels in the lagoon are extremely low and the lagoon water is often anoxic. This a significant contributing factor to the sulphurous odours in the lagoon. Artificial aeration of the lagoon, using one of three technology options to increase oxygen levels, was investigated. This short-term measure could be implemented near the Erica Road outfall or at the Otto du Plessis road bridge where the channel is relatively deeper and narrower. Due to the low contact time between the flowing water and the aerators, oxygen levels may be increased to a maximum of 4 mg/L – low in absolute terms but an improvement on the current situation.

Modelled effects of aeration

DISSOLVED OXYGEN	▲	**Increases in dissolved oxygen levels in the lagoon relative to the status quo, reducing the amount of time the lagoon is in an anoxic state**
NUTRIENTS & ALGAE	◯	**No effect on nutrients**

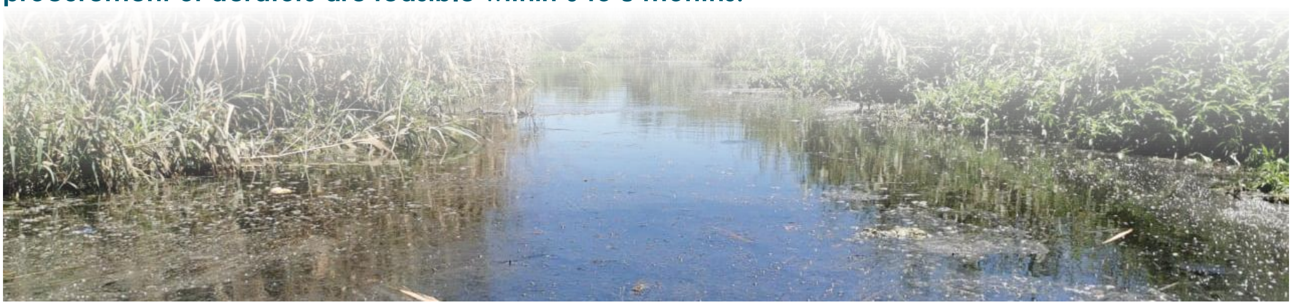
Typical jet (top) and vertical axis (bottom) aerator types



Costs are estimated at between **R 1.5 and R 4.1 million** for the equipment, installation and supporting infrastructure depending on the technology option, with the most cost-effective being the relatively low-power floating aspirator-type aerator. Operational costs vary considerably depending on the electrical supply options, with an anticipated range of between R 45 000 and R 550 000 per month without generator backup.

The proposed aeration must be seen as a pilot project, and there is a risk that only minimal improvement to the situation will result, especially in respect of odour. Initially, the disturbance of sediments and release of stored gases may in fact increase the odour in the lagoon area and increase algal growth. The mechanical equipment may be subject to theft or vandalism, and its operation will be affected by power availability during load-shedding. Other more minor impacts include noise, obstruction to boating, disturbance of habitats and possible resuspension of sediments.

Although oxygen levels in the lagoon will only be significantly and lastingly improved by reducing the oxygen demand in the Potsdam effluent, it is recommended that artificial aeration be trialled as a short-term measure to increase the oxygen concentrations in the water and contribute to reducing odours and other impacts. Authorisations may however be required from the authorities, increasing the timeframes to as much as 8-9 months for implementation. If pursued, aeration of the lagoon may alleviate some of the symptoms of the pollution, but will not have a lasting effect. **It is recommended that aeration be pursued if authorisation processes and the installation of electrical supply and procurement of aerators are feasible within 6 to 8 months.**



Seawater flushing of the lagoon

The Milnerton Lagoon is fed year-round by treated effluent flows from the Potsdam WWTW, which keeps salinity much lower than it would historically have been. The mouth is also very constrained by surrounding infrastructure, and flushing by seawater at high tides is limited. It is proposed that seawater be pumped into the estuary, either from an offshore intake or from a field of beach wells installed at a point upstream of the Woodbridge, to increase salinity levels in the lagoon, improve flushing of sediments, and reduce nutrient concentrations and algal growth.

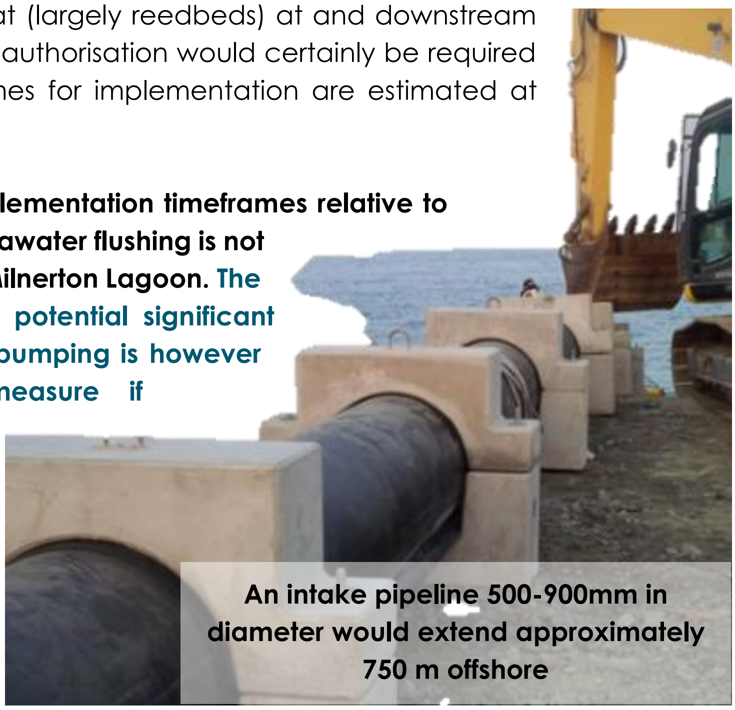
Modelled effects of seawater pumping

Parameter	Effect	Parameter	Effect
DISSOLVED OXYGEN	**No significant effects on oxygen levels in the lagoon**	TEMPERATURE	**Water temperature reduced by cooler seawater**
NUTRIENTS & ALGAE	**Nutrient levels reduced, along with algal concentrations – due largely to the effect of dilution**	SALINITY	**Salinity increased in the lagoon**

Estimated costs for this intervention depend on the rate of seawater pumping required; modelled scenarios considered a rate of 100 to ~500 litres per second and found the greatest effect with the higher rate of pumping. The installation costs of implementation are estimated at **R10 million** for beach wells with a 100 L/s flow rate, and **R 70 million** for an offshore intake with a 500 L/s flow rate. Operational costs are not included in this estimate.

Risks include changes to freshwater habitat (largely reedbeds) at and downstream of the outfall in the lagoon. Environmental authorisation would certainly be required from the national authority, and timeframes for implementation are estimated at between three and four years.

Due to the high capital cost and long implementation timeframes relative to the intended improvements at Potsdam, seawater flushing is not a <u>short-term</u> remediation measure for the Milnerton Lagoon. The combination of low ecological risks and potential significant ecological benefits means that seawater pumping is however recommended as a medium-term measure if financially feasible.



An intake pipeline 500-900mm in diameter would extend approximately 750 m offshore

Other options investigated

Diversion of Potsdam effluent to a new marine outfall

The most significant driver of poor water quality in the lagoon during 2022 was the treated effluent discharged by the Potsdam WWTW. This option proposes the redirection of final effluent (after treatment) to a marine outfall replacing the current outfall into the estuary. The outfall would extend approximately 1.8 km offshore to a diffuser at 18 m depth and would discharge all effluent (currently between 30 and 40 Ml/day, increasing after expansion of the WWTW) to Table Bay.

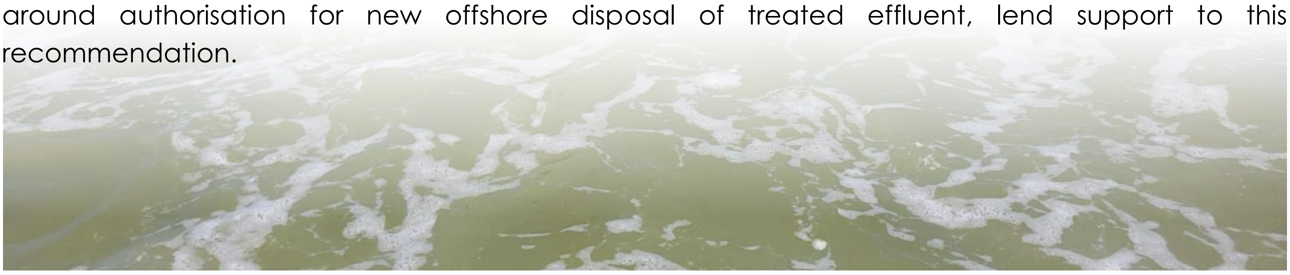
Removing Potsdam inflows would remove a nutrient-dense perennial water source and significantly reduce the loading of pollutants in the lagoon [noting that capital upgrades planned for the works are expected to improve suspended solids discharge significantly]. Models suggest this would result in an improvement in water quality flowing downstream and slow the rate of future organic sludge accumulation in the estuary. However, this option would also seriously alter the hydrodynamic functioning of the system. The Potsdam flows perform an important function in keeping the estuary mouth open in summer, which enables tidal flushing. Removing these flows entirely may cause the mouth to close during the summer unless actively managed through dredging. In the absence of summer flows from the Diep River catchment, this may result in a situation where contaminated stormwater and untreated sewage continues to enter the lagoon and is not flushed out to sea. A further risk posed by summer mouth closures is that they presents a flooding risk to the nearby areas during the winter months, as the water levels will rise if the mouth is not artificially breached.

Modelled effects of a new marine outfall

DISSOLVED OXYGEN	▲▲	**Significant improvement in dissolved oxygen, as oxygen demand is reduced**
NUTRIENTS & ALGAE	▼▼	**Nutrient levels reduced, along with algal concentrations**

The construction of a new marine outfall would have significant risks in respect of regulatory approvals – public concern has been expressed recently in respect of existing marine outfalls (albeit discharging primary effluent treated to a much lesser degree than would be the case at Potsdam). Environmental authorisation would be required from the national authority, in addition to a coastal waters discharge permit. Timeframes for implementation are estimated at between three and four years at minimum, and costs are estimated at **R 190 million**, higher than any other option.

Based on the potential serious alteration of the mouth state during the summer months and the resulting need for intensive mouth management, a marine outfall is not recommended as a remediation measure for the Milnerton Lagoon. The high capital cost and long implementation timeframes relative to the intended improvements at Potsdam, together with the regulatory risks around authorisation for new offshore disposal of treated effluent, lend support to this recommendation.



Treatment Wetlands

The potential for constructed treatment wetlands to improve the quality of runoff before it enters Milnerton Lagoon was investigated. Treatment wetlands do not perform well with very high pollutant loadings, and would not be appropriate for catchments where the bulk of stormwater runoff is grey or black water. In the present situation, treatment wetlands might be used only for polishing of stormwater runoff from catchments with lower pollution levels, such as Killarney and parts of Montague Gardens. Given the high level of existing impacts on natural wetlands within the estuarine area, it is not proposed that existing natural wetlands should be used for treatment.

Areas available for this purpose include reedbeds no. 2 and 3, located between Potsdam WWTW and the Theo Marais Canal. These would first require extensive remediation to remove contaminated sediments and re-establish reedbed vegetation. This is a strongly recommended intervention, as these historic ponds have been contaminated by a history of receipt of raw sewage overflows during surcharges at the WWTW, and are assumed to have played a role in actively contributing to pollution loads into Milnerton Lagoon. It is recommended that the ponds should be redesigned to include capacity for the isolation of WWTW overflows within contained areas of the wetland, rather than allowing such surcharges to spread over into and contaminate the entire wetland. Stormwater flows from the refinery area might also reasonably be treated within this wetland, provided that suitable ongoing management can be implemented. Additional treatment wetlands might also be created in the area between reedbed no. 3 and Koeberg Road, although this area is currently in use for various stockpiling purposes and may be required as a dewatering site for dredging of the lagoon.

Remediating the Potsdam reedbeds is recommended as an essential contamination remediation measure, but it is not short-term, is likely to require environmental and water use authorisation, and would not in itself be likely to bring about any measurable change in the current condition of the Milnerton Lagoon.

Potsdam reedbeds where remediation is recommended

Biofiltration

Retention and filtration of polluted runoff using large-scale filters packed with biochar or similar media has shown promise in small-scale trials implemented downstream of informal settlements elsewhere in the Western Cape. Such filters mimic processes that are found in young wetland systems, but are dependent on in-situ conditions and variability in water quality and retention time to achieve the best performance.

It is proposed that a gravity-fed diversion channel be implemented on an experimental basis to divert a small proportion of the Milky Way stormwater flows through a biofiltration system before its discharge to the lagoon. Costs are estimated at approximately R 400 000. The implementation of a such an experimental system may serve to contribute to the incremental reduction of cumulative impacts arising from urban runoff, while increasing the state of knowledge of such systems through a comprehensive monitoring programme. A pilot project is recommended at the Milky Way diversion channel. Biofiltration systems are not, however, anticipated to have any measurable effect on water quality in the Milnerton Lagoon.

Inoculation

Microbial bioremediation is the use of microorganisms to degrade and remove pollutants. Because of the proprietary nature of, and scarcity of detailed technical information regarding commercial preparations of microbes and enzymes used for bioremediation, this study could not rely on published literature to recommend products for implementation. Prospective suppliers were asked via a formal, public process to submit information to assist in determining which commercially available products may assist with the breakdown of organic matter in sediments into non-toxic substances that can easily be flushed out of the water body with natural flow. Submissions were assessed to determine whether evidence was provided of their efficacy in saline or brackish environments, whether they demonstrated effectiveness in flowing systems with short retention times, and whether the information provided was scientifically and technically robust. Three of the 12 submissions received by the City show some promise on these bases and were subjected to ecotoxicity testing to determine any potential ecological risk. **Two products are recommended for in-situ testing as the basis for a final determination in respect of microbial bioremediation implementation in the lagoon.** An experimental design for in-situ testing of the selected products is set out in this report, for testing of their efficacy under in-situ conditions.

Summary and recommendations

Water quality in the Milnerton Lagoon is a complex problem impacted by multiple pollution sources. Current efforts to address these sources of pollution are the subject of a suite of existing plans including the Lower Diep River Transversal Action Plan (dated April 2021), the Diep River Estuarine Management Plan (dated October 2022), and the Potsdam WWTW Improvement Plan (dated February 2023). **It is strongly recommended that the various plans and responses in preparation by the City be aligned, including through the incorporation of the findings of this Remediation Plan into the respective catchment- and estuary-scale planning.**

Table 0-1 summarises the recommended actions in respect of each option assessed, while Table 0-2 sets out in more detail the results of the assessment and the reasons for the recommendations presented here.

Table 0-1. Summary of recommended options

Intervention	Recommendation	
Improve Potsdam functioning (including remediation of maturation ponds)	✓ ▶▶	Implement with urgency (underway as of April 2023)
Potsdam capital works (expansion and upgrade)	✓ ▶▶	Implement with urgency
Dredging	✓	Proceed with design and approvals, to implement in parallel with WWTW capital works
Aeration	✓ ▶▶	Implement **pilot** as soon as possible
Seawater flushing	✓	Not a short-term measure; proceed with design for medium-term implementation
Marine outfall	✗	**Not recommended** due to high project costs and long timeframes
Biofilters	✓	**Recommended** non-critical intervention – implement a **pilot project**
Treatment wetlands	✗ ✓	Not feasible at scale – but **remediate existing reedbeds** at Potsdam
Inoculation	🔬 ✓	**Recommended for *in situ* trials** before implementation

Next steps

A high-level implementation plan is presented in section 15, and the measures recommended for implementation should be pursued with urgency, including where relevant the undertaking of feasibility and detailed designs, submission of application for authorisations, budgeting, and procurement.

Table 0-2. Summary of options, high-level risks, costs and benefits for assessment

Option / Factor	1: Dredging	2: Aeration	3: Seawater flushing	4: Marine outfall	5: Biofilters	6: Treatment wetlands	7: Inoculation	Improve Potsdam functioning
Intended outcome	**Remove organic sediment to reduce oxygen demand and nutrient availability**	**Increase oxygen concentration in the water column to reduce odour**	**Increase oxygen concentration and salinity in water column, increase flushing of sediments**	**Remove largest source of pollution by bypassing the lagoon to discharge treated effluent at sea.**	**Reduce pollutant loading in selected influent stormwater systems**	**Reduce pollutant loading in selected influent stormwater systems**	**Speed breakdown of organic sediments to reduce oxygen demand**	**Return Potsdam WWTW to functioning at licensed effluent standards**
Expected benefits	Increased oxygen levels; improved scour;	Short-lived and localised effect in dissolved oxygen	Increased salinity; reduction in nutrients, algae; reduction in ammonia toxicity	Significantly reduced nutrient pollution and increased salinity in the lagoon; reduced flow and scour	Unlikely to alter situation significantly	Unlikely to alter situation significantly; remediation of reedbeds will provide for improved WWTW management	*May contribute to reduction in sediment buildup in shallow margins*	Significantly reduced suspended solids and ammonia concentrations
Modelled efficacy	**High** for oxygen levels **Negligible** for nutrients and algae	**Low positive** for oxygen levels	**Low** for oxygen levels **Moderate** for nutrients **High** for algae Reduces temperature	**Low** for oxygen levels **High** for nutrients and algae – greatest effect on eutrophic state	*Not modelled*	*Not modelled*	*Not modelled*	**High** for oxygen levels, nutrients and algae
Estimated costs	R 77 154 000 to R 133 137 000	R 1 524 000 to R 4 145 000 [capital] R 45 000 to R 550 000 per month [running]	R 10 000 000 to R 70 000 000	R 190 000 000	R 400 000 [Pilot]	*Not priced here – subject of separate project by W&S under contract 194C*	*Not priced here*	*Not priced here*
Minimum time to start	24 months	9 to 12 months	20 to 30 months	24 to 30 months	3 months	12 months	3 months	Various interventions already **completed or underway**
Time to achieve effect	36 months	Immediate	36 months	48 months	12 months	24-36 months	*Pending testing*	3 months
Ecological risks	Disturbance of benthic and shoreline habitats	None anticipated	Some loss of freshwater habitat (largely reedbeds) at and downstream of outfall	Impacts on benthic habitat along route of outfall pipeline; marine pollution impacts to be determined	None anticipated	None anticipated	***Unknown***; *to be determined by ecotoxicity testing*	None anticipated
Risk of failure	**Low** (but will require maintenance dredging)	**Moderate**	**Moderate**	**Moderate** (due largely to regulatory risk)	**High**	**High**	**Moderate**	**Low** (but will require ongoing maintenance)
Other negative impacts	Availability of disposal sites not confirmed Noise, disturbance, and increased turbidity during implementation	Noise and obstruction Possible increase in entrainment of sediments	Short-term disturbance of shoreline and benthic habitat along pipeline route and at intake / outfall	Reduced scour and flow into lagoon; summer mouth closures	None anticipated	None anticipated	*Product-dependent: to be determined by testing*	None anticipated
Approvals required	Environmental authorisation Water use authorisation	Environmental authorisation Water use authorisation	Environmental authorisation Coastal permits	Environmental authorisation Coastal Waters Discharge Permit	None (to be confirmed with authorities)	Environmental authorisation Water use authorisation	None (to be confirmed with authorities)	None

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Abbreviations

DEA&DP	Provincial Department of Environmental Affairs and Development Planning
DFFE	Department of Forestry, Fisheries and the Environment
DO	Dissolved Oxygen
DWS	National Department of Water and Sanitation (previously the Department of Water Affairs and Forestry – DWAF)
EFZ	Estuarine Functional Zone
EMP	Estuarine Management Plan
ICMA	National Environmental Management: Integrated Coastal Management Act (24 of 2008)
MASL	Metres Above Sea Level
MLRA	Marine Living Resources Act (18 of 1998)
MMP	Maintenance Management Plan
NEMA	National Environmental Management Act (107 of 1998)
NEMBA	National Environmental Management Biodiversity Act (10 of 2004)
NEMP	National Estuarine Management Protocol
NEMPAA	National Environmental Management Protected Areas Act (57 of 2003)
NWA	National Water Act (36 of 1998)
RQOs	Resource Quality Objectives
TBNR	Table Bay Nature Reserve
WWTW	Wastewater Treatment Works

1 INTRODUCTION

1.1 Background

The Milnerton Lagoon, located at the mouth of the Diep River in Cape Town, South Africa, is part of an urban estuary affected by high levels of pollution and other human impacts. Water quality in the lagoon declined significantly between 2019 and 2022, and deteriorated rapidly after May 2022.

The effects of poor water quality in the estuary are tangible; the lagoon frequently produces a sulphurous odour and water is discoloured by high levels of suspended solids. The closures of the lagoon to aquatic sports, and of the adjacent beach to recreational users have impacted on both adjacent residents and on businesses relying on tourist visitors. Ecological impacts have also been observed in the lagoon, with a series of fish kills and a near-complete loss of biota in the lagoon downstream of the wooden bridge. A high level of community activism has resulted from this situation, with a number of stakeholder groups and ratepayers associations raising concerns regarding the management of the ongoing pollution of the system.

The lagoon is affected by a build-up of nutrient-rich, organic solids which have settled out on the bed of the waterbody in layers of varying thickness. Conditions within these layers and the water above them are anoxic (i.e. without oxygen) leading to the production by bacteria of various sulphur-containing compounds that result in the characteristic odours. The natural hydrodynamic functioning of the estuary is constrained by development around the mouth and by road and rail infrastructure crossing the estuary. These factors constrain flows and alter the movement of sediment, reducing the ability of the system to flush accumulated sediments, nutrients, and contaminants entering the lagoon from its catchment. It must be noted that bathymetric surveys show that sediment levels prior to 2021 are similar to those surveyed in 2023, indicating that organic sediment build up in the lagoon is a historic problem.

In a context of significant historical and ongoing impacts on the estuary from multiple sources, the most significant single contributor to pollution in the estuary in the three-year period leading up to this report was the Potsdam Wastewater Treatment Works (WWTW), which receives sewage flows from the large and rapidly expanding urban areas of Parklands, Table View, Milnerton, Century City, and adjacent suburbs. The works has a nominal treatment capacity of 47 megalitres per day, but average inflows to the works have increased steadily since 2017 and frequently exceed this volume. Until 2019, treated effluent discharged from the WWTW into the Diep River estuary was largely compliant with licensed standards, but a deterioration in final effluent quality occurred between 2019 and 2022. These changes largely explain the recent observed deterioration in the ecological condition of Milnerton Lagoon and the associated odour problems resulting from the combination of a high and unmet oxygen demand and a high organic pollutant load. Prior to 2019 Potsdam WWTW was largely compliant with its license conditions. The organic sediment described above is therefore likely to be a result of a gradual build-up over time from multiple pollution sources over the last few decades. Other significant sources of pollution of the lagoon include sewage-contaminated runoff from informal areas in Du Noon and informally densified areas in Joe Slovo and Phoenix, as well as from periodic spills and overflows throughout the urban catchment when pump station failures or pipe blockages occur.

The data show that there was an exponential increase in sediment and oxygen demand loading into the lagoon in 2021 and 2022, and in particular after May 2022. This appears to have led to a 'tipping point' in lagoon conditions, and ecologically toxic conditions now predominate in the lagoon with little to no dissolved oxygen, high suspended solids loading, and acutely toxic levels of un-ionised ammonia. The most recent and significant contributing factor appears to be the repeated contamination in 2022 of the Potsdam WWTW maturation ponds and the difficulties associated with procuring equipment to bypass and clean these ponds.

A review of the Estuarine Management Plan for the Diep River estuary, completed in 2022 by the City of Cape Town and Infinity Environmental and approved by the provincial MEC in March 2023, identified a number of potential short-term interventions for the lagoon intended to address water quality impacts in parallel with the reduction in pollution inputs. These interventions required further investigation and assessment to determine their feasibility and potential efficacy.

Recent and ongoing water quality problems in this system have been identified for intervention in a Mayoral Priority Programme. Addressing the source of pollution in the catchments is critical and is the subject of the Potsdam Improvement Plan and the Lower Diep Pollution Improvement Plan already in implementation by the City. Remediation of the accumulated contamination and ecological deterioration which have already occurred has also been identified as a priority. In response to this need, the City of Cape Town in late 2022 appointed a professional team to prepare this Water Quality Remediation Plan for the Milnerton Lagoon.[*]

1.2 Terms of reference

1.2.1 Overview

Infinity Environmental (Pty) Ltd and PRDW South Africa were appointed by the City of Cape Town under a professional services framework contract to prepare a **"Remediation Plan for the Milnerton Lagoon section of the Diep River Estuary, including short-term options to address the build-up of nutrient rich and contaminated sediments in the lagoon and the Diep River channel, and the associated water quality and odour issues."**

The primary focus of the work is on determining the potential effects, feasibility, and costs of short term and strategic remediation measures targeted at water quality and habitat improvements within the Milnerton Lagoon.

Remediation options identified for assessment included:

- Dredging of the Milnerton Lagoon and/or the upper channel near the Potsdam WWTW to increase scour and flushing and remove nutrient-rich sediment (refer to section 5);
- Aeration of the lagoon to address anaerobic conditions (refer to section 6);
- Pumping of seawater into the Milnerton Lagoon to increase salinity (refer to section 7);
- Rerouting of Potsdam effluent to an outfall at the lagoon mouth or an ocean outfall (refer to section 8).
- Constructed reedbeds or off channel treatment trains for additional treatment of runoff (refer to sections 9 and 10); and
- Bioremediation of sediments through microbial inoculation (refer to section 11).

[*] For the purposes of this study, the Milnerton Lagoon is defined as that portion of the Diep River system that lies downstream of the Otto Du Plessis road bridge.

1.2.2 Professional team and authors of this report

The professional team encompasses a wide range and significant depth of expertise in coastal engineering, hydrological, sediment, and hydrodynamic modelling, water quality modelling, freshwater and estuarine ecology, and urban water resource management and bioremediation.

- Prof Gerrit Basson, Eddie Bosman, and Andreas Brooks of ASP Tech undertook the sediment transport, hydrological and hydrodynamic modelling and prepared conceptual designs for dredging and related interventions.
- Hanief Ally, in association with ASP Tech, developed water quality and hydrodynamic modelling to test proposed interventions.
- Dr Liz Day of Liz Day Consulting undertook the water quality assessments and assessed treatment wetlands and related interventions, as well as contributing to the assessment of bioremediation options.
- Dr Kevin Winter of the University of Cape Town and Future Water Institute assessed bioremediation and biofiltration options.
- Dr Barry Clark of Anchor Environmental provided estuarine function and ecological assessments, as well as contributing to the assessment of bioremediation options.
- Gus Hojem of PRDW prepared estimations of costs and feasibility for dredging, marine outfalls and pumping interventions. Specialist inputs in respect of electrical supply were provided by Brent Samson of B2A Consulting Engineers.
- Jeremy Rose of Infinity Environmental coordinated the specialist team and contributed to various sections of this report. Project support and inputs to this report were provided by Tom Smyth, Kirsten Barratt, and Kimberleigh Reddiar of Infinity Environmental.

1.3 Objectives and approach

The approach to the scope of work is, as far as possible, based on quantitative assessment of the potential effects of proposed interventions using hydrodynamic and water quality modelling. This is coupled with an in-depth assessment of existing and new data, as well as qualitative and expert review of interventions not tested by modelling.

Remediation as described in this report is envisaged as a set of short-term [<3 years to implement] interventions that are aimed at improving management and repairing ecosystem function. The restoration continuum (Figure 1-1) conceptualises 'remediation' as lying between the reduction of impacts and the initiation of full ecosystem recovery.

It must be noted as a starting premise that short-term remediation measures in this area are not expected to achieve any real desired ecological or associated human health and/or aesthetic outcome unless there is a significant change at a catchment scale, reducing the routing of major pollutants into the Diep River and its associated watercourses.

The focus of this plan, as indicated above, is to identify, assess, prioritise, and recommend for implementation measures that will contribute to ameliorating the **odour**, **water quality**, and **in-lagoon habitat** impacts of ongoing pollution in the Milnerton Lagoon. They are short-term measures focused on addressing symptoms, rather than causes. They cannot substitute for or replace the urgent steps required of the City of Cape Town to manage and mitigate the major causes of pollution.

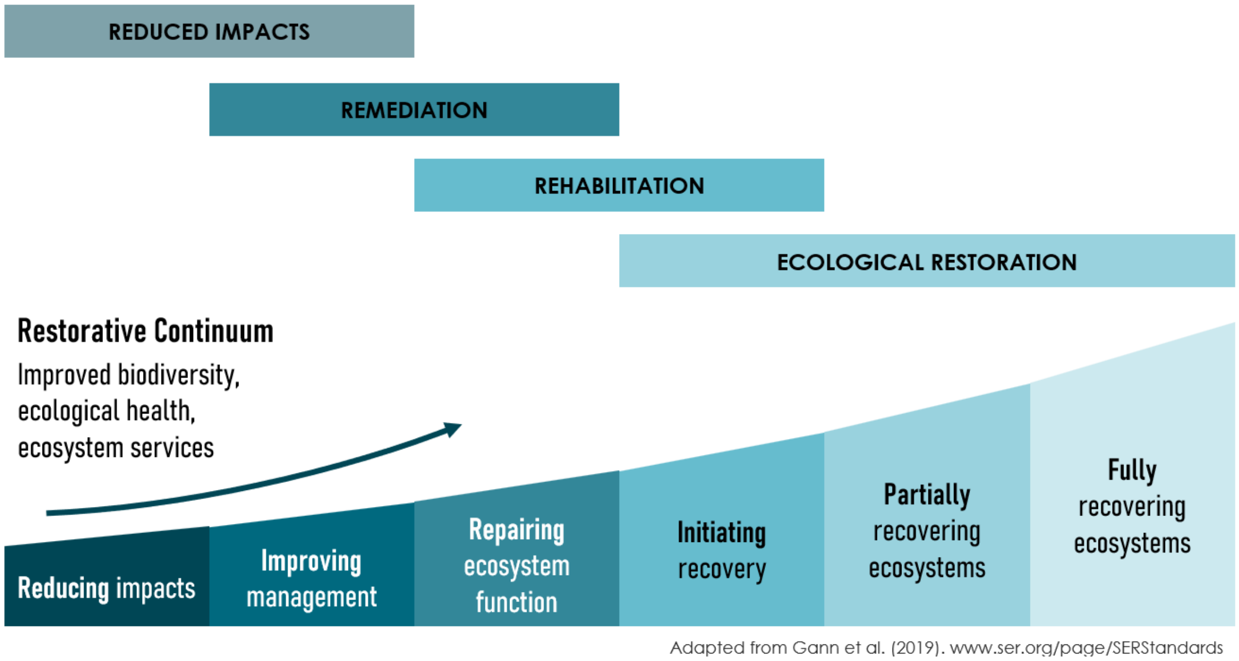


Figure 1-1. Remediation on the restoration continuum

This Remediation Plan must be seen in the context of a broader set of responses to pollution in the catchment, including the City's Water Quality Improvement Programme and Transversal Action Plan for the Lower Diep River, the planned Potsdam WWTW capital upgrades, and the more routine operational and maintenance responses to pollution already underway. Required interventions to manage the sources of pollution are summarised in section 3, although these are not the primary focus of the Remediation Plan and are beyond the scope of the detailed assessments carried out in respect of other proposed remediation actions. Several of these interventions were either already underway or in various stages of planning at the time of finalisation of this report in 2023.

1.4 Methodology and data collection

This Remediation Plan is based on a combination of long-term, existing data and new data collected for the purposes of the assessment and modelling. New data collection included:

- A full-coverage bathymetric survey and 50 metre grid sediment depth quantification of the lagoon and river channel conducted by Tritan Survey;
- Ad hoc water quality sampling at catchment inflows to the lagoon in November-December 2022;
- Water quality sampling during a full daily tidal cycle at spring tide in November 2022;
- Sediment quality classification in terms of waste disposal norms and standards for surface samples collected in the lagoon during December 2022;
- Sediment granulometric grading for surface samples collected in the lagoon during December 2022 and January 2023.

Modelling was conducted using the following models:

- Hydrodynamic: DHI Group Mike21C two-dimensional (horizontal) model
- Water quality: HEC RAS one-dimensional model

Further details of these methodologies are provided in the relevant sections of the report. Primary data, including the bathymetric survey, water and sediment quality results, and model outputs, are included in Annexures A through G.

1.5 Structure of this report

This Remediation Plan is structured as follows:

Chapter 1	**Background and introduction**
Chapter 2	**Situation Assessment**
Chapter 3	**Causes of impacts and current efforts to mitigate them**
Chapter 4	**Legislative context**
Chapter 5	Option 1: **Dredging**
Chapter 6	Option 2: **Aeration**
Chapter 7	Option 3: **Seawater flushing**
Chapter 8	Option 4: **Marine outfall**
Chapter 9	Option 5: **Biofilter supplementary stormwater treatment**
Chapter 10	Option 6: **Wetlands for supplementary stormwater treatment**
Chapter 11	Option 7: **Microbial bioremediation**
Chapter 12	**Hydrodynamic modelling of options**
Chapter 13	**Water quality modelling of options**
Chapter 14	**Risk, Cost and Benefit Analysis**
Chapter 15	**Recommendations and implementation plan**

1.6 Acknowledgements

Contributions to or comments on a draft version of this report are gratefully acknowledged from:

- Ben De Wet, City of Cape Town Catchment Planner
- Darryl Colenbrander, City of Cape Town Head: Coastal Policy Development
- Graham van Niekerk, City of Cape Town District Manager: North Operations (Wastewater treatment)
- Gregg Oelofse, City of Cape Town Manager: Coastal Management
- Michael Killick, City of Cape Town Director: Bulk Services
- Nicola Garcia, City of Cape Town Senior Professional Officer: Bulk Services
- Nisreen Hoosain, City of Cape Town Professional Officer: Technical Services
- Rajan Moodley, City of Cape Town Manager: Wastewater Branch
- Richard Nell, City of Cape Town Head: Strategy & Specialist Support, Catchment, Stormwater & River Management
- Suretha Dorse, City of Cape Town Senior Environmental Professional, Biodiversity Management Branch
- Werner Rössle, City of Cape Town: Head: Operations (North Section)

2 SITUATION ASSESSMENT

2.1 Estuarine context of the Milnerton Lagoon

Milnerton Lagoon lies within the Diep River Estuarine Functional Zone (EFZ) †, as indicated in Figure 2-1. The EFZ extends from just downstream of the Malibongwe Road bridge over the Diep River in the north, to the outlet of the estuary into the Atlantic Ocean at Milnerton Beach. It incorporates all areas below the 5 m contour including Rietvlei and Flamingo Vlei to the west; the Zoarvlei wetland‡ to the south; the coastal dunes west of Marine Drive; and the beach around the estuary mouth. Clark et al. (2018) estimate that the EFZ, excluding urban development, covers 834 ha, with a total open water area of approximately 229 ha. The Diep River estuary, save for a few occasions during Cape Town's 2015-2017 "Day Zero" drought, is permanently open to the sea. It is located in the cool temperate region of the Western Cape, entirely within the City of Cape Town but with a catchment area of 1 495 km^2 that extends over four municipalities.

The Diep River Estuarine Management Plan (EMP) (City of Cape Town and Infinity Environmental 2022) includes measures for the management of the entire area falling within the EFZ. Since the EFZ includes many different habitat types, variously affected by a range of ecosystem drivers and management objectives and pressures, the EMP has divided the EFZ into six distinct zones, also indicated in Figure 2-1 and summarised as follows (adapted from City of Cape Town and Infinity Environmental (2022):

1. **Upper channel**: This zone extends from the upper extent of the Estuarine Functional Zone (EFZ) of the Diep River (some 150m downstream of the Malibongwe Drive bridge) to the Blaauwberg Road Bridge, upstream of the discharge point for the Potsdam WWTW: this area comprises a wide river channel, edged (mainly) by *Phragmites australis* and *Typha capensis* reedbed, within a broader floodplain that includes depressional wetlands and relic saltmarsh areas;
2. **Middle channel**: The zone extends from the Blaauwberg Road Bridge to the Otto du Plessis Road Bridge, including the discharge point for the Potsdam WWTW - this zone comprises an excavated channel, that allows flows from the river upstream as well as inflows from the WWTW and stormwater channels (e.g. the Theo Marais channel) to be routed along the eastern edge of the EFZ, to the Otto Du Plessis Drive bridge – immediately downstream of Blaauwberg Road bridge, secondary channels divert some river flow upstream of the diversion channel into the eastern Rietvlei reedbeds;
3. **Rietvlei**: The zone is located west of the middle channel and comprises a mosaic of open water and seasonal wetland habitats including an extensive salt marsh, which dries out in the summer and is shallowly inundated in winter;
4. **Flamingo Vlei**: This zone includes two deep artificial lakes/ vleis, also referred to in places as the northern and southern deepwater lakes. The Bayside canal discharges into the north-western corner of the northern vlei;

† Estuarine Functional Zone (EFZ): Listing Notice 3 of Government Notice (GN) 324 of 2017 under the National Environmental Management Act (NEMA), Environmental Impact Assessment (EIA) Regulations (2017) defines an EFZ as "*the area in and around an estuary which includes the open water area, estuarine habitat (such as sand and mudflats, rock and plant communities) and the surrounding floodplain area, as defined by the area below the 5 m topographical contour (referenced from the indicative mean sea level)*".

‡ Zoarvlei wetland, although once part of the Diep River Estuary (Grindley and Dudley 1988) today is discharged via a stormwater culvert onto Milnerton Beach (Cerfonteyn and Day 2011)

5. **Milnerton Lagoon**: This zone consists of the lower part of the EFZ between the Otto du Plessis Road Bridge and the mouth of Lagoon Beach. **This is the only part of the EFZ that currently conforms to the legal definition of an "estuary"**, [§] as provided in the National Environmental Management: Integrated Coastal Management Act 24 of 2008 (as amended 2015), although it is understood that, under natural circumstances, the functional estuary would have extended much further upstream. The lagoon, in a confined channel stabilised by road embankments and bridges, has a maximum width of 150 m. The lagoon mouth naturally migrates between a gabion structure and concrete wall to the south, and a natural raised area about 250 m to the north. **Milnerton Lagoon is the focal area for this report**.
6. **Developed zone**: This zone consists of the transformed and developed section which comprises 33% of the EFZ. The zone consists of residential areas; infrastructure (e.g. roads, railways, stormwater conveyance and attenuation areas); industrial land use; commercial areas; the WWTW; grassed public spaces; sports facilities; and golf courses.

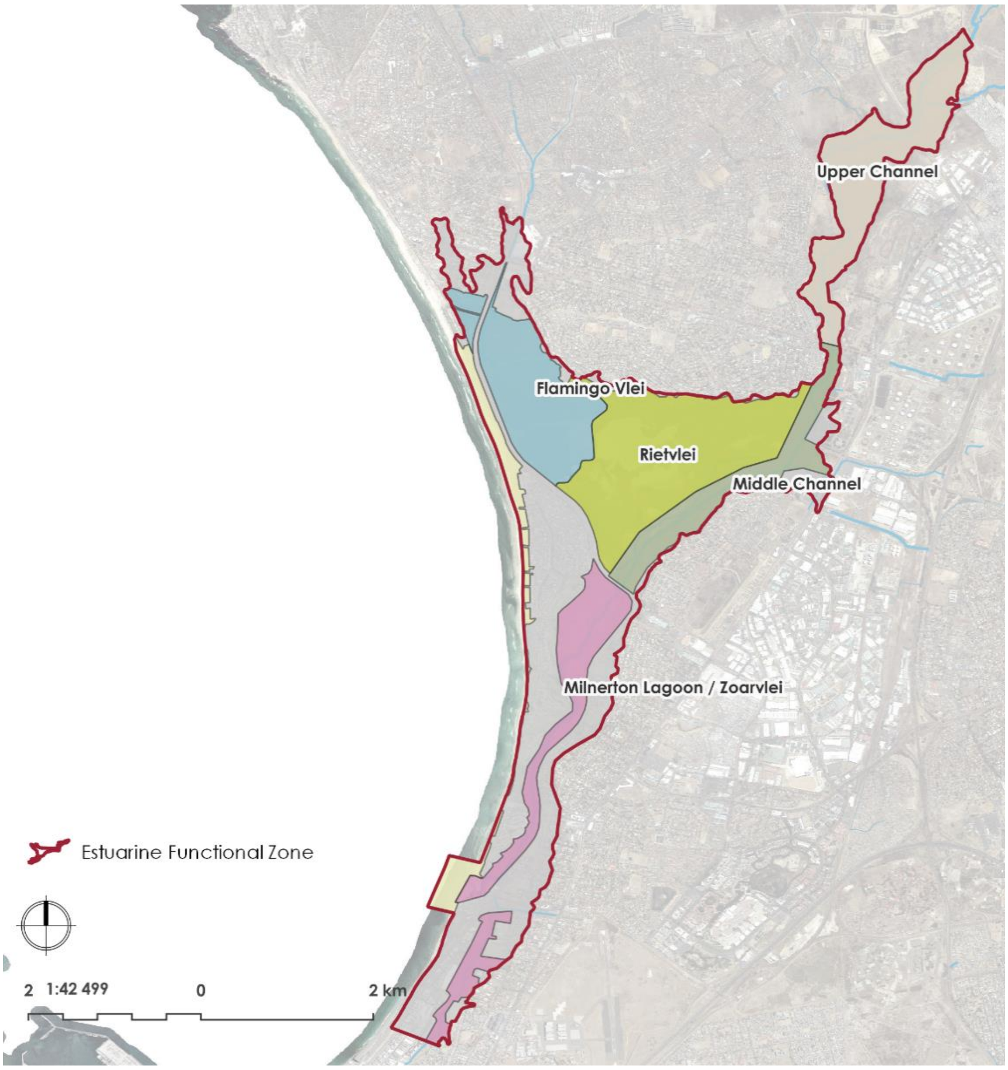


Figure 2-1. Management zones in the Diep River Estuarine Functional Zone, adapted from the Diep River Estuarine Management Plan of City of Cape Town and Infinity Environmental (2022)

[§] The National Environmental Management: Integrated Coastal Management (Act 24 of 2008 (as amended 2015)) defines an estuary as follows: **'estuary'** means a body of surface water— *(a)* that is permanently or periodically open to the sea; *(b)* in which a rise and fall of the water level as a result of the tides is measurable at spring tides when the body of surface water is open to the sea; or *(c)* in respect of which the salinity is higher than fresh water as a result of the influence of the sea, and where there is a salinity gradient between the tidal reach and the mouth of the body of surface water''

2.2 Diep River catchment characteristics and land use

The Diep River's catchment extends far north and east of the boundary of the City of Cape Town and includes parts of the Swartland Local Municipality (West Coast District) as well as portions of the Drakenstein and Stellenbosch Municipalities (Cape Winelands District).

The river rises in the Riebeek Kasteel mountains north of Malmesbury and flows through agricultural areas (mainly vineyards and wheat) before passing through the town of Malmesbury, where it receives treated effluent from the Malmesbury WWTW. Downstream of Malmesbury it flows through primarily agricultural areas, where its condition is impacted by abstraction from numerous on- and off-channel dams; poor water quality from crops and feed-lots; and extensive invasion of its banks by alien vegetation (both eucalyptus and various invasive acacia species).

The Diep River is joined in the Philadelphia area by its largest tributary, the Mosselbank River, which flows in from the east, rising in the Kraaifontein area of the City of Cape Town and passing through mainly residential and urban / peri-urban areas in its upper reaches and then mainly farms and smallholdings including agricultural feedlots in its lower reaches (Cerfonteyn and Day 2011). The Mosselbank River is a naturally seasonal, low-nutrient river system that does however now receive perennial effluent inflows from the Kraaifontein and – in winter – Fisantekraal WWTWs.

Downstream of the Malibongwe Drive / M12 road bridge, and just before the start of the EFZ at the 5 metres above sea level (masl) contour, the river passes into a burgeoning urban environment. Although formal settlement lies largely outside of the wide 1:100 year floodplain, invasion of the floodplain by informal settlements in the Du Noon area has burgeoned in the past decade, as shown in the sequence of images in Figure 2-2. Most of the informal settlements within the floodplain occur in the area between Malibongwe Drive and the railway bridge, and extend well into the floodplains of both the Diep River and its minor tributary, the Kleine Stink River, which enters the Diep River within this zone, passing under Malibongwe Drive some 780 m south east of the Diep River crossing.

Some 620 m downstream of the Blaauwberg Road bridge, the remnant waterbodies and surrounding undeveloped areas form a broad triangle, extending across Rietvlei and Flamingo vlei towards the coast to the west and edged by the WWTW; Theo Marais Park; and then dense urban development. Otto Du Plessis Drive itself forms first the western boundary of the upper part of this area, and, downstream of the Otto Du Plessis road bridge, the western edge of Milnerton Lagoon.

No other natural tributaries enter the Diep River between the railway bridge and the estuary outlet, with all point source inflows in these reaches today emanating from piped or channelled (and in some cases highly polluted) stormwater flows and treated effluent from the Potsdam WWTW – these are described in more detail in Sections **2.3** (Hydrology) and **2.5** (Water Quality).

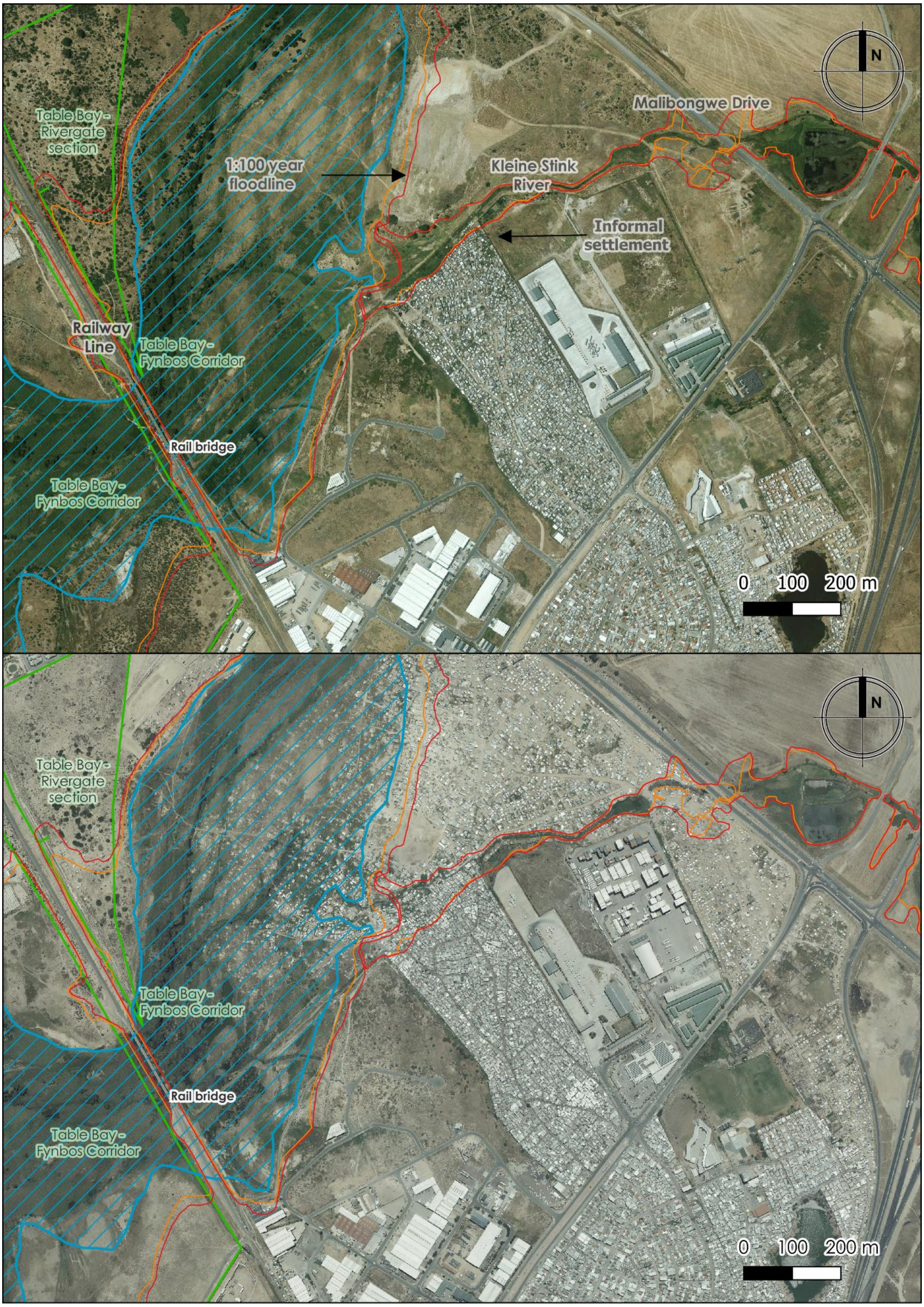


Figure 2-2. Aerial images from 2012 (top) and 2021 (bottom) showing invasion by informal settlements into the Diep River and Kleine Stink River 1:100 year floodplains (1:100 year floodlines in red; 1:50 in orange)

2.3 Hydrology

The Diep River Estuary lies in an area of the Western Cape characterised by winter rainfall and dry summers, which results in rapid evaporation during the summer. Mean annual precipitation for the entire catchment is approximately 500 to 600 mm (Jackson et al., 2008).

Early literature suggests (e.g. Scott and Millard 1954) that the Diep River and its tributaries were naturally seasonal, with the Diep River and Rietvlei wetlands drying up completely around December, when the river stopped flowing. Over the summer period, the waves built up a bar across the mouth, leaving a shallow stretch of water largely isolated from the sea, outside of high tides. These conditions persisted through summer and autumn until the return of the wet season when river inflows led to a steady rise in water level in the whole system, leading to the mouth bursting open around June. Throughout the rest of the winter and spring, the system was open to the sea and under tidal influence.

Since then, the hydrological functioning within the catchment has been subject to significant human-induced change. Flows entering the estuary from the catchment are presently 37.3 million cubic metres per year (Mm^3/a), significantly reduced from natural levels of 60.8 Mm^3/a (a 39% reduction) due mainly to agricultural abstraction of water (Clark et al, 2018). However, the estuary receives supplementary flows in the form of treated wastewater from the Potsdam WWTW, totalling 20.7 Mm^3/a. Total freshwater inputs are, therefore, similar to pre-development levels at 95% of the reference flow. The Potsdam WWTW has obtained environmental authorisation to increase its treatment capacity from 47 000 m^3 per day to 100 000 m^3 per day, discharging up to 75 m^3 per day. If fully utilised, this additional discharge would increase supplementary flows by a further 10.3 Mm^3/a.

Hydrology in the catchment is strongly seasonal, with flow in the river reducing to zero in summer months. Hydrology in the estuary in the summer months is driven, therefore, primarily by the effluent discharge rather than inflows from the river.

The Diep River EFZ is fed by flows entering at various points. These include:

1. The **Diep River** itself.
2. The **Kleine Stink River**.
3. Treated effluent from the **Potsdam WWTW**, discharged into an earth channel along the eastern boundary of Rietvlei. This channel, referred to hereafter as the bypass channel, enters the lagoon immediately downstream of the Otto du Plessis bridge. The main channel of the Diep River splits upstream of the bypass channel, with minor channels passing into extensive reedbeds west of the bypass, and the bypass channel lying to the east. In summer, inflows from the WWTW comprise almost the whole flow in the EFZ downstream of Blaauwberg Bridge, excluding sewage overflows. In high flow events, river water from the upstream catchment overtops into the bypass channel, Rietvlei and Flamingo Vlei – at such times, treated effluent from the WWTW would be dispersed into the broader Rietvlei and Flamingo Vlei areas.
4. The **Bayside Canal**, which discharges stormwater from the Table View area into the north-western part of Flamingo Vlei. Flows from this canal vary and are significantly lower in summer. Flamingo Vlei is inundated by (and therefore connected to) the main stem Diep River channel only in flood conditions.
5. The **Theo Marais Canal**, which discharges just downstream of the Potsdam outfall and carries stormwater flows from the Montague Gardens area as well as additional effluent, at times, from the WWTW, when the reedbeds between Theo Marais and the WWTW are inundated by overflows from the WWTW. The reedbeds themselves have been contaminated by a history

of receipt of sewage overflows, which affects the quality of water passing through them. The Theo Marais channel also receives runoff from the remediated Duikersvlei channel (previously a highly polluted system impacted by a former AECI fertiliser factory upstream) and the Theo Marais canal itself when the nearby Koeberg sewage pump station fails, as sewage overflows enter this channel upstream of the Duikersvlei inflow.

6. The **Erica Road** outfall, which carries stormwater including sewage from residential areas Phoenix and Joe Slovo Park where informal densification has taken place and sanitation infrastructure is insufficient for current population numbers; the outfall discharges these flows into the left (eastern) bank of the lagoon. These stormwater flows were, at the time of this report, being pumped (as far as possible) into an adjacent sewer manhole. Flow logging provided by the City for April 2023 indicates that the flows in the stormwater pipe were approximately 70 L/s during this time period, or 6 megalitres per day (Figure 2-4).
7. Other various minor **stormwater conduits**.
8. **Sea water** flows when the mouth is open.

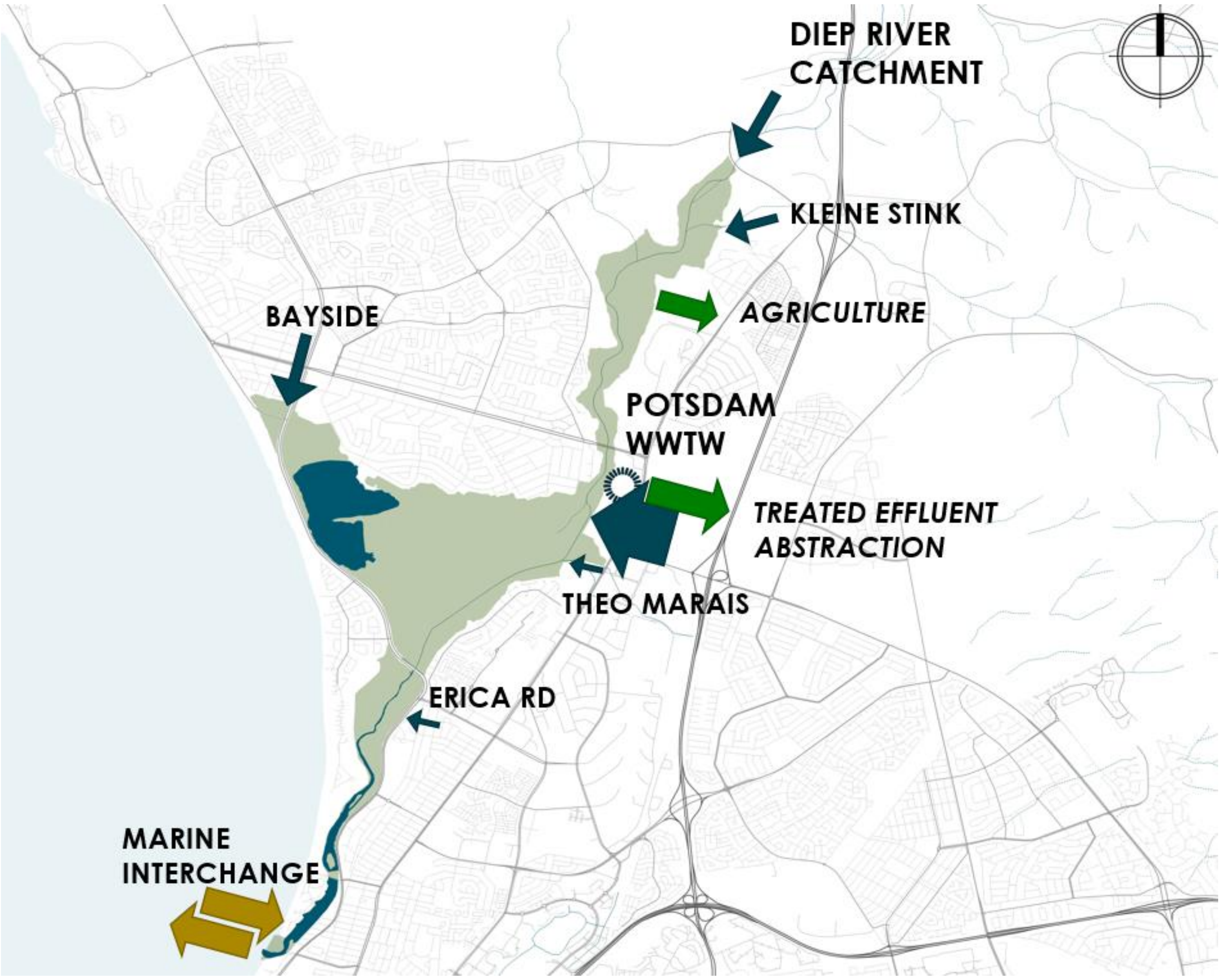


Figure 2-3. Flows into and out of the Diep River EFZ

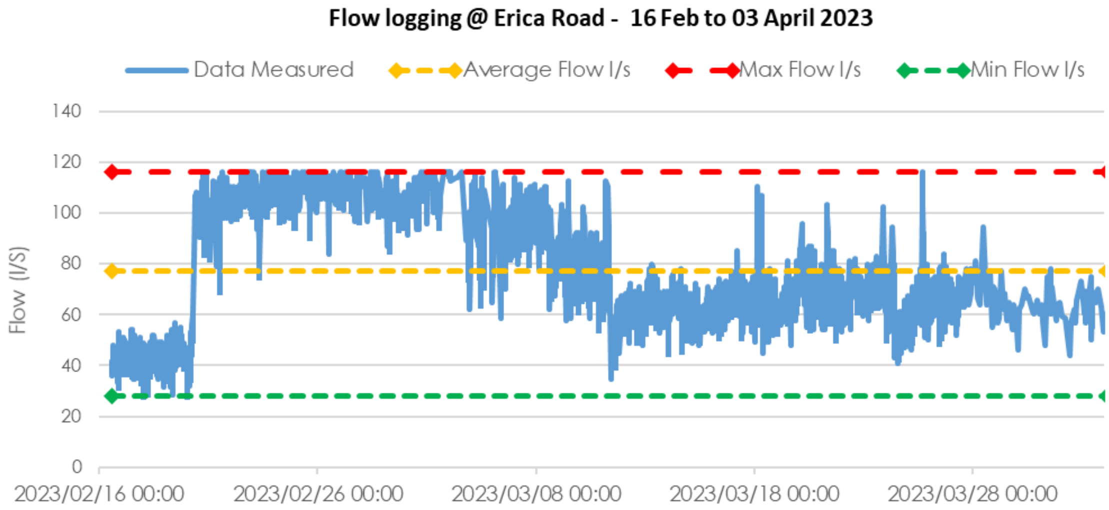


Figure 2-4. Stormwater flow rates at Erica Road outfall during March 2023 (supplied by CCT)

Flood peaks for the Diep River catchment were determined in a previous study: *Remediation plan for Flamingo Vlei: specialist assessments to inform proposed dredging of recreational water bodies* (Infinity Environmental and others, 2023). Table 2-1 summarises the flood peaks for the Diep River catchment determined in the study and the recommended flood peaks for each return period. For the purposes of this study, all flood peaks have been used without consideration for the effects of future climate change. Figure 2-5 shows a typical flood hydrograph that was determined from DWS flow gauging station G2H042 situated further upstream within the catchment on the Diep River, the peak of the flood hydrograph was scaled to the Q2-year and Q10-year floods respectively.

Table 2-1. Calculated flood peaks for the Diep River catchment (Infinity Environmental and others, 2023)

Annual Recurrence Interval (years)	2	5	10	20	50	100
Emprical (Kovacs)	-	-	-	-	2211	2666
SDF method	349	637	869	1100	1475	1808
Unit Hydrograph	300	415	532	666	881	1097
Probabilistic@G2H042	34	120	234	404	748	1128
Probabilistic Scaled to Otto du Plessis	37	129	252	435	806	1215
SHETRAN Probabilistic	105	275	399	600	804	949
Recommended	**105**	**275**	**399**	**600**	**806**	**1215**

The flood hydrograph shown in Figure 2-5 was routed through Rietvlei to determine the flood hydrographs used in this study. The routed Q2-year and Q10-year flood hydrographs used in this study are also shown. Rietvlei has large open flood plains which act as a detention pond and change the shape of the flood hydrograph. The routed flood peak for the Q2-year flood was 72.6 m^3/s and the Q10-year flood peak was 358 m^3/s.

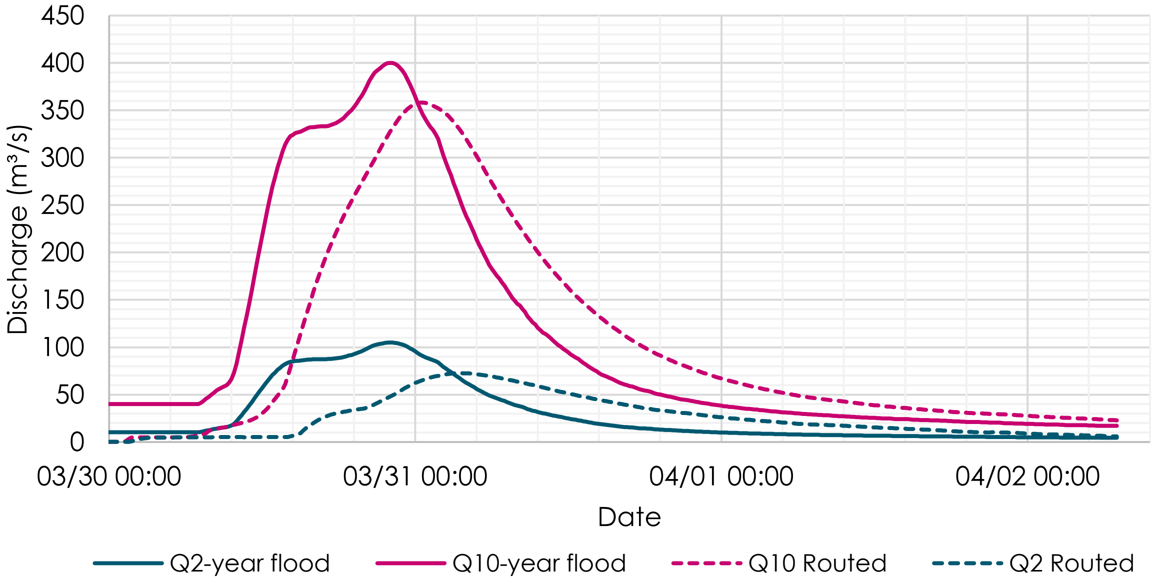


Figure 2-5. Q2-year and Q10-year flood hydrographs at the inlet and outlet of Rietvlei

Figure 2-6 shows a typical 2-week spring tide cycle as used in the hydrodynamic modelling section of this study.

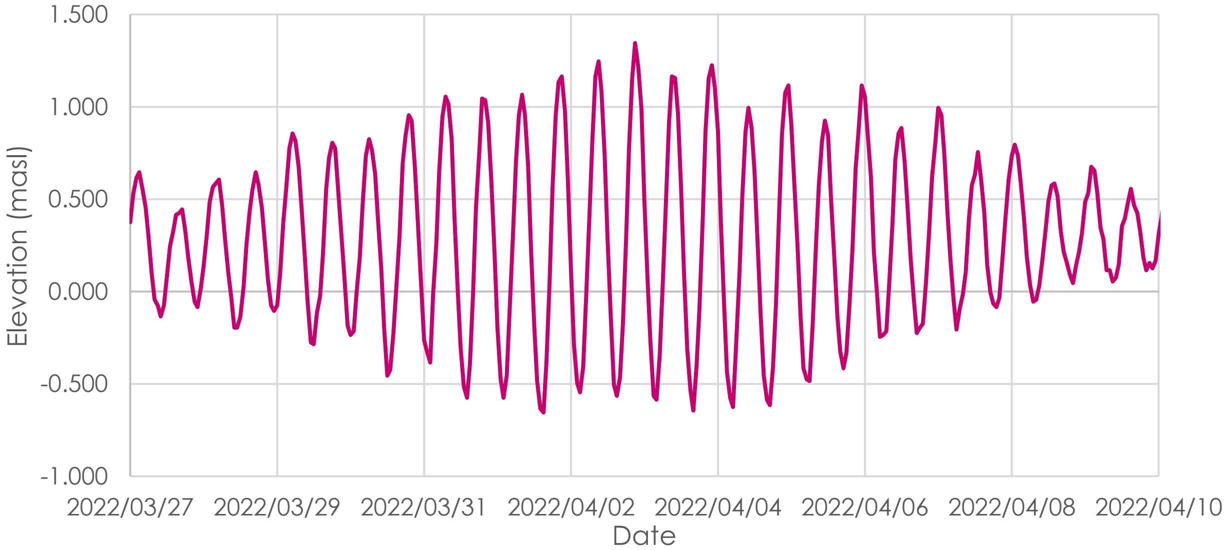


Figure 2-6. Observed tidal cycles at the Milnerton Lagoon mouth

2.4 Infrastructure context

The Milnerton Lagoon is at the mouth of the Diep River estuary, wholly surrounded by urban development and at the lowest point of the catchment containing Cape Town's most rapidly expanding new urban growth areas (Figure 2-7). It receives urban stormwater as well as untreated or poorly treated effluent from informal residential and industrial areas, as well as periodic spills from sewerage infrastructure and pump stations.

2.4.1 Stormwater

The stormwater catchment draining directly to the lagoon includes the areas of Bothasig, Richmond Park, Edgemead, Century City, Montague Gardens, Killarney Gardens, Du Noon, and Milnerton (Figure 2-8). Tableview, Parklands, and Sunningdale also drain to the estuary more broadly, with their stormwater discharging to the Flamingo Vlei or Rietvlei areas before entering the Diep River itself. The subcatchments identified as having significant impacts on water quality include the areas with highest densities of informal or less-formal housing. Du Noon / Doornbach discharges stormwater, grey water and sewage to the Kleine Stink and Diep River channels directly, while runoff and sewage overflows from Joe Slovo / Phoenix travel via a detention pond at Milky Way to an outfall at Erica Road. At this outfall, a temporary overpumping measure is currently implemented to transfer contaminated runoff and raw sewage from the stormwater system to sewer. It is reported that between five and seven megalitres per day is pumped in this way (Figure 2-4).

2.4.2 Potsdam WWTW

The Potsdam WWTW treats sewage from a catchment area that includes Milnerton, Edgemead, Bothasig, Tableview, Blouberg, Du Noon and part of Melkbosstrand. One section of the current plant was completed in 1997 with a treatment capacity of 17 megalitres (Ml) per day. The plant was expanded a decade later, adding an additional 30 Ml/day of treatment capacity. These sections operate as independent trains and are known as the '97 and '08 plants respectively. They utilise a biological nutrient removal activated sludge treatment process. After treatment and clarification, the treated effluent passes through a series of maturation ponds with a retention time of approximately two days at average dry weather flows. At the outlet from these ponds, effluent is (by design, though not currently in practice) disinfected by ultraviolet (UV) irradiation before discharge to the Diep River bypass channel and ultimately the Milnerton Lagoon.

The plant diverts a portion of the treated effluent for reuse in industry and in irrigation of parks, schools, golf courses, etc. At times the volume of effluent diverted for this purpose exceeds the volume discharged to the river, and the plant has prioritised interventions to improve the quality of effluent for supply to end users. Volumes treated by the plant have increased steadily over time, and it regularly exceeds its design hydraulic capacity. The existing WWTW is unable to cope with the volume and concentrations of influent it currently receives and this, coupled with maintenance issues and the effects of frequent power cuts, has resulted in a number of failures and resultant collapses in final discharge effluent quality. Plans to expand the plant to more than double its capacity have been in progress for more than a decade, and contracts for the construction of the expansion were awarded in March 2023 (see Section 3.2).

Photograph 2-1. Potsdam WWTW

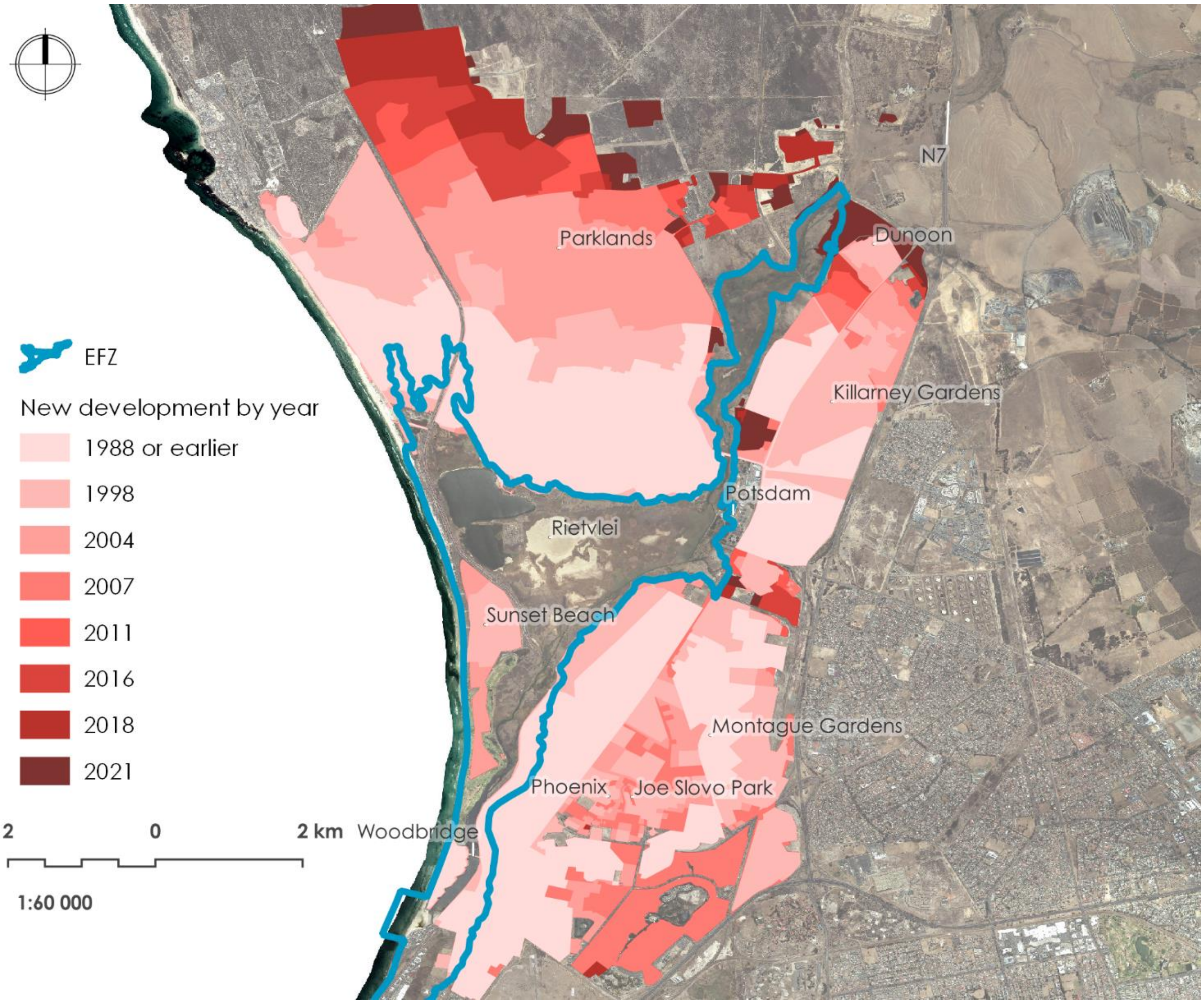


Figure 2-7. Development in the catchment since 1988 (from Infinity Environmental & CCT, 2022)

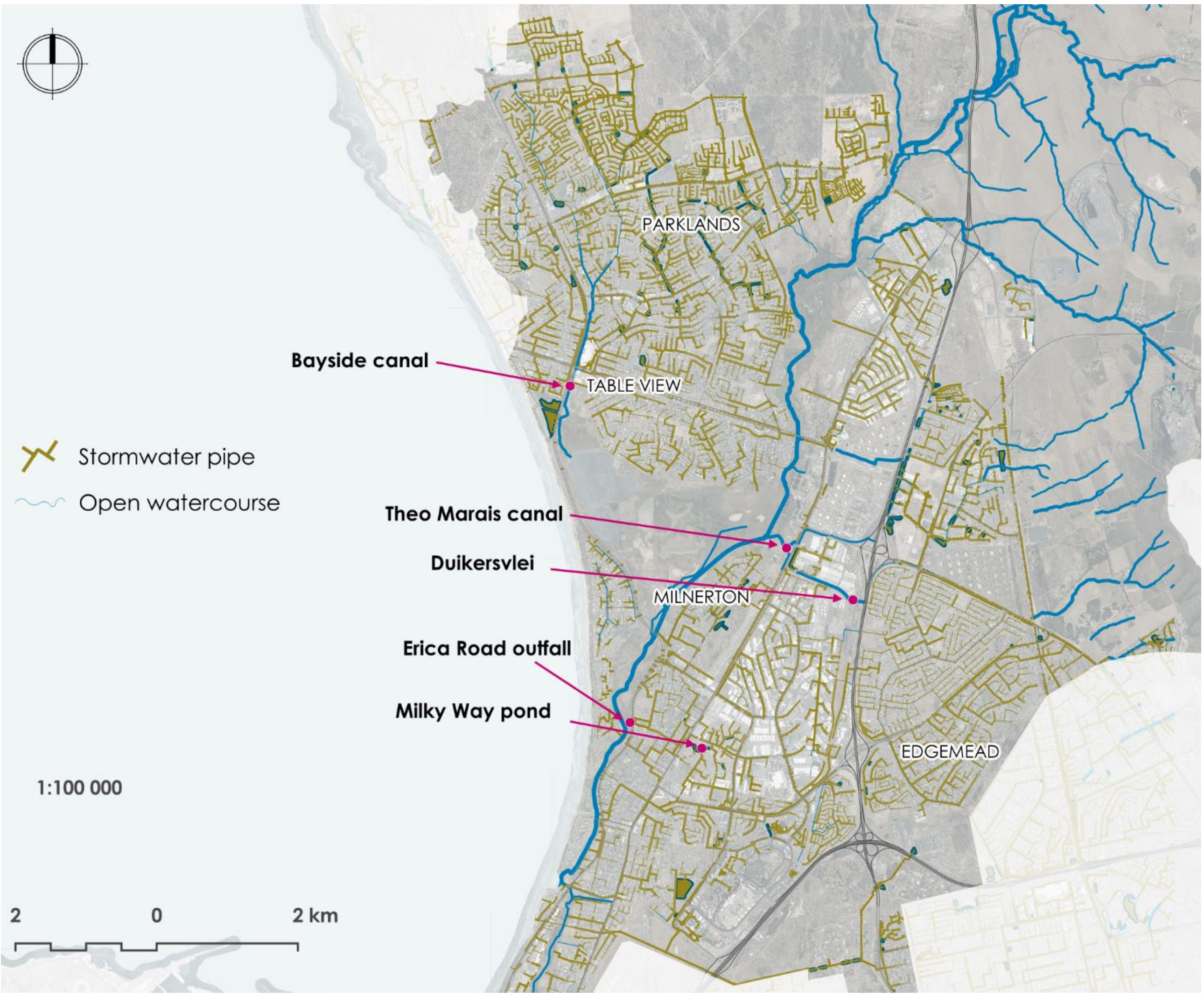


Figure 2-8. Stormwater infrastructure in the catchment, with points referred to in this report highlighted

2.5 Water quality assessment

2.5.1 Approach to water quality assessment in this report

Milnerton Lagoon, the focus of this study, lies at the downstream end of a catchment that includes natural river flows, stormwater runoff from urban areas and treated and untreated effluent. In order to understand the current condition of the lagoon and the key drivers of degradation, it is important to examine water quality in the broader inflows. This section presents both historical and once-off water quality data for inflows into and through the EFZ, with once-off water quality assessments intended to supplement long-term data and provide site-specific insights into conditions and particular challenges pertaining to key parts of the catchment at the time of sampling.

Once-off water quality samples were collected on 24 November 2022 from 11 sites along the Diep River and selected tributaries and significant stormwater channels / effluent outlets that enter the Diep River. These samples were analysed at BEMLAB (Somerset West) and used to complement interpretation of longer-term datasets from the City. The once-off nature of these assessments means that they cannot in themselves be interpreted with high confidence.

In addition, data for final treated sewage effluent that enters the Diep River system from the authorised Potsdam WWTW ("WWTW") was also sourced from the City, along with flow data, that allowed calculation of loading of key water quality parameters into the downstream system.

2.5.2 Water quality data availability

The City of Cape Town routinely collects water quality samples for analysis from a number of points along the Diep River including within the EFZ, as well as from key point-source discharges into the system, such as the WWTW and various major stormwater inflows. These water quality data go back as far as 1988 (Day et al 2020), although sampling frequency, and the number and locations of different sampling points has changed over time, with data from considerably more routine and *ad hoc* sampling points having been added to the City's water quality database over time.

The present assessment has focused on water quality data from the last six years only (i.e. 2017 to 2022) although it draws on findings from previous studies assessing a longer-term dataset. Specifically, data from the period of January 2017 to September 2022, when the present study commenced, were accessed from the City.

2.5.3 Locations of monitoring points

Figure 2-9 shows the locations of the City's routine water quality monitoring points as well as the *ad hoc* November 2022 sampling points included in the present study.

Table 2-2 describes the locations of the various sites in more detail. They are listed in order from upstream to downstream, and only those sites that are currently monitored and that lie on the main stem of the Diep River and could have a potentially significant impact on water quality and general ecosystem health in Milnerton Lagoon have been included – that is, water quality in Flamingo Vlei, Bayside Canal and the western part of the catchment feeding into Flamingo Vlei have not been considered here, although it is recognised that these parts of the catchment contribute significantly to ecosystem condition in Flamingo Vlei and are indeed linked with the main Diep River channel inflows during flood conditions.

Table 2-2. Locations of City's routine and current study ad hoc (November 2022) sampling points indicated in Figure 2-9 and referenced in subsequent sections.

Site code	Location	Routine City versus ad hoc (this study) site
***Ad hoc* sites sampled on 24 November 2022 for this study**		
Diep1 (D1)	33°48'30.77"S; 18°31'48.78"E: On the Diep River channel upstream of inflows from the Kleine Stink River and other channels from Du Noon informal settlement	*Ad hoc*
Kleine Stink (KS)	33°48'29.42"S; 18°31'57.98"E: Kleine Stink channel within Du Noon informal settlement	*Ad hoc*
Diep 2	33°48'38.83"S; 18°31'47.63"E: Diep River at railway bridge downstream of all inputs from Du Noon settlement	*Ad hoc*
Diep 3	33°49'6.74"S; 18°31'34.92"E: In stormwater outlet from industrial area into Diep River	*Ad hoc*
WWTW1	33°50'33.93"S; 18°31'18.34"E: Effluent at SPE upstream of maturation ponds	*Ad hoc*
WWTW2	33°50'26.31"S; 18°31'13.19"E: Final effluent just upstream of outlet into diversion channel	*Ad hoc*
TM-D	33°50'46.78"S; 18°31'0.05"E : Theo Marais channel in stormwater channel downstream of potential outlet from WWTW reedbed	*Ad hoc*

Site code	Location	Routine City versus ad hoc (this study) site
TM-Up	33°50'50.18"S;18°31'9.41"E : Theo Marais channel upstream of outlet from WWTW reedbed – Duikersvlei channel not flowing at time of sampling	*Ad hoc*
ER_RD	33°52'11.56"S; 18°29'44.17"E: Stormwater channel into Milnerton Lagoon at outlet west of Erika Road	*Ad hoc*
MWay	Stormwater channel into upstream of Erica Road, immediately downstream of Milky Way crossing	*Ad hoc*
City's routine monitoring data accessed for this study		
RTV06	Diep River downstream of N7 road bridge	City routine monitoring site
RTV01	Diep River at Blaauwberg Road bridge	City routine monitoring site
RTV11	Diep River (diversion channel) downstream of formal Potsdam WWTW inflow	City routine monitoring site
RTV12	Theo Marais channel downstream of the confluence with the Duikersvlei channel AND downstream of inflows from the WWTW reedbeds	City routine monitoring site
RTV03	Duikersvlei channel	City routine monitoring site
RTV08	Theo Marais channel upstream of the confluence with the Duikersvlei channel	City routine monitoring site
RTV05	Confluence of outlet from Rietvlei and bypass channel at Otto du Plessis Drive bridge	City routine monitoring site
RTV18	Milnerton Lagoon upstream of Woodbridge Island bridge, opposite Broad Rd and downstream of Erica Road outfall	City routine monitoring site
RTV09	Milnerton Lagoon at Woodbridge Island bridge (Loxton Rd)	City routine monitoring site
RTV10	Diep River estuary at mouth of Milnerton Lagoon	City routine monitoring site

2.5.4 Water quality variables considered in this report

Although the City analyses water samples for a relatively wide range of physical, chemical and microbiological variables, the following were considered both the most relevant to interpretation of ecological conditions in the lower Diep River and Milnerton Lagoon within the EFZ and to identification of point source inflows of key concern from both an ecological and a human health perspective. The following variables have been considered in this report:

- Electrical conductivity (EC) (as a surrogate measure for salinity);
- pH (primarily because of its role in determining ammonia toxicity);
- Major nutrients, which drive plant productivity; determine both habitat availability and quality; determine the trophic state of aquatic ecosystems and the frequency of occurrence of toxic algal blooms; and which form part of important nutrient cycling processes, which dictate *inter alia* rates of sediment formation and quality; and oxygen demand. The nutrients considered comprise:
    - Total inorganic nitrogen (TIN), calculated by addition of nitrogen in nitrate, nitrite and total ammonia (NH_4-N);
    - Orthophosphate (PO_4-P)
- Potential toxicants (ammonia [NH_3]; low dissolved oxygen [DO] and microcystin toxins);
- Indicators of risk to intermediate contact recreational use of Milnerton Lagoon by human users (that is, Microcystin toxins; *Escherichia coli* bacterial concentrations; and Enterococci bacteria).

Note that Total Suspended Solid (TSS) data for river and lagoon water have not been included in these analyses – this is because the magnitude of TSS is dependent on flow regime, with more solids being suspended as flow rates increase. The City does not however measure flow rate, making these data difficult to compare. TSS data for the WWTW have however been included, as these can be linked to particular measured discharge, from which loading rates can be calculated.

Note also that Chemical Oxygen Demand (COD) values downstream of RTV05 (Otto Du Plessis Drive) have been excluded, as these represent measurements in relatively saline environments, in which COD determinations are inaccurate. Moreover, the City's COD data record is patchy for the study, and does not provide a useful basis for statistical or other analysis. Only once-off COD data collected along the river course as far as RTV05 are thus referenced in this report, to complement long-term City DO and total ammonia nitrogen records.

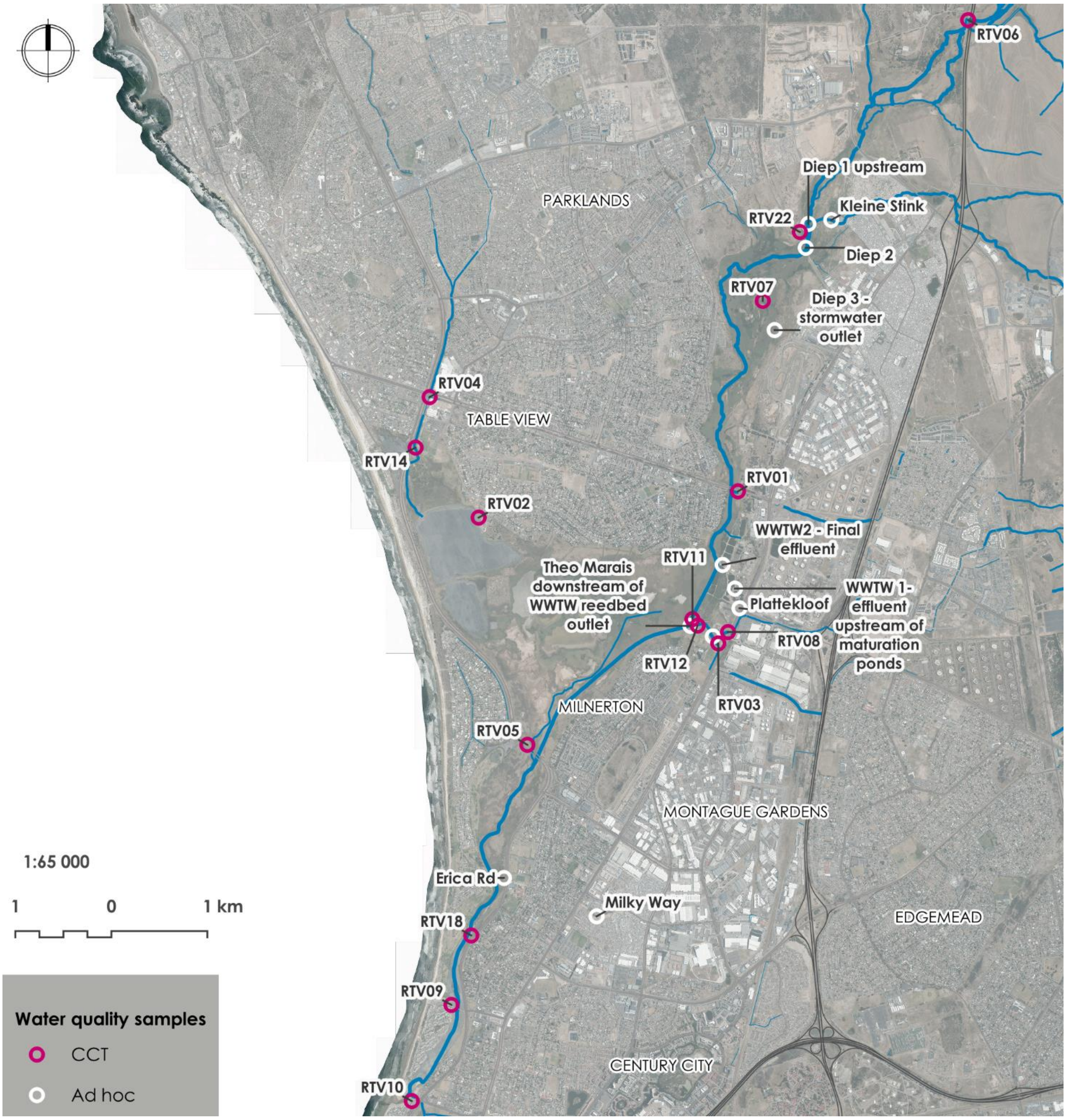


Figure 2-9. Locations of City's routine and current study *ad hoc* (November 2022) sampling points as referenced in subsequent sections – see Table 2-2 for description of site locations

2.5.5 Limitations and assumptions

Reporting on the water quality components of this report is subject to the following assumptions and limitations:

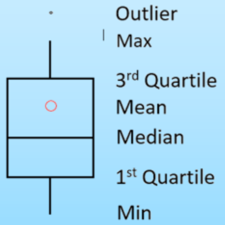
- No data on Contaminants of Emerging Concern (CECs) were available or considered for the affected aquatic ecosystems – it is however stressed that CECs are likely to play an increasing but as yet-unrecognised or quantified role in aquatic ecosystem condition, and particularly those ecosystems (including Milnerton Lagoon) that receive discharges from sewage effluent, including hormones and other pharmaceutical products that are not treated by standard effluent treatment procedures;
- The City does not routinely analyse its treated effluent discharges or watercourses for heavy metals – previous studies have however highlighted the fact that both estuarine sediments and tissues from various fish species indicate that the Diep River estuary has been impacted by receipt of various heavy metals at potentially toxic levels to different levels of the estuarine food chain (Jackson et al 2005; Hutchings and Clark 2010; Gihwala *et al*. 2021);
- The water quality data presented in this study are presented in terms of calendar years (January to December) rather than hydrological years (April to March), as per the City's Inland water quality reporting programme. This was because the hydrological and sedimentology models were set up to annual cycles, and for the purposes of current reporting, this aspect had little relevance;
- Routine City water quality data for the 2022 year included data up to end of September 2022 only – data for subsequent months in 2022 were only available in early 2023, by which time preliminary data assessment had been carried out already.

2.5.6 Assessment of City water quality data

The City's water quality data for the routine monitoring sites listed in Table 2-2 and illustrated in Figure 2-9 are summarised in Figure 2-10 to Figure 2-27, which include both time series data, showing patterns in data at individual sites over time, differentiated between summer and winter, and summary data, showing mean annual concentrations or values, with standard deviations around the mean or, in the case of un-ionised ammonia, electrical conductivity (EC) and pH, maximum values, since these pertain to specific toxicity and/or ecological thresholds, explained in more detail in the following sections.

Box 2-1. Interpretation of box plots

The box plots used to reflect water quality in the Milnerton lagoon and its catchment display data as follows:



2.5.6.1 Major nutrients

Orthophosphate and total inorganic nitrogen (TIN) data are presented in Figure 2-10 to Figure 2-16 with TIN data also presented as a ratio of total ammonia nitrogen (NH_4-N) to TIN, to illustrate the dominant form of nitrogen in the aquatic ecosystems, as indicative of the degree of nitrification [**] taking place.

Orthophosphate data suggest the following:

- All sites were consistently above the lower threshold for hypertrophic ecosystems with regard to phosphorus nutrients. This range has been identified in the City's Inland Water Quality Reporting system (e.g. Day et al 2020) as "Unacceptable", and the fact that most data fell

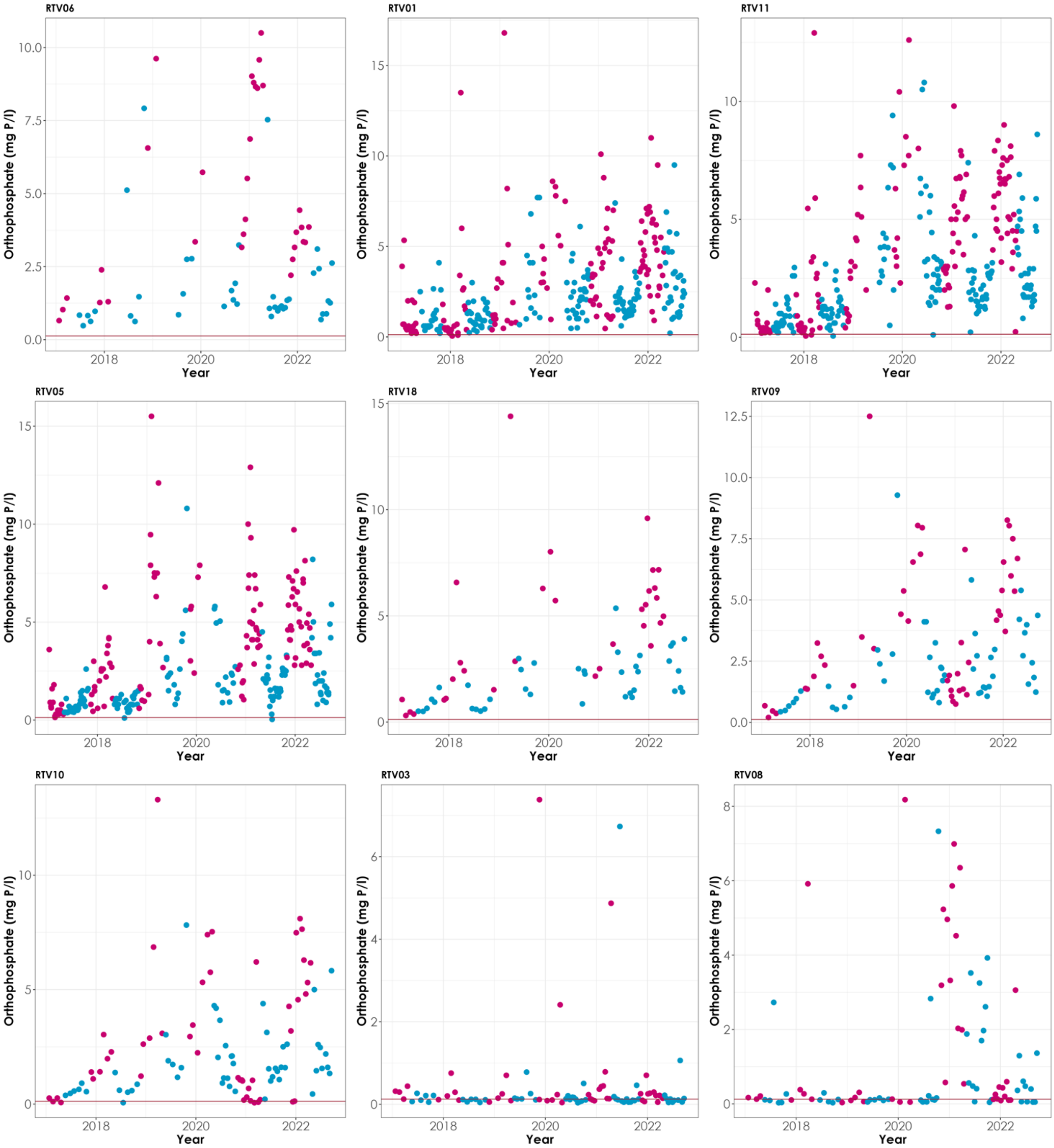
[**] Nitrification is the biological oxidation of ammonia to nitrite followed by the oxidation of the nitrite to nitrate; alternatively the direct oxidation of ammonia to nitrate

within this range over the assessed period points to a watercourse that has been impacted at a catchment scale by phosphorus enrichment over a significant period of time.

- All datapoints that were classified as "better than" hypertrophic represented samples from inflowing stormwater systems, via the Theo Marais channel (downstream of the WWTW inlet into the diversion channel). There were three exceptions to this pattern in the dataset, namely RTV01 and RTV11 in the main channel (Blaauwberg Bridge and downstream of the WWTW inlet) in January 2018; and RTV11 in August 2018.
- Water quality was consistently worse in summer, when phosphorus concentrations were elevated, presumably largely as a result of evapoconcentration and reduced dilution of pollutant inflows by river water under low (or no) flow conditions.
- Mean annual orthophosphate concentrations between the N7 (RTV06) and Blaauwberg Bridge (RTV01) increased substantially in 2022 at the downstream site, which is affected by inflows from the (expanding) Du Noon informal settlement as well as inflows from the (also highly impacted) Kleine Stink River. Data for previous years suggested variable (but polluted) water quality in these reaches, with no marked deterioration between the two sites. This highlights the impact of upstream land use in contributing to phosphorus enrichment (agricultural impacts [feedlots; fertilisers]) as well as inflows from other WWTWs (e.g. in the Diep River's Mosselbank catchment) – the data show no improvement in water quality with distance downstream, indicating sustained inflows of polluted water within these reaches as well (e.g. from the Du Noon informal settlements and other polluted stormwater inflows).
- The river reaches between RTV01 and RTV11 receive treated effluent from the WWTW. Over the period 2019 to 2022, there was a marked increase in orthophosphate concentrations in these reaches, which can reasonably be attributed to inflows from the WWTW. What the data do not indicate, but which is clear from site assessments, is the fact that while summer orthophosphate concentrations are elevated at upstream sites, although generally lower than those at RTV11, orthophosphate **loading** is significantly greater as a result of inflows from the WWTW rather than from inflows further upstream. Site assessments in the present study in November 2022; December 2022 and January 2023 all indicated no perceptible flow downstream of Du Noon (at the railway bridge) and trickle flow only at Blaauwberg Bridge (RTV01), in stark contrast to the permanent inflows from the WWTW, in the region of 36 million L / day – this issue is unpacked in more detail in Section 3.2.
- Concentrations of this nutrient were generally lower in the Theo Marais channel than at RTV11. The standard deviation data shown in Figure 2-11 do however suggest high variability, particularly at RTV12 – this is presumed to be indicative of variable levels of inflows of polluted effluent into the channel as a result of intermittent pollution events, including sewage overflows into the Theo Marais channel as a result of failure of the nearby Koeberg sewage pump station as well as from overflows of stormwater or treated effluent into the channel via the (contaminated) WWTW reedbeds.
- Figure 2-11 indicates a pattern of slight improvement with distance downstream from RTV11, with mean annual orthophosphate concentrations at RTV05 (Otto Du Plessis Drive bridge) decreasing slightly albeit not significantly in most years (excluding 2019).
- With distance into the lagoon, orthophosphate concentrations generally initially increase – data for RTV18, downstream of the Erica Road outfall, showed higher mean annual concentrations of this nutrient, with a sharp rise in the 2022 monitoring period, which carried through all the way to the mouth of the lagoon at RTV10, despite potential dilution by marine waters at the mouth, indicating severe eutrophication and potential contributions from in-lagoon sediments as well in these reaches. Prior to 2022, lagoon mouth concentrations were generally much lower than at upstream sites and considerably lower than at RTV11.

- Generally, orthophosphate data show significant sources of phosphorus into the Milnerton Lagoon area, likely to contribute to substantial aquatic plant growth such as algae, including at times blue-green algal growth. The frequency of testing[††] and results of Microcystin analyses shown in Figure 2-12 suggest that there has been a clear increase in the frequency of blue-green algal blooms in the lagoon over the past three years, albeit Microcystin toxicity on all of these occasions has been very low.

[††] Microcystin analyses are costly and the City undertakes these tests only when blue-green algal blooms are evident, and not as a routine water quality test (see Day et al 2020). The frequency of testing thus can be used as a surrogate for the number of observed algal blooms



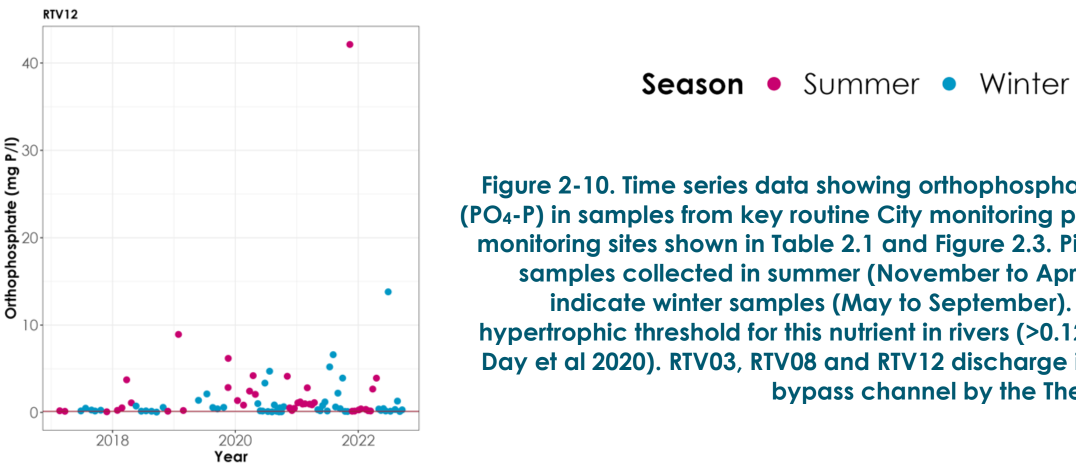


Figure 2-10. Time series data showing orthophosphate concentrations (PO_4-P) in samples from key routine City monitoring points – positions of monitoring sites shown in Table 2.1 and Figure 2.3. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September). Red line illustrates hypertrophic threshold for this nutrient in rivers (>0.125 mg P/L – as per Day et al 2020). RTV03, RTV08 and RTV12 discharge into the Diep River bypass channel by the Theo Marais channel

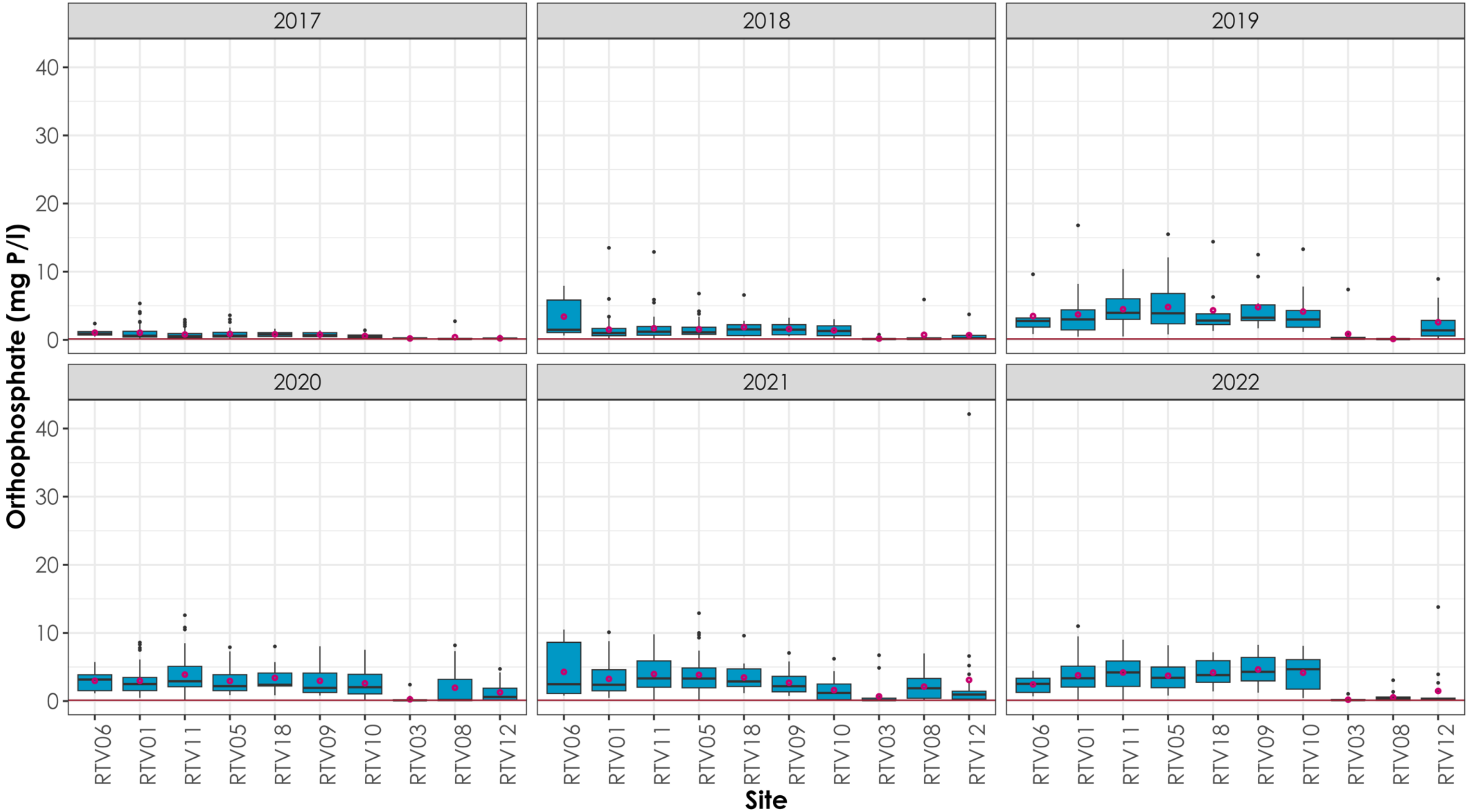


Figure 2-11. Summary data showing mean, median, range and outliers in annual orthophosphate concentrations (PO_4-P) in samples from key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2.1 and Figure 2.3. Red line illustrates hypertrophic threshold for this nutrient in rivers (>0.125 mg P/L – as per Day et al 2020). Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December. Inset to 2021 graph shows RTV12 outlier at 42 mg P/L in 2021 dataset

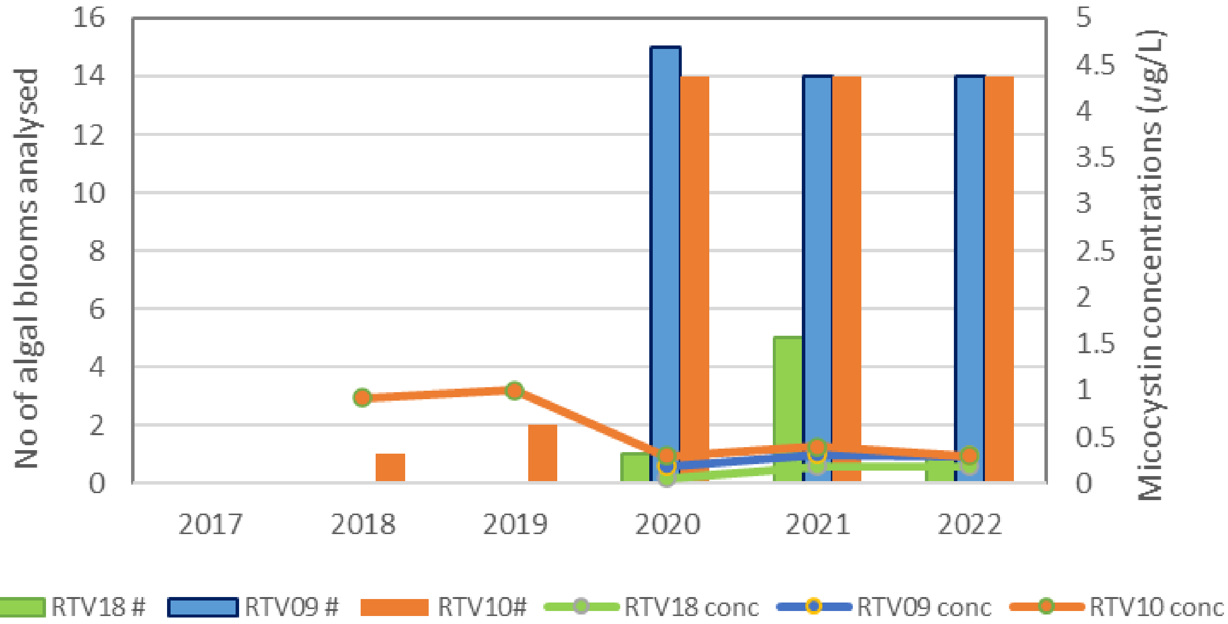


Figure 2-12. Number (No) of Microcystin analyses per year in Milnerton Lagoon between January 2017 and September 2022, with secondary axis showing maximum Microcystin concentrations

Nitrogen nutrients are reflected in TIN data shown in Figure 2-13 **and** Figure 2-14**,** as well as in data reflecting the proportions of total ammonia nitrogen in TIN (Figure 2-15 and Figure 2-16). These data suggest the following:

- Nitrogen enrichment increased dramatically with distance downstream of the N7 crossing (RTV06) and was generally greatest in summer months when dilution of polluted inflows by river water was least likely;
- Mean and median TIN concentrations increased significantly between 2017 and 2022, with data from 2019 and 2022 showing the highest median and mean annual concentrations downstream of RTV06 – the latter site showed an increase in 2018 but thereafter its TIN concentrations remained relatively stable and low, compared to downstream river reaches, suggesting that TIN inflows are associated mainly with urban inflows and land use, and that upstream inputs are largely ameliorated by in-channel biological and chemical processes;
- After 2017, mean TIN values were always in the eutrophic range for this nutrient in the main river reaches as far as RTV05 (Otto Du Plessis Road bridge), and fell within the City's "Unacceptable" water quality category (see Day et al 2020);
- The influence of inflows from the WWTW (upstream of RTV11) as well as from the Theo Marais channel (RTV12) on downstream water quality can be inferred from Figure 2-14, where elevated median annual concentrations at RTV12 (downstream end of the Theo Marais channel) were linked to elevated concentrations at RTV05 (2019) suggesting relatively high discharge and associated loading from RTV12 compared to the upstream RTV11 in the bypass channel, whereas when concentrations were elevated only at RTV08, upstream of RTV03, it appears that actual loading was probably low, resulting in a reduction in TIN concentrations at RTV05 downstream (e.g. 2018, 2020, 2021 and 2022 data). Polluted inflows into the Theo Marais channel at RTV08 are likely to have included sewage overflows during Koeberg pump station failures; while elevated concentrations at RTV12 alone would probably reflect inflows from the contaminated WWTW reedbeds;

- Inflows from the Duikersvlei channel (RTV03) were generally always associated with low levels of nitrogen enrichment, indicating that past remediation of this once-highly nitrogen enriched channel (see Cerfontein and Day 2010) has been successful;
- 2019 and 2022 TIN data showed markedly elevated TIN concentrations throughout the Diep River and Milnerton Lagoon downstream of RTV06. While inflows from the Du Noon area (reflected in part in data for RTV01) do indicate significant pollution. Loading of TIN at the WWTW and periodically from inflows from the Theo Marais channel, via the above pollution pathways, is most likely to drive elevated TIN concentrations downstream, particularly during summer when upstream flows are very low;
- Ratios of total ammonia nitrogen to TIN (and) indicate a marked and progressive increase in the proportion of total ammonia nitrogen in TIN over the past six years downstream of RTV06. Median data shows that almost all of TIN was in its ammonium form. This indicates very low levels of nitrification in the system, associated with high chemical and biological oxygen demand and low dissolved oxygen availability. This pattern is particularly evident in data for sites RTV18; RTV09; and RTV10 (all in the Milnerton Lagoon) in 2022 in particular, and correlates well with reported hydrogen sulphide odour and visible bubbling of (assumed) methane and hydrogen sulphide gases from decomposing sediments in the lower lagoon area.

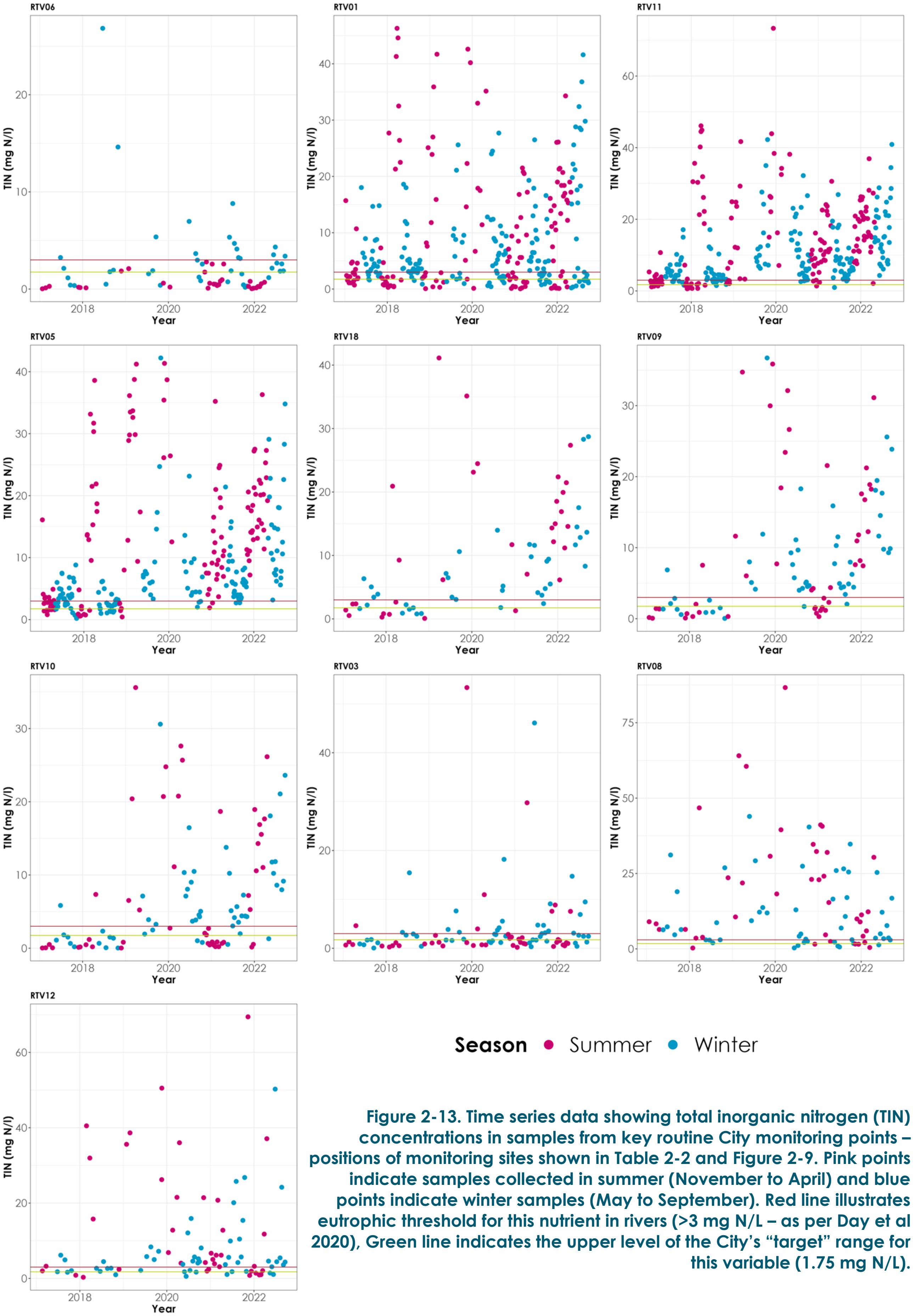


Figure 2-13. Time series data showing total inorganic nitrogen (TIN) concentrations in samples from key routine City monitoring points – positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September). Red line illustrates eutrophic threshold for this nutrient in rivers (>3 mg N/L – as per Day et al 2020), Green line indicates the upper level of the City's "target" range for this variable (1.75 mg N/L).

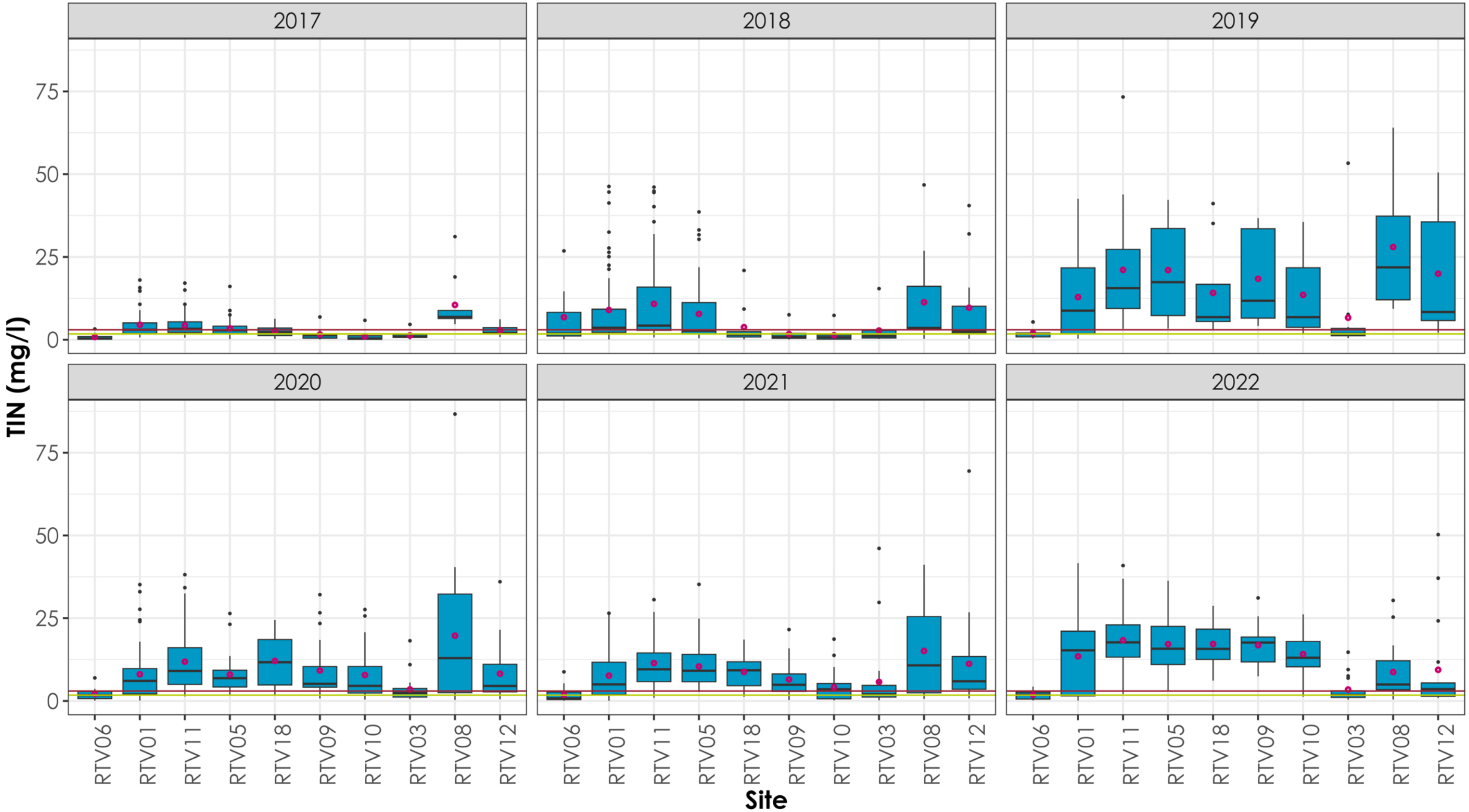


Figure 2-14. Summary data showing mean, median, range and outliers in annual TIN concentrations in samples from key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Red line illustrates eutrophic threshold for this nutrient in rivers (>3 mg N/L – as per Day et al 2020), Green line indicates the upper level of the City's "target" range for this variable (1.75 mg N/L). Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December.

Figure 2-15. Time series data showing ratios of total ammonia nitrogen to TIN in samples from key routine City monitoring points – positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September)

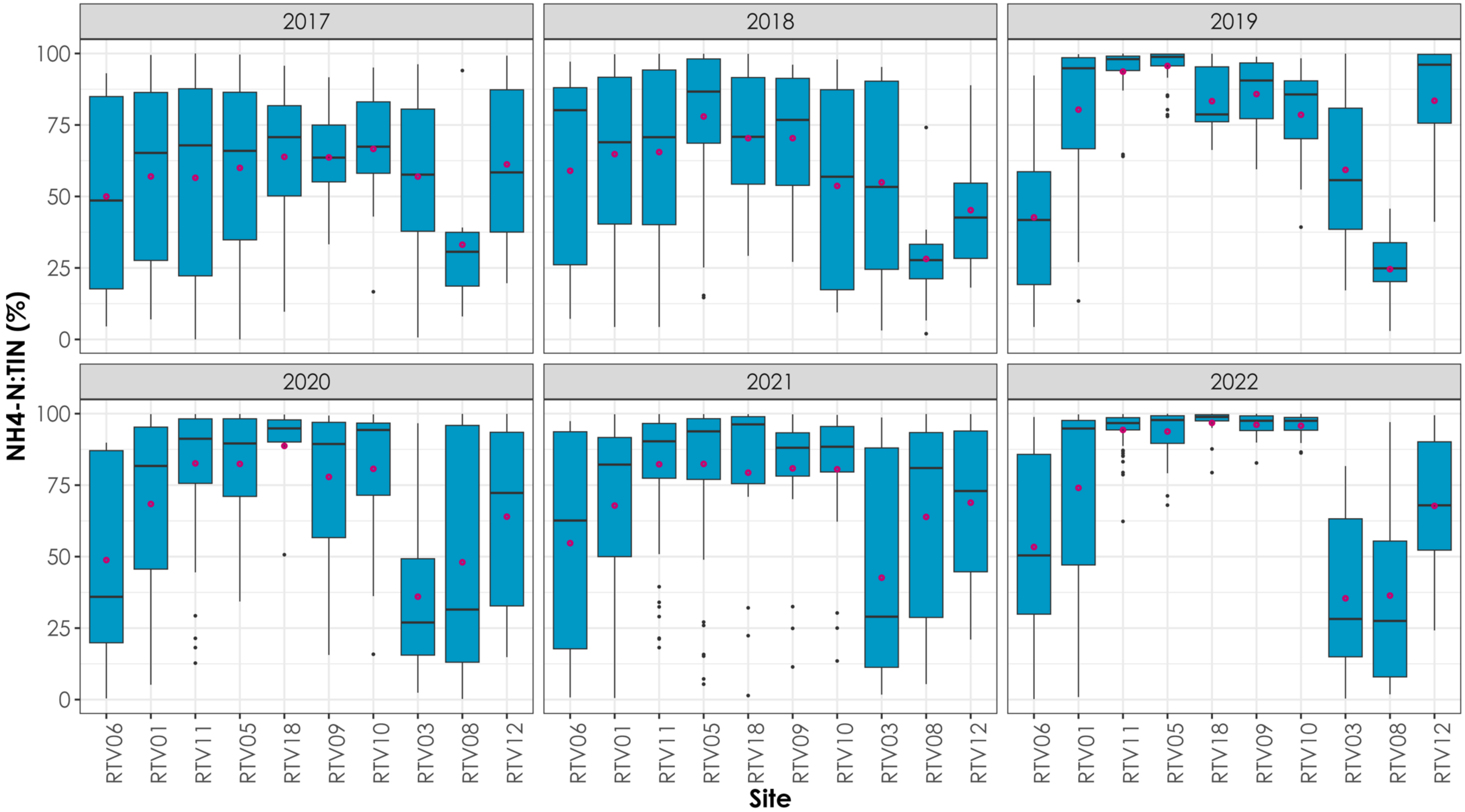


Figure 2-16. Summary data showing annual median, range and outliers in ratios of total ammonia nitrogen to TIN in samples from key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December

2.5.6.2 Dissolved oxygen

Dissolved oxygen concentration data (Figure 2-17)[‡‡] are important informants of river, wetland and estuarine condition. They do however need to be interpreted with caution. This is because dissolved oxygen (DO) concentration is affected by temperature, which means that the time of day that the measurement is taken is important, when comparing data between sites. DO concentrations may also increase diurnally under eutrophic and hypertrophic conditions, as a result of elevated photosynthesis, but drop markedly at night, when photosynthesis is absent and respiring plants have a high oxygen demand. In addition, DO data for all sites reflect shallow sub-surface sampling, and do not reflect DO concentrations at the base of the water bodies, where DO is likely to be considerably lower as a result of organic decomposition.

Bearing in mind the above considerations, the City's long-term DO data suggest that:

» DO concentrations in all river and lagoon reaches between the N7 bridge (RTV06) and RTV18 fell mainly below the City's Target concentrations throughout the period 2017 to 2022.
» RTV11, RTV05 and RTV18 were usually the worst performing sites, with most sites lying below the Unacceptable threshold.
» DO concentrations generally increased in the lower lagoon area downstream of RTV18, with RTV09 and RTV10, with most measured samples at these sites lying within the Target range.
» Of the three sites contributing to water quality in the Theo Marais outlet into the bypass channel, RTV08 and RTV03 generally performed better than did RTV12, with the latter (downstream) site also being affected, at least at times, by runoff from the contaminated WWTW reedbed area.

» Summary data in Figure 2-18 are valuable in showing:
- The persistent drop (usually to within the Unacceptable range) of DO at RTV05, presumably indicating the high chemical and biological demand associated with inflows from the WWTW as well as from the Theo Marais channel;
- The accompanying ongoing deterioration in DO concentrations at sites RTV18, RTV09 and RTV10 – although RTV09 and RTV10 were usually higher in DO concentrations than RTV18 (and above critical thresholds), presumably as a result of tidal flushing, the data show a major decrease in median and mean DO concentrations at these sites over the past six years, and in particular over the 2022 monitoring period.

Reduced DO in the lower estuary over the past few years is consistent with increased ammonium nitrogen: TIN ratios; and increased reporting on hydrogen sulphide odours in this area, all indicative of high COD and BOD that exceeds available oxygen.

[‡‡] Higher levels of DO are considered beneficial while lower levels are damaging. Therefore, having DO concentrations below the unacceptable level is actually unfavourable, unlike the other nutrients discussed in this section where lower concentrations are preferred.

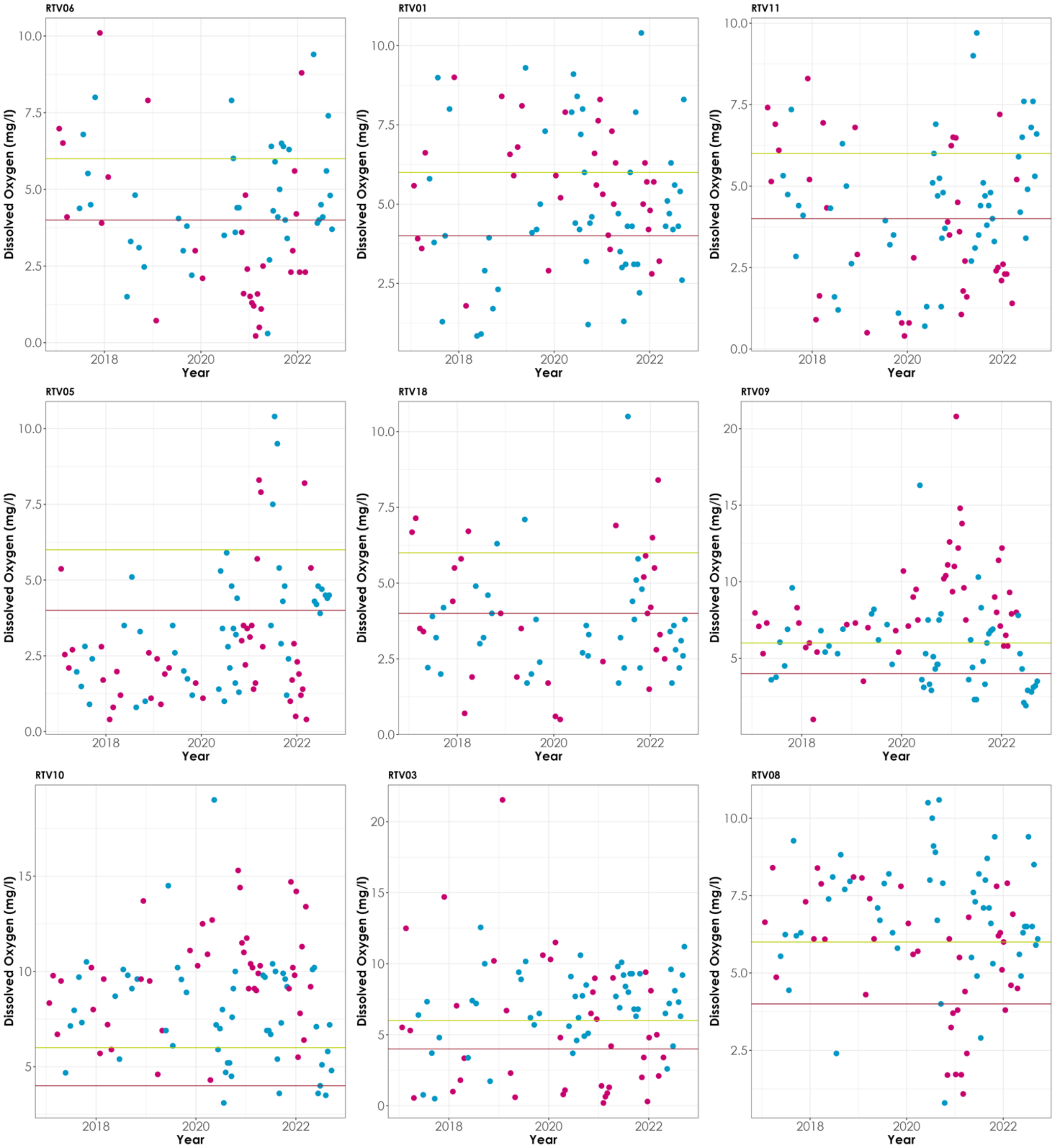


Figure 2-17. Time series data showing dissolved oxygen concentrations as measured at key routine City monitoring points – positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September). Green line indicates the threshold for the City's target DO concentrations (> 6 mg/L). Red line indicates the threshold value for unacceptably low DO concentrations (< 4 mg/L) (as per Day et al 2020).

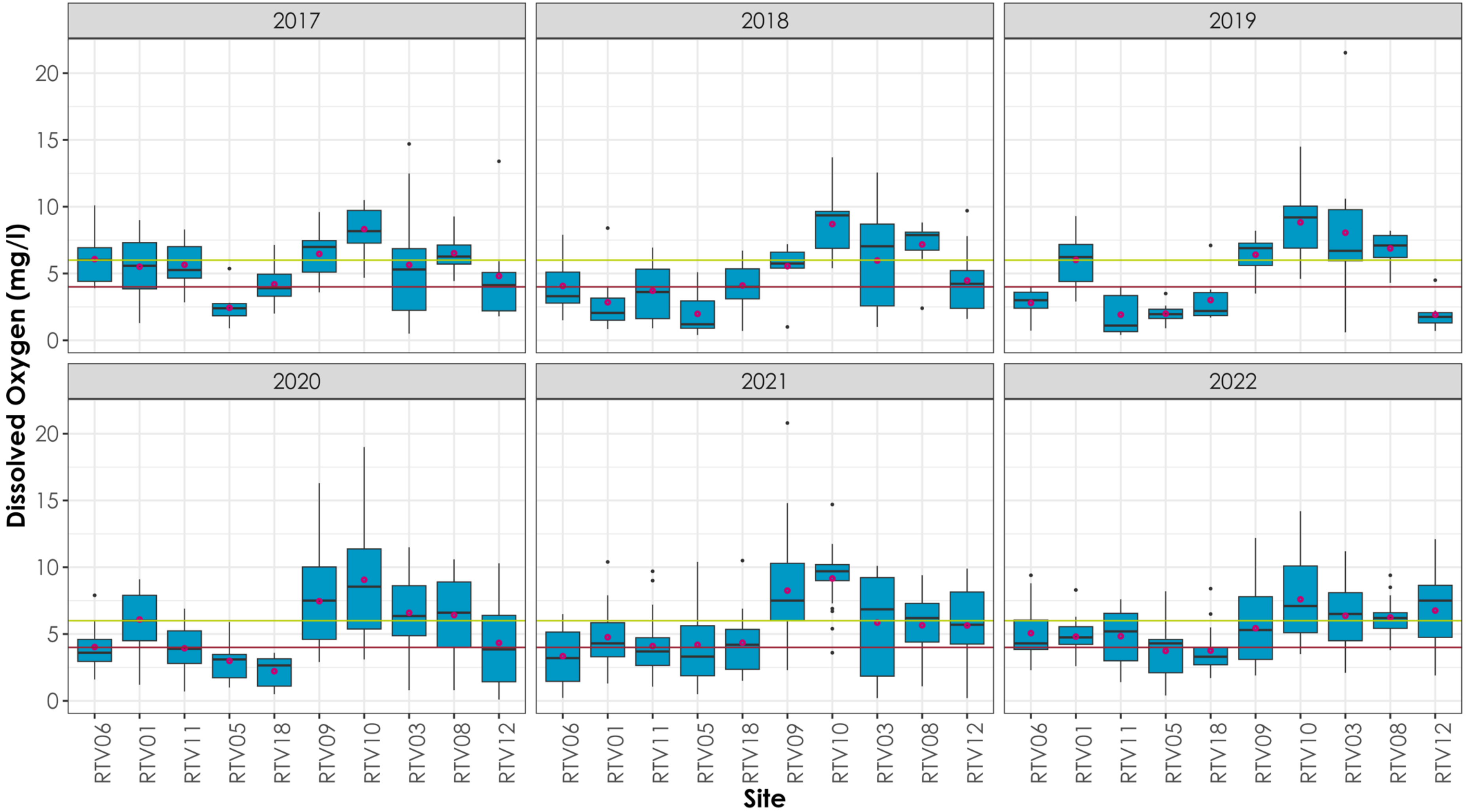


Figure 2-18. Summary data showing annual mean, median, range and outliers in dissolved oxygen concentrations measured at key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December. Green line indicates the threshold for the City's target DO concentrations (> 6 mg/L). Red line indicates the threshold value for unacceptably low DO concentrations (< 4 mg/L) (as per Day et al 2020).

2.5.6.3 Ammonia toxicity

Ammonia (NH_3) is a component of the reported total ammonia nitrogen referenced in Section 2.5.6.1. Total ammonia includes both ionised (NH_4^+) and un-ionised (NH_3) forms. Of these, the latter can be toxic, even at very low concentrations. The proportion of NH_3 in total ammonia is largely determined by pH and temperature, although salinity also plays a role. As pH and temperature increase, specifically as pH increases above pH 8.0, so does the proportion of NH_3 increase. DWAF (1996) provides a tabulated conversion to calculate NH_3 as a percentage of total ammonia.

pH data for monitoring sites within the study area (Figure 2-19 and Figure 2-20) show that:

- pH values were with few exceptions below pH 9 at all sites
- pH was generally slightly elevated in the lower lagoon compared to the river / upstream diversion channel.

Figure 2-19. Time series data showing pH in samples from key routine City monitoring points– positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September).

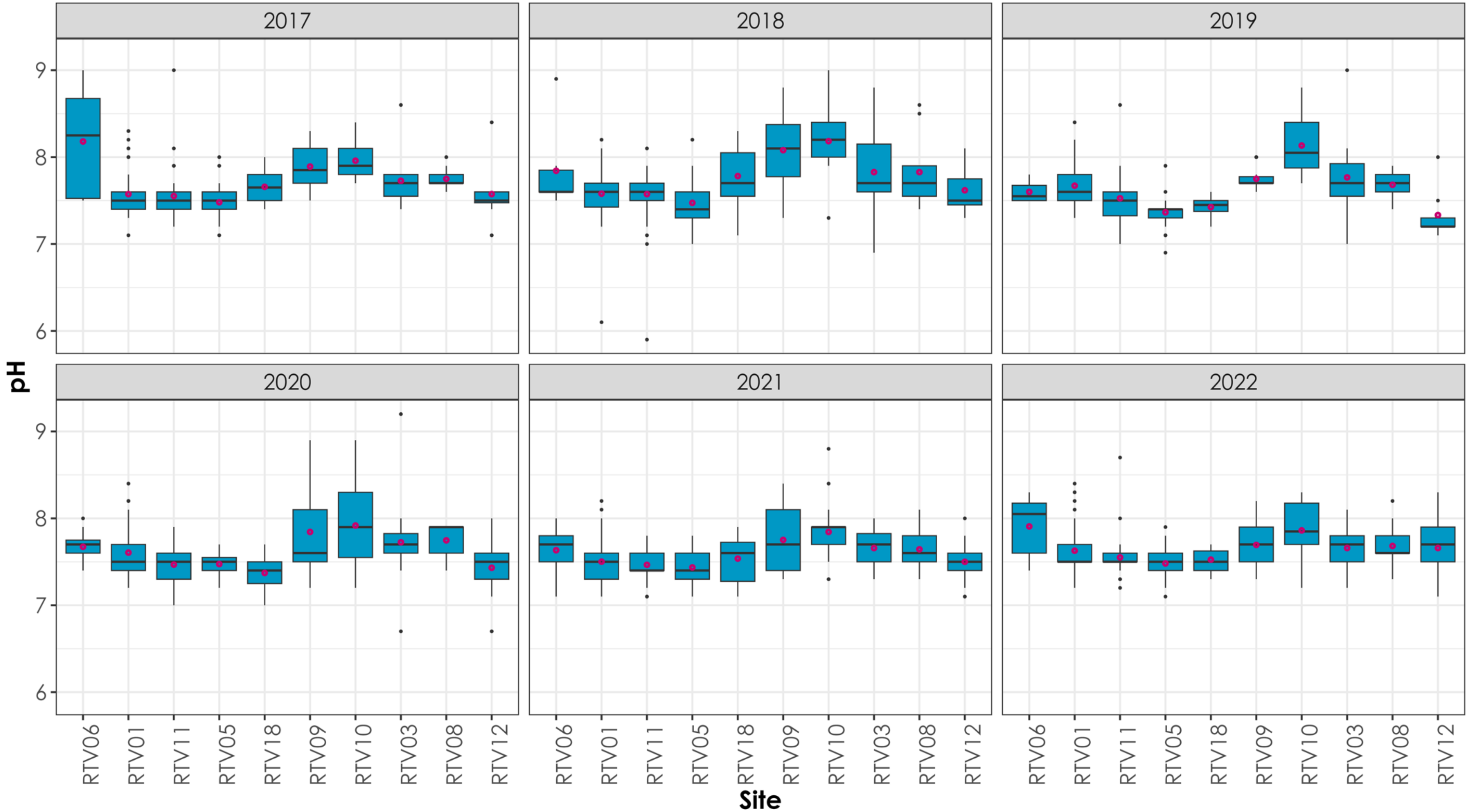
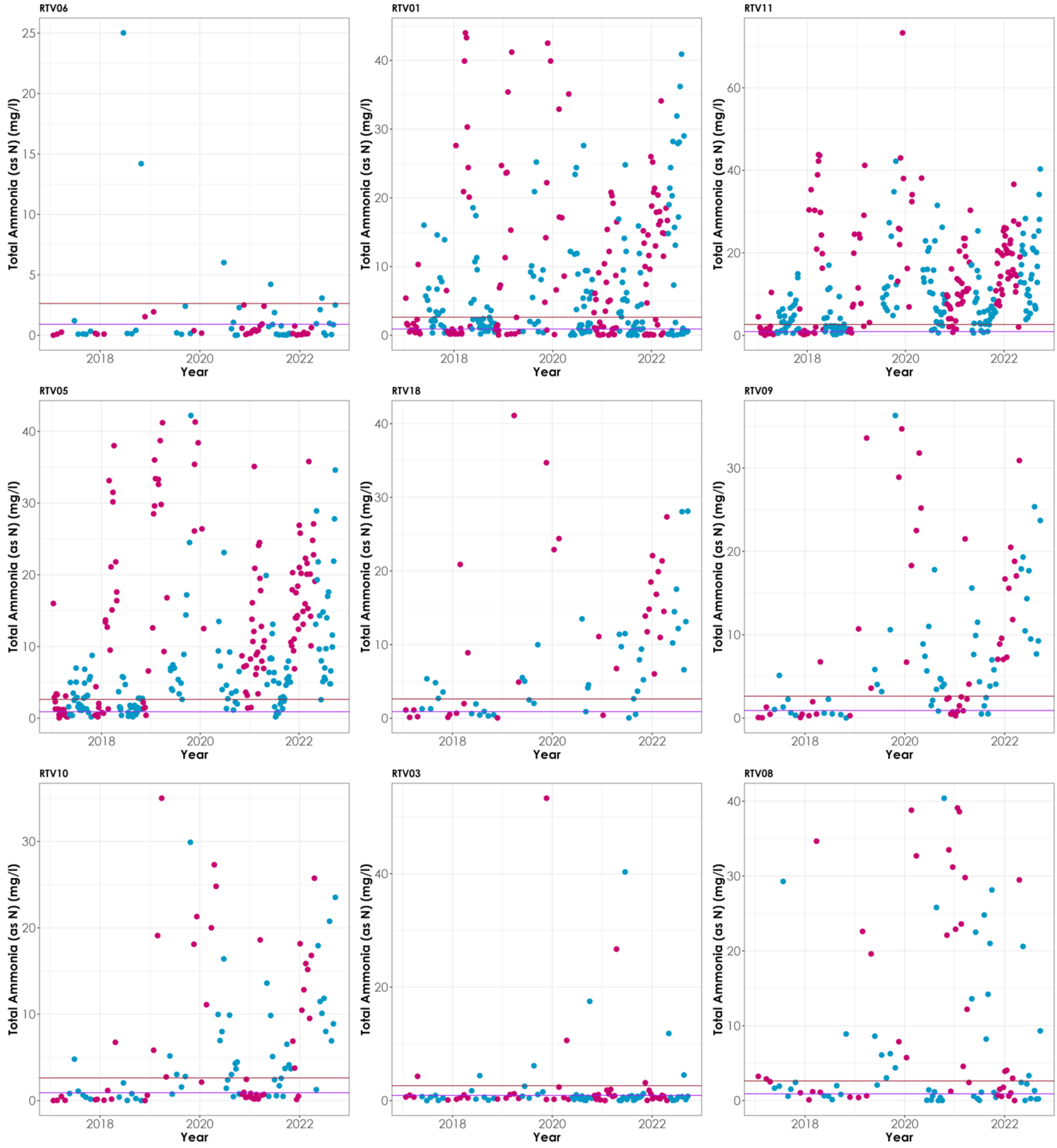


Figure 2-20. Summary data showing annual mean, median, range and outliers in pH measured at key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Sites listed in order from upstream to downstream. Sites RTV12, RTV03 and RTV08 included on right hand side of the graphs are inflows into the main channel, via the Theo Marais channel. Note that 2022 annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December.

Total ammonia data for monitoring sites within the study area, interpreted with regard to likely pH maxima (Figure 2-21 and Figure 2-22) suggest that:

- There has been a significant increase in potential ammonia toxicity throughout the study area since 2017.
- Mean and median total ammonia concentrations increased markedly in 2018 at all sites between RTV06 and RTV05 – thereafter however, total ammonia concentration decreased consistently at the upstream site (RTV06) but median, mean and inter-quartile range values increased markedly compared to 2017 values, and generally lay within the range of acute toxicity at pH 8 and 8.5.
- These data suggest that ammonia concentrations downstream of RTV06 were frequently in the range associated with acute toxicity to aquatic organisms (DWAF 1996) and thus likely to have contributed to overall ecosystem degradation downstream. This is largely as a result, it is assumed, of the high and un-met chemical and biological oxygen demand of the system through these reaches.
- High levels of total ammonia were periodically recorded from the upper Theo Marais channel (RTV08) – these are assumed to relate mainly to sewage overflows from the Koeberg pump station, but could also reflect episodic (illegal) polluted inflows from the upstream catchment.



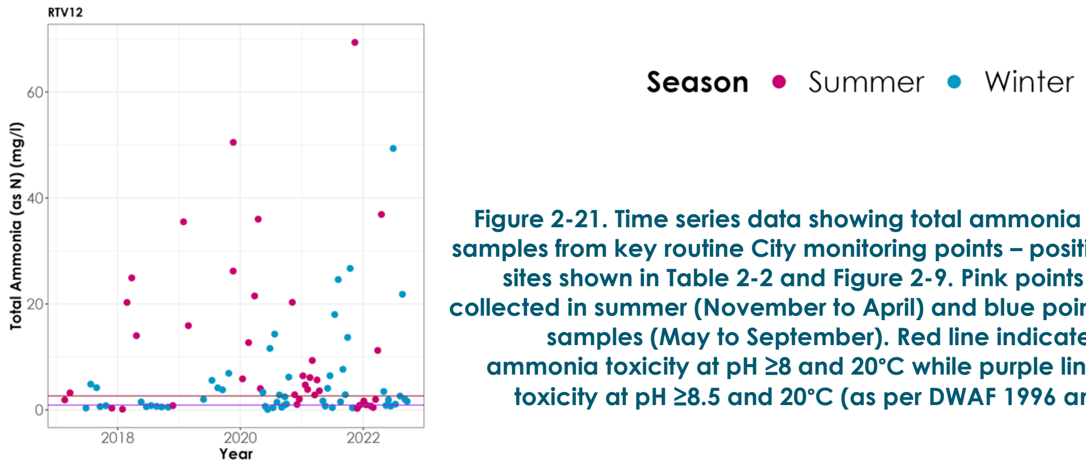


Figure 2-21. Time series data showing total ammonia concentrations in samples from key routine City monitoring points – positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September). Red line indicates assumed acute ammonia toxicity at pH ≥8 and 20°C while purple line indicates acute toxicity at pH ≥8.5 and 20°C (as per DWAF 1996 and Day et al 2020)

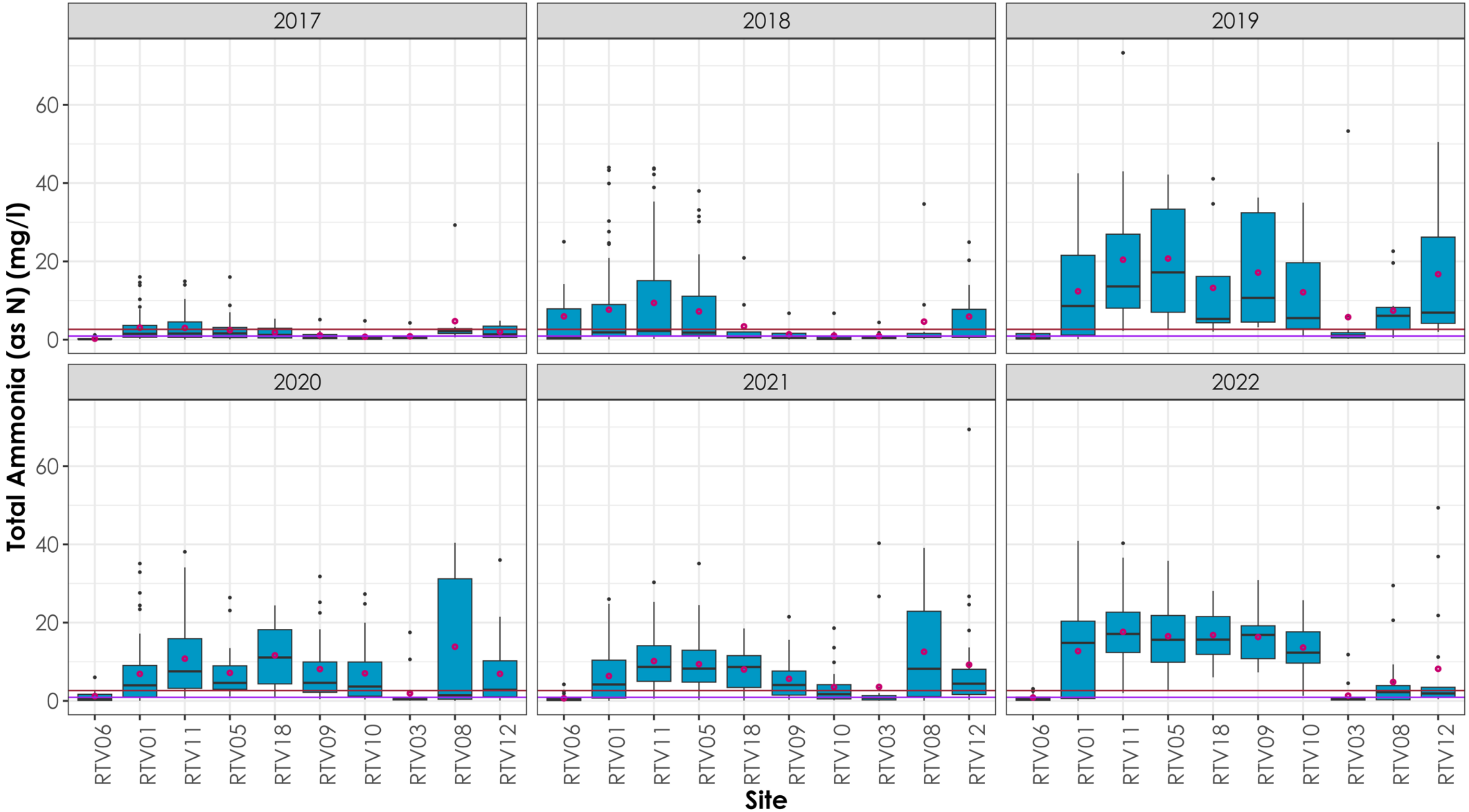


Figure 2-22. Summary data showing the annual mean, median, range and outliers in total ammonia concentrations in samples from key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September and are arguably best-case data, as they exclude the drier summer months of October to December. Red line (total ammonia concentration 2.63 mg N/l) indicates assumed acute ammonia toxicity at pH ≥8 and 20°C while lower purple line (total ammonia concentration 0.91 mg N/l) indicates acute toxicity at pH ≥8.5 and 20°C (as per DWAF 1996 and Day et al 2020)

2.5.6.4 *Escherichia coli* bacteria as indicators of human health issues

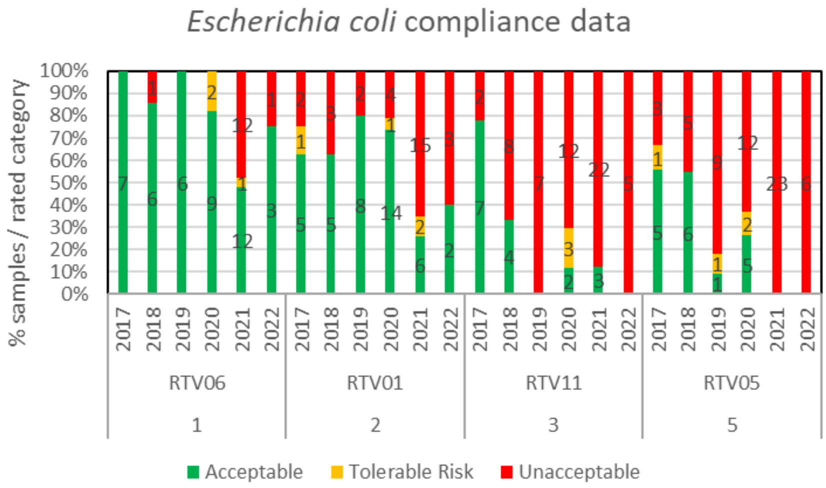
Escherichia coli (*E*. coli) data are presented in Figure 2-23 as compliance data (with regard to the City's rating ranges as presented in Day et al [2020]) rather than as standard box plots or time series data. This is because of data reporting issues, with the City's data being reported as "greater than" a wide range of values, from ">100" to "> 1 000 000" cfu / 100 ml (colony-forming units/100ml). This means that it is difficult to work numerically with the data. Instead, any data reflected as ">" have been relegated to the City's "Unacceptable" category for Intermediate Recreational Use, as specified in its Inland Water Quality Reporting schedules (e.g. Day et al 2020). In most cases, the ">" data are stated above this threshold, but where it lies below the threshold, it has conservatively been assumed that it could lie well above. In addition, and prompting the above conservative approach, there have been concerns over data accuracy, affecting *E. coli* data for the period from at least April 2021 to January 2022. These data have been discarded in this study.

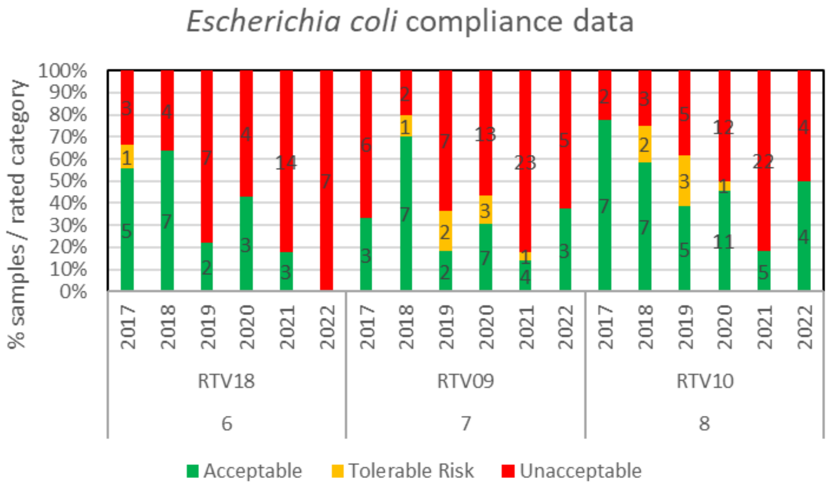
The data presented in Figure 2-23 are thus shown as compliance-type data, which reflect the proportion of samples that fall within and outside of limits of acceptability, as outlined in the City's Inland Water Quality Reporting guidelines of Day et al (2020). Note that these limits are intended to indicate human health risk for intermediate recreational use – in the current context, it is clear that not all of the water bodies evaluated (e.g. Theo Marais channel) are intended for such uses. They do however impact on downstream waters.

The data suggest:

- A general deterioration in water quality at all sites monitored over the past six years – this is indicated by a decreasing proportion of compliant data (i.e. data within the Acceptable range) – this indicates increased sources of faecal contamination;
- Decreased compliance at RTV06 upstream in the 2021/ 2022 period is assumed to stem from livestock inputs in the agricultural areas upstream, rather than from human sewage sources, although this is speculative;
- The significant decrease in compliance at RTV01 downstream over the same time period is however assumed to reflect in part inflows from RTV06 as well as a substantial increase in the extent of informal settlements within the Diep and Kleine Stink River floodplains over this period;
- Increased proportion of *E. coli* concentrations lying within the Unacceptable range ;
- Compliance at RTV11 and RTV05 also decreased significantly since 2017, particularly in 2021/ 2022 (although it is noted that few samples were useable for this period and sample size is low) – nevertheless, non-compliant inflows from the WWTW appear to be an issue, despite the chlorination or UV disinfection there;
- Data for RTV18 also reflect poor and decreasing compliance – this site lies downstream of the Erica Road stormwater inlet, and its data reflect inflows of untreated sewage from informal settlements upstream;
- Generally improved compliance at RTV10 reflect both distance downstream of upstream sources of contamination as well as die-off of *E. coli* in more saline water – other components of raw sewage are however likely to have persisted;
- Of the three sites passing into and along the Theo Marais channel, RTV08 (Theo Marais channel upstream of Duikersvlei) showed higher levels of compliance than the downstream reaches, albeit a major increase in the proportion of non-compliant data in the last three years – main sources of sewage into this canal are likely to be from the frequently overflowing Koeberg pump station nearby (Mr Ben De Wet; City of Cape Town; pers. comm. to Liz Day);

- Poor compliance was shown in the Duikersvlei channel – flows from this channel are generally much lower than from the Theo Marais channel, and downstream loading is thus likely to be low;
- The lower reaches of the Theo Marais channel performed poorly and on a decreasing trajectory with regard to compliance with *E. coli* thresholds – water quality in this channel is affected by upstream flows from RTV03 and RTV08 but also from stormwater and overflows from the WWTW to the north, which pass through an extensive reedbed, understood to be contaminated by years of receipt of waste water and other effluent. This is likely to exert considerable influence on the quality of water discharged through it – effluent was being discharged into the channel from this reedbed during three of four site visits carried out to this area between November 2022 and February 2023, under dry conditions.





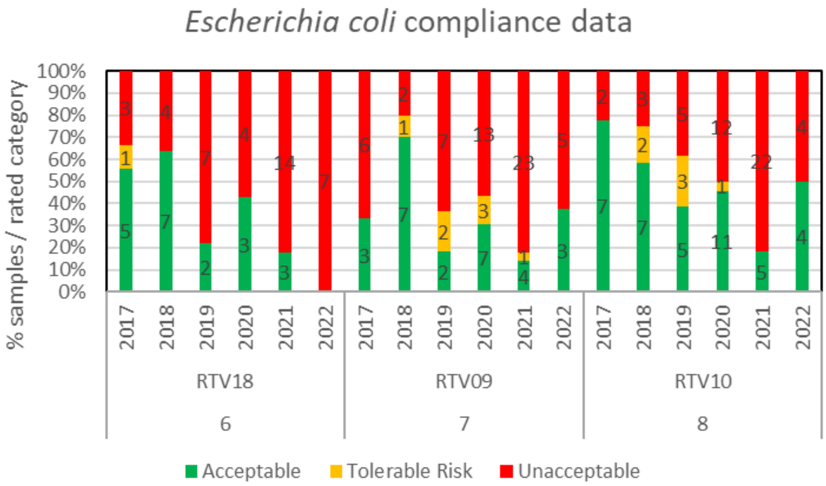


Figure 2-23. Summary data for *E. coli*, presented as compliance data with regard to the City's thresholds of acceptability for safe intermediate contact recreation. Threshold values set in Day et al (2020) as follows: Acceptable: ≤2500 cfu; Tolerable Risk: >2500 to ≤4000 cfu; Unacceptable Risk: >4000 cfu. Data exclude data between April 2021 and January 2022. Sites presented in order from upstream to downstream, with inflows downstream of RTV11 from the Theo Marais Channel presented separately (RTV03; RTV08 and RTV12)

2.5.6.5 Electrical conductivity

Electrical conductivity (EC) data are important in that they can reflect both natural seasonal water quality cycles in the river and provide a surrogate indicator for salinity in the estuary – that is, in Milnerton Lagoon downstream of Otto Du Plessis Drive. The lagoon has a long history of receipt of treated effluent from the Potsdam WWTW, which has *inter alia* resulted in long-term freshening of the estuary, contributing to a significant decline in its value as a nursery for marine fish and habitat for once-abundant estuarine fauna such as sand prawns, even prior to the more recent and current other water quality issues affecting the lagoon (Peak Practice 2008; Cerfonteyn and Day 2010; Infinity & City of Cape Town 2022).

The Diep River EMP (2022) sets various water quality Resource Quality Objectives (RQOs) for the system, including an RQO for salinity in Milnerton Lagoon as follows: Average salinity=20 ppt (≈ EC 3200 mS/m) up to a maximum of 35 ppt (≈ EC 5300 mS/m). The data shown in Figure 2-24 and Figure 2-25 and suggest the following:

- Summer time EC levels in the Diep River in the upper reaches of the EFZ (RTV06) showed a pattern of elevated salinity, rising well into the range of a brackish river system (± 200 – 1700 mS/m) during these low flow conditions. This pattern was particularly strong in 2017, during the regional drought.
- RTV01 also showed seasonal variability in EC, but with reduced summer / winter range – this is assumed to reflect the increasing influence of inflows of grey water runoff and sewage inflows from expanding unserviced or poorly serviced informal settlements upstream, as well as inflows at times from industrial effluent discharged into the stormwater system (see Cerfonteyn and Day 2010).
- With distance downstream, seasonal changes in EC became less evident, presumably because they are drowned by perennial inflows from the WWTW, which comprises almost the entire run of river flow into the estuary in summer.
- High variability in data at RTV18; RTV09; and RTV10 are likely to reflect tidal range rather than seasonal trends.
- While EC levels are clearly elevated in Milnerton Lagoon compared to the upstream river reaches, mean EC falls well short of the RQOs for this variable, as seen in summary data in Figure 2-25.
- Summary annual EC data also highlight the fact that salinity increases in Milnerton Lagoon downstream of RTV05, with marked increases at RTV09 and RTV10. These data support suggestions made elsewhere in this report that the high levels of organic sediment accumulation in the lagoon in the vicinity of the Woodbridge Island bridge, are driven by the saline / freshwater interface in this area.

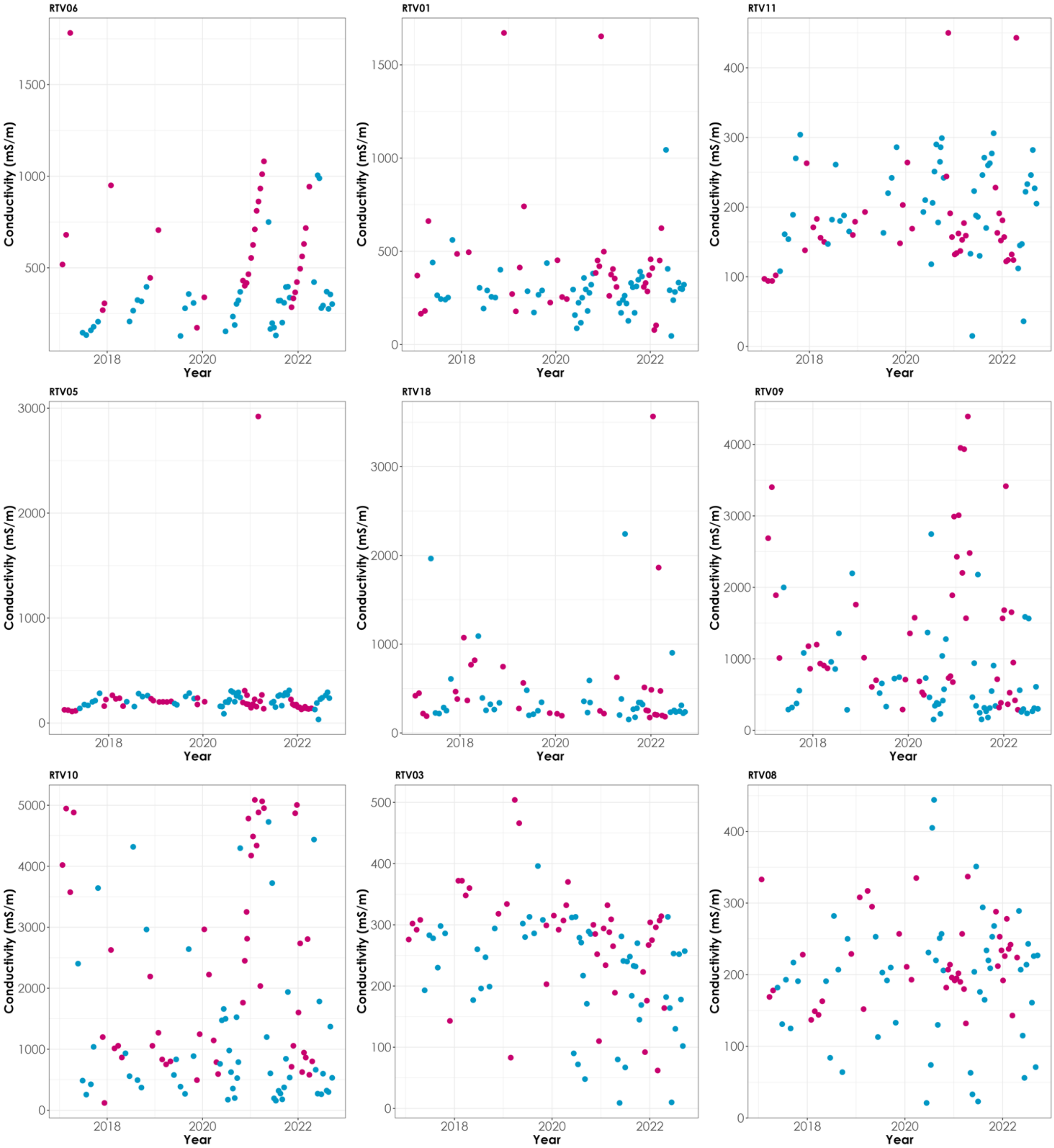


Figure 2-24. Time series data showing electrical conductivity (EC) data in samples from key routine City monitoring points. Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Pink points indicate samples collected in summer (November to April) and blue points indicate winter samples (May to September).

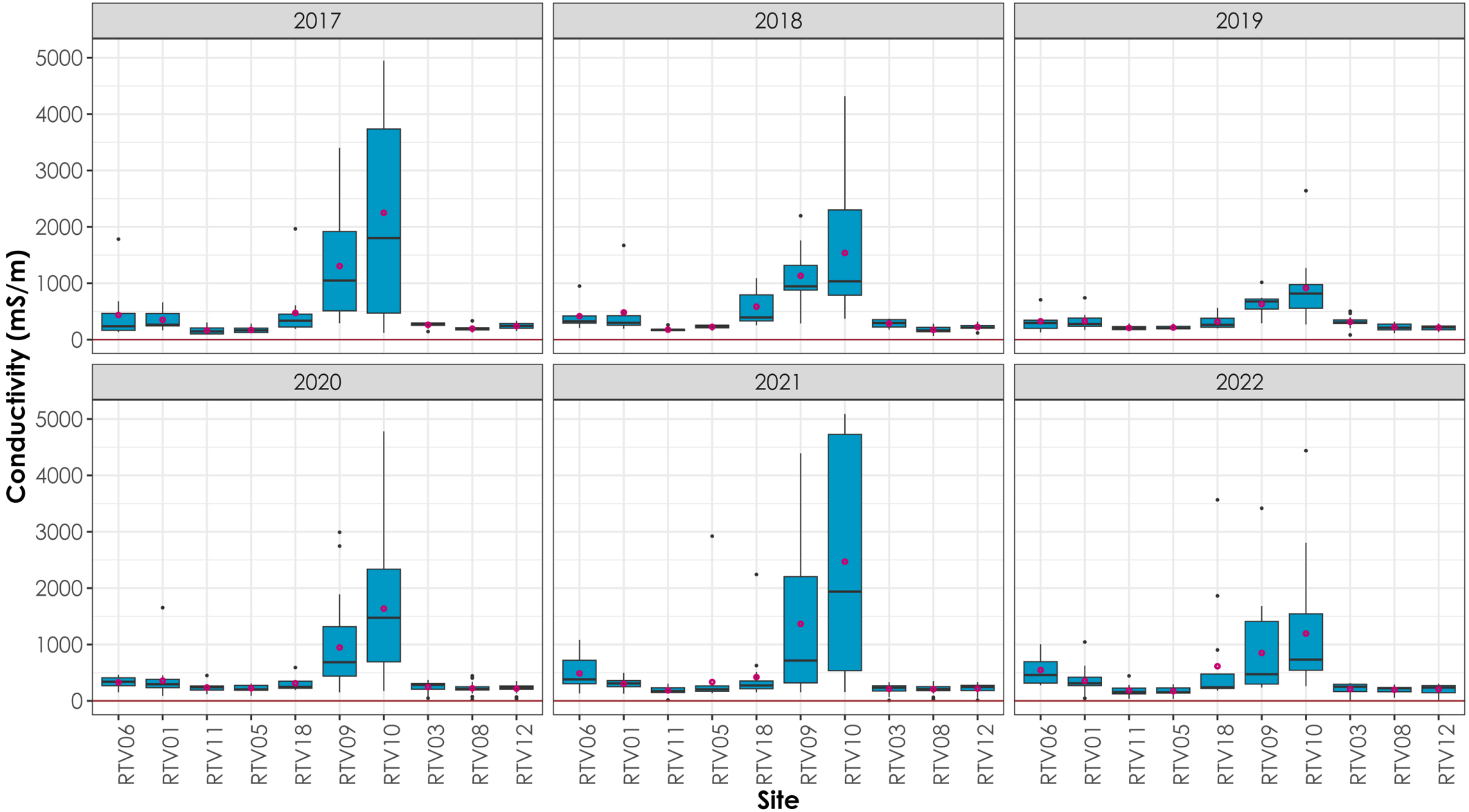


Figure 2-25. Summary data showing the annual mean, median, range and outliers in electrical conductivity data for samples from key routine City monitoring points (see Box 2-1). Positions of monitoring sites shown in Table 2-2 and Figure 2-9. Sites listed in order from upstream to downstream – sites RTV12, RTV03 and RTV08 included on right hand side of the graphs – these are inflows into the main channel, via the Theo Marais Channel. Note that 2022 mean annual data extend only to end of September – this means that they exclude the drier summer months of October to December, and may thus be artificially lower in this year.

2.5.7 Findings of the ad hoc November 2022 assessment in present study

Table 2-3 presents summary data collected during the once-off site walk-over on 24 November 2022. Site locations are indicated in Figure 2-9. Additional sampling focused on Milnerton Lagoon itself, where data were collected hourly through a full spring tide cycle, to inform the water quality model presented in Section 13 and the aeration proposal presented in section 6.

Bearing in mind the limitations of once-off data only, the data in Table 2-3, interpreted in conjunction with the photographic illustrations in Table 2-4, provide the following supplementary information over and above that suggested by long term data already presented:

- Copper (Cu), lead (Pb) and zinc (Zn) are at concentrations of concern in watercourses and inflows in several parts of the study area, with copper at D1 potentially derived from upstream agricultural practices (RTV06), while zinc was probably linked to inflows from urban land use.
- Orthophosphate concentrations were very high in all of the river channels, and assumed to be associated with high levels of sewage effluent as well as other catchment inputs –stormwater from more industrial areas (TM-UP and D3), less likely to be affected by human settlements, had lower but still elevated concentrations – note that the lower threshold for hypertrophic rivers is as low as 0.125 mg P/ L (see Day et al 2020).
- TSS was elevated at all sites with evident high organic composition.
- Water quality in the linked Milky Way / Erica Road stormwater system was highly polluted with regard to all assessed variables, although concentrations were lower than in the treated final effluent sample for orthophosphate; total ammonia; and TIN; and in the case of the Milky Way site upstream, for TSS as well. COD and DO concentrations were both worse in the stormwater discharges than the WWTW effluent. A deterioration in most water quality variables between the upstream and downstream site suggests further inflows of polluted sources downstream of Phoenix as well. Loading of Milnerton Lagoon from these sources is however likely to be well below that resulting from the 2022 WWTW treated effluent discharges.
- Total ammonia concentrations were very high at all sites except the two industrial effluent streams (D3 and TM-UP) and total ammonia comprised all or nearly all of the measured TIN in all cases (barring the above two sites). This reflects the very high chemical (and presumably biological) oxygen demand in the system. COD values were also very high and the **aquatic** ecosystems clearly did not have the capacity to deal with the organic loading present at this time, through normal oxygen-demanding nitrification. If sustained, such a scenario would result in the build-up of (undecomposed) anoxic organic sediments in areas where flow is slow enough for sediments to settle. This would also lead to the production of gases such as hydrogen sulphide and methane under anaerobic conditions – the former being associated with foul smells. All these conditions were apparent in the Milnerton Lagoon at the time of the assessment.

Hydrogen sulphide and sulphur cycles

Sewage effluent usually includes significant sulphur in various forms. Under aerobic conditions, organic waste is broken down to form sulphates, which can be taken up by plants. Under anoxic conditions, decomposition results in the formation of hydrogen sulphide (H_2S), which is an odorous gas. Kadlec (2020) notes that other odoriferous volatile forms of sulphur that are common in sewage effluent are methanethiol (CH3SH) and dimethyl sulphide ($(CH_3)_2S$). All of these could contribute to strong odours under anaerobic conditions. See section 2.6.4 for further details.

- Finally, the quality of the final effluent at the time of sampling (until Sep 2022) is also of relevance to discussions in this document:

- Table 2-3 shows that the final effluent lay far above General Effluent Limits for most assessed variables, with the exception of orthophosphate and pH, for both of which generous limits have been set. In particular, concentrations of total ammonia; total suspended solids (TSS); and COD were all nearly or at least an order of magnitude above the General Limits. During summer, treated WWTW effluent discharges most of the flow into the Milnerton Lagoon. **This has significant implications for estuarine condition**. The issue is unpacked further in Section 3.2 in light of the City's long-term effluent data and loading calculations.
- Elevated TSS in final effluent water has a secondary implication which is also of great concern for downstream aquatic ecosystems. Potsdam WWTW has been designed to disinfect its final effluent with UV radiation. This is intended to kill or render impotent harmful pathogens including *E. coli* bacteria. Under conditions of high TSS, which do not allow effective UV penetration, the plant has had to change its disinfection process and was, at the time of both sampling and of writing this report, disinfecting by chlorination. Chlorine and ammonia in water can form one of three forms of chloramine, which are persistent chemicals with potentially high levels of toxicity to aquatic organisms. The high concentrations of total ammonia in both treated effluent and in the bypass channel itself suggest that it would be very susceptible to chloramine formation and associated toxicity – note that the City does not currently test for chloramines.
- A second important aspect highlighted by the data is the difference in water quality between the sample upstream of the maturation ponds (WWTW) and the final effluent sample. The upstream sample was at least compliant with General Effluent limits with regard to COD, before it was passed into the maturation ponds. The once-off comparative data show that the concentrations of all measured variables **increased** after passage through the maturation ponds. Conversations with the plant management team indicated that contamination of the ponds occurred in early 2022, and they have been used in this state since then, pending availability of a pump to allow the ponds to be bypassed while they are cleaned. Cleaning of the ponds began only at the end of March 2023, and is scheduled for completion by the end of June 2023.

Site	Location	Cd (mg/L) [ppm]	Cu (mg/L) [ppm]	Pb (mg/L) [ppm]	Zn (mg/L) [ppm]	P (mg/L) [ppm]	TSS (mg/L) [ppm]	NH3 as N (mg/L) [ppm]	TIN (mg/L) [ppm]	COD (mg/L) [ppm]	DO (mg/L) [ppm]	EC (mS/m)	pH
Diep1	Diep River Upstream Kleine Stink	<0.05	0.06	<0.05	<0.05	6.27	313	30.36	30.36	151	2.39	282	7.6
Diep2	Diep River Downstream of Kleine Stink	<0.05	<0.05	<0.05	<0.05	4.99	540	66.66	66.66	167	2.27	192	7.9
KS	Kleine Stink in Dunoon	<0.05	<0.05	<0.05	<0.05	5.02	263	48.75	48.75	232	1.9	163	7.4
Diep3	Outlet at Killarney refuse site	<0.05	0.08	<0.05	0.51	0.23	33	0.31	2.62	13	5.62	33	9.3
Milky Way	Stormwater channel at Milky Way - upstream of Erica Road	<0.05	<0.05	<0.05	<0.05	3.9	303	43.1	43.1	311	0.96	129	7.4
Erica Road	Stormwater outfall into lagoon - downstream of Milky Way	<0.05	<0.05	<0.05	0.29	5.5	430	52.77	52.77	341	0.95	155	7.6
TM-UP	Theo Marais channel upstream of reedbed inflow	<0.05	0.06	<0.05	0.2	0.44	160	0.43	1.57	31	3.59	125	7.5
TM-DOWN	Theo Marais channel downstream of reedbed inflow	<0.05	<0.05	<0.05	0.19	5.46	207	38.54	39.27	190	1.61	148	7.1
WWTW1	Sump / outlet at P8 (not final effluent)	<0.05	<0.05	<0.05	0.06	3.74	300	67.01	67.01	42	4.1	154	7.6
WWTW2	Downstream Final effluent	<0.05	0.1	<0.05	0.17	8.18	390	74.72	74.72	188	1.5	190	7.2
DWS General effluent limits		0.005	0.01	0.01	0.1	10	25	3	18	75	-	-	5.5-9.5
DWAF (1996) Acute toxicity thresholds		*0.003-0.013	*0.0016-0.012	*0.004-0.016	0.036	-	-	-	-	-	-	-	-
COLOUR CODING:		Detection limit above toxicity threshold				Above acute toxicity or effluent limit					Below acute toxicity or effluent limit		

Table 2-3. Results of *ad hoc* water quality sampling in November 2022. Site locations as indicated in Figure 2-9 DWS General Effluent Limits for WWTW discharges are included in the table, to assist in interpretation of final effluent quality at the time of sampling. DWAF (1996) aquatic ecosystem toxicity thresholds are also included for interpretation of watercourse data.

Table 2-4. Photo illustrations of different parts of the EFZ relevant to water quality issues – photos as of November 2022

<table>
<tr><td>
Photo 2-1. Pedestrian crossing over permanent main channel of the Diep River (site Diep 1) upstream of main inflows from Du Noon and associated informal settlements - note dwellings on floodplain in background</td><td>
Photo 2-2. Ponding of water on Diep River floodplain – water collects downstream of open drains from the informal settlements</td></tr>
<tr><td>
Photo 2-3. One of the Kleine Stink channels through Du Noon</td><td>
Photo 2-4. Ponded water in main channel of Diep River downstream of Du Noon at railway bridge – no discernible flow from here</td></tr>
<tr><td>
Photo 2-5. Stormwater outlet into Diep River near Killarney refuse site (Diep 3)</td><td>
Photo 2-6. Potsdam WWTW showing outlet into bypass channel (circled)</td></tr>
</table>


Photo 2-7. Turbid water in bypass channel downstream of Potsdam WWTW outlet


Photo 2-8. Pipe overflow inlet from WWTW reedbeds into Theo Marais channel


Photo 2-9. Detention pond upstream of sewage pump station on Milky Way, in Phoenix, downstream of Jo Slovo Park (a suburb with high levels of informal backyard settlements within formal housing)


Photo 2-10. Open stormwater channel downstream of Milky Way culverts – this water passes out via the Erica Road culvert downstream


Photo 2-11. Polluted stormwater lowflows are pumped out of the open outfall at the Erica Road culvert and discharged into the adjacent sewer manhole


Photo 2-12. Erica Road culvert showing sand bags to assist in over-pumping to the adjacent sewer

2.5.8 WWTW effluent water quality and loading

Effluent discharge and water quality data for the WWTW were provided by the City and used to assess changes in the quality of effluent and the loading of key variables into the downstream aquatic ecosystem (the bypass channel and Milnerton Lagoon). These are presented in Figure 2-26 and Figure 2-27. The data indicate:

- Up until 2019, the WWTW largely met General Effluent Limits for key variables shown in Figure 2-26
- From 2019 onwards, there was a clear deterioration in final effluent quality, with elevated total ammonia, COD, orthophosphate and TSS.
- Over this period, the proportion of ammonia in TIN also rose substantially, indicating a deterioration in WWTW processes that facilitate nitrification – this would be linked to the elevated COD levels.
- In 2022, there was a dramatic reduction in final effluent quality, with clear increases in COD, TSS and total ammonia – well above compliance levels for three variables.
- Orthophosphate concentrations also increased, although they largely fell within General Limits. The latter are however very generous limits, and allow for orthophosphate concentrations two orders of magnitude (i.e. one hundred times) greater than the City's threshold of Unacceptability from an ecological perspective.
- It should be noted that General Effluent limits assume a degree of dilution of effluent sources in downstream water resources. In the case of Milnerton Lagoon, during at least summer, WWTW effluent in fact comprises the majority of flow into the lagoon with the remainder of the flow arising from raw sewage and greywater inflows.

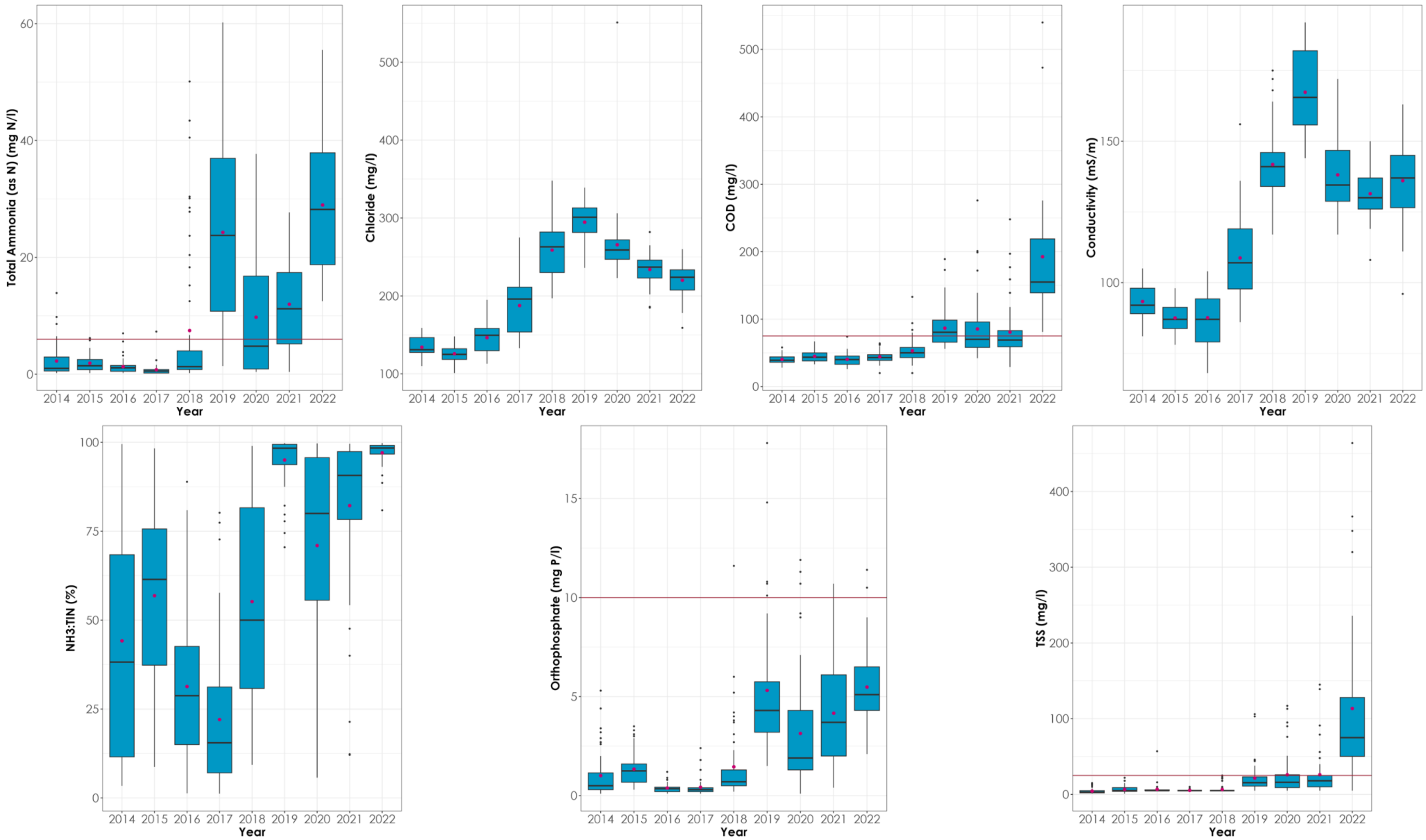


Figure 2-26. Summary data showing the annual mean, median, range and outliers for a range of water quality variables measured in final effluent from the Potsdam WWTW (site code: "Potsdam Final Effluent Combined"). Red lines indicate General Effluent Limits for different variables.

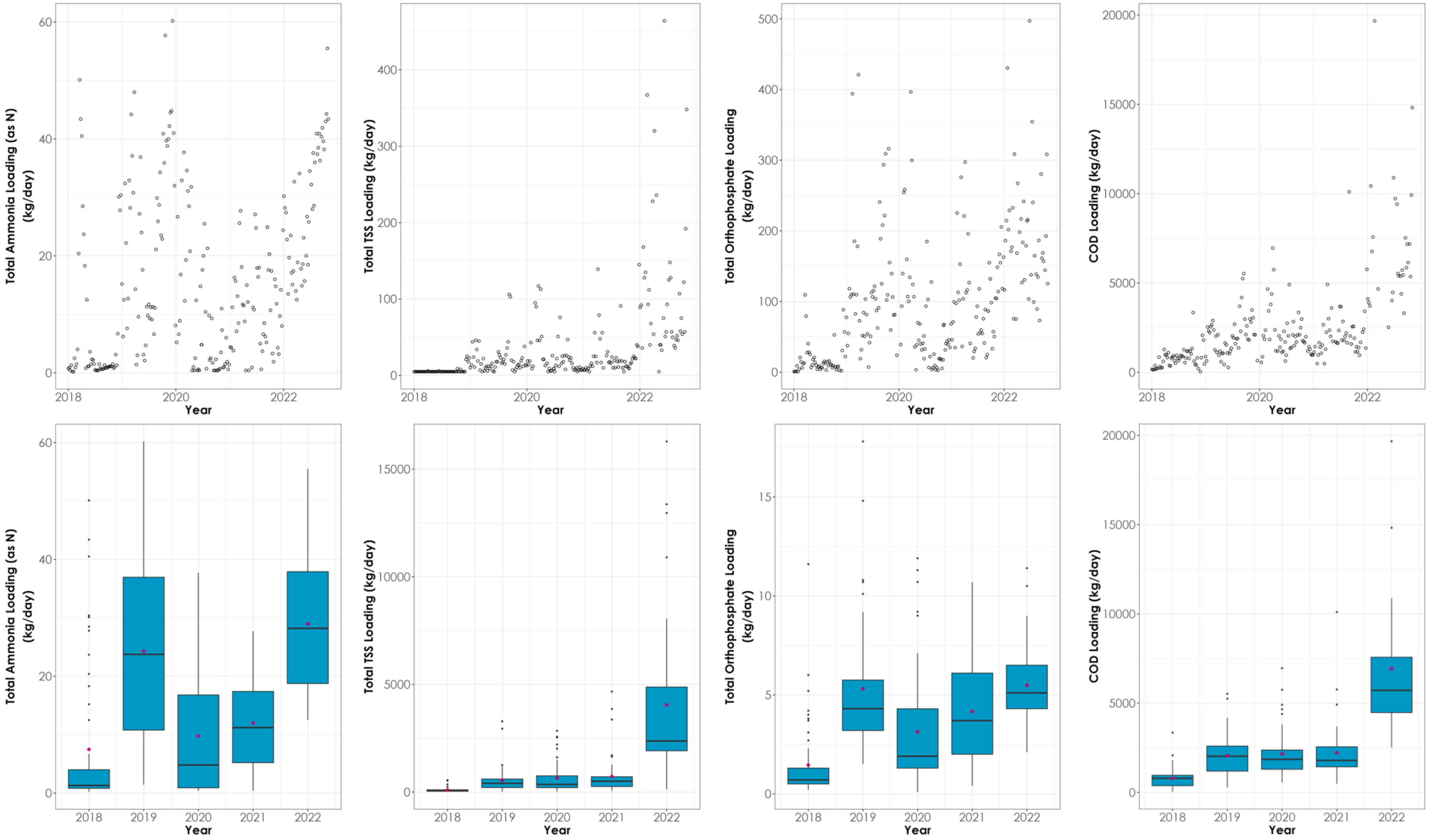


Figure 2-27. Summary data showing the annual mean, median, range and outliers for a range of water quality variables measured in final effluent from the Potsdam WWTW (site code: "Potsdam Final Effluent Combined"), and time-series data for the same period.

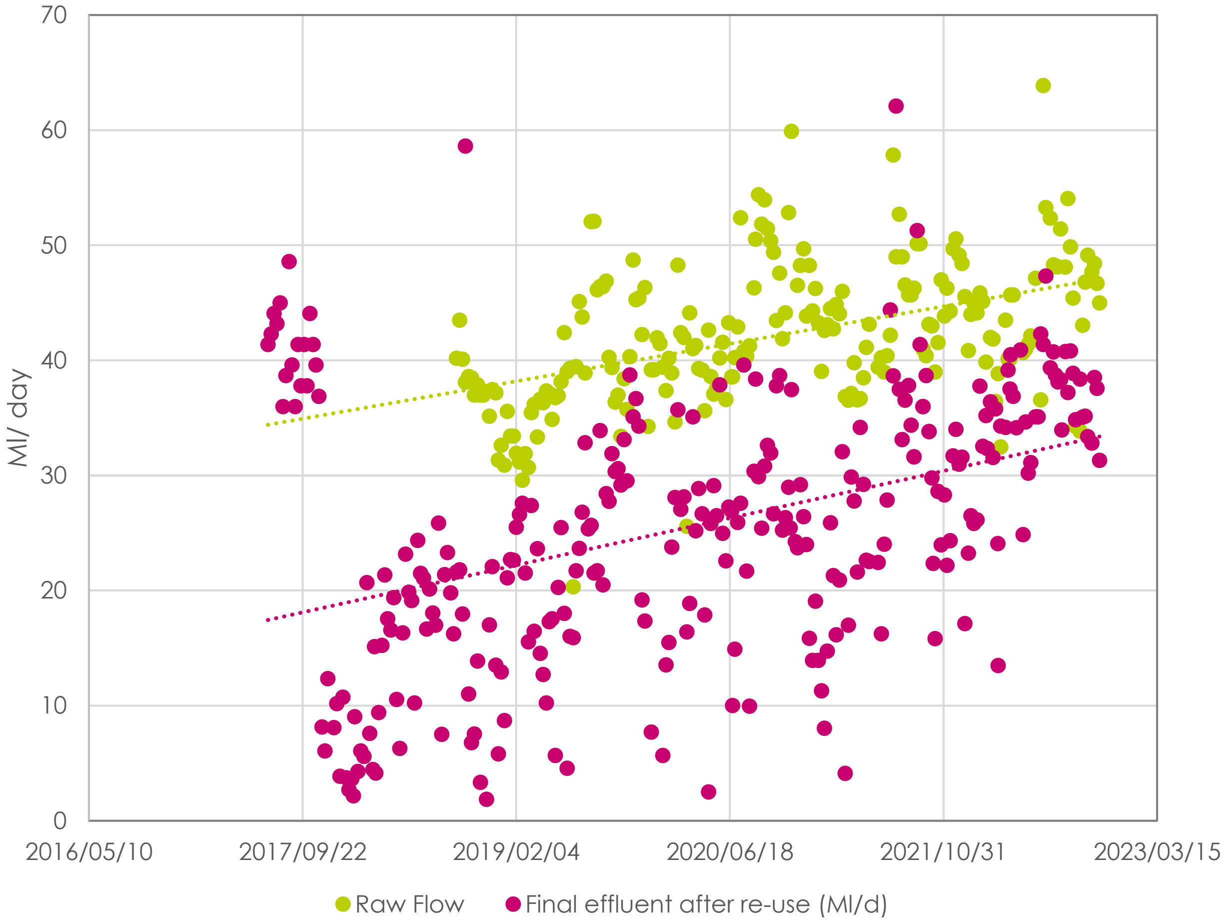


Figure 2-28. Final effluent discharge volumes from the Potsdam WWTW (site code: "Potsdam Final Effluent Combined"). Graph compares total raw flows with final effluent after abstraction of effluent for re-use. Final effluent volumes are calculated / estimated and are not metered. Data supplied by City of Cape Town.

Figure 2-27 shows loading data for the WWTW final effluent, calculated from final effluent flows, after allowance for diversion of some treated effluent for re-use (Figure 2-28). Data were available from July 2017 only (effluent after re-use) and September 2019. These data together allow the following conclusions to be drawn:

» Raw volumes have increased since 2017, with mean annual raw between July 2021 and end of June 2022 calculated as 40.1 ML/ day (32.4 ML/day excluding re-use); compared with 32.6 and 19.5 ML/ day respectively in the 2017/2018 period;
» This means that even without a deterioration in effluent quality, nutrient loading of the downstream estuary would have increased over this period;
» Loading data (Figure 2-27) indicate significant increases in loading of TSS, total ammonia, orthophosphate and COD into the Milnerton Lagoon as a result of passage of residual treated effluent, after abstraction for re-use – these marked increases largely explain the recent observed deterioration in the ecological condition of Milnerton Lagoon and the associated odour problems, resulting from a high, un-met COD from a large, permanent organic load.

2.6 Sediment quality and characteristics

2.6.1 Previous assessments

Two comprehensive studies on sediment quality in the Milnerton Lagoon have been conducted by Anchor Environmental in recent years – one in 2010 (Hutchings & Clark) for the provincial government and one in 2021 (Gihwala et al.) for the City of Cape Town. These studies provide important context on sediment contamination levels in Milnerton Lagoon over the past 13 years.

The 2010 study assessed levels of trace metal contamination in both the sediment and biota of the Diep estuary. Twenty-five sample sites were used in this study, with biota including fish, crabs, polychaete worms and insect larvae being collected along with the sediment samples. These samples were analysed for the full spectrum of trace metals. The metal content of sediment was normalised against aluminium concentration in order to allow for spatial comparisons. Enrichment factors were also calculated for all eight heavy metals (Cd, Cu, Cr, Fe, Mn, Ni, Pb, and Zn), based on records from a study conducted in 1992.

This study concluded the trace metal content of sediments from the Diep estuary exceeded both South African and international sediment quality guidelines. This was not the case for all elements at all sites, but each sample site was found to exceed guidelines for at least one metal.

The 2021 monitoring survey of the Diep River Estuary found that the sediment texture was primarily coarse and sandy, with higher mud content at certain sites due to decreased hydrodynamic flow caused by biotic or anthropogenic obstructions. Total organic carbon (TOC) and total organic nitrogen (TON) levels and trace metal concentrations mirrored the patterns observed for mud content and accumulated in sediments further upstream in the estuary. Significant differences between the Milnerton Lagoon and Diep River areas were detected for five metals (Al, As, Cr, Fe, and Ni). The Diep River areas had statistically significant higher concentrations of these five metals than the lagoon. Only three elements did not exceed sediment quality guidelines (South African and international) (Cr, Pb, and Hg). Over the past 32 years, there were substantial increases in Cd, Ni, and Zn within the Diep River Estuary. Most trace metals in the sediments of the Diep River Estuary have become enriched compared to historical surveys, with Cd, Ni, and Zn concentrations relatively high compared to other local and international estuaries, which is concerning due to their known ecotoxicological effects.

The 2021 benthic macrofauna survey conducted in Milnerton Lagoon recorded a significant decline in diversity compared to historical reports (including the 2010 study), with only six different taxa. Trace metal pollution levels in the Diep River Estuary sediments were found to be elevated above toxic levels, particularly in areas past the Otto du Plessis Bridge and near the Blaauwberg Bridge, potentially caused by effluent from wastewater treatment works, stormwater, and industrial wastewater.

In summary, both of these reports definitively conclude that there are high levels of heavy metal contamination in the sludge layer of Milnerton lagoon.

2.6.2 Sediment sampling 2022

Limited new sediment sampling and analysis were undertaken as part of this study, results of which are presented in this section. Full results are presented in **Annexure A**. The intention was not to reproduce the previous, more in-depth sampling, but rather to inform the design of the proposed dredging remediation options, including disposal options.

2.6.2.1 Methodology

The sediment from the bottom of the lagoon and river channel was sampled in December 2022 to understand the current sediment quality and disposal options for dredging. Samples were collected by Tritan Survey (refer to section 1.4 and **Annexure B**). Tritan Survey utilised a weighted gravity corer consisting of an acrylic 50 mm pipe fitted into a stainless steel 75 mm diameter pipe with a 30 kg lead weight. Samples could be collected from the top and bottom of the core thus retrieved, allowing for differentiation of sediment quality at different depths in the sediment column. Particle size distribution and specific gravity tests were conducted by Immerteq and by Steyn & Wilson. Sludge classification samples were analysed by A. L. Abbott & Associates (Pty) Ltd.

Figure 2-29. Sediment sampling locations in Milnerton Lagoon (2022) as well as Hutchings and Clark (2010) samples sites for comparison.

2.6.2.2 Guidelines

South Africa currently does not have a framework of guidelines for sediment quality in aquatic systems, so alternate guidelines were used for comparison purp oses. The South African Solid Waste Standards were used in conjunction with two classes of the Australia New Zealand Sediment Toxicant Guideline framework to provide a baseline against which to compare sediment quality data. These sets of guidelines are summarised below:

1. **Australia New Zealand Sediment Toxicant Guidelines:** One of the few frameworks that clearly identifies differing levels of concern for heavy metal sediment contamination in natural systems.
    a. **Default Guideline Value (DGV):** DGVs indicate the concentrations below which there is minimal risk to environmental and human health.
    b. **Thresholds of major concern (GV-High):** GV-High indicate concentrations at which toxicity-related adverse effects to humans or the environment would likely be observed.
2. **Total Concentration Threshold (TCT):** TCT limits are prescribed in Norms and Standards for the Assessment of Waste for Landfill Disposal (GN. No. 36784 of 2013). Developed for solid waste management, the TCT limits may be applied by landfill sites prior to acceptance of dredged material for disposal.
    a. **TCT0:** Waste with contaminant concentrations below all TCT0 limits (Type 4 waste) can be disposed of in regular non-lined landfills (Class D). These limits are based on the South African Soil Screening Values for commercial/industrial land determined by the Department of Environmental Affairs' (DEA) "Framework for the Management of Contaminated Land" (March 2010) that are protective of water resources (SSV1) or alternatively the State of Victoria value for fill material.
    b. **TCT1:** Waste with contaminant concentrations below TCT1 (and above TCT0 limits) is considered Type 2 or 3 waste depending on the leachable content of contaminants and must be disposed of in a Class B or C landfill. Waste with concentrations above the TCT1 (and below TCT2 limits) is considered Type 1 waste and must be disposed of in a Class B landfill. TCT1 Limits are derived from the soil screening values for commercial/industrial land or the State of Victoria values were used.
    c. **TCT2:** Limits are calculated by multiplying the TCT1 value by a factor of four (as used by the State of Victoria EPA, Australia). Waste with concentrations above the TCT2 is considered Type 0 waste and must be disposed of in a Class A (i.e., hazardous waste) landfill.

2.6.3 Sludge classifications for disposal

To inform disposal options for the organic sediments, samples were submitted for classification in terms of the 2006 wastewater sludge classification system. Triplicate samples were analysed by A.L. Abbott & Associate, a SANAS accredited laboratory. The classification system includes microbiological parameters (faecal coliforms and helminth ova), physical and stability indicators, and chemical characteristics (nutrients, metals and organic pollutants).

Key findings are as follows:

2.6.3.1 Physical characteristics

pH of the sediment (Figure 2-30) was very slightly alkaline (generally between pH 7-7.5).

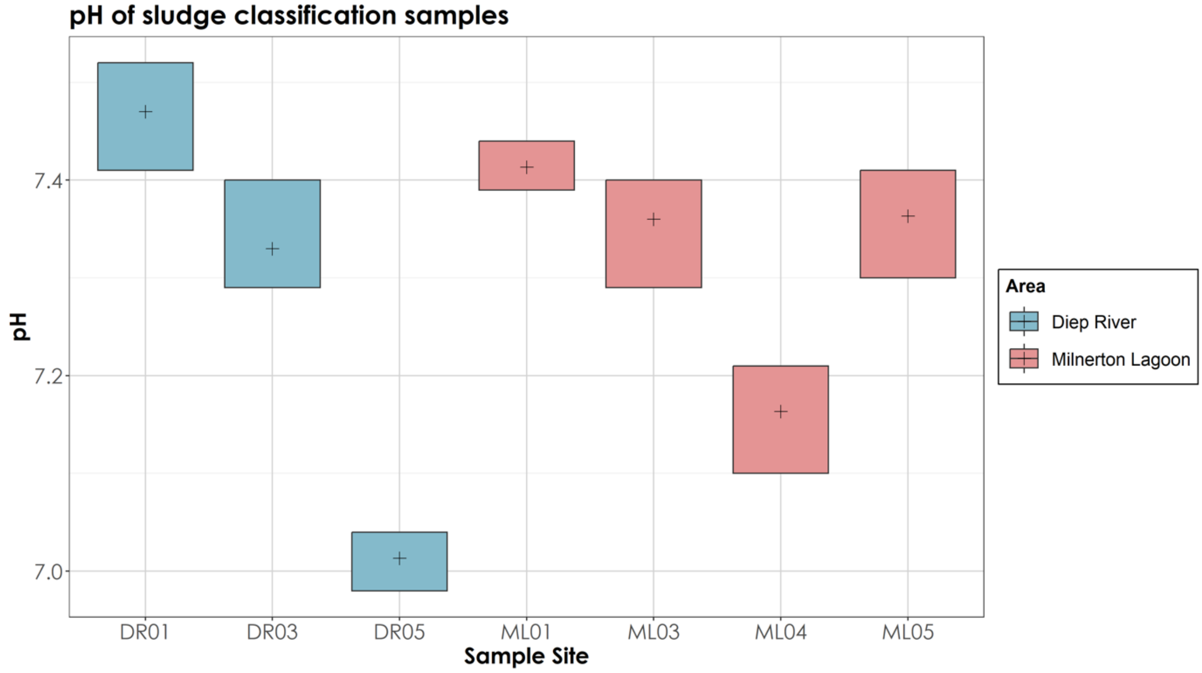


Figure 2-30. pH range of the sediment at each site. The mean value is indicated by the cross.

Faecal coliform bacteria were only found in the two out of the three Diep River sites (but in all three replicates at these two sites). Both sites were found to have faecal coliforms greatly in excess of the DGV limits (Figure 2-31).

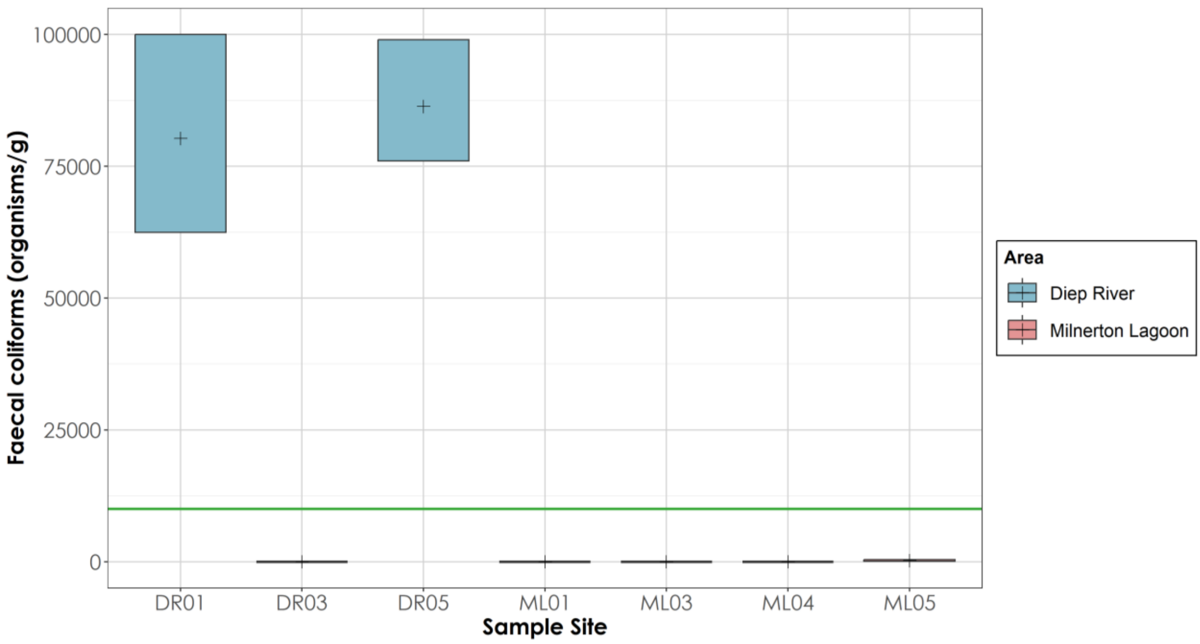


Figure 2-31. Faecal coliform content range of the sediment at each site. The mean value is indicated by the cross. The green line indicates the DGV limit.

2.6.3.2 Nutrients

The TCT and DGV guidelines do not specify limits for the nutrients that were tested (Total Kjeldahl Nitrogen, Total Phosphate, and Potassium).

Total Kjeldahl Nitrogen content (Figure 2-32) was consistently higher in the Diep River area than the Milnerton Lagoon area.

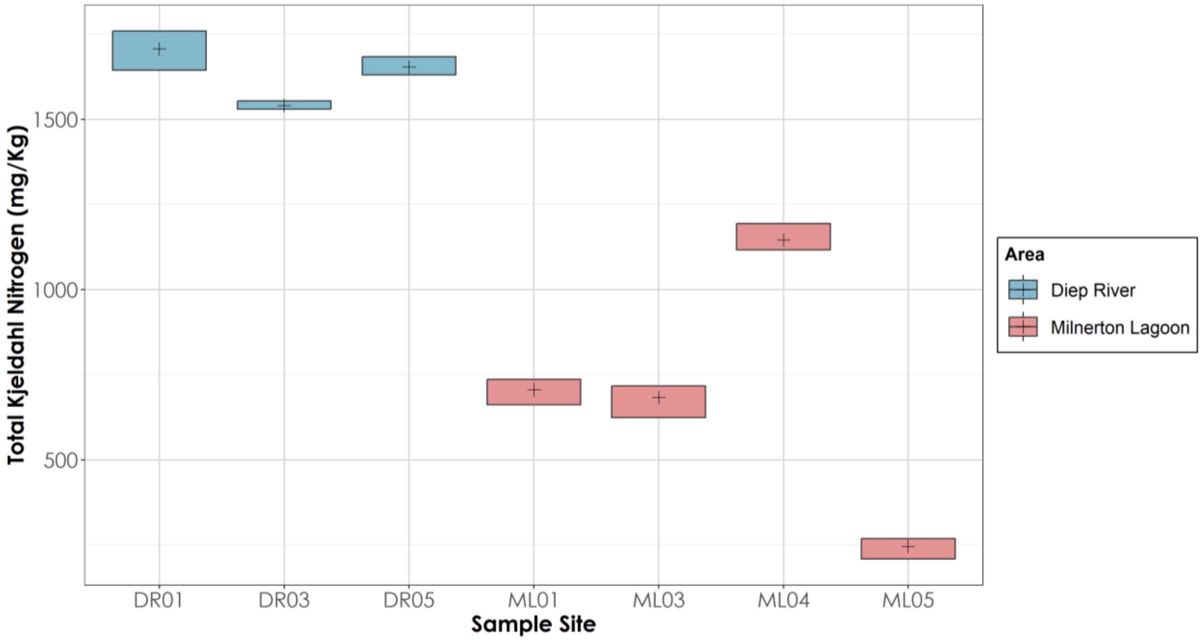


Figure 2-32. Range of total Kjeldahl nitrogen content of samples taken at each site. The mean value is indicated by the cross.

Total phosphate content (Figure 2-33) varied across the ML and DR sites. The range of concentrations was similar between the two areas.

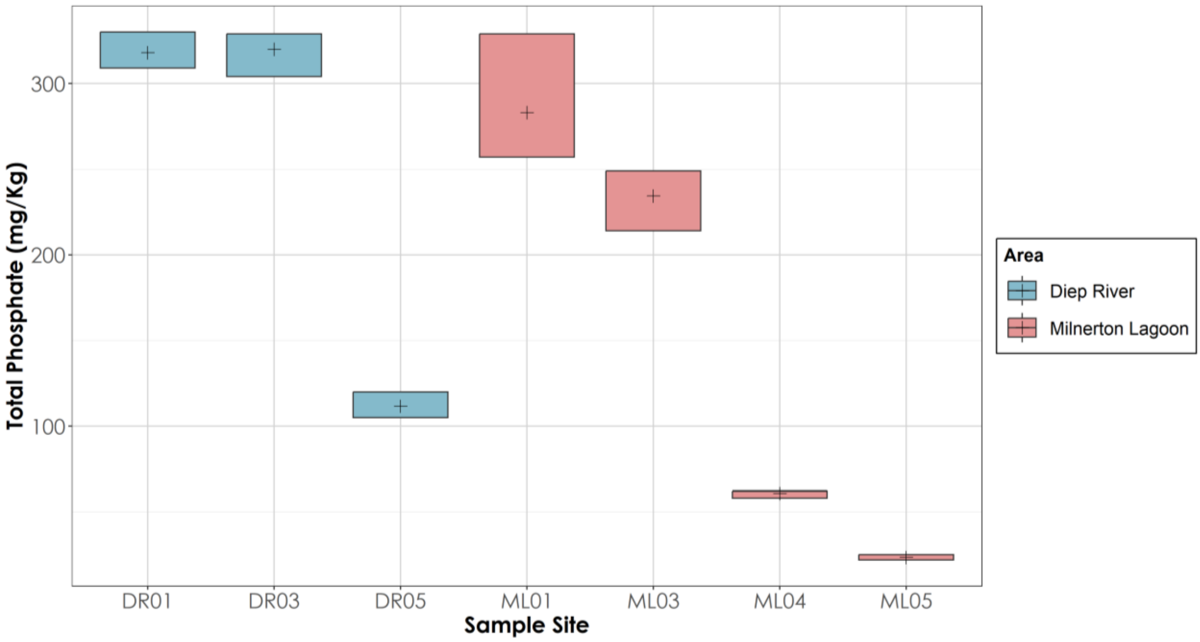


Figure 2-33. Range of total phosphate content of samples taken at each site. The mean value is indicated by the cross.

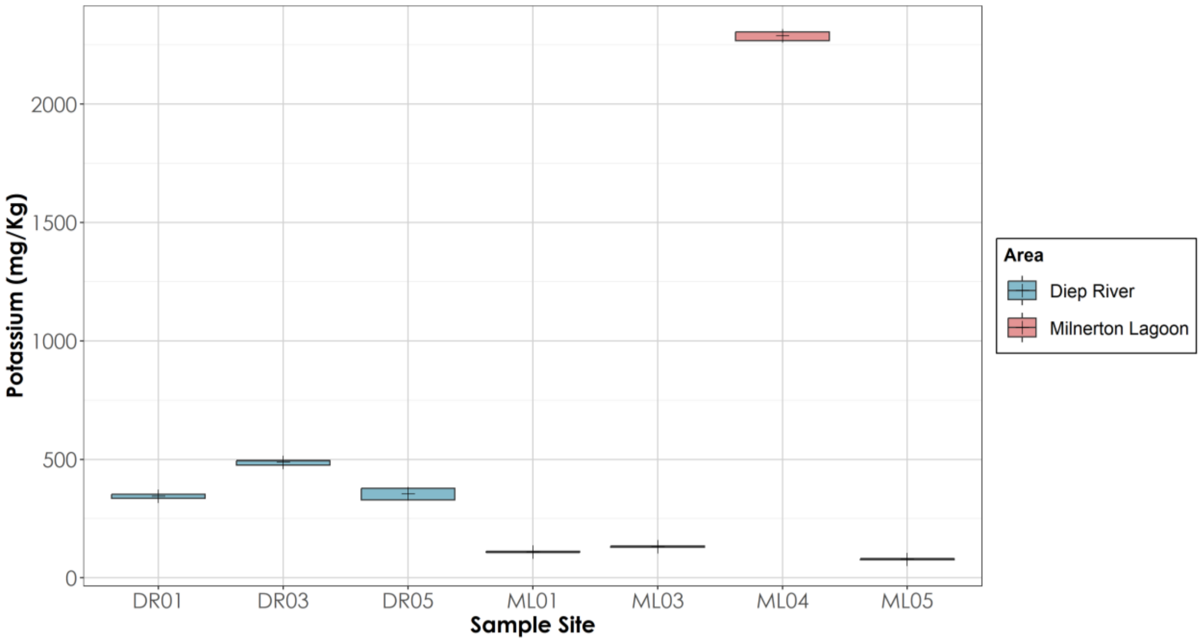


Figure 2-34. Range of potassium content of samples taken at each site. The mean value is indicated by the cross.

2.6.3.3 <u>Metals</u>

Heavy metal contamination in water and sediment can have serious ecological consequences as a result of its toxicity to biological organisms and tendency to bioaccumulate. Not only is this a concern for ecological reasons, but also for human health due to the potential of consumption of contaminated drinking water and fish. Sediment was tested for eight heavy metals – arsenic, cadmium, chromium, copper, mercury, nickel, lead, and zinc. All data below are presented in the form of box and whisker plots that contain the entire data range (each sample site had three replicates). The cross in each box represents the mean value.

Arsenic (As) is a heavy metal occurring in the form of oxides, sulphides and other compounds in contaminated water and soil due to weathering rocks as well as anthropogenic sources (Gavhane et al., 2021) such as pesticides and wood preservatives (Ali et al., 2013). As levels did not exceed the detection limits at any sample sites. In contrast, Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021) detected As in almost all their samples.

Cadmium (Cd) contamination is generally associated with fossil fuel combustion and waste disposal. Cd was detected in the majority of samples. It was detected at levels well below the guideline thresholds (Figure 2-35). Levels of Cd were in general much lower than found by Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021).

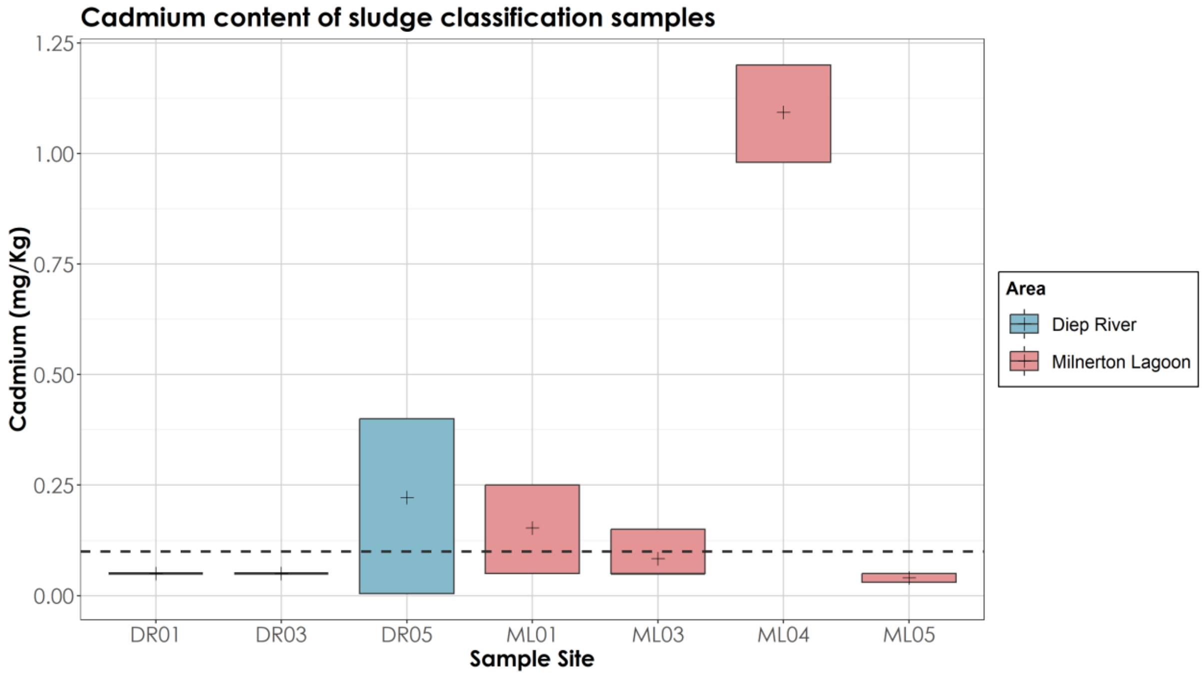


Figure 2-35. Box and whisker plots indicating the range of the Cd levels. The dashed line indicates the detection limit.

Chromium (Cr) is a heavy metal associated primarily with metallurgical industries (Gavhane et al., 2021). Cr was detected in most samples, but in concentrations well below the guideline limits and thresholds (Figure 2-36). Chromium speciation was not conducted as part of this assessment. Levels of Cr were in general much lower than found by Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021).

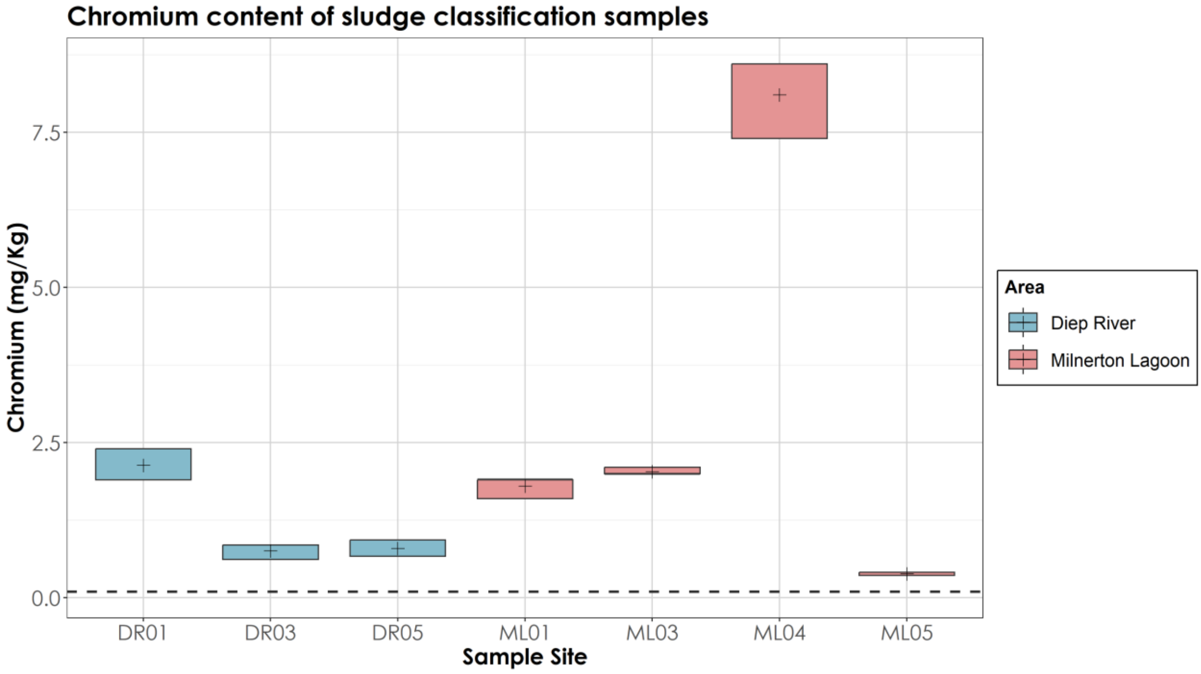


Figure 2-36. Box and whisker plots indicating the range of the Cr levels. The dashed line indicates the detection limit

Copper (Cu) was detected in all samples, generally at very low levels concentrations. Cu levels of all samples except ML04 fell below all guideline thresholds. ML04 exceeded the TCT0 guidelines (Figure 2-37). Levels of Cu were in general much lower than found by Hutchings and Clark (2010). Note that Gihlwala, Dawson & Clark (2021) did not test samples for copper.

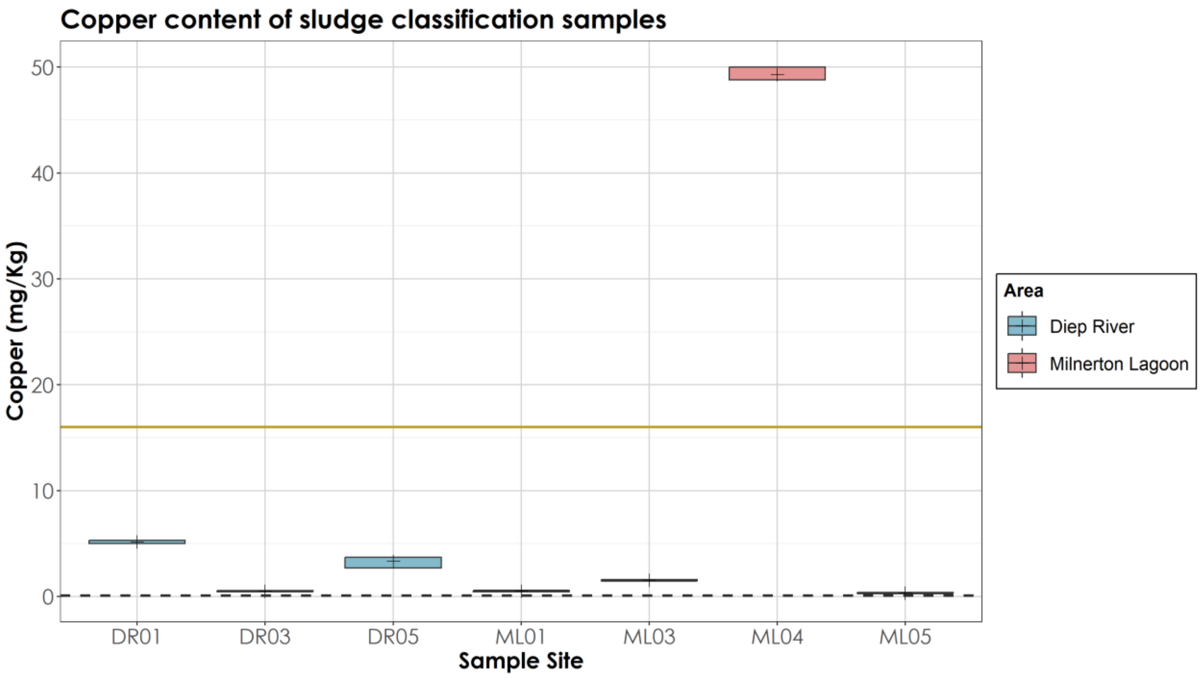


Figure 2-37. Box and whisker plots indicating the range of the Cu levels. The dashed line indicates the detection limit. The yellow line indicates the TCT0 limit.

Lead may derive from a number of anthropogenic sources, including (historically) petrol fuel as well as batteries, herbicides and insecticides. It was detected in all sample sites (Figure 2-38), although no samples were found to exceed guidelines values. Levels of lead were in general much lower than found by Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021).

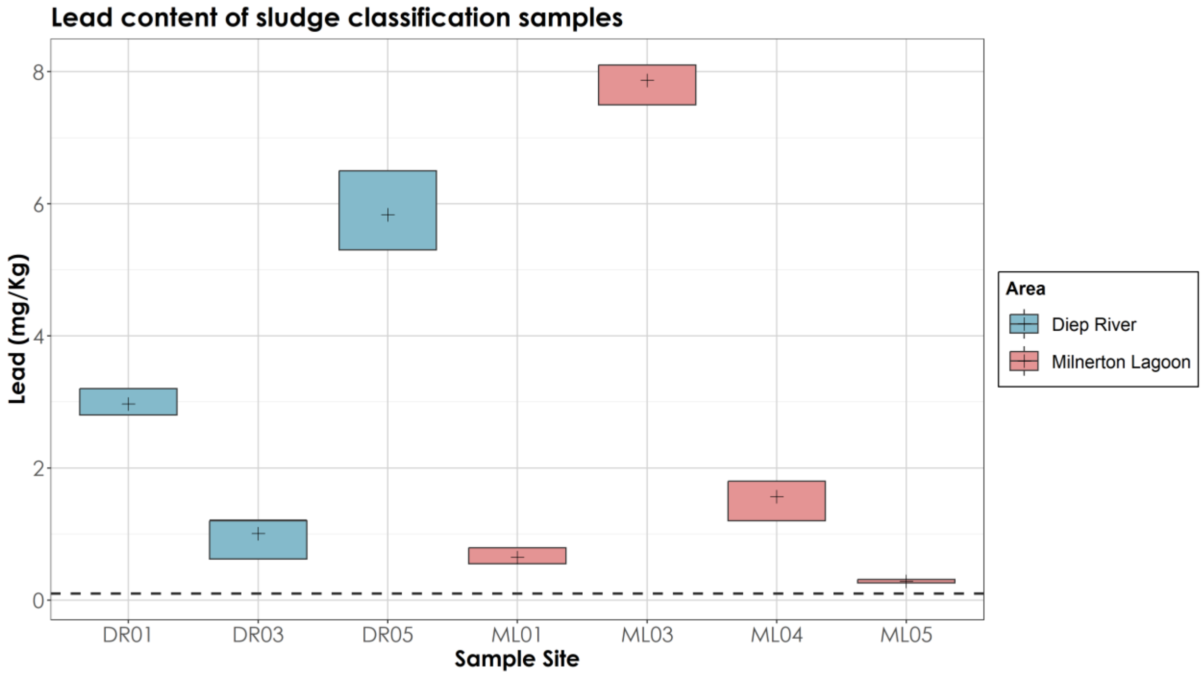


Figure 2-38. Box and whisker plots indicating the range of the Pb levels. The dashed line indicates the detection limit.

Mercury (Hg) was not detected in any samples. In contrast, both Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021) detected mercury across the lagoon (although these at low concentrations).

Nickel (Ni) was detected in three samples – one in the Diep River Area and two in the Milnerton Lagoon. No samples exceeded guideline values or limits (Figure 2-39). Where detected, nickel concentrations were lower than found by Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021).

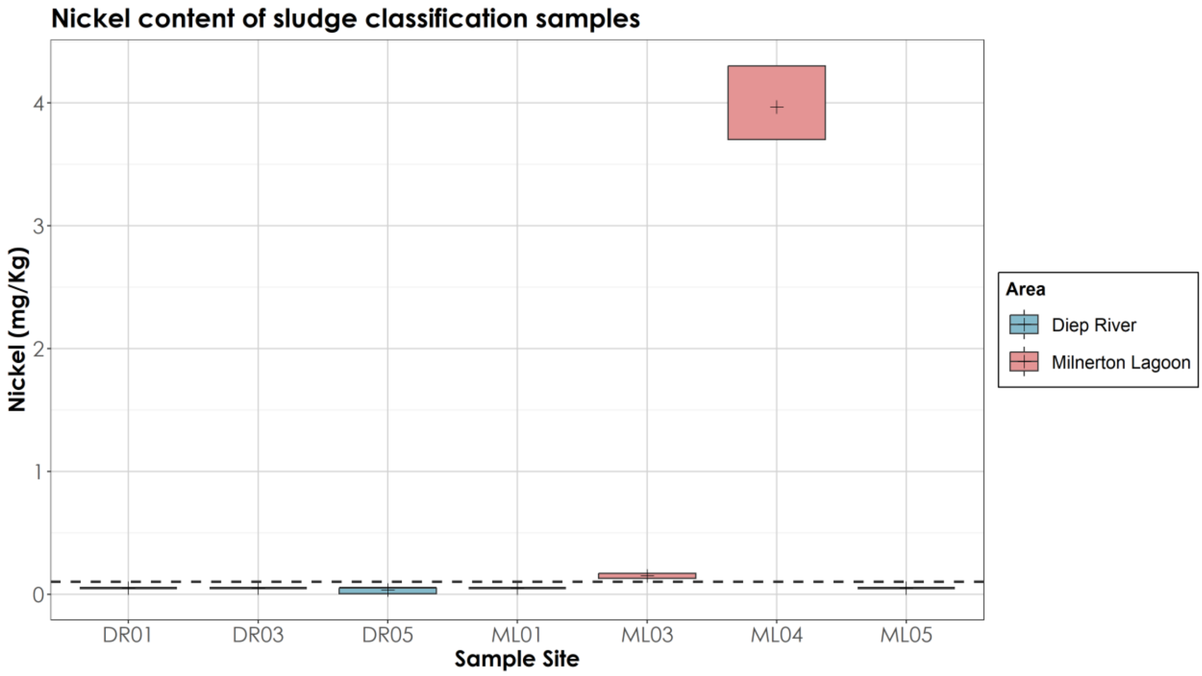


Figure 2-39. Box and whisker plots indicating the range of the Ni levels. The dashed line indicates the detection limit.

Zinc (Zn) may derive from a number of anthropogenic sources, including galvanised materials as well as fertilisers (it is an essential metal for plant growth) and pesticides. It was detected in all sample sites (Figure 2-40), but did not exceed guideline limits or thresholds. Levels of zinc were in general lower than found by Hutchings and Clark (2010) and Gihlwala, Dawson & Clark (2021).

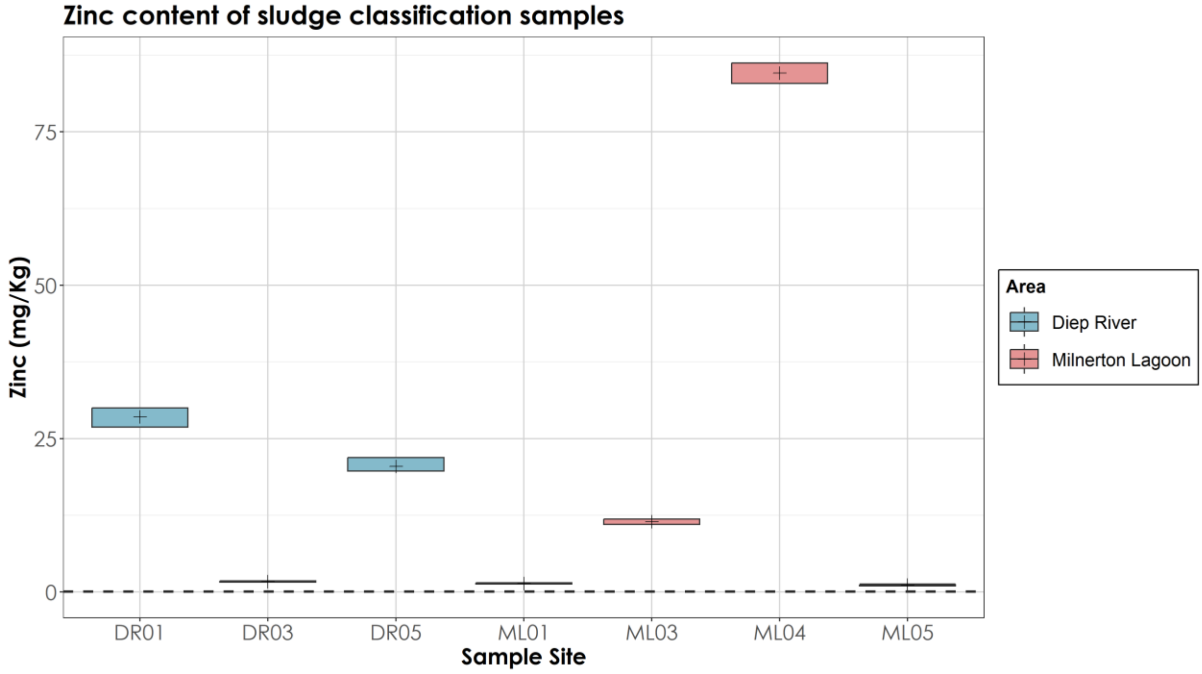


Figure 2-40. Box and whisker plots indicating the range of the Zn levels. The dashed line indicates the detection limit.

These most recent results show significantly lower contaminant levels in sediments in Milnerton lagoon. Given the very limited sampling effort, high spatial variation in sediment, and difference in methodologies, no conclusions are drawn regarding changes since the more rigorous analyses by Hutchings & Clark (2010) or Gihwala et al. (2021). The classification of the material in terms of sludge guidelines, however, is useful in determining potential options for disposal of dredged material.

The classification of sludges (in terms of the 2006 waste water guidelines) considers microbiological parameters (faecal coliforms and helminth ova), physical and stability indicators, and chemical characteristics (nutrients, metals and organic pollutants). On this basis a three-part classification is assigned that guides the beneficial use or disposal options for the material. Classifications for the Milnerton lagoon sludges are set out in the tables below.

Table 2-5. Sludge classifications for Milnerton Lagoon surface sediments, December 2022

	DR 01	DR 03	DR 05	ML 01	ML 03	ML 04	ML 05
Pollutant Class	B	A	A	A	A	A	A
Stability Class	1	1	1	1	1	1	1
Microbiological Parameters	a	a	a	a	a	a	a
Classification	B1a	A1a	A1a	A1a	A1a	A1a	A1a

All samples were classified as **B1a** or **A1a**, suggesting that they may be suitable for the following uses in line with the 2006 guidelines:

- **Disposal** with general restrictions or requirements, although beneficial use is recommended if possible.
- **On-site mono-fills or lagoon disposal**, with surface and groundwater monitoring for faecal coliforms and *E. coli* but no requirement for lining of ponds.
- **Landfill cover material**, with restrictions in relation to the microbiological parameters.
- **Agricultural beneficial use**, with general restrictions.

Not all parameters set out in the Norms and Standards for the Assessment of Waste for Landfill Disposal (GN. No. 36784 of 2013) were tested for, and it is expected that the Vissershok landfill site will require such testing and classification prior to acceptance of the dredged material. One parameter (copper) exceeded the TCT0 threshold in samples from the lagoon. The City of Cape Town undertakes the required analyses in terms of the Norms and Standards internally, and it is recommended that the Solid Waste Management Department confirm the parameters to be tested and the number of samples prior to further sampling in respect of possible disposal of dredged material.

2.6.4 Sediment bacterial processes

Hydrogen sulphide producing bacteria are also known as sulphate-reducing bacteria (SRB). These bacteria are obligate anaerobes that use sulphate as a terminal electron acceptor in their metabolic processes (Mara & Horan, 2003). This means that they use sulphate in the place of oxygen to produce energy from organic matter. The by-product of this process is hydrogen sulphide, which is a toxic and foul-smelling gas (Mara & Horan, 2003). In the case of Milnerton lagoon, it is this gas that causes a distinct rotten-egg-like smell at times. SRB play an important role in the cycling of sulphur, as they are responsible for the conversion of sulphate into sulphide (Mara & Horan, 2003).

The presence of some hydrogen sulphide producing bacteria in sediment is normal, but large populations of these bacteria can have a range of negative impacts on aquatic ecosystems (Mara & Horan, 2003). As hydrogen sulphide is toxic to many organisms (including fish, invertebrates, and plants), high levels of hydrogen sulphide in sediment can lead to the death of benthic organisms. Additionally, this leads to the exacerbation of anoxic conditions in the sediment. As SRB are obligate anaerobes, they thrive under anoxic conditions, thus forming a positive feedback loop whereby their population (and resultant negative effects on the ecosystem) continues to increase as dissolved oxygen concentrations continue to drop.

The impact of hydrogen sulphide production on the environment can be further amplified by physical processes such as sediment resuspension. When sediment is resuspended, hydrogen sulphide can be released into the water column, leading to the development of sulfidic water conditions (Mara & Horan, 2003). These conditions can be harmful to aquatic organisms and can also lead to foul odours and the deterioration of water quality.

The presence of hydrogen sulphide producing bacteria in sediment can be influenced by a range of factors, including the organic matter content of sediment, the availability of sulphate, and the presence of other electron acceptors such as iron and manganese (Mara & Horan, 2003). In addition, anthropogenic activities such as the discharge of poorly treated sewage and agricultural runoff can significantly impact the growth and proliferation of SRB in sediment (Mara & Horan, 2003) – clearly the case in the Milnerton Lagoon.

2.7 Estuarine Ecology

2.7.1 Fish

Estuaries are considered critically important nursery habitat for fish. Indeed, the Diep Estuary represents some 10% of the nursery area for fish on the West Coast, including species such as the white steenbras *Lithognathus lithognathus* (Peak Practice 2008, Viskich *et al*. 2016).

Prior to the development of greater Cape Town, the Diep Estuary mouth was dynamic and deep but prone to closure during dry years. The fish assemblage was fairly diverse with 23 indigenous species having been recorded in the estuary and Rietvlei wetlands. Fish recruitment into Rietvlei was likely to have only been during wet years when the entire area was inundated with water. Fish would have survived in the deeper ponds and vleis once the waters receded and their residence in Rietvlei may have been for as long as 10 years.

However, there are clear declines in fish species richness over time, from 12 species reported by Millard & Scott (1954) to five reported by Viskich et al. (2016). Of these five species, the most abundant was the opportunistic southern mullet, or harder *Chelon richardsonii*, which moves between estuarine and marine environments (Whitfield 1994, Lamberth et al. 2010, Viskich et al. 2016). In addition, two species reported by Viskich et al. (2016) and aliens, the mosquito fish Gambusia affinis, and Tilapia sparrmanii, leaving only three of the original native species present in the system in 2014.

These changes are likely linked to changes in water quality, specifically increased ammonia levels linked to malfunctions in the Potsdam wastewater treatment works, and severely reduced dissolved oxygen concentrations, which are frequently below the 2 mg/l threshold for the survival of aquatic species. While many estuarine-associated species are adapted to hypoxia, an increased frequency of low oxygen events (anoxia) has almost certainly negatively impacted benthic fish communities (Lamberth et al. 2010). Stagnant, low oxygen conditions likely contributed to the massive fish kill reported by the City of Cape Town on 3 March 2022. Roughly 500 dead fish located at the mouth of the Milnerton Lagoon, consisting of juvenile Mullet species, were removed by a team from the Table Bay Nature Reserve in early 2022 (Figure 2-41).

The permanent water bodies excavated in the Rietvlei section may actually provide fish with a refuge from the poor conditions in the Diep Estuary/Milnerton Lagoon (Gammon and Clark). Frequent perturbations in the estuary have seen almost complete switches in the fish assemblage occurring over short time periods. There has been a complete loss of the benthic gobies *Caffrogobius salhanha*, *Psammogobius knysnaensis* and *C. nudiceps* as well as the extirpation of the burrowing sandprawn *Callianassa kraussii* and other invertebrate species. Loss of this food source has contributed to a drastic decline in the number of important linefish species in the system such as the juveniles of the white steenbras *Lithognathus lithognathus* and white stumpnose *Rhabdosargus globiceps*.

Figure 2-41. The fish kill at the mouth of Milnerton Lagoon, reported to the City of Cape Town on 3 March 2022.

A subsequent seine and gillnet survey of the lagoon and Flamingo Vlei, conducted on the 16th of August 2022 by Anchor (Gammon and Clark 2022), collected only five individual fish in the system, of which two were small Southern mullet *Chelon richardsonii*, caught right at the estuary mouth. The remaining three consisted of a single larger southern mullet, banded tilapia *Tilapia sparrmanii* (invasive), and sharptooth catfish *Clarias gariepinus* (invasive), which were caught in two gillnets deployed in Flamingo Vlei overnight. These results suggest that there has been an almost complete collapse of the fish community in the system linked to the prevailing poor water quality conditions.

Tissue samples from all fish caught in the survey were submitted to an analytical laboratory (Scientific Services) for trace metal determination. Levels of lead (Pb) in all fish, and arsenic (As) in one, were found to exceed the recommended upper limits in foodstuffs, suggesting that consumption of these fish would pose a risk to human health.

2.7.2 Benthic macrofauna

Benthic macrofauna comprise all invertebrate animals that occur in or on sediments and can be retained on a 1.0 mm sieve (i.e. are usually visible with the naked eye) (Figure 2-42). They are a fundamental part of the estuarine food web and are important as processors of organic particles. These organisms are frequently monitored as bio-indicators to detect changes in the health of marine environments. This is because most of them are relatively sessile and cannot easily avoid contaminants (as mobile biota can), nor are they carried passively by ocean currents (as may happen to planktonic organisms). Additionally, they are relatively easy to sample quantitatively, and they exhibit a range of tolerances to environmental stress and pollution (Warwick 1993, Dutertre et al. 2013).

Figure 2-42 Examples of benthic macrofauna from the West Coast of South Africa, including crabs (A), amphipods (B and C) and isopods (D).

The most recent study on the benthic macrofaunal community in the system was conducted in July 2021 by Gihwala *et al*. (2021), in which a total of 728 macrofaunal organisms from six different taxa were recorded from sampling transects between the mouth of the lagoon up until the Otto du Plessis Bridge. This represents a dramatic decline in diversity since the earliest survey by Millard & Scott (1954), who recorded 47 species within Milnerton Lagoon. Surveys conducted by Weil in 1974 (unpublished data) and in (Viskich et al. 2016) yield 23 species each, which suggests that species richness have been on a downward trajectory from some time. Only six species (*Capitella capitata*, *Ficopomatus enigmaticus*, *Scolelepis squamata*, *Kraussillichirus kraussi*, *Melita zeylanica* and *Hymenosoma orbiculare*) were recorded in all three surveys between 1954 and 2014 (out iof of a total of 69 taxa species, Viskich et al. 2016). From these, only the polychaete *C. capitata* and the brachyuran *H. orbiculare* were observed in the present study. *C. capitata* (which is characteristic of highly polluted and degraded environments) dominated all samples and constituted 79% of the abundance. Furthermore, of the 28 successful macrofauna samples, only 13 of them contained any macrofaunal organisms, with organisms being absent in half of the samples collected further upstream than 1.5 km of the mouth.

The 2021 study also measured trace metal pollution levels in the Diep River Estuary sediments but did not conduct field or laboratory ecotoxicity studies to determine toxic effects on biota found in this estuarine system. Nonetheless, the analyses of various physical and chemical parameters in sediment collected in the Diep River Estuary in June 2021 provided evidence that some trace metal elements were elevated well above levels considered to be toxic to living organisms (according to international and local quality guidelines) and highlighted sites past the Otto du Plessis Bridge and near the Blaauwberg Bridge as areas of concern. Furthermore, the large absence of benthic organisms indicates a severely degraded system and no longer pristine compared to historical surveys (Van Niekerk et al. 2019).

Finally, an important consideration is that the 2021 study predates the rapid worsening of water quality in the system since 2022, discussed in Section 2.5 and 2.6 of this report, which would suggest that macrofaunal abundance and diversity in the estuary may have actually declined further that is evident from these data.

2.7.3 Estuarine vegetation

Milnerton Lagoon and the lower estuarine area are highly disturbed, with the only remaining "natural vegetation" of environmental significance downstream of the Woodbridge comprising a thin strip of dunes between the Woodbridge Island development (which itself is classified as "transformed habitat of no known conservation significance") and the beach itself. There is also a semi-vegetated dune towards the mouth of the estuary on the south bank. Typical species of this dune vegetation includes Blombos *Metalasia muricata*, *Osteospermum moniliferum* (previously *Chrysanthemoides monilifera*) and Thesium species interspersed with arum lilies (DWS 2017). Submerged/aquatic macrophytes within the lower estuary include Ruppia species, *Stuckenia pectinata*, filamentous algae Cladophora/Enteromorpha and the invasive water hyacinth (DWS 2017).

The estuarine channel upstream of the Milnerton lagoon is dominated by freshwater species, with water hyacinth covering in a thick surface blanket of the macrophyte. The banks of the serpentine area of the estuary are dominated by invasive grasses that encroach onto the open water and thrive under the eutrophic, fresh conditions. Between the main channel and golf course to the west, and R27 on the east, are vegetated areas comprising saltmarsh species (Figure 2-43).

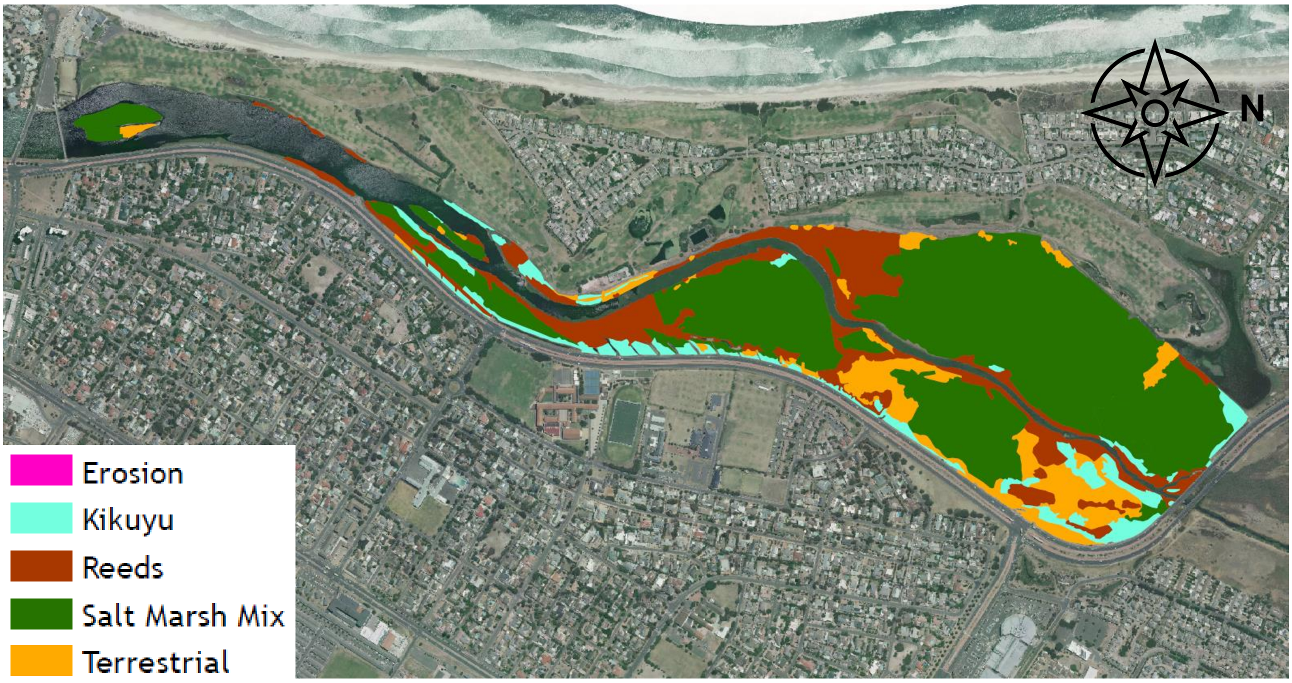


Figure 2-43. Estuarine vegetation mapping 2018 (CCT Biodiversity Management)

2.7.4 Algae

Microalgae are generally defined as unicellular algae that either live suspended in the water column (phytoplankton), on rock or sediments in the estuary (benthic microalgae), or on plants (epiphytic microalgae). Estuarine phytoplankton communities are typically comprised of flagellates and diatoms, while benthic microalgae communities are dominated by blue-green algae and diatoms. Microalgal productivity is the most important determinant of overall biomass of estuarine biota, with most trophic pathways originating in microalgal rather than macrophyte (plant) productivity i.e. microalgae are an important source of food for fish and microfauna (Turpie & Clark 2007b). As such, microalgae are an important component of the estuary biota.

The 2017 RDM rated the microalgae health in the system relative to natural baseline conditions (DWS 2017). This report noted that nutrient input and an overall decrease in salinity would likely lead to reduced species richness in the system, and cause a shift from diatoms and Chlorophytes to Cyanophytes (blue-green algae) which occur under nutrient-rich, freshwater conditions (DWS 2017). The current high nutrient levels would likely have resulted in an increase in microalgal biomass, while the permanent inundation due to the WWTW flows would result in a loss of benthic microalgal habitat (DWS 2017).

While there is little information available on microalgae species in the Diep Estuary, diatoms including Coscinodiscus, Rhizosolenia, Biddulphia, Thalassiosira and Skeletonema costatum have been reported in Rietvlei (Grindley & Dudley 1988). Of particular concern to the ecological functioning of the system are Cyanophyceae, which form toxic Microcystis blooms that have led to extensive fish kills in the system (Haskins 2013).

2.7.5 Birds

The Diep Estuary system (including Rietvlei) is considered one of the most important areas for water birds in the region and is recognised as an Important Bird and Biodiversity Area (IBA) by Birdlife International (an IBA is an area identified using an internationally agreed set of criteria as being globally important for the conservation of bird populations). Rietvlei has also been ranked sixth out of the 65 coastal wetlands in the southwestern Cape and ranks sixth or seventh out of 42 estuaries in

South Africa based on their conservation value for waterbirds (Ryan et al. 1988, Turpie 1995, City of Cape Town 2016).

While most studies of bird abundance and species richness for the area is focused on Rietvlei, rather than the lower estuary, Scott (1954) did report kelp gull *Larus dominicanus*, Hartlaub's gull *Chroicocephalus hartlaubii*, common tern *Sterna hirundo* and Cape shoveler *Spatula smithii*, amongst others. Viskich et al. (2016) reported all these species, as well as species absent from Scott (1954), specifically predominantly freshwater species such as the red-knobbed coot Fulica cristata and African darter *Anhinga rufa*.

Coordinated Waterbird counts (CWAC) surveys have been conducted at two sites in the lower Diep Estuary in proximity to Woodbridge Island, however, the last survey was conducted in 2009 so these results are not considered current. More recent surveys have been conducted in Rietvlei and the surrounding wetlands, yet these have been highly sporadic and are not considered meaningful for determining trends in the bird population with time. In total, 76 waterbird species have been sighted in the lower Diep Estuary since CWAC surveys began in 1985.

Table 2-6. Total CWAC species present in the lower lagoon seen between 1985 and 2009.

Species Name	Taxonomic name	Species Name	Taxonomic name
African Black Duck	*Anas sparsa*	Grey-headed Gull	*Chroicocephalus cirrocephalus*
African Darter	*Anhinga rufa*	Hadada Ibis	*Bostrychia hagedash*
African Fish Eagle	*Haliaeetus vocifer*	Hartlaub's Gull	*Chroicocephalus hartlaubii*
African Jacana	*Actophilornis africanus*	Hybrid duck	*Anas hybrid*
African Marsh Harrier	*Circus ranivorus*	Intermediate Egret	*Ardea intermedia*
African Oystercatcher	*Haematopus moquini*	Kelp Gull	*Larus dominicanus*
African Sacred Ibis	*Threskiornis aethiopicus*	Kittlitz's Plover	*Charadrius pecuarius*
African Snipe	*Gallinago nigripennis*	Lesser Flamingo	*Phoeniconaias minor*
African Spoonbill	*Platalea alba*	Little Egret	*Egretta garzetta*
African Swamphen	*Porphyrio madagascariensis*	Little Grebe	*Tachybaptus ruficollis*
Bank Cormorant	*Phalacrocorax neglectus*	Little Stint	*Calidris minuta*
Black-crowned Night Heron	*Nycticorax nycticorax*	Little Tern	*Sternula albifrons*
Black-headed Heron	*Ardea melanocephala*	Malachite Kingfisher	*Corythornis cristatus*
Blacksmith Lapwing	*Vanellus armatus*	Mallard Duck	*Anas platyrhynchos*
Black-winged Stilt	*Himantopus himantopus*	Marsh Sandpiper	*Tringa stagnatilis*
Cape Cormorant	*Phalacrocorax capensis*	Pied Avocet	*Recurvirostra avosetta*
Cape Shoveler	*Spatula smithii*	Pied Kingfisher	*Ceryle rudis*
Cape Teal	*Anas capensis*	Purple Heron	*Ardea purpurea*
Cape Wagtail	*Motacilla capensis*	Red-billed Teal	*Anas erythrorhyncha*
Caspian Tern	*Hydroprogne caspia*	Red-knobbed Coot	*Fulica cristata*
Common Greenshank	*Tringa nebularia*	Reed Cormorant	*Microcarbo africanus*
Common Moorhen	*Gallinula chloropus*	Ruff	*Calidris pugnax*
Common Ringed Plover	*Charadrius hiaticula*	Sanderling	*Calidris alba*
Common Sandpiper	*Actitis hypoleucos*	Sandwich Tern	*Thalasseus sandvicensis*

Species Name	Taxonomic name	Species Name	Taxonomic name
Common Tern	*Sterna hirundo*	South African Shelduck	*Tadorna cana*
Crowned Cormorant	*Microcarbo coronatus*	Spur-winged Goose	*Plectropterus gambensis*
Curlew Sandpiper	*Calidris ferruginea*	Three-banded Plover	*Charadrius tricollaris*
Domestic Duck	*Anas platyrhynchos*	Unidentified Duck	*N/A*
Egyptian Goose	*Alopochen aegyptiaca*	Unidentified Tern	*N/A*
Giant Kingfisher	*Megaceryle maxima*	Unidentified Wader	*N/A*
Glossy Ibis	*Plegadis falcinellus*	Water Thick-knee	*Burhinus vermiculatus*
Great Crested Grebe	*Podiceps cristatus*	Western Cattle Egret	*Bubulcus ibis*
Great White Pelican	*Pelecanus onocrotalus*	Western Yellow Wagtail	*Motacilla flava*
Greater Crested Tern	*Thalasseus bergii*	White-breasted Cormorant	*Phalacrocorax lucidus*
Greater Flamingo	*Phoenicopterus roseus*	White-fronted Plover	*Charadrius marginatus*
Greater Painted-snipe	*Rostratula benghalensis*	White-winged Tern	*Chlidonias leucopterus*
Grey Heron	*Ardea cinerea*	Wood Sandpiper	*Tringa glareola*
Grey Plover	*Pluvialis squatarola*	Yellow-billed Duck	*Anas undulata*

Site visits by Anchor in December 2020, and February 2022 confirmed that the estuary is an important feeding and roosting area for many bird species, including greater flamingos *Phoenicopterus roseus*, white-breasted cormorants *Phalacrocorax lucidus* and pied avocets *Recurvirostra avosetta*. The exposed tidal mudflats appear to be used as sites for roosting and foraging (Figure 2-44; Table 2-7).

Table 2-7. Bird species observed in the lower Diep Estuary in December 2020 and February 2022.

Common name	Scientific name	Common name	Scientific name
Greater flamingo	*Phoenicopterus roseus*	Common tern	*Sterna hirundo*
White-breasted cormorant	*Phalacrocorax lucidus*	Sandwich tern	*Thalasseus sandvicensis*
African oystercatcher	*Haematopus moquini*	White-fronted plover	*Charadrius marginatus*
Grey heron	*Ardea cinerea*	Pied avocet	*Recurvirostra avosetta*
Black-winged stilt	*Himantopus himantopus*	Curlew sandpiper	*Calidris ferruginea*
Red-winged starling	*Onychognathus morio*	Pied kingfisher	*Ceryle rudis*
European starling	*Sturnus vulgaris*	Great egret	*Ardea alba*
Cape teal	*Anas capensis*	Little egret	*Egretta garzetta*
African black duck	*Anas sparsa*	Egyptian goose	*Alopochen aegyptiaca*
Yellow-billed duck	*Anas undulata*	Kelp gull	*Larus dominicanus*
Cape shoveler	*Spatula smithii*	Hartlaub's gull	*Chroicocephalus hartlaubii*

Figure 2-44 Bird species seen in the Milnerton Lagoon (February 2022)

3 CAUSES OF IMPACTS AND CURRENT EFFORTS TO MITIGATE THEM

3.1 General and catchment-scale factors

The Diep River estuary is a highly modified estuarine system occurring entirely within South Africa's oldest and most populous city. In this context the persistence of estuarine processes and habitat and the management of the estuary for biodiversity, recreation and functionality, are beset by several challenges. The estuary mouth is constrained by development on both sides, and can no longer migrate north and south as was historically the case. The estuary is in an urban area and surrounded by development. Infrastructure, particularly roads and rail, crosses the estuary. Four road bridges, a pedestrian bridge and a rail bridge cross the estuary. These factors constrain flow in the estuary and alter the movement of sediment, reducing the ability of the system to flush accumulated sediments, nutrients, and contaminants during high flow periods.

As for any estuary, the quality and quantity of inflows from the river catchment is a major determinant of estuarine health and function. In the Diep River catchment, extensive agriculture covers more than half the land surface. This increases siltation in the river, contributes to high nutrient loads, and lowers the flow due to abstraction for irrigation purposes. Urban areas in the broader catchment, including Malmesbury and Klapmuts, are a source of runoff to the river. The latter includes stormwater runoff with contaminant loads from roads, gardens, and industrial areas, but also sewage where failing or blocked sewer pipes spill into the stormwater system. The Diep River estuary receives runoff from at least 4 140 hectares of urban areas in its immediate surroundings within the City of Cape Town, including industrial areas, informal settlements and a golf course. The estuary is also the discharge point for treated effluent from more than 75 000 households and significant industrial areas. An increasing number of residents depend on the sewer system in the catchment and the effluent cannot feasibly be treated or discharged elsewhere than at Potsdam.

Water quality in the Milnerton Lagoon is a complex problem impacted by multiple pollution sources, the most significant of which are described below.

Current efforts to address these sources of pollution are also summarised, based on information provided by the City in the Lower Diep River Transversal Action Plan (dated April 2021), the Diep River Estuarine Management Plan (dated October 2022), and the Potsdam WWTW Improvement Plan (dated February 2023). **It is strongly recommended that the various plans and responses in preparation by the City be aligned, including through the incorporation of the findings of this Remediation Plan into the respective catchment- and estuary-scale planning.**

3.2 Potsdam wastewater treatment works

Loading data presented in Section 2.5.8 indicate that, at least in summer, the Potsdam WWTW was the single most significant contributor to water quality pollution of Milnerton Lagoon during the five years leading up to the publication of this report. During summer, its flows comprise almost the entire discharge into the lagoon, barring inflows during storm events and other smaller effluent discharges. Potsdam performance is thus critical to estuarine condition and function.

Final effluent data show that the WWTW has over the past six years, and most notably since May 2022, contributed water with very high chemical oxygen demand (COD) loading and concentration, exemplified by the high proportion of ammonia nitrogen in total inorganic nitrogen. These inflows also included high loads of mainly organic sediments that deposit in the lagoon. While the bulk of the sediments present in the lagoon are the result of decades of deposition rather than of recent origin, the data show an exponential increase in sediment and COD loading into the lagoon in 2021 and 2022, and it is suggested that a tipping point has been reached in which the aerobic decomposition of organic material is prevented by the high COD and low (to zero) dissolved oxygen levels in flows from the WWTW. These conditions also facilitate the production of odoriferous gases including hydrogen sulphide within the accumulating sediments. Ecologically toxic conditions are likely to predominate in the lagoon as a result, with little to no oxygen in lagoon waters; un-ionised ammonia likely to be present at acute toxicity levels; and chloramines likely to be present at high (but unmonitored) levels.

3.2.1 Capital upgrades

The backdrop to this Remediation Plan is an ongoing programme of major capital upgrades to the Potsdam WWTW, intended to increase both its treatment capacity and the effectiveness of treatment. These works will include a membrane bioreactor plant to provide an additional 53 megalitres (Ml) of daily treatment capacity, taking the total capacity of the works to a total of 100 Ml per day. An ultrafiltration plant will be installed at the existing 08 Plant to improve its management of suspended solids. The maturation ponds currently causing significant water quality issues will be completely removed and the area rehabilitated. Two new concrete lined emergency ponds with return pumps will be constructed in their place for raw sewage overflow management.

Delays have been experienced by this capital programme since its inception, with the first tender award in 2010 having been successfully appealed and subsequently cancelled. The reallocation of capital budgets to water resource projects associated with the drought between 2015 and 2018 also impacted on these timeframes. New contracts were awarded in late 2022, and two appeals against these awards were dismissed in March 2023. The Potsdam upgrade project has two components: the mechanical and electrical contract for the design, supply, installation and commissioning of mechanical and electrical infrastructure and the civil contract for the civil construction of infrastructure and buildings, including demolition works. The two contracts for the new construction were approved by Council in March 2023 following a section 33 process. Full commissioning of the new plant is programmed for the end of 2027, with various aspects coming online at earlier dates – including the ultrafiltration plant from 2025.

If the Potsdam WWTW works are concluded on schedule and function as expected, the Milnerton Lagoon would experience improved water quality from approximately 2025 onward, reaching optimal effluent standards by 2028. **These capital upgrades are strongly supported, and should be pursued with urgency.**

3.2.2 Recent issues and efforts at the WWTW

The existing WWTW is unable to cope with the loading of influent it currently receives and this, coupled with maintenance issues and the impacts of loadshedding, has resulted in deterioration of final discharge effluent quality. An overall increase in raw sewage flows to the WWTW has been measured in recent years, often exceeding its design capacity of 47 ML/day.

In 2021, challenges in managing the distribution of flows between the 97 and 08 Plants had resulted from failures of concrete in the 97 Plant settled sewage sump. These failures resulted in damage to the pumps that allow for pumping of flows from the 97 Plant to the 08 Plant, and thus in the hydraulic overloading of the 08 Plant. The settled sewage pumps were recently refurbished, allowing better distribution of flows between the 97 and 08 Plants.

The secondary treated effluent at the 08 Plant flows into maturation ponds 1-4 (Figure 3-1) and was historically diverted to Reedbed 1 and then to long pond. The ponds are now bypassed via a pumped line to either Pond 7 or the treated effluent pond or to UV treatment. The installed pump is not capable of pumping all 08 Plant secondary treated effluent, however, and a second 10-inch pump is required to fully bypass the maturation ponds for cleaning.

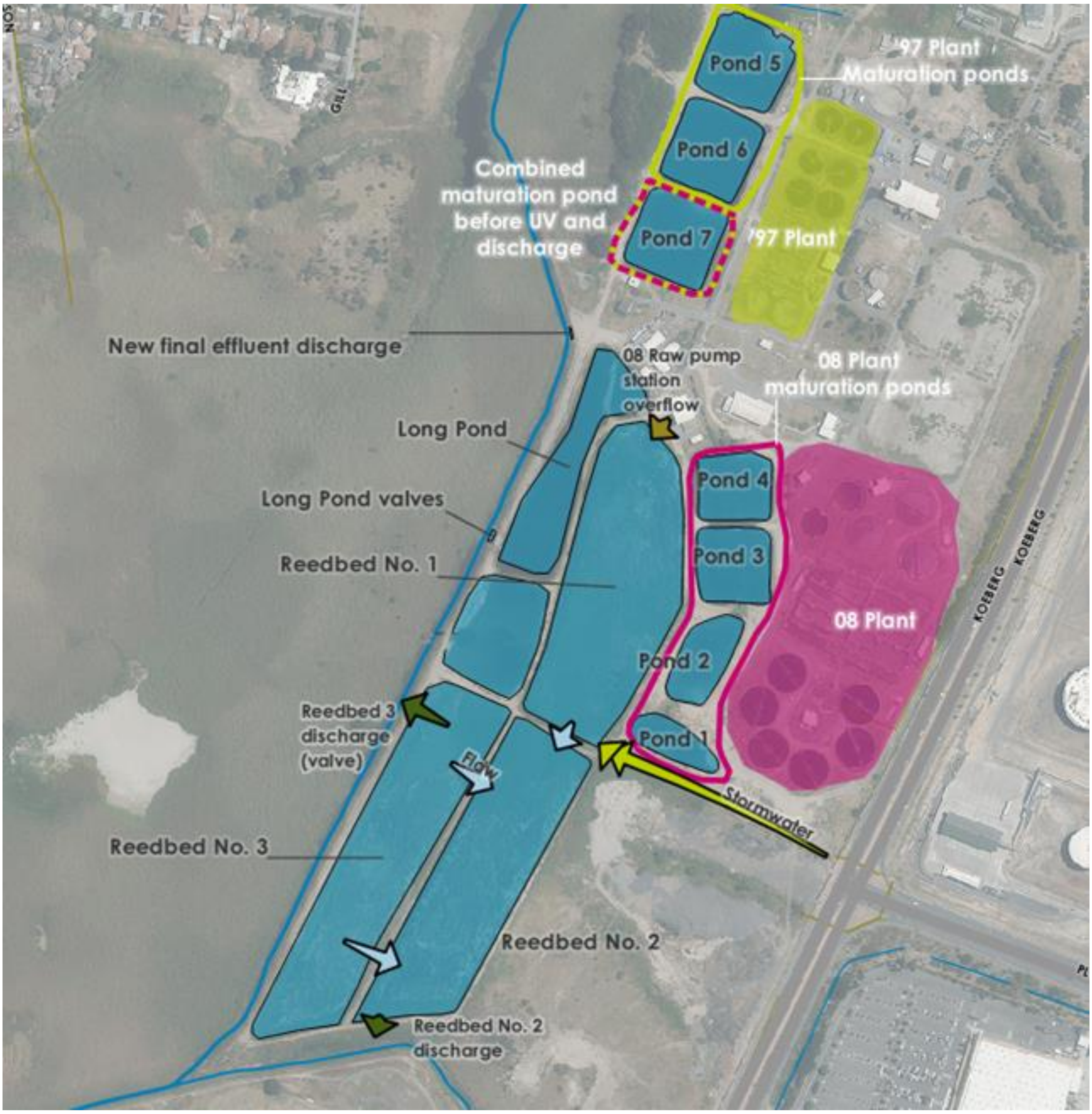


Figure 3-1. Maturation ponds and reedbeds at the Potsdam WWTW

Originally, the plant was designed so that after UV disinfection, the treated effluent supplied to end users would be drawn off from Wally's Pond with excess stored in Long Pond or allowed to overflow into the river via overflow weirs on Long Pond. Due to a history of contamination of Long Pond and difficulty in cleaning it due to its lack of lining, flows within the plant have been rerouted by installing isolation valves at the long pond discharge point and installing a new discharge point immediately after UV treatment.

Mechanical wear of the plant's belt filter presses has resulted in a loss of reliability of this dewatering equipment, and consequent frequent failures of other treatment processes due to the high concentration of solids. All three Return Activated Sludge pumps at the '08 Plant broke down in January 2022, resulting in excessive solids overflow into the maturation ponds normally used for final settling and clarifying of effluent before discharge – and just after the annual cleaning of these ponds had been concluded. The ponds were cleaned again in February-March 2022, but another failure of the pumps in March 2022 resulted in the recontamination of these ponds with sludges. In May 2022, a mini-substation failed resulting in an electrical failure and therefore a failure of the mechanical surface aerators at the 08 Plant. The substation was repaired in mid-2022, but the lack of aeration resulted in a loss of the biological processes that are central to treatment. In November 2022, failure of the return activated sludge pump in the 97 Plant also caused contamination of the 97 secondary settled sludge tanks and the 97 plant maturation ponds. A further failure of the beltpress motor control centre occurred in early 2023.

Although each individual breakdown or failure was responded to and repairs implemented, the nett effect was a serious deterioration in final effluent quality and exponential increase in the loading of pollutants discharged to the estuary.

The most recent cause of the spike in poor water quality in the Milnerton Lagoon appears to be the contamination in 2022 of the Potsdam WWTW maturation ponds and the delays associated with cleaning and reinstating them – exacerbated by loss of the biological treatment processes associated with reduced aeration.

Contamination of the maturation ponds has resulted in re-contamination of treated effluent passing through the maturation ponds with suspended solids, nutrients and bacteria prior to discharge, and is believed to be a significant contributing factor to the spike in water quality issues. The high turbidity also resulted in an inability to utilise the UV irradiation system for disinfection of final effluent, and chlorination of effluent is being implemented instead in an attempt to reduce bacterial counts in line with licensed standards and water quality guidelines. Treated effluent intended for reuse is at present not permitted to flow through the maturation ponds but is instead bypassed directly to the pump station to be pumped offsite.

Work to drain and clean the maturation ponds was delayed for several months, due to delays in procuring equipment including a long-boom excavator, dump trucks, and pumps to bypass flows. A pump large enough to pump effluent around the maturation ponds and allow for their remediation was not available until the end of March 2023, having been in use as a backup pump for the Koeberg sewage pump station. The required contracts were in place by late March 2023, and cleaning of the ponds had commenced at the time of finalisation of this report, with completion scheduled for end-June 2023. Bypassing and remediation of the contaminated maturation ponds is probably the single most effective action that the City could have taken to alleviate the break-down in estuarine function in late 2022. The Milnerton Lagoon could well have been on a trajectory of recovery by mid-2023 if the pump had been available at the time of maturation pond contamination.

Bypassing and remediation of the maturation ponds is a critical intervention now underway that could result in an improvement in Milnerton Lagoon – or at least allow other in-lagoon measures to achieve a long-term improvement. The delays in obtaining the required equipment were significant, and **it is recommended that the City procure sufficient standby pumps of the required specification to provide for future incidents of this nature at Potsdam or elsewhere.**

3.3 Contribution of runoff through contaminated WWTW reedbeds

Extensive reedbeds some 6.2 ha in area are located south of the WWTW, between the works and the Theo Marais canal (Figure 3-1). These were not assessed in any detail in this study, but discussions with Dr Ewart-Smith (Freshwater Consulting Group) who has conducted recent surveys of the reedbed indicate that they are contaminated with high levels of anoxic organic sediments.

The reedbeds were used historically to treat stormwater, discharging into the Theo Marais canal. Reed bed No. 1 receives stormwater from a small catchment including Koeberg Road, the refinery, adjacent commercial areas and the '08 Plant secondary settling tanks area. All 08 Plant maturation ponds (ponds 1, 2, 3 & 4) can drain into Reedbed No. 1. The '08 Plant raw sewage pump station also overflows into Reedbed No. 1 when all three of its pumps fail, and the reedbeds have been polluted by these frequent overflows.

In order to reduce the frequency of discharge into the Theo Marais channel (and thence into the Diep River system), a valve has been installed on the outlet pipe. In theory, this allows accumulated effluent in the reedbed to be pumped back into the WWTW for treatment. A sump has also been installed upstream of the valve, to prevent the ingress of stones and other material that might prevent closure of the valves. Reedbed 2 has been isolated from Reedbed 1, and an isolation valve installed at the Theo Marais outlet of Reedbed 2. However, this valve has been subject to frequent leaks and during the course of this study, effluent was flowing from the reedbeds into the channel on three of four site visits. The quality of this effluent is likely to be very poor, given its exposure to additional contaminated sediments with a high COD.

Reedbed remediation and potentially separation of the reedbeds into isolated zones that contain contaminated inflows rather than allowing them to re-contaminate the whole reedbed are necessary interventions. These measures are not rapid and will not on their own improve lagoon quality, but form part of the capital upgrades and should be pursued with urgency.

3.4 Sewage inflows as a result of pump station failure

Figure 3-2 shows the locations of pump stations and sewers in the vicinity of Milnerton Lagoon. Gravity fed sewers convey raw sewage to low points, from where they are pumped in rising mains, eventually passing into the WWTW. When pump stations fail as a result of sewer blockages, mechanical failure or power outages, sewage builds up, either at the pump station itself, or in the downstream sewers, from where it overflows from the lowest lying manholes. This sewage then drains into the stormwater system and in this way passes into the Diep River system / Milnerton Lagoon.

Under load-shedding conditions, pump stations that do not have back-up generators cannot operate, and even those that do have generators are often unable to keep running through sustained (Stage 6 or higher) power outages. This means that the frequency of pump station overflows has increased as a result of load shedding, the intensity of which has increased over the past six to eight months.

In addition, pump stations are prone to blockages from inappropriate waste that is disposed of into the sewers, and includes bricks, concrete, plastic waste and rags, all of which can block pumps and sewers. The volume of raw sewage inflows into the lagoon as a result of pump station failure has not been quantified. It is however regarded as a significant City-wide risk, affecting water quality in all catchments.

In the Diep River catchment, the Koeberg pump station, abutting the Theo Marais Channel, is regarded as one of several that are particularly problematic. When it blocks, sewage overflows from the sewer into the channel. Upgrades to this pump station were however about to commence at the time of writing this report (Mr B. De Wet; City of Cape Town; pers. comm. to Liz Day). These will include construction of a litter grid and sand trap, to reduce the risk of blockage by debris.

The above measures are necessary and should ideally be rolled out at all problematic pump stations prone to blockage, while ideally all pump stations should also be equipped with back-up generators. They are however not rapidly implementable (other than provision of additional generators) and on their own would not provide immediate relief to the lagoon ecosystem.

Figure 3-2. Sewage pump stations and sewers in the vicinity of Milnerton Lagoon

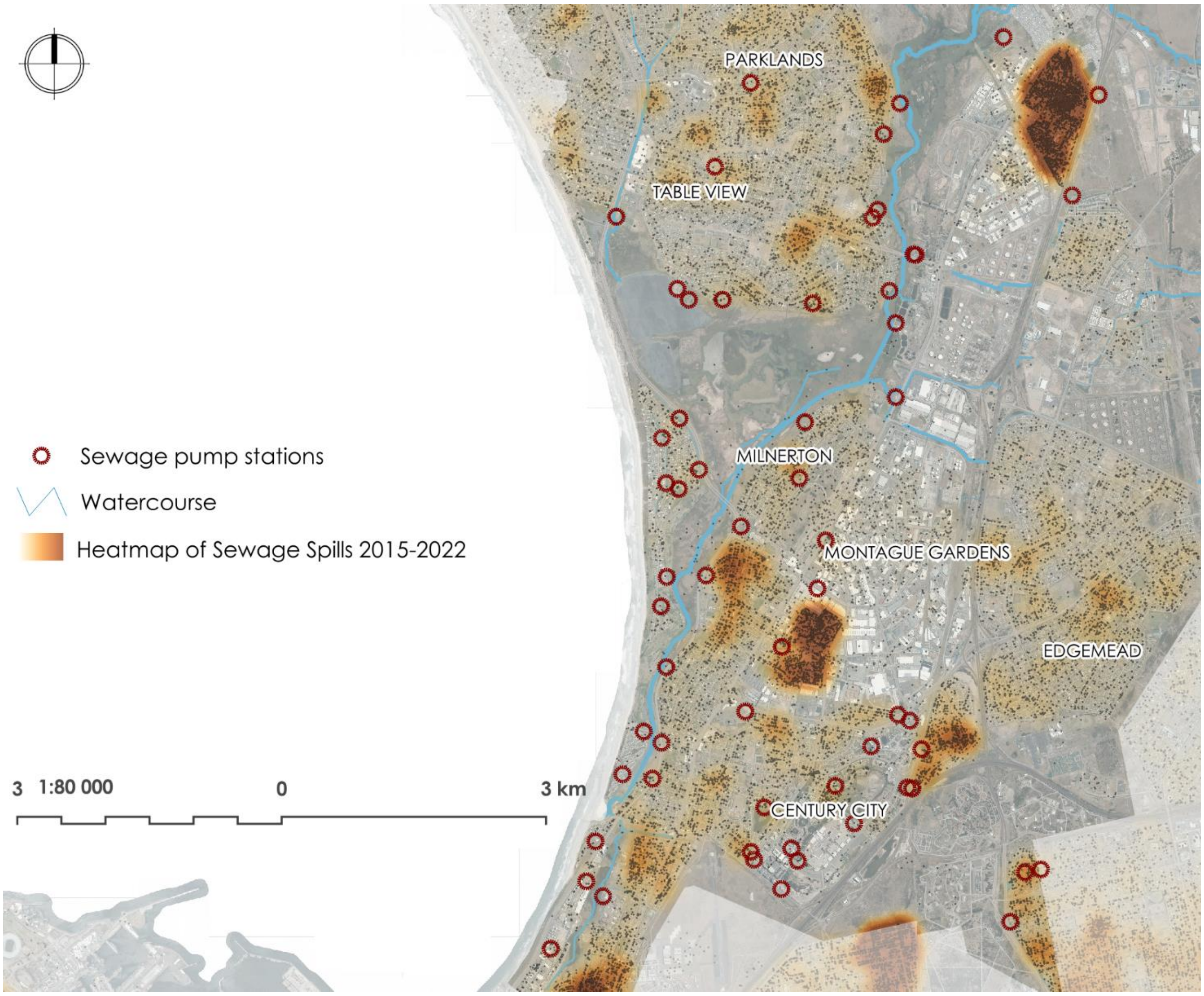


Figure 3-3. 'Heat map' of sewage spill and overflow events recorded in the catchment between 2015 and 2022

3.5 Polluted runoff from informal and backyard settlements

The Diep River catchment as a whole includes extensive areas of informal settlements, both within and outside of formal housing areas (see Section 2.2). The largest of these comprise the Du Noon area and Joe Slovo. In the latter, informal dwellings outnumber formal dwellings by three or four times.[§§] Backyard dwellings and other forms of informal densification, often with limited access to formal sewerage services and in areas where the sewer network has insufficient capacity to manage the daily flow, also occur within many areas of Phoenix, Brooklyn and other parts of the catchment. In addition, levels of solid waste collection in these areas are poor, compared with the volume of solid waste generated by a large, poorly resourced population. Security concerns further hamper collection of refuse in some areas. Solid waste thus accumulates in the streets and is also disposed of by residents into the sewers and stormwater system. As a result, runoff discharged from these areas includes permanent discharges of sewage and other waste, and stormwater after rainfall events flushes further contamination from a dirty catchment area. This runoff discharges into the Milnerton

[§§] Cerfonteyn and Day (2009) estimated at that time that population in the Joe Slovo area was at least three times that of its services design capacity, and this number is likely to have increased considerably.

Lagoon from upstream of Blaauwberg Road (Du Noon area) and via other stormwater inlets, including the Erica Road stormwater system, described in Section 2.5.7.

Runoff from these areas certainly contributes to the poor condition of Milnerton Lagoon, with the passage of raw sewage into these watercourses in particular reflecting unacceptable management practice. Diversion of low flows into the sewers and thence to the WWTW is clearly a necessary intervention – and one that was recommended and designed by Cerfonteyn and Day (2009). No actual interventions other than recent over-pumping at the Erica Road discharge point have however been undertaken since then, and the source area of untreated sewage has simply expanded. Designs are however currently being considered for a permanent low-flow diversion between the Erica Road pipeline and the sewer on the eastern (upstream) side of Otto Du Plessis Road, as well as upstream of the Milky Way pump station (Mr De Wet, pers. comm to Liz Day).

Importantly, clean river runoff and wash off from clean parts of the catchment should be separated from polluted wash-off areas and point source inflows – this means that stormwater diversions to sewers need to be in place as close to source as possible. Diversion of the Kleine Stink River around the Du Noon area would also decrease the volume of effluent requiring diversion and thus increase the efficacy of such interventions. While there is clear urgency to proceed with implementation of these measures, it is also clear that as long as the WWTW continues to perform as poorly as it does at present, implementation of these measures is unlikely to result in any immediate measurable ecological benefit.

Diversion of low flows from the Milky Way pond and Du Noon into sewer is a necessary intervention but not capable of rapid implementation and would not provide immediate relief to the lagoon ecosystem. The low flow diversion at Erica Road outlet is in progress and programmed for completion by the end of June 2023.

3.6 Other sources of polluted runoff

The previous sections highlighted the key point sources of pollution into the Milnerton Lagoon. There are however numerous other land uses and practices that contribute to ongoing pollution of downstream water resources and aquatic ecosystems, including the wetland, estuarine and marine environments. These were identified and to some extent quantified in the catchment-wide study by Cerfonteyn and Day (2009). Since then, there appear to have been few interventions that address these issues, and it is likely moreover that these sources have been exacerbated since then. Overflows from problem pump stations and low-lying sewer manholes in the vicinity of the wetlands and other watercourses appear particularly problematic.

While a catchment-scale pollution abatement strategy is clearly required and recommended, such interventions will not bring about immediate relief to the condition of the Milnerton Lagoon or facilitate the efficacy of short-term in-lagoon interventions considered in this report.

4 LEGISLATIVE CONTEXT

4.1 Environmental legislation regulating pollution

The Constitution of the Republic of South Africa, 1996 establishes a right to 'an environment that is not harmful to … health or well-being', and requires that the environment be 'protected, for the benefit of present and future generations, through reasonable legislative and other measures that –

i. prevent pollution and ecological degradation;
ii. promote conservation; and
iii. secure ecologically sustainable development and use of natural resources while promoting justifiable economic and social development.'

The environmental right is given effect by a suite of environmental legislation including the overarching National Environmental Management Act, 1998 (NEMA), and the Specific Environmental Management Acts regulating various aspects of environmental management.

Pollution control and remediation are central to the principles of environmental management set out in the NEMA, which include requirements that –

» pollution and degradation of the environment are avoided, or, where they cannot be altogether avoided, are minimised and remedied.
» the environment is held in public trust for the people, and the beneficial use of environmental resources must serve the public interest and the environment must be protected as the people's common heritage.
» the costs of remedying pollution, environmental degradation and consequent adverse health effects and of preventing, controlling or minimising further pollution, environmental damage or adverse health effects must be paid for by those responsible for harming the environment
» sensitive, vulnerable, highly dynamic or stressed ecosystems, such as coastal shores, estuaries, wetlands, and similar systems require specific attention in management and planning procedures, especially where they are subject to significant human resource usage and development pressure.

These principles are applicable to the City of Cape Town's management of the Diep River estuary, and the Milnerton Lagoon specifically.

4.2 Duty of care

Section 28 of the NEMA establishes a general 'duty of care' for the environment, stating that:

> *'Every person who causes, has caused, or may cause significant pollution or degradation of the environment must take reasonable measures to prevent such pollution or degradation from occurring, continuing, or recurring, or, in so far as such harm to the environment is authorised by law or cannot reasonably be avoided or stopped, to minimize and rectify such pollution or degradation of the environment.'*

This legal duty is imposed on persons in control of land or premises, who are required to take 'reasonable measures' to prevent and control pollution. These reasonable measures may include measures to 'remedy the effects of the pollution or degradation'.

The National Water Act, 36 of 1998, which was promulgated to provide for the protection of the quality of water resources also provides for a duty of care. Section 19 requires that –

> *'An owner of land, a person in control of land or a person who occupies or used the land on which –*
>
> *(a) Any activity or process is or was performed or undertaken, or*
>
> *(b) Any other situation exists,*
>
> *which causes, has caused or is likely to cause pollution of a water resource, must take all reasonable measures to prevent any such pollution from occurring, continuing or recurring.'*

The measures referred to may include measures to—

(a) cease, modify or control any act or process causing the pollution;
(b) comply with any prescribed waste standard or management practice;
(c) contain or prevent the movement of pollutants;
(d) eliminate any source of the pollution;
(e) remedy the effects of the pollution; and
(f) remedy the effects of any disturbance to the bed and banks of a watercourse.

These legal instruments impose a clear duty on the City of Cape Town to prevent pollution of the Milnerton Lagoon, and where such pollution or degradation have already occurred, to remedy the effects thereof by remedial measures.

4.3 Diep River Estuarine Management Plan

An Estuarine Management Plan for the Diep River Estuary (City of Cape Town and Infinity Environmental, 2022) was adopted by the City's Council in December 2022 and approved by the provincial MEC in in terms of Section 9 of the National Estuarine Management Protocol in April 2023. The plan acknowledges the systemic pressures arising from multiple land uses within the wider catchment area of the Diep River and adopts a transversal approach towards addressing these pressures. A set of specific objectives, for water quality, ecology, hydrology and other factors is established and a total of 47 priority actions are set out in the EMP for the 2022-2026 period. Many of the actions identified in the EMP relating to the water quality of the Milnerton Lagoon are further investigated and assessed in this Remediation Plan.

4.4 Authorisations

4.4.1 Existing approvals

The Potsdam WWTW holds a **water use licence** from the Department of Water and Sanitation in terms of section 40 and section 21 of the National Water Act, 1998. The licence permits the discharge of effluent treated to specified water quality standards to the Diep River, and the onsite management of sludge. The licence conditions also stipulate a single discharge point for final effluent, and require notification of the Department of Water and Sanitation should there be any deviations from licence conditions.

The City of Cape Town has an approved **Maintenance Management Plan** (MMP) for maintenance activities in the Diep River estuary. The MMP for the Diep River estuary, adopted by the Department of Environmental Affairs and Development Planning in terms of the Environmental Impact Assessment

Regulations, 2014 (as amended), allows the City,and its appointed contractors to undertake a range of routine stormwater maintenance tasks within the EFZ, including opening of the estuary mouth and maintenance dredging, without the need for environmental authorisations.

4.4.2 New approvals required

Statutory approvals may be required for implementation of some of the remediation options set out in this report, as set out in sections 5 through 11. These include the following:

Environmental authorisation may be required in terms of section 24 of the NEMA and the Environmental Impact Assessment Regulations, 2014 (as amended by GNR 326 of 2017). Listed activities defined in three Listing Notices (GNR 324, 325 and 327 of 2017) require environmental authorisation, which will be granted or refused by the competent authority based on the findings of an environmental impact assessment (EIA). An EIA may be conducted as a Basic Assessment,[***] with a duration of 5 to 8 months, or a Scoping and EIA, with a duration of 9 to 12 months, depending on the activities to be undertaken. The competent authority will be the provincial Department of Environmental Affairs and Development Planning unless activities occur below the low water mark of the sea, in which case the national Department, Fisheries and the Environment is the competent authority.

Water use authorisation may be required in terms of sections 21, 39 and 40 of the National Water Act, 1998, for water uses defined as 'altering the bed, banks, course or characteristics of a watercourse', and/or 'impeding or diverting flow in a watercourse'. As an estuary, the Milnerton Lagoon may or may not be considered a 'watercourse' and is the subject of overlapping legislative instruments. It is likely that the upper reaches, especially the channel upstream of the Otto du Plessis Bridge, would be considered a watercourse, while the lower lagoon is an 'estuary' not subject to the National Water Act. If a water use authorisation is required, it may take the form of a water use **licence,** or it may be a **generally authorised** water use in terms of the general authorisations published for section 21 (c) and (i) uses – namely, GN 1198 of 2009, which authorises government departments to rehabilitate wetlands, or GN 509 of 2016, which authorises water uses assessed to pose a low risk to the water resource.

A coastal waters discharge permit may be required in terms of section 69 of the National Environmental Management: Integrated Coastal Management Act, 24 of 2008, for one of the remediation options proposed (refer to section 8).

Section 14.1.9 summarises the listed activities and associated authorisation requirements in respect of the proposed remediation options. Section 15.3 describes potential alternative mechanisms for authorisation and implementation.

[***] A Basic Assessment is a regulated process that assesses potential environmental impacts, obtains input from stakeholders, landowners, government departments, and members of the public. The Basic Assessment Report must detail mitigation measures and consider the relative impacts of alternatives to the proposed development. A recommendation is made to the competent authority whether to grant authorisation. An Environmental Management Programme to manage and monitor the impacts of the development must be compiled. Public participation includes notifications to neighbours and government departments, and media advertisements.

5 REMEDIATION OPTION 1: DREDGING

5.1 Scope and purpose

The bed of the Milnerton Lagoon is covered with a layer of organic, nutrient-rich fine sediment originating from the catchment and from poorly treated effluent. The material, referred to in this report either as 'nutrient-rich sediment' or as 'sludge' is oxygen-poor and contributes to poor water quality and odour in the lagoon.

The purpose of dredging the Milnerton Lagoon is simply to remove these settled sediments from the lagoon for disposal or reuse elsewhere. Dredging may also accomplish other hydrodynamic outcomes, including improving exchange of seawater at the mouth. Dredging will not, however, reduce the inflow of pollutants and sediment from upstream and cannot be seen as more than a temporary intervention in the absence of catchment interventions and improvements to the WWTW functioning.

This section presents the quantification of the contaminated material, the physical characterisation of the material to be dredged, the volumes to be dredged to remove the contaminated material and the costs for dredging, dewatering and disposal of the dredged material as well as an estimate of the time required for the dredging operations. Recommended dredging and dewatering methods are also presented.

5.2 Historic and present bed bathymetry and bed material

The last time dredging was carried out in Milnerton Lagoon (between the mouth and Woodbridge Island concrete bridge) was in the 1980's with the development of Woodbridge Island development – refer to Figure 5-1. A maximum dredging depth of approximately -1.5 metres above sea level (masl) in local zones was recorded in the 1987 bathymetric survey by the CSIR (1987).

The natural flushing of the estuary by floods (say floods larger than 50 m^3/s – see Figure 5-3 for an example of an approximate 100 m^3/s flood) from the Diep River catchment has not occurred since winter 2012 (Figure 5-2).

Sediment and sludge have accumulated in the estuary to an approximate mean level of +0.2 masl in the lower estuary (downstream of the Woodbridge Island bridges) and to an approximate mean of +0.4 masl in the upper estuary (from the Woodbridge Island bridges to approximately 1 700 m upstream of the bridges).

Figure 5-1. Bathymetry of Milnerton Lagoon, 26 June 1987 (CSIR, 1987) showing deeper location of dredged zones (mainly red shaded zones) during construction of Woodbridge Island Development in the 1980's (insert 1985 photograph from Independent Newspapers Archive).

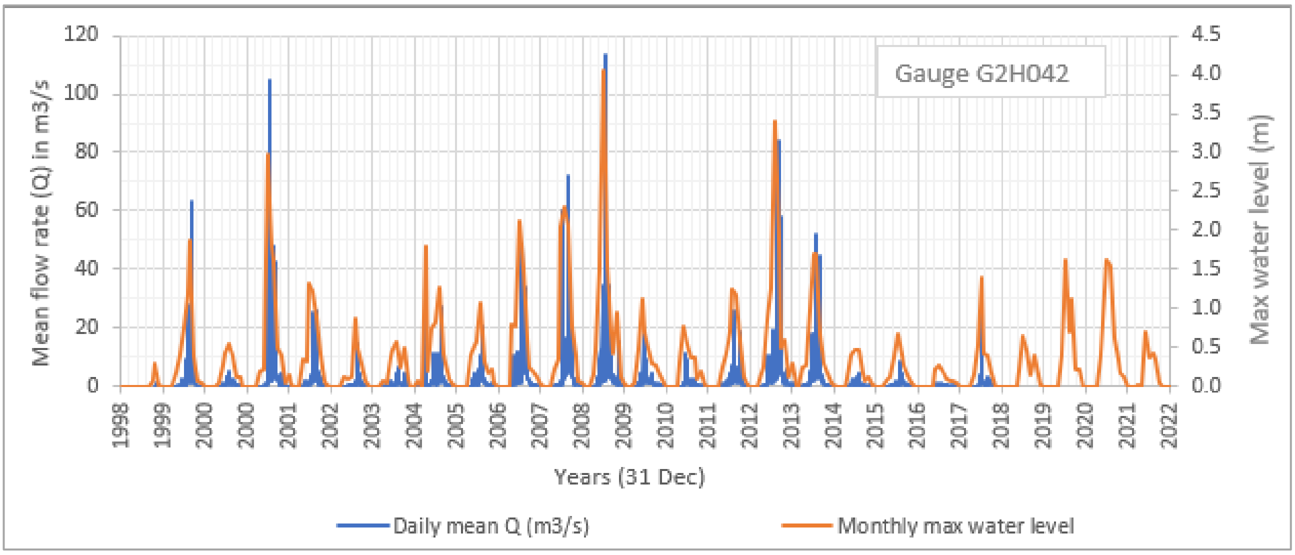


Figure 5-2: Flow history in the Diep River (at DWS gauging station G2H042) since end of 1998 indicating lack of larger floods since winter 2012.

Figure 5-3: Before and after 2001 flood of about 100 m³/s showing scoured lower estuary and estuary mouth.

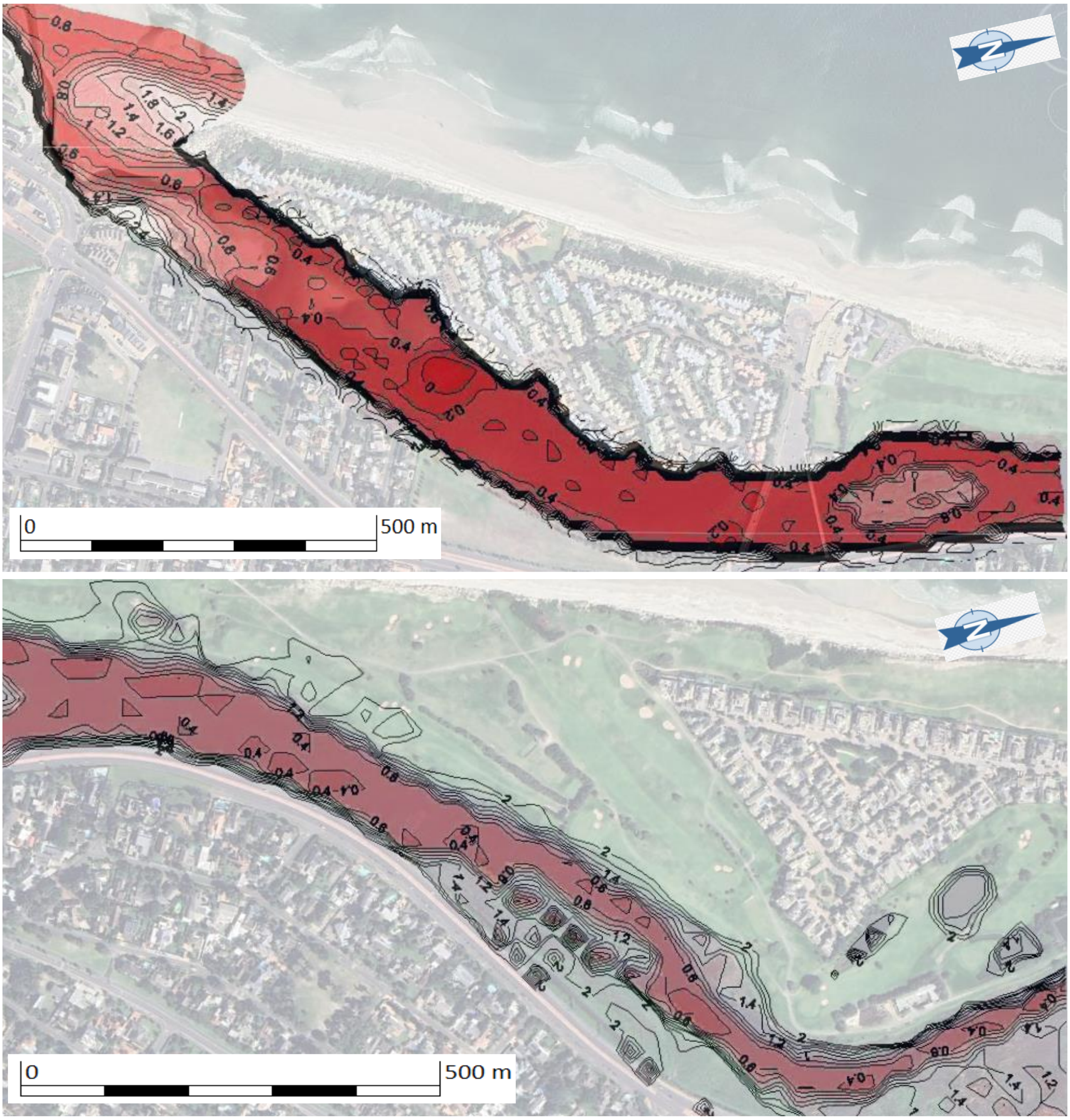


Figure 5-4: Bathymetry of Milnerton Lagoon according to the December 2022 Tritan (2023) survey showing mean bed levels of +0.2 masl in the lower estuary (downstream of the Woodbridge Island bridges) and to an approximate mean of +0.4 masl in the upper estuary (from the Woodbridge Island bridges to approximately 1700 m upstream of the bridges).

Cross sections, showing all available survey data (1993, 1999, 2004, 2021 and December 2022) of the estuary from the mouth to Potsdam WWTW are presented in Figure 5-5 to Figure 5-8. The sludge depths and thicknesses based on probe samples are also shown in these figures in red. The sludge thicknesses obtained from the probes were used to decide on the required dredging depths to remove the contaminated sludge layer.

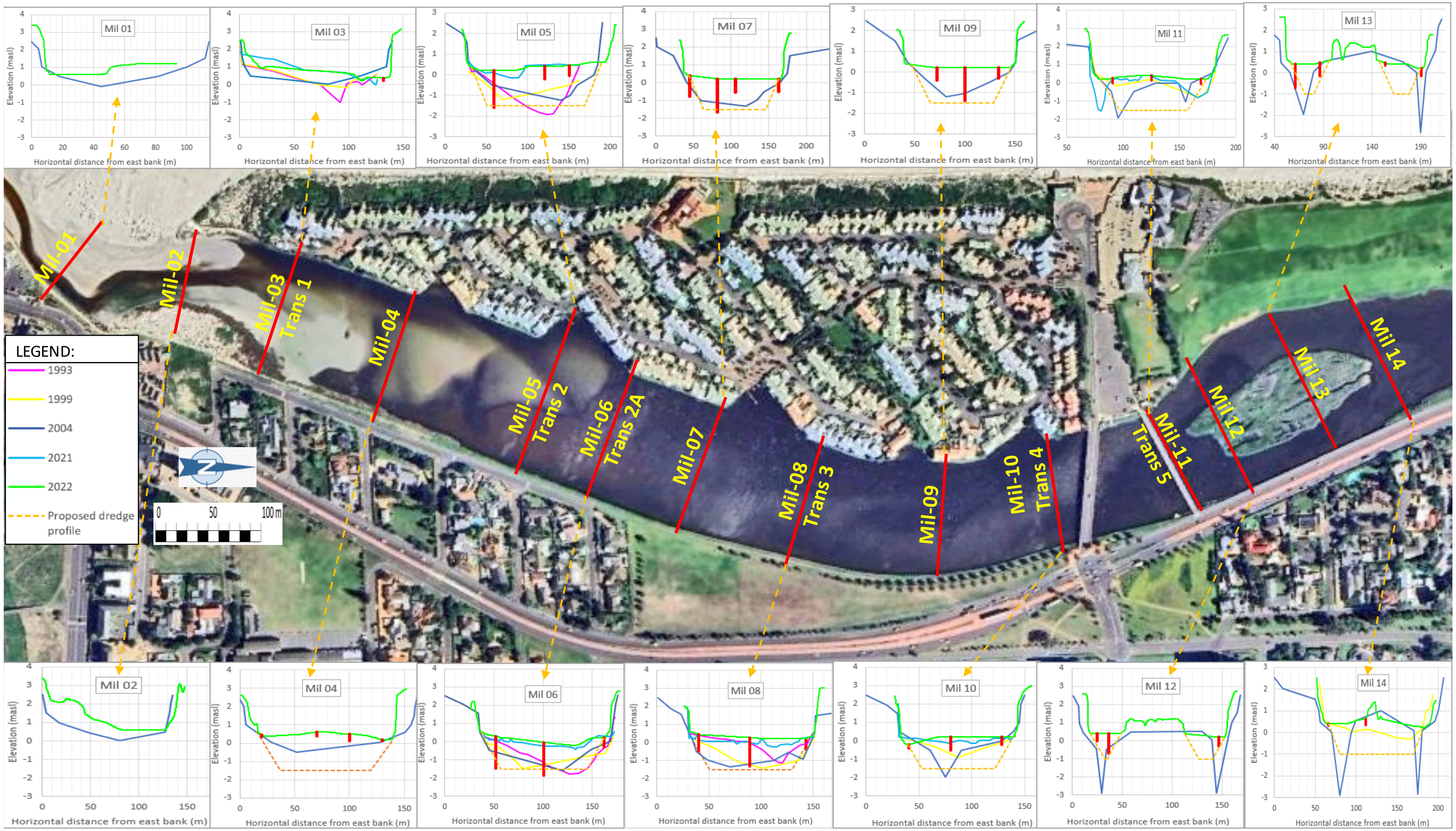


Figure 5-5: Cross section profiles (Mil 01 to Mil 14) obtained from all historic and recent December 2022 Tritan survey. Sludge thicknesses based on the December 2022 core sampling of the bed are shown in red. Tentative dredging profiles with 1:10 side slopes are also indicated.

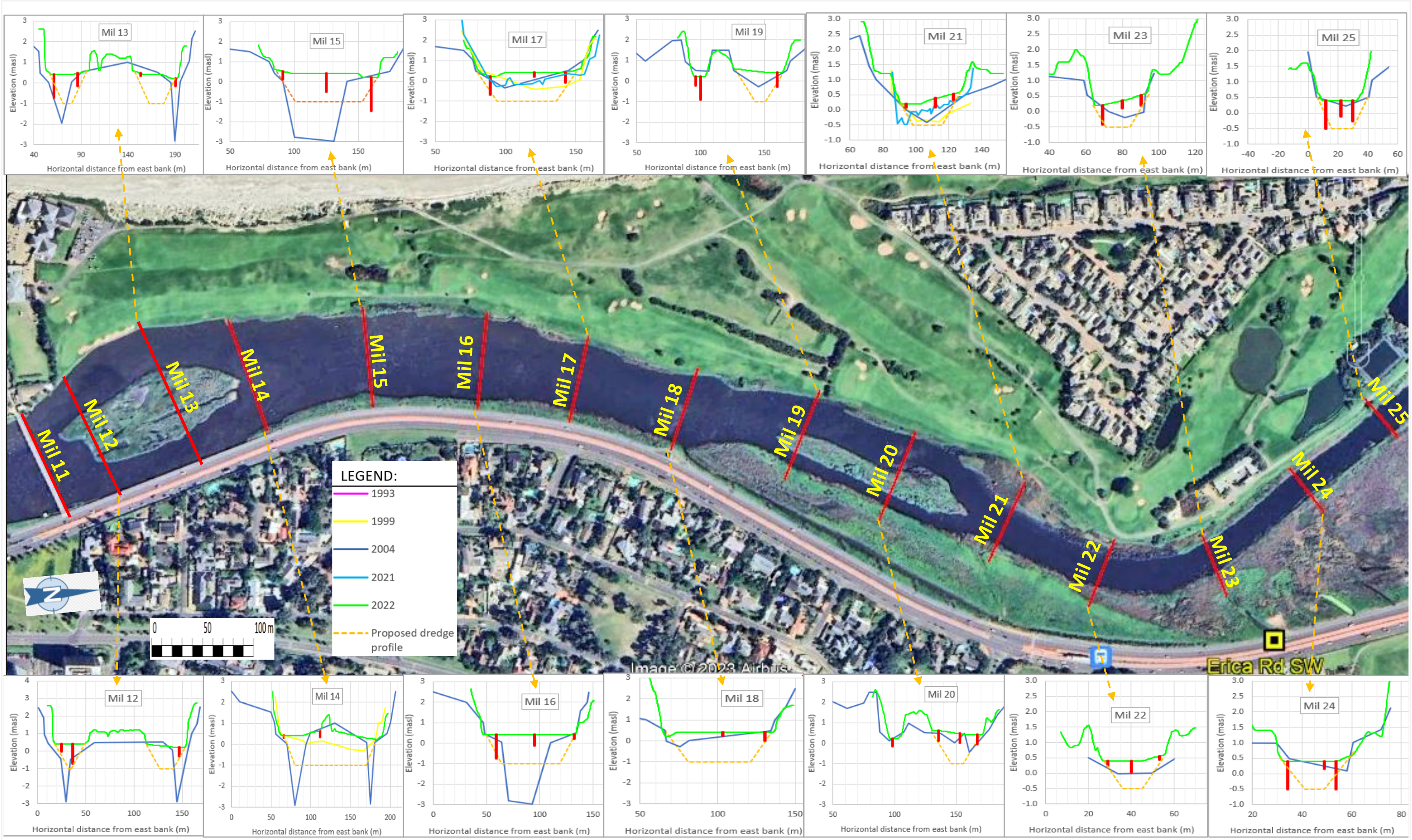


Figure 5-6: Cross section profiles (Mil 12 to Mil 25) obtained from all historic and recent December 2022 Tritan survey. Sludge thicknesses based on the December 2022 core sampling of the bed are shown in red. Tentative dredging profiles with 1:10 side slopes are also indicated.

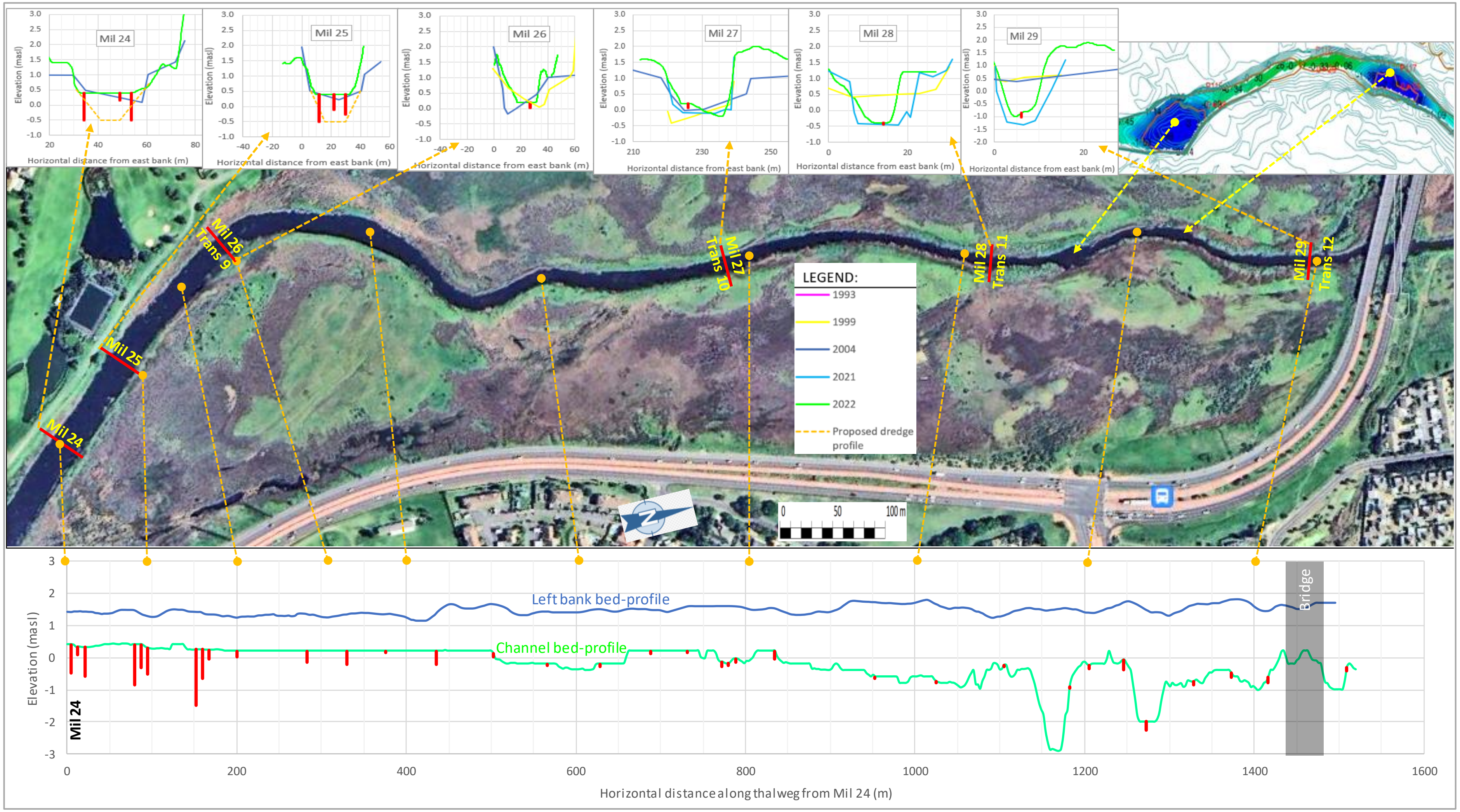


Figure 5-7: Cross section profiles (Mil 24 to Mil 29) obtained from all historic and recent December 2022 Tritan survey. Sludge thicknesses based on the December 2022 core sampling of the bed are shown in red.

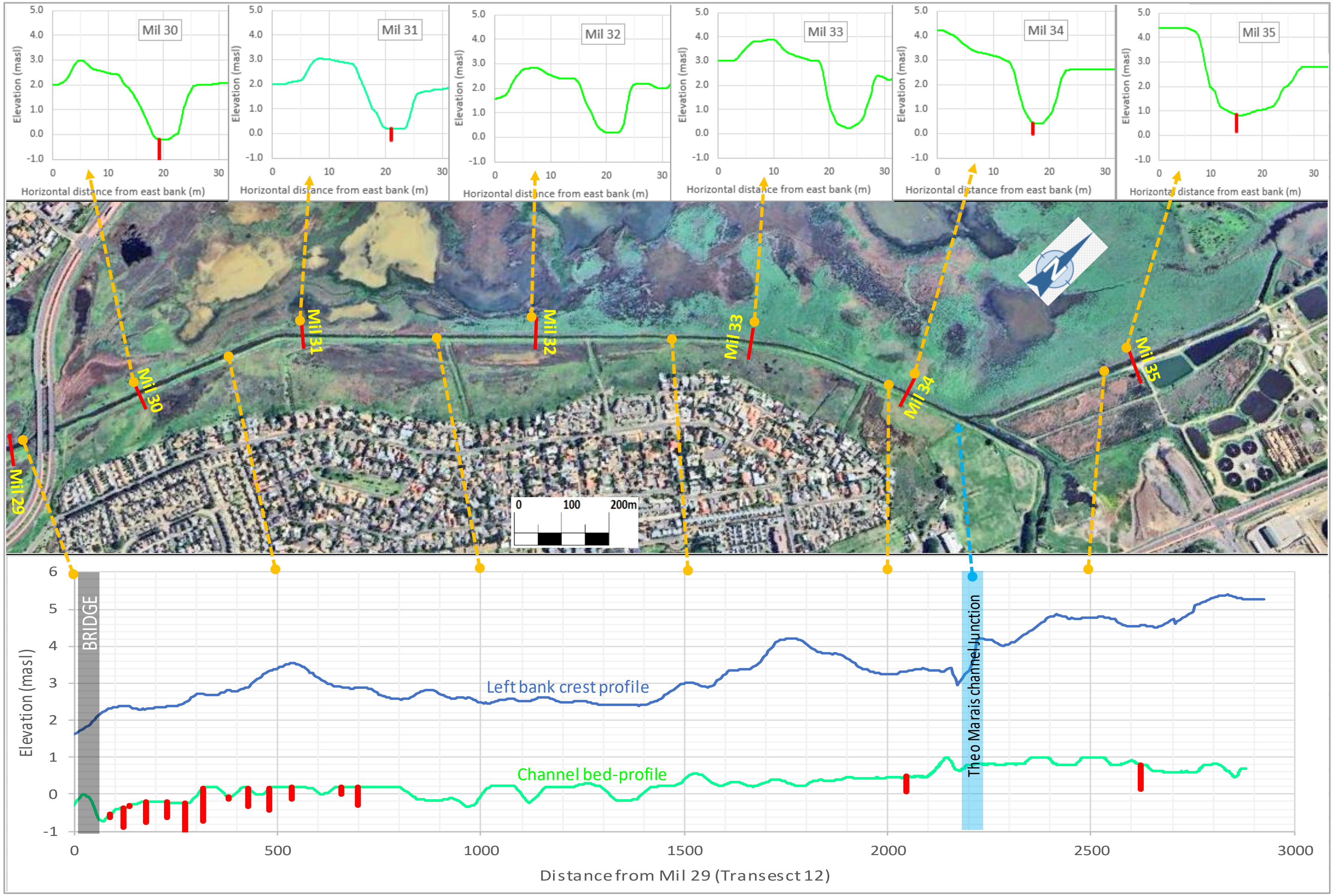


Figure 5-8: Cross section profiles (Mil 30 to Mil 35) obtained from the recent December 2022 Tritan survey. Sludge thicknesses based on the December 2022 core sampling of the bed are shown in red. Tentative dredging profiles with 1:10 side slopes are also indicated.

5.2.1 Dredge material grain size and moisture content

To characterize the nutrient-rich sediment on the bottom of the Milnerton Lagoon for the purpose of removing the sediment by dredging, core samples were taken at locations as indicated in the figure in Figure 5-9. The particle size distribution (PSD) and specific gravity (SG) values of the sampled sediment are also shown in Figure 5-9. Refer to **Annexure C** for full data.

	D50 (mm)	Clay (%)	Silt (%)	Sand (%)	Gravel (%)	SG
DR01-B	0.3	1	0	97	2	2.656
DR01-T	0.3	1	0	92	7	2.684
DR02-B	0.42	1	0	93	6	2.625
DR03-T	0.32	1	3	90	6	2.6445
DR03-B	0.5	1	1	93	5	2.68
DR04-T	0.55	1	1	67	31	2.632
DR05-T	1	1	0	57	42	2.652
ML01-B	0.2	1	2	95	2	2.677
ML01-T	0.18	1	0	99	0	2.635
ML02-B	0.45	1	0	90	9	2.584
ML02-T	0.42	1	0	93	6	2.642
ML03	0.6	1	1	94	4	2.59
ML03-B	0.6	1	1	86	12	2.612
ML04-T	0.65	1	0	87	12	2.632
ML04-B	0.7	1	0	91	8	2.684
ML05	0.5	1	0	97	2	2.596

Figure 5-9. Particle size distributions and specific gravity values for core sampled sediment (December 2022). '-T' denotes samples from the top of the core sample, while '-B' denotes samples from the bottom.

The analysis of the sediment indicated the grain size to be predominantly in the sand fraction. The photographs taken of some of the sediment core samples (Figure 5-10) indicate the sediment to be a mixture of sand and black, sludge-like material whose consistency varies from a fluid (Figure 5-10(a) and (b)) to a more cohesive material such as in Figure 5-10(d). The core shown in Figure 5-10 (e) shows the black sludge-like material also occurs in layers, separated by sandy material.

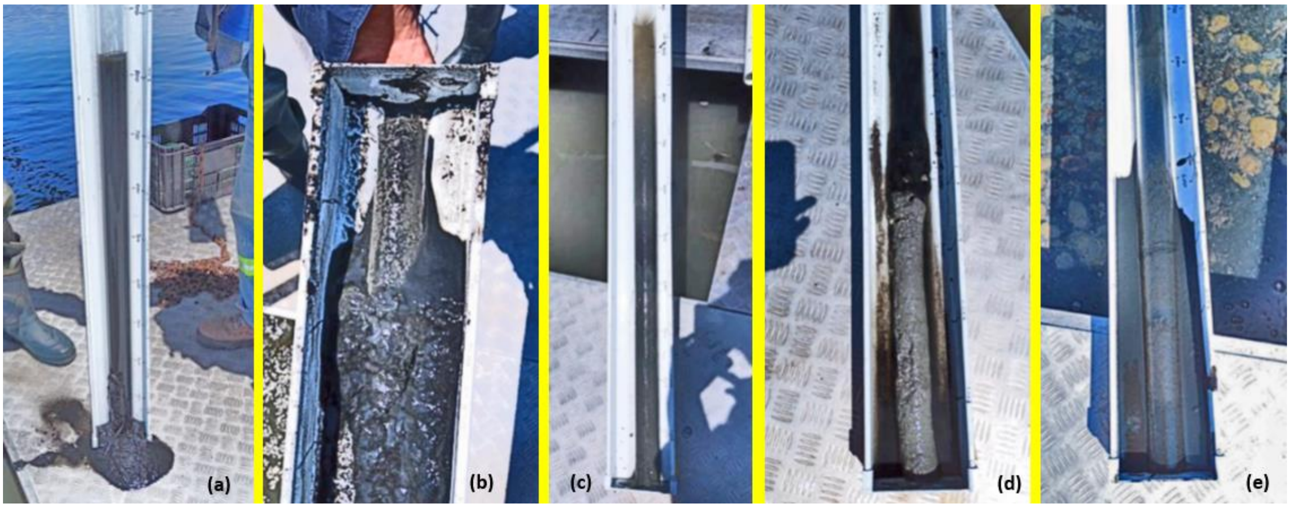


Figure 5-10: Photographs of some of the core samples of the Milnerton Lagoon bed material

Additional samples of the bed material are shown in Figure 5-11 with their respective values of D50, silt & clay and moisture content. These samples confirm the bed material to be a mixture of sand (lighter material) and sludge (black material).

Figure 5-11: Additional samples of sludge material taken on the south side (A), between (B) and on the north side (C) of the Woodbridge Island bridges

Based on Tritan Survey's bathymetric surveys (Figure 5-4) and bed probe sampling (Figure 5-5, to Figure 5-8) during December 2022/January 2023 (Tritan, 2023) a sludge thickness isopach was derived by Tritan as shown in Figure 5-12 with an estimated nett sludge volume of 136 550 m³.

Sludge thickness (m)			Volume (m3)	% of total
0	to	0.25	44 347	32.5
0.25	to	0.5	31 719	23.2
0.5	to	1	38 333	28.1
1	to	2	22 151	16.2
Total			**136 550**	

Figure 5-12: Tritan Survey's derived sludge isopach contours of sludge thicknesses and estimated nett sludge volumes per thickness ranges. Sample locations for the additional samples shown in Figure 5-11 are indicated.

5.3 Dredging phases, volumes, cost, and implementation time

To provide for the possibility of not having sufficient funding available to implement all the required Milnerton Lagoon dredging in one contract, the areas to be dredged have been divided into three areas/phases. The priority phase (Phase 1) is between the estuary mouth and Woodbridge Island wooden bridge. The three areas as shown in Figure 5-13. Figure 5-14 to Figure 5-16 show Phases 1 to Phase 3a in more detail. Phase 1 dredging could commence with the dredging of a channel in the mouth to lower the sill in the mouth (similar to the effect of the 2001 flood – refer Figure 5-3) and to enable possible flushing of a portion of the contaminated sludge in the lower estuary as well as to enable a larger tidal exchange for a limited period, until the equilibrium sill level has regained (usually in a relative short period). The three phases could be implemented either in one contract or separately depending on available funds – the most cost-effective way would be to implement all three phases in one contract.

The decision on dredge depths for the calculation of dredge volumes were based on the core samples taken from the lagoon bed, as shown in Figure 5-5 to Figure 5-8. On the edges of the dredging perimeters as shown in Figure 5-14 to Figure 5-16 a dredging slope of 1:10 has been allowed for to ensure the stability of the existing lagoon banks. The proposed method of dewatering the dredge material is to pump it into Geotextile tubes. The proposed dewatering sites are discussed in Section 5.4. Drained water from the Geotextile tubes will flow back into the lagoon with due cognisance of the quality of the effluent with respect to adherence to regulatory requirements. For the disposal of the dewatered silt/sludge, it has been assumed that it could be beneficially used as capping material or disposed of at the Vissershok waste disposal site. The estimated costs for the three phases implemented either by three separate contracts or through a single contract are shown in Table 5-1.

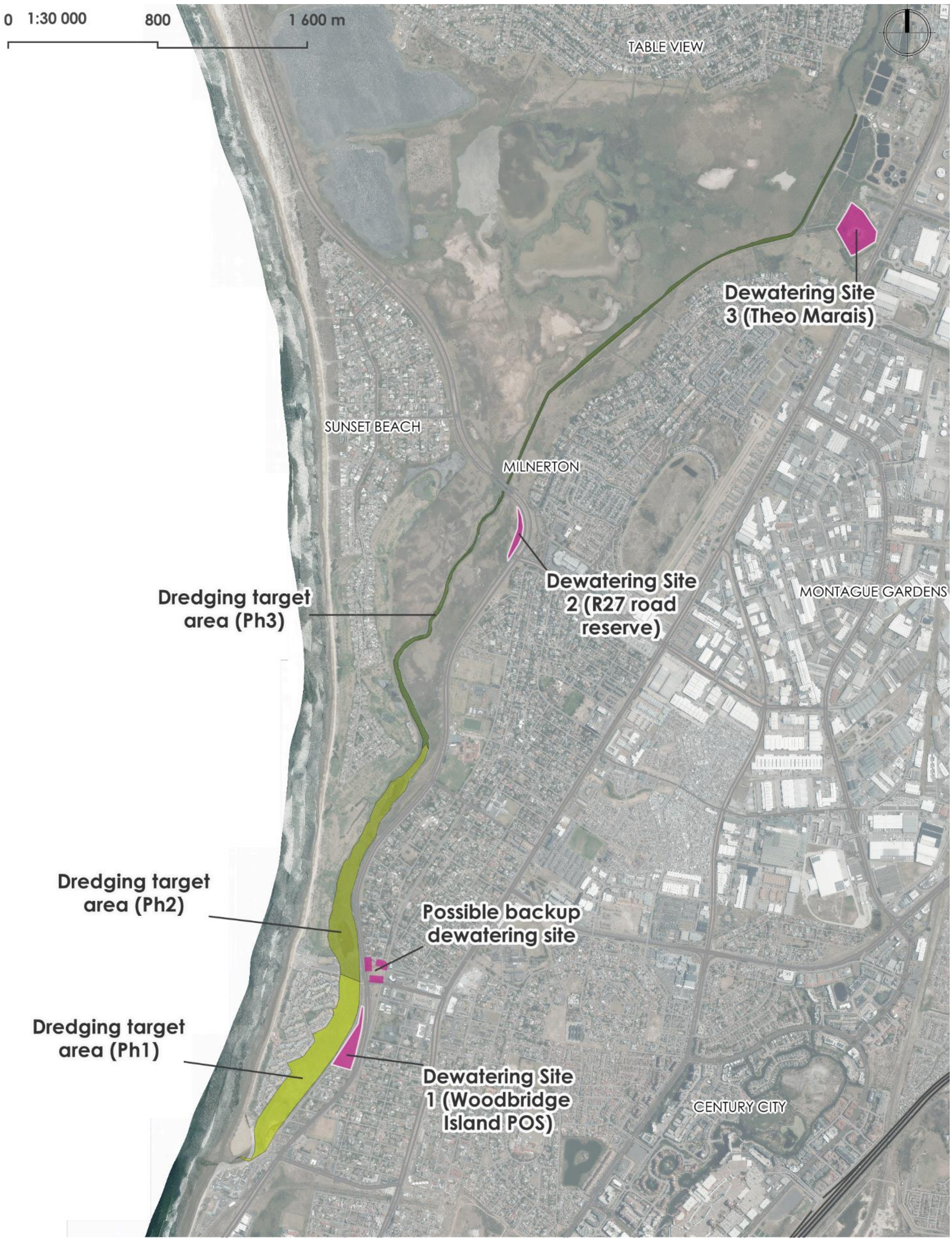


Figure 5-13: Proposed three dredging areas/phases for the removal of contaminated sludge from Milnerton Lagoon.

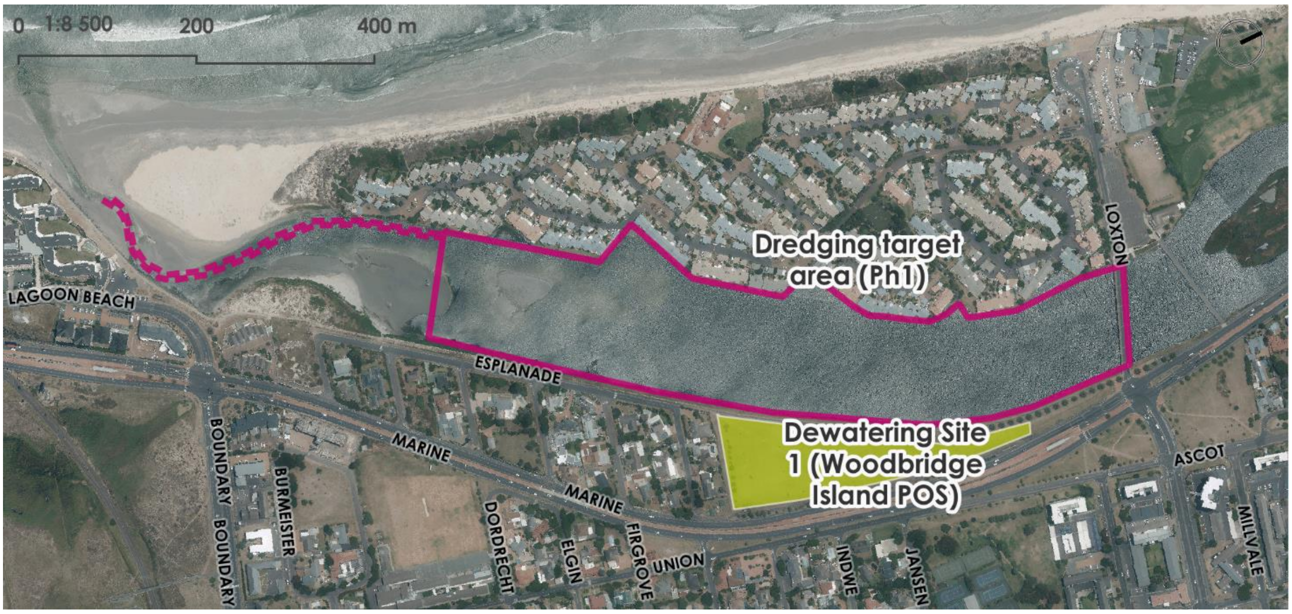


Figure 5-14: Proposed Phase 1 dredging area. The proposed first dredging action is the dredging of a canal in the mouth area (dotted line).

Figure 5-15: Proposed Phase 2 dredging for the removal of contaminated sludge.

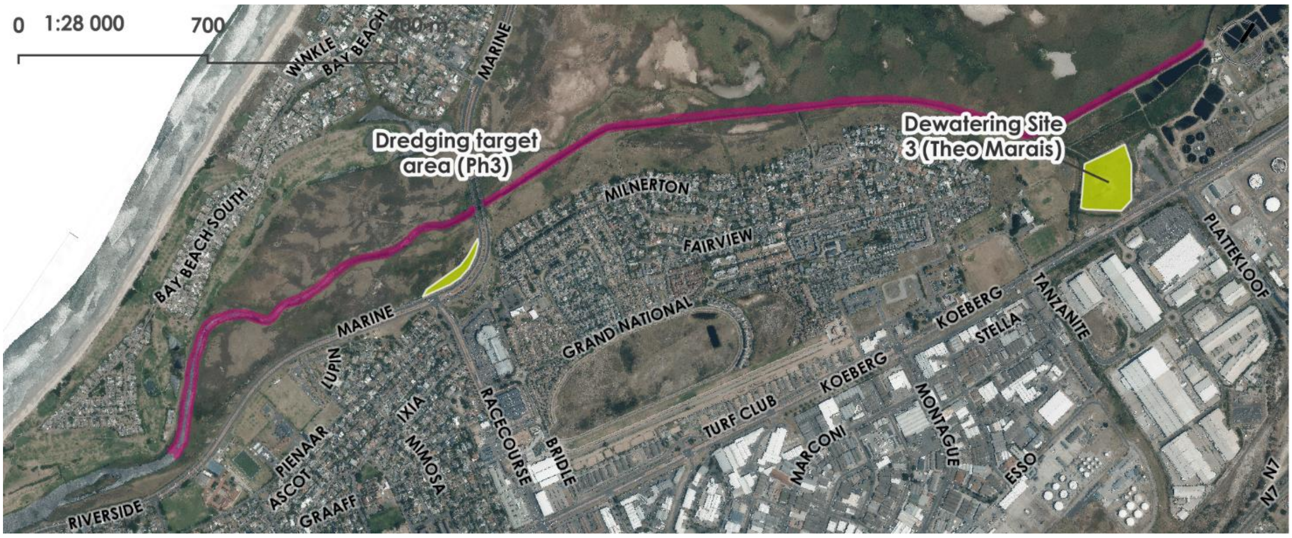


Figure 5-16: Proposed Phase 3a dredging for the removal of contaminated sludge.

Table 5-1: Dredge volumes, dewatering areas, contract durations and high level cost estimates for dredging.

Item	Volumes, Dredging Costs and Durations	Unit	Phase 1 (Below Woodbridge)	Phase 2 (Above Woodbridge)	Phase 3 (Diep River Canal)	Phase 1 + Phase 2	All Areas
1	**General Details**						
1.1	Dredge Volumes (insitu volume)	m^3	100 800	70 200	20 800	171 000	191 800
1.2	Dredge plan area	m^2	84 000	78 000	26 000	162 000	188 000
1.3	Dredge Depth	m MSL	-1	-0.5	-0.4		
1.4	Average dredge thickness	m	1.2	0.9	0.8	1.1	1.0
1.5	Dewatering platform area	m^2	14 000	12 000	6 000	14 000	18 000
1.6	Approx. dewatering platform volume	m^3	22 400	19 200	9 600	22 400	28 800
1.7	Filling cycles required for dredge volume	No.	5	4	2	8	7
1.8	Number of geotextile tubes required (730m^3/tube)	No.	138	96	28	234	263
1.9	Average dredge rate (insitu sludge vol)	m^3/month	18 000	18 000	12 000	18 000	16 000
2	**Contract Duration**						
2.1	Appointment, Site and equipment establishment	Months	4	4	4	4	4
2.2	Dredge duration	Months	6	4	2	10	12
2.3	Lag in the dewatering and offsite disposal	Months	2	2	2	2	2
2.4	Demobilisation	Months	1	1	1	1	2
	Total Dredging Contract Duration	Months	**13**	**11**	**9**	**17**	**20**
3	**Cost Estimate**						
3.1	Fixed P&G's (incl. Mob & Demob of equipment)		R14 170 000	R14 260 000	R11 230 000	R15 170 000	R16 990 000
3.1	Time related P&G's		R5 670 000	R4 905 000	R3 930 000	R7 425 000	R8 544 000
3.2	Dredge and pump slurry to dewatering site		R20 200 000	R14 000 000	R4 200 000	R34 200 000	R38 400 000
3.3	Dewatering of dredge slurry		R14 700 000	R10 300 000	R3 000 000	R25 556 000	R28 000 000
3.4	Off-Site Disposal (Vissershok charge not included)		R15 400 000	R10 600 000	R3 200 000	R27 360 000	R29 100 000
	Sub- Total		**R70 140 000**	**R54 065 000**	**R25 560 000**	**R109 711 000**	**R121 034 000**
3.5	Contingency (10%)		R7 014 000	R5 407 000	R2 556 000	R10 971 000	R12 103 000
	Total Dredging Cost (excl. VAT)		**R77 154 000**	**R59 472 000**	**R28 116 000**	**R120 682 000**	**R133 137 000**

The dredging contract durations and cost estimates in Table 5-1 are based on the following assumptions:

- Dredging the sediment/sludge with a cutter suction dredger and pumping the slurry to a dewatering site.
- Dewatering the dredge slurry using geotextile tubes. Each tube having a volume of 730 m^3.
- The material will be left to dry in the tubes for approximately one month before it is carted off site.
- The geotextile tubes are sacrificial and will be cut open so the material can be loaded onto trucks and transported to the disposal site.
- All dried dredge material will be transported to and disposed of at the Vissershok disposal site.
- Dredging will be carried out during the daylight hours.
- Monthly dredge volumes average from 12 000 m^3 to 18 000 m^3
- There is very limited shrinkage of the dredge material (i.e. The dried vol is similar to the insitu volume).
- A 10% contingency is included.

Cost estimates include:

- Fixed and time related P&G's including mob and demob of dredging equipment, slurry pipelines, construction of dewatering platforms and site establishment.

Cost estimates exclude:

- VAT
- Professional fees
- Environmental studies and approvals
- Stakeholder and public engagement
- Any charges for disposing of the material at the Vissershok landfill site

5.4 Dredging and dewatering methods

5.4.1 Dredging and dewatering methods

A cutter-suction type dredger is considered the appropriate type for the removal of the contaminated material. The dredger could either be a cutter-suction type dredger that has a revolving crown type cutter head and is manoeuvred by means of two spuds such as that shown in Figure 5-17 or a cutter suction dredger with a cylindrical revolving cutter head (auger type) and is manoeuvred by two adjustable star type under water driving wheels such as the dredger shown in Figure 5-18.

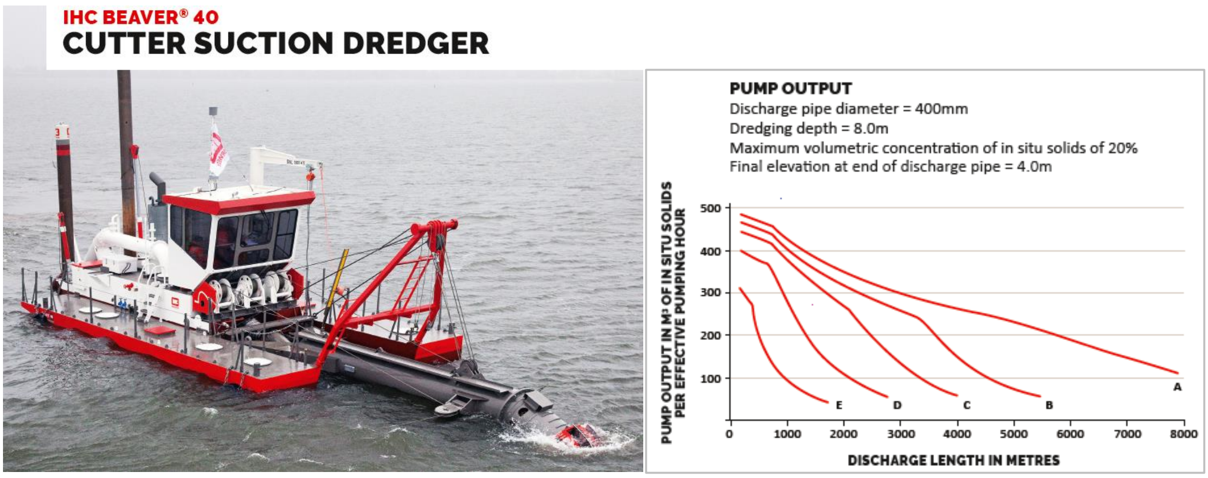


Figure 5-17: Example of a cutter-suction dredger with a crown type cutter head and two spuds for manoeuvring and its production performance curves.

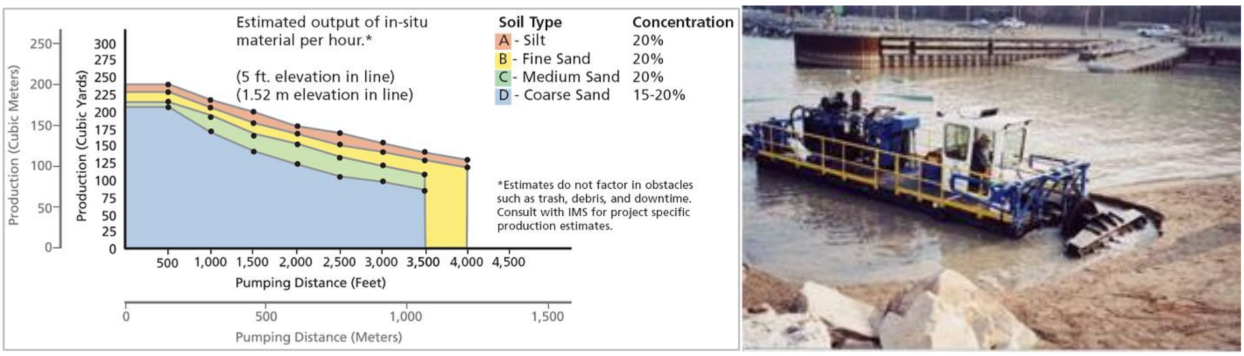


Figure 5-18: Example of a cutter-suction dredger with an auger type cutter head and two adjustable under water star type driving wheels for manoeuvring and production performance curves.

These types of dredgers can be either diesel or electric driven and should be able to pump the dredged slurry to the proposed dewatering locations without a booster pump.

5.5 Proposed Dewatering Method

The three most common methods for dredged material dewatering and containment are:

- The Confined Disposal Facility (CDF) method. This is the conventional method using settling ponds.
- The GeoPool dewatering method. Which essentially consist of free-standing reservoirs with permeable walls lined with nonwoven geotextile to assist with the drainage of the slurry which is pumped into the reservoirs. This is a relatively new method developed in the USA.
- The Geotextile tube method. The dredged slurry is pumped into large bags or tubes made from a permeable woven geotextile. The free water is drained from the slurry and the material consolidates and dries in the bag/tube. This is also a relatively new method developed in the USA in the mid-1990s.

Since the contaminated material on the bottom of Milnerton Lagoon is a sludge type material the Geotextile tube dewatering method is considered the most appropriate method because of its contaminant retaining and sediment consolidation characteristics.

The Geotextile tube method is a slurry dewatering method developed in the 1990's and is especially suitable as an environmentally friendly method to contain contaminated material. It involves the permeable containment of fills and wastes using specially engineered textiles. These specially engineered textiles enable the passage of *water* while at the same time retain the solids component of the dredged material and any contaminants present. The containment units are supplied in a range of shapes and sizes depending on the application. These can be tubular shaped, bag shaped, mattress shaped.

An example of the efficiency of the Geotextile tube dewatering method to retain contaminated material of dredged contaminated slurry is shown in Table 5-2 for a dredging project in the Melaka River in Malaysia.

Table 5-2. Example of Geotextile tube containment efficiency of contaminants of contaminated dredged material (Tencate, 2013)

Contaminant	Raw contaminated sediments	Geotube® filtered effluent water
Phosphorus	5 - 220 ppm	0.05 – 0.9 ppm
Nitrogen	2.5 - 65 ppm	0.3 - 0.6 ppm
TSS	43,000 - 160,000 ppm	2 - 50 ppm
BOD	140 - 360 ppm	2 - 5 ppm
COD	350 - 1,200 ppm	5 - 15 ppm
ppm – parts per million (by weight) TSS – Total Suspended Solids BOD – Biological Oxygen Demand COD – Chemical Oxygen Demand		

The principle of the Geotextile tube application is illustrated schematically in Figure 5-19 below. The dredged slurry is pumped directly in the Geotextile tubes in a sequential manner while dewatering through the geofabric walls of the tube and compaction of the remaining sediment takes place.

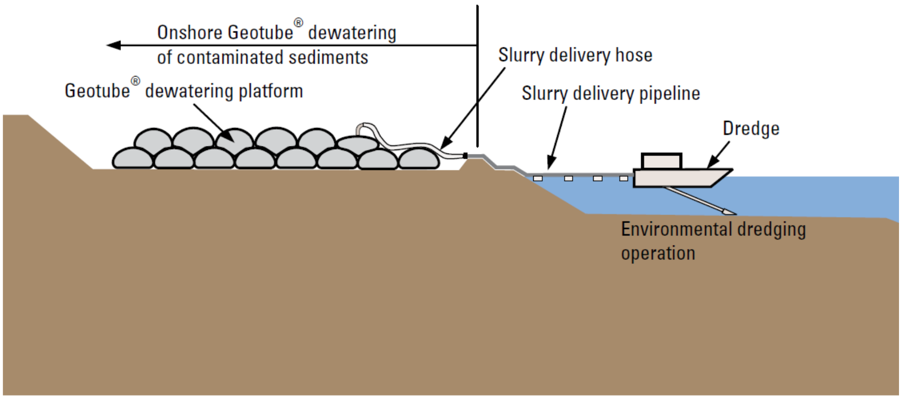


Figure 5-19: Schematic diagram of Geotextile tube application

After filling of the Geotextile tubes (one at a time) in a series of layers, as shown in Figure 5-20, the Geotextile tube mound can then be covered with topsoil and planted.

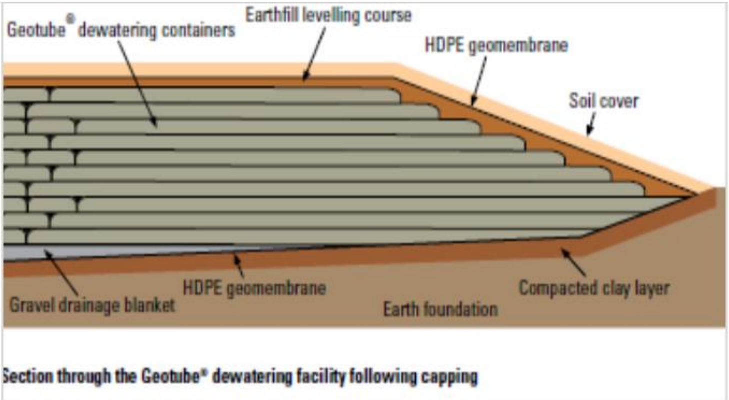


Figure 5-20: An example of a first layer of filled Geotextile tubes (left) and the multi-layering of Geotextile tubes with topsoil cover (right).

An example of the covering of the Geotextile tube mound with topsoil and vegetation afterwards is illustrated in Figure 5-21 below.

Figure 5-21: An example of Geotextile tubes being covered with topsoil (left) and after vegetation has been established (right).

An illustration of the consistency of dewatered sediment (in this case dewatered sewage sludge) is shown in Figure 5-22. There are many existing environmental dredging and remediation projects internationally where the Geotextile tube has been successfully applied. It is recommended that this method should be considered as the preferred method for dewatering the dredged slurry from the main Milnerton Lagoon waterbody. **The reason for recommending the Geotextile tube dewatering method for the Milnerton Lagoon project is its efficient contaminant retaining and sediment consolidation characteristics.**

For the Milnerton Lagoon dredging project it is envisaged that dredged material be dewatered with the Geotextile tube method in suitable dewatering sites shown in Figure 5-23 to Figure 5-25 and that the dewatered material from the Geotextile tubes be hauled to the neatest disposal site (a provisional site is the Vissershok waste disposal site, the proximity of which to the Milnerton Lagoon dewatering sites is shown in Figure 5-26).

Figure 5-22: Excerpt from Fowler et al (2002): An example of sewage sludge dewatered with the Geotextile tube method indicating material consistency – suitable to be transported by truck.

A video illustration of the Geotextile tube operation can be found at the following links:

https://www.youtube.com/watch?v=p3QT_w1mK64 ; https://www.youtube.com/watch?v=SJpuWyYtles

https://www.youtube.com/watch?v=aRm4voQjpfA&t=69s

Figure 5-23: Geotextile tube dewatering site for dredging Phases 1 & 2. Total volume of this layout shown, is approximately 22 000 m³.

Figure 5-24: Geotextile tube dewatering site for dredging the lower part of Phase 3. Total volume of this layout, is approximately 16 800 m³.

Figure 5-25. Geotextile tube dewatering site for dredging Phase 3. Total volume of the layout shown, is approximately 10 000 m³ for each of the two branches shown which is much more than required for Phase 3.

Figure 5-26: Haul route for carting dewatered material from dewatering sites 1, 2 and 3 to Vissershok waste disposal site as a provisional dredged material disposal site

5.6 Legal and regulatory approvals required

5.6.1 National Environmental Management Act, 1998

The National Environmental Management Act (NEMA, Act 107 of 1998) requires an environmental authorisation to be obtained prior to the commencement of certain activities which have been identified as potentially harmful to the environment. These activities are listed in three listing notices published under the NEMA Environmental Impact Assessment Regulations, 2014 (GNR 326 of 2017, as amended). Activities listed in Listing Notice 1 and Listing Notice 3 require a basic assessment as part of the environmental authorisation application process and activities listed in Listing Notice 2 require a scoping and environmental impact assessment as part of the environmental authorisation application process. The competent authority is likely to be the Department of Environmental Affairs and Development Planning.

Listed activities potentially applicable to proposed dredging of the lagoon and Diep River channel, and requiring a Basic Assessment, would include:

- **19 of Listing Notice 1:** The infilling or depositing of any material of more than 10 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from a watercourse
- **19A of Listing Notice 1:** The infilling or depositing of any material of more than 5 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from … an estuary or a distance of 100 metres inland of the highwater mark of the sea or an estuary, whichever distance is the greater
- **12 of Listing Notice 1:** The development of … infrastructure or structures with a physical footprint of 100 square metres or more; where such development occurs— (a) within a watercourse
- **14 of Listing Notice 3:** The development of infrastructure or structures with a physical footprint of 10 square metres or more; where such development occurs— (a) within a watercourse; (b) in front of a development setback; or (c) if no development setback has been adopted, within 32 metres of a watercourse, measured from the edge of a watercourse; **i. Western Cape** i. Outside urban areas: (aa) A protected area identified in terms of NEMPAA.

5.6.2 National Water Act, 1998

In terms of the National Water Act certain water uses require prior authorisation from the Department of Water and Sanitation.

Water uses applicable to dredging include section 21 (i) of the Water Act: "altering the bed, banks, course or characteristics of a watercourse" and 21 (c): "impeding or diverting the flow of water in a watercourse". If the risk posed by a proposed activity is Low (as determined by a freshwater specialist on the basis of the outcomes of the application of a risk assessment matrix), the Department of Water and Sanitation will register a Generally Authorised use, which is typically a three- to six-month process. If the risk is medium or high, a Water Use Licence Application is required, typically involving a 12- to 24-month timeframe. The type of authorisation required, and the prospects of approval are risk-dependent, and therefore design-dependent.

5.6.3 National Environmental Management: Waste Act, 2008

The National Environmental Management: Waste Act (NEM:WA) stipulates that no waste management activity may be undertaken except in accordance with prescribed standards or an issued waste management licence, depending on the activity. Amongst other activities, the importation, exportation, generation, accumulation, storage, collection, handling, reduction, re-use, recycling, recovery, trading, transportation, transfer, treatment and disposal of waste are all included in the scope of waste management activities. In the list published under the NEM:WA which lists the activities which require a waste management licence, a distinction is made between activities which require a basic assessment in the application process and activities which require a scoping and environmental impact assessment in the application process. Additionally, the NEM:WA requires prescribed norms and standards to be adhered to in relation to certain waste management activities which do not require a waste management licence.

The treatment of general waste using any form of treatment is listed under the NEM:WA as a waste management activity which has or is likely to have a detrimental effect on the environment. Depending on the capacity of the waste treatment facility, a waste management licence must be acquired prior to the construction of such a facility, and either a basic assessment or a scoping and environmental impact assessment must form part of the waste management licence application.

Clarity from the competent authority is required as to whether dredged material is to be considered as 'waste' in terms of the Act, and additionally whether the dewatering and handling of such material is to be considered 'treatment' of waste. Should both be the case, it is possible that the proposed dredging and subsequent handling of dredged material may require a waste management licence, including a basic assessment or a scoping and environmental impact assessment. This requirement is believed to be unlikely, but will require confirmation during the pre-application phase of the Basic Assessment to be undertaken for the proposed dredging.

5.6.4 Coastal Waters Discharge

The Department of Forestry, Fisheries and the Environment was consulted in respect of the possible applicability of a coastal waters discharge permit in terms of section 69 of the National Environmental Management: Integrated Coastal Management Act, 24 of 2008 to dewatering of dredged material into the lagoon. Section 69 (1) provides that 'no person may discharge effluent that originates from a source on land into coastal waters' or an estuary without authorisation [emphasis added]. Section 1 of the Act defines effluent as 'any liquid discharged into the coastal environment as waste'. The proposed dewatering will certainly produce runoff / eluent water, but we note that the source of this runoff will be the estuary itself, rather than a source on land. Subsection (6) provides that a person who discharges effluent into coastal waters 'may only do so to the extent that it is not reasonably practicable to return any freshwater in that effluent to the water resource from which it was taken' – the latter being the intention of the proposed dewatering activities. The Department responded on 30 May 2023 to confirm that 'This Department is of the opinion that the proposed activities will not require a Coastal Waters Discharge Permit ("CWDP") in terms of section 69 of the National Environmental Management: Integrated Coastal Management Act, 2008 (Act No. 24 of 2008) (the "ICM Act"). Written authorisation from this Department will thus not be required prior to the commencement of the said proposal.'

5.7 Potential ecological risks and benefits associated with dredging the lagoon sediments

Dredging of the lagoon sediments is likely to present numerous ecological benefits. Once the bulk of the contaminated, organic rich surficial sediment has been removed from the Milnerton Lagoon, the reduction in Sediment Oxygen Demand (SOD) and Biological Oxygen Demand (BOD), and subsequent increase in dissolved oxygen levels in the overlying water column should allow for recolonisation of the estuary by a wide variety of invertebrate and fish species (mostly marine species), which should have large positive knock-on effects, such as restoring food supplies for larger fish and birds.

Potential risks associated with the dredging removal of habitat for benthic species (mostly invertebrates) and mortality of any fauna currently present in the sediments; mobilisation of sediment and associated contaminants (organic matter, trace metals, methane and hydrogen sulphide gas) into the water column causing turbidity plumes which will reduce light penetration, increase Biological Oxygen Demand (BOD) and reduce dissolved oxygen levels in the water column, potentially exacerbating the existing water quality and odour problems. It is also likely that vegetation along the banks and floating in the water will also be disturbed by dredging equipment.

These concerns are seen as fairly minor for several reasons. Firstly, dredging will be a relatively short-term intervention and so negative impacts will not persist for long. The paucity of invertebrate life in the sediments means that disruptions by dredging activities will have a very low impact. Additionally, short-term reductions in DO will likely have minimal impact on local fish and invertebrates, again due to the paucity of fauna in the estuary. Turbidity plumes are also unlikely to have serious consequences, both due to the high current turbidity in the area, as well as the lack of submerged macrophytes living in the dredging areas. The disturbance of vegetation along the banks and floating in the water column are also of relatively low concern, as the banks are densely vegetated with grass and freshwater reeds, which are of low conservation value. Furthermore, the primary floating macrophyte is invasive water hyacinth *Eichhornia crassipes*, which is a major problem in the estuary, and thrives under the continuous eutrophic conditions. Exposure of clean, uncontaminated sediment as a result of dredging operation and subsequent improvements in water quality in the estuary will rapidly compensate for any disturbance created.

In summation, the proposed dredging is predicted to have a substantial positive ecological benefit for the health of the Diep Estuary, with very few associated risks that should terminate once the dredging activities conclude.

5.8 The option of the City purchasing and operating a dredger

There are a few waterbodies within the greater Cape Town metropole area, that require dredging. Due to the high cost and extended durations required for most dredging campaigns, the City has queried the option of purchasing and operating its own dredger. Below are some considerations for a City run dredging operation:

- The high capital cost of purchasing a dredger.
- A dredger requires significant supporting plant and equipment, which may vary between operations. This includes, support vessels, pipelines, booster pumps, generators, refuelling equipment, landside facilities, etc.
- Dredging operations involve a number of activities other than just dredging, which are typically managed by a contractor:
    - Managing of the dredged slurry (e.g. dewatering, handling, transporting, stockpiling, etc) is likely to be more complicated than the actual dredging operation and this will vary from site to site.
    - Long slurry pipelines need to be installed and managed.
    - Dewatering and earthmoving equipment may be required at the disposal site.
- Suitably trained and experienced staff will need to be employed for the dredging operation and to maintain the equipment.
- An inexperienced dredging team could result in low production rates (e.g. pumping water without achieving any production).
- There is less incentive to achieve the dredge volumes required.
- A significant portion of the dredging cost is the fuel (or electricity) component, and an inefficient operation could result in a significantly more expensive operation.

In conclusion, dredging and handling of dredge material can be a complicated operation requiring a variety of plant and equipment that needs to be operated and maintained by an experienced dredging team. If not properly managed, a City dredging operation is likely to take longer, be more expensive and have higher risks for the City. It is recommended that this work be carried out by an experienced specialist dredging contractor.

5.9 Summary and recommendations for dredging as a remedial option.

The following is a summary of the pertinent points of dredging as a possible remedial option:

- The proviso for dredging as a remedial option towards improving the water quality in the Milnerton Lagoon is that the high pollutant loading from the Potsdam WWTW should be resolved prior to commencement of dredging.
- Field surveys (including bathymetry core sampling of the bed) of the Milnerton Lagoon between the estuary mouth and the Potsdam WWTW have been carried out during December 2022 and January 2023. This information was used to define the extent and thickness of the contaminated sludge layer on the lagoon bed.
- Based on the recent and historic transects of the lagoon (since 1993) the dredging extent has been defined. Dredging depths of between 0.75 and 1.2 m are proposed to remove the contaminated sediment.
- Dredging operations to remove the nutrient-rich sediment have been divided in three phases with the priority phase (Phase 1) stretching from the estuary mouth to the Woodbridge Island wooden bridge. Phasing may be necessary for both budgeting and different degrees of urgency for nutrient-rich sediment removal of zones along the 6.8 km length of the lagoon and upper Rietvlei estuary.
- The estimated dredging costs for the three proposed phases is approximately R 133 million if all phases are done in a single contract and R165 million if the three phases are done with three separate consecutive contracts. The reason for the difference is the costs for mobilisation and demobilisation of the dredging plant for each contract if done separately.
- The dredging method proposed is a cutter-suction type (either crown type cutter head or auger type cutter head). The suction action is necessary to limit increase in suspended sediment at the cut face in order to obtain minimum negative impact on the ecology of the lagoon. The time required to implement the dredging of the three phases with a single contract is estimated to be 20 months while the total time required if the dredging is done by separate contracts is 33 months.
- The Geotextile tube method for the dewatering of the dredged contaminated material is proposed due to its filtering/contaminant-containing and silt compaction characteristics.
- Three dewatering sites have been identified for the Geotextile tube dewatering process. Since the areas are relatively small cyclic filling and emptying operation will have to be applied where a footprint of a specific Geotextile tube will be used 2 to 8 times. The dewatering sites will have to be reinstated after the dredging operation. Environmental approval for the proposed sites A and B will have to be obtained as well as for utilising Vissershok waste disposal site for the disposal and possible beneficial use of the material as capping material to the waste dumps.
- For the dredging operations the necessary regulatory and legal approvals will have to be obtained before commencement of the dredging operations.

6 REMEDIATION OPTION 2: AERATION

6.1 Scope and purpose

The suspended material in the Diep River settles mainly in the lower reaches of the Milnerton Lagoon and is decomposed there by microbes during aerobic activity. The excessive accumulation of particulate organic matter in sediment has caused bacteria to proliferate to such a degree they are using oxygen at a rate faster than it can be resupplied by atmospheric reaeration. This process causes a low dissolved oxygen concentration in the water column and is associated with high biological oxygen demand (BOD) which in turn causes the sulphate reducing bacteria to switch from aerobic to anaerobic respiration. This process produces sulphite which is then further reduced to sulphide, to which a hydrogen attaches to form hydrogen sulphide (H_2S), the smelly compound which causes the odour problem.

Aeration of the lagoon is proposed as a means to artificially increase the oxygen concentration in the water, thereby addressing the odour problem by creating conditions in which aerobic bacteria can survive and decompose the organic material in preference to sulphur-respirating anaerobic respiration.

6.2 Water quality, oxygen and odour

Ad hoc water quality sampling was carried out in the Milnerton Lagoon in November-December 2022. Figure 6-1 shows the water quality sampling points for this study. The sampling was performed in summer at spring tide for the typical worst-case scenario, namely low flow and high temperatures. Water quality was sampled at low river flows during summer on the 24 November 2022 for one tidal cycle, to assess the water temperature, turbidity, electrical conductivity, dissolved oxygen concentration, total suspended solids (TSS), nutrients, chemical oxygen demand and various heavy metals. Temperature, electrical conductivity and dissolved oxygen were measured in situ using a Hach HQ4300 portable multi-meter with gel pH, conductivity, and dissolved oxygen electrode. Turbidity was measured on-site using a Hach 2100Q Portable Turbidimeter. Nutrients, TSS, COD and metals were analysed by EPL laboratories. Separately, water samples were collected from the bottom of the water column on 8 December 2022 and analysed by AL Abbott & Associates for biological oxygen demand (BOD). Water quality results are presented in the following figures and **Annexure D**.

6.2.1 Water temperature

The major influences on lagoon water temperature are the seawater temperature, the temperature of discharge effluent from the WWTW as well as solar heating and evaporative cooling of the system.

Figure 6-2 shows that for low river flows the entire lagoon is subjected to fluctuating water temperatures that mimic those of the tidal flow in and out of the system. At high tides the water temperature in the lagoon is the coolest as the penetration of seawater cools the system, and as the tide drops the upstream water flow increases towards the mouth. This upstream water would have been heated by solar radiation and results in increased water temperatures in the lagoon as it flows towards the ocean. Thus, the highest water temperature in the lagoon is achieved some time after low tide.

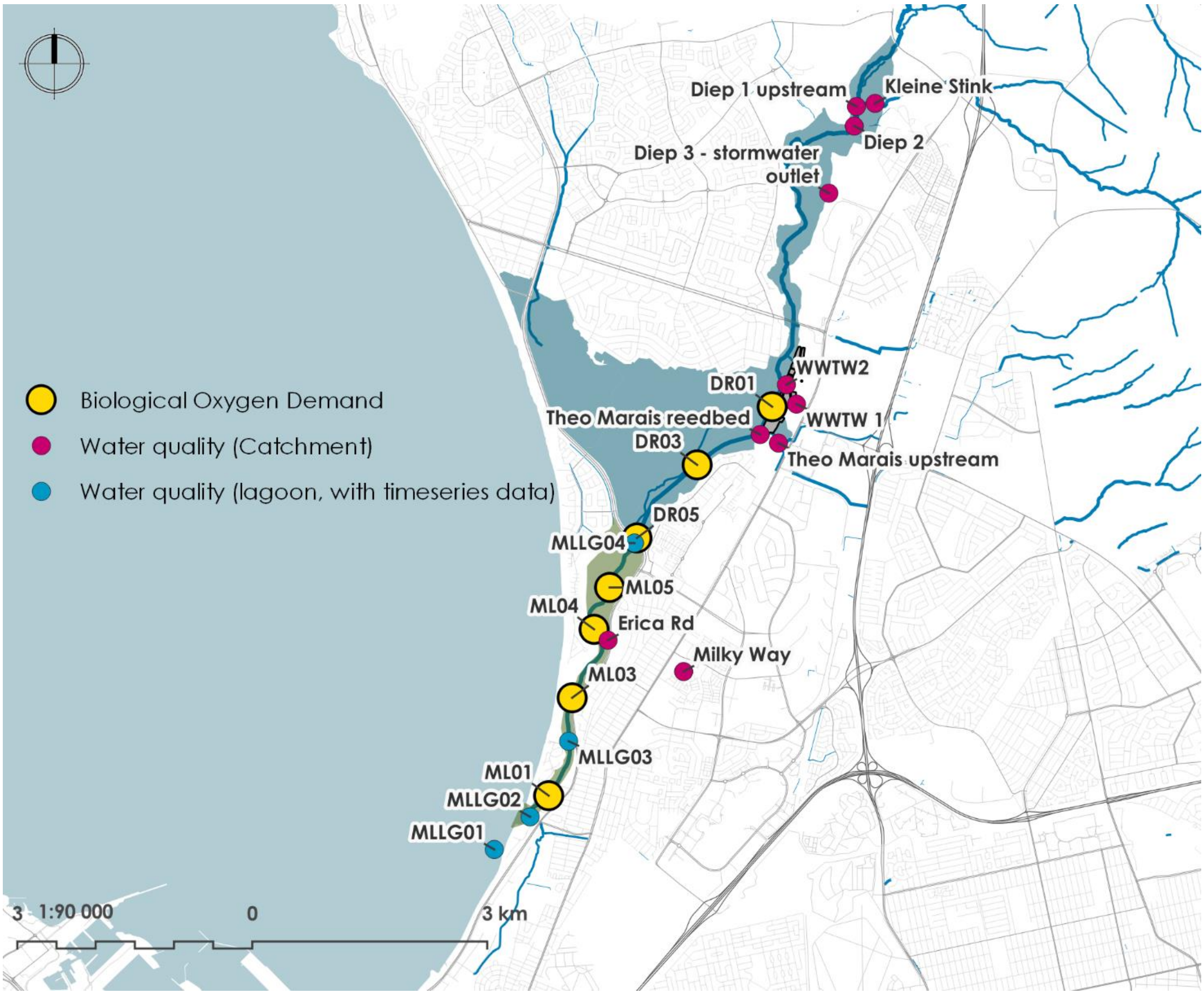


Figure 6-1: The water quality sampling sites (pink and blue) and BOD sampling sites (yellow)

In the upper reaches of the lagoon close to Otto du Plessis Drive, the river channel is narrow and relatively fast flowing when compared to the lower reaches. After Erica Road the channel width increases and flow velocities drop off, resulting in increased water temperatures as a result of solar radiation absorption. This increase in water temperatures lowers the oxygen saturation and would increase bacterial decomposition, exacerbating the odour nuisance. The Atlantic Ocean is relatively cold and the inflowing tide cools the water by up to 5°C which would result in lower biological rates.

6.2.2 Dissolved oxygen

Figure 6-3 shows the data collected for dissolved oxygen concentrations in the lagoon. Dissolved oxygen concentration in the lagoon is very low and reflective of anoxic conditions between Woodbridge and Otto du Plessis bridge. It is well below the target water quality guidelines of 80% – 120% saturation (DWAF, 1999). The DO concentration for 100% air saturated water at sea level is 8.4 mg/l at 25°C and increases to 10 mg/l at 15°C.

The low dissolved oxygen level in the lagoon is attributed to the high oxygen demands on the system in the form of high sediment oxygen demand (SOD) in the river coupled with high COD and BOD in effluent discharged by the Potsdam WWTW. The tidal change has temporal effects on the dissolved oxygen concentrations in the lower reaches but does not affect the DO substantially upstream of Woodbridge, even during a spring high tide.

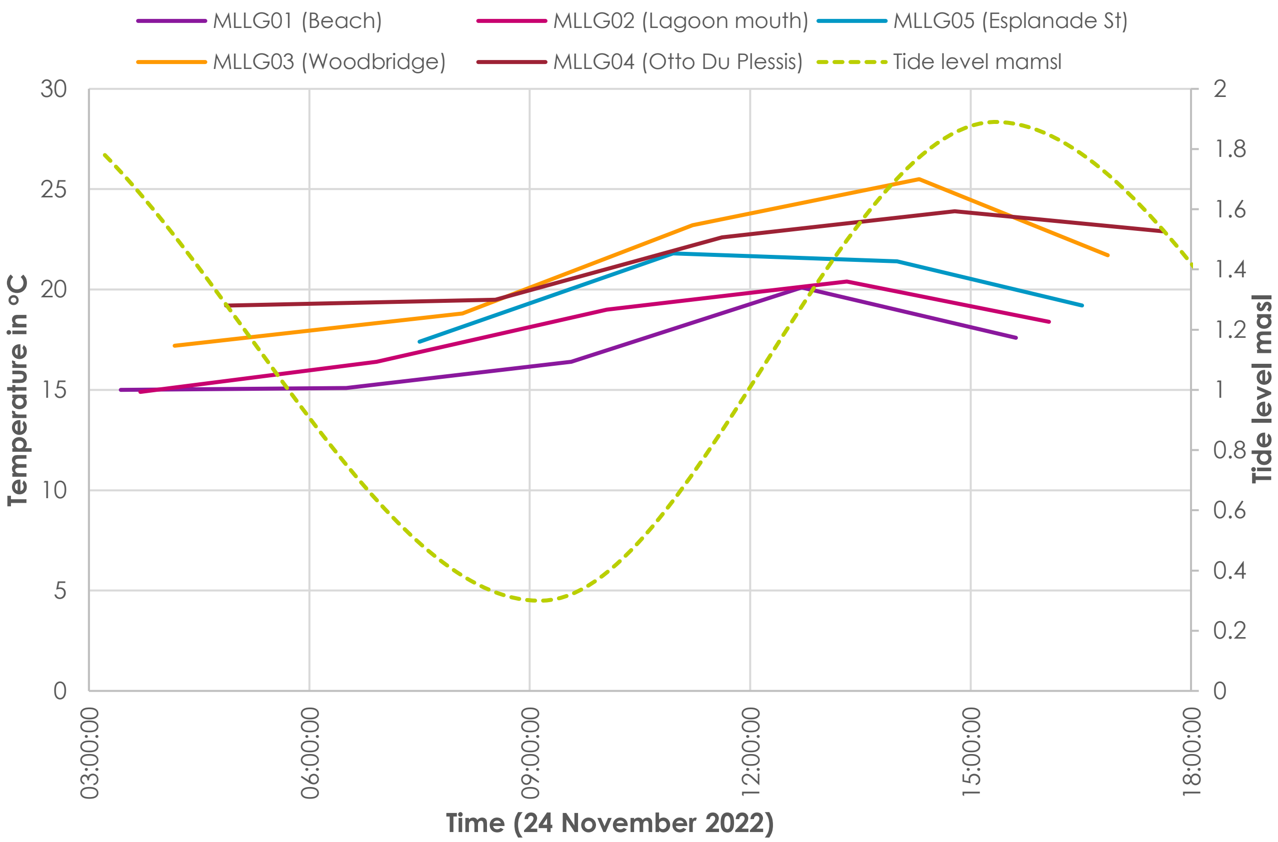


Figure 6-2: Sampled tidal water temperatures in Milnerton Lagoon (Tidal levels on the right-hand side axis)

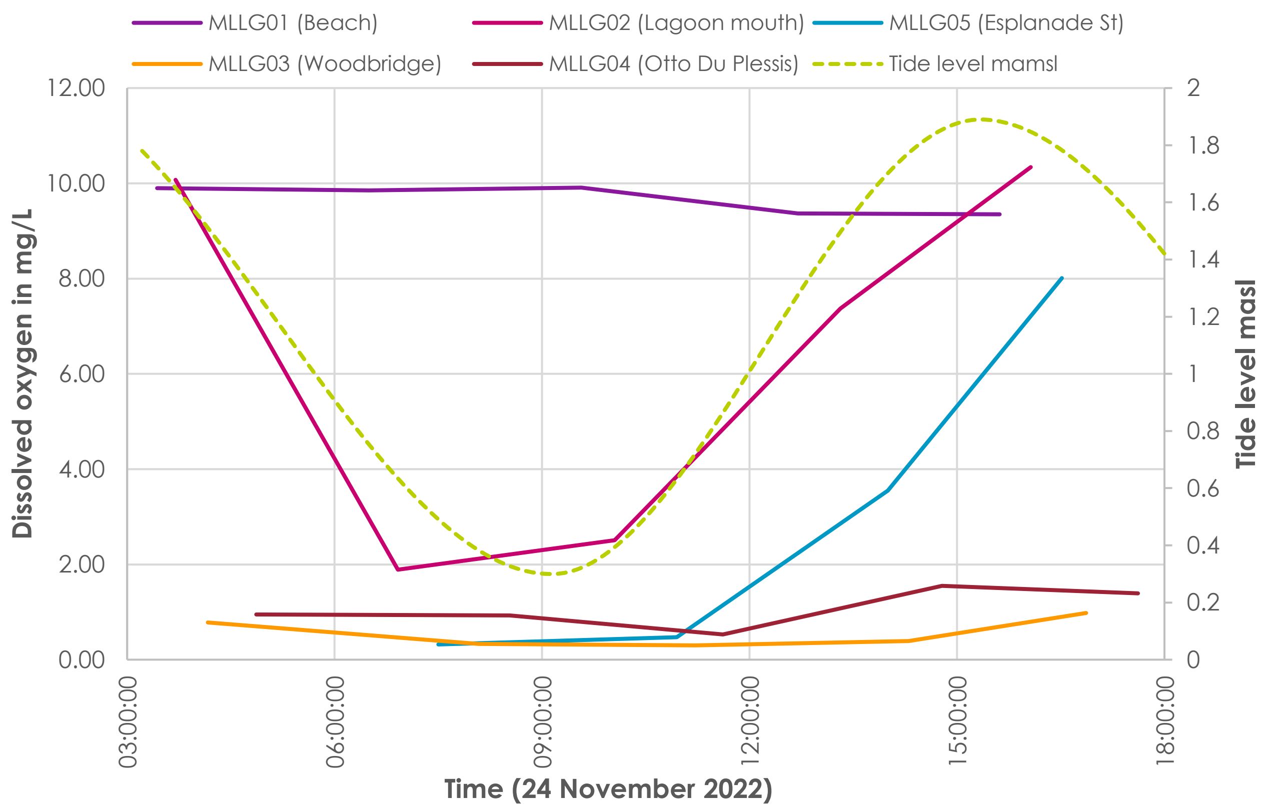


Figure 6-3: Sampled dissolved oxygen concentrations in Milnerton Lagoon (Tidal levels on the right-hand side axis)

6.2.3 Total suspended solids

Figure 6-4 shows the total suspended solids in the lagoon. Suspended solids are highest closer to the mouth as a result of the lower flow velocities, most likely due to a combination of deposited river sediments and tidal marine sediments entering and leaving the system. Further upstream the TSS load on the system is lower, albeit still very high as the DWS recommended background TSS is a maximum of 100 mg/l and the target water quality range (TWQR) is to be limited to <10% of the background range (DWAF,1996).

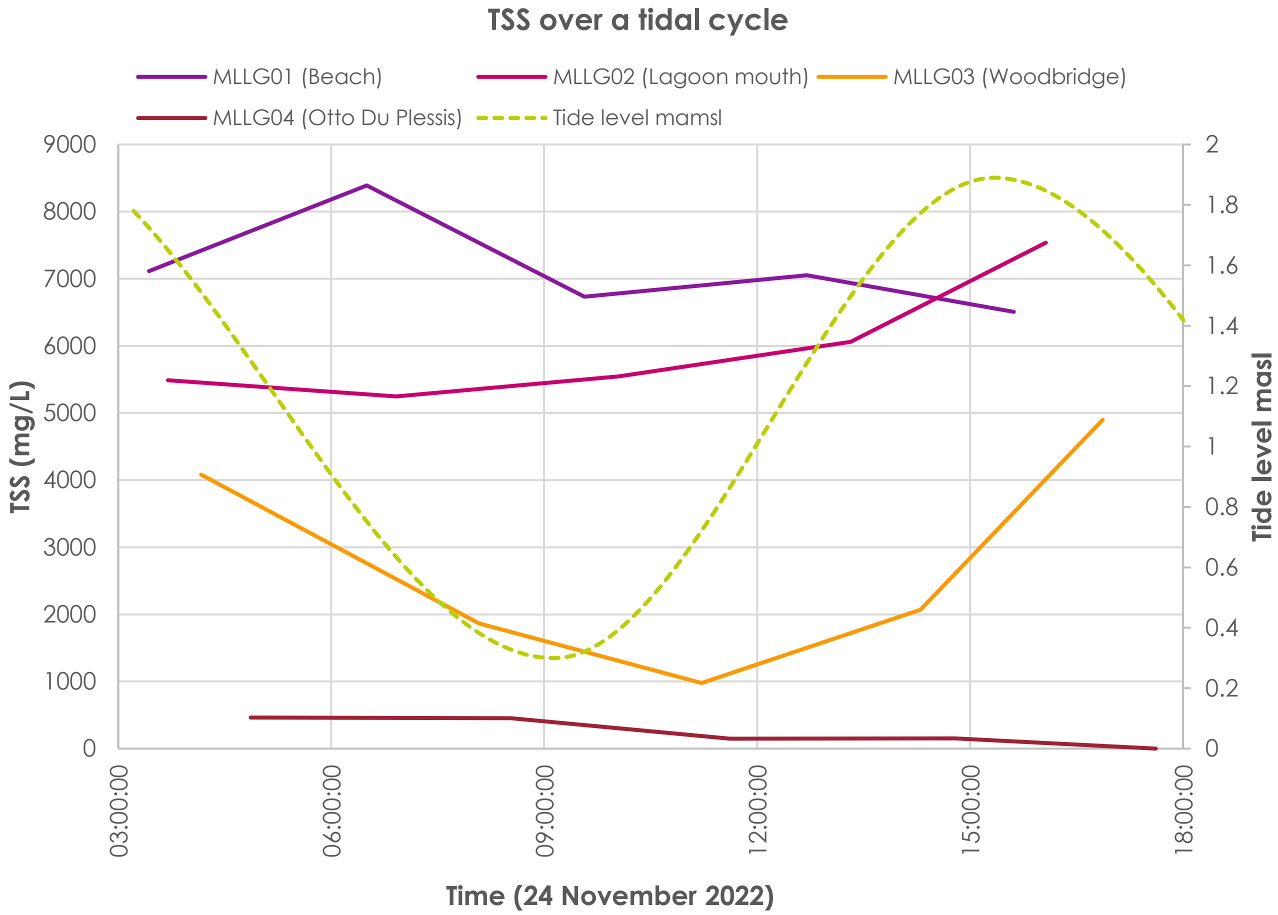


Figure 6-4: Total Suspended Solids in Milnerton Lagoon

6.2.4 Biological oxygen demand

BOD is defined as the amount of dissolved oxygen consumed by organisms in water to break down organic material present, and is often used as the surrogate for the degree of organic pollution of the water. It is also used to gauge the effectiveness of wastewater treatment plants, as well as the short-term impacts on the oxygen levels of the receiving water.

Sludge is the accumulation of organic materials that accumulate, mixed with inorganic material such as sand, clay, or silt. As these organic materials settle, they begin a decomposition process. Decomposition requires oxygen, which results in an anoxic layer of water and sludge on the bottom. In this anoxic layer, aerobic decomposition cannot take place. The sludge layer will continue to grow as more organic materials is deposited. This layer also hosts unwanted anaerobic bacteria that produce hydrogen sulphide which results in the rotten egg smell. Sampling results of the BOD performed in the lagoon from upstream to downstream are illustrated in Figure 6-5. Results indicate that the BOD in the system is extremely high in the lagoon area (ML03 TO ML05).

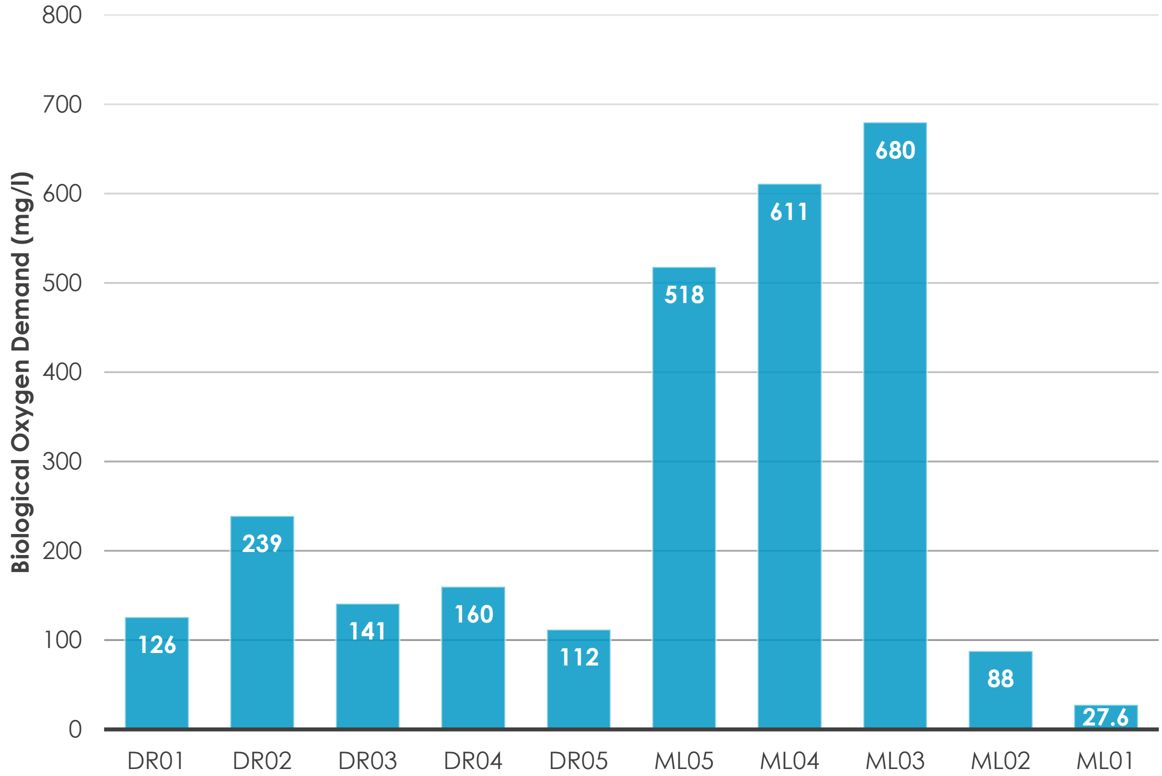


Figure 6-5: Sampled BOD in the lagoon on 14 December 2022

BOD analysis is similar in function to chemical oxygen demand (COD) analysis, in that they both measure the quantity of organic compounds in water. COD is a measure of the oxygen equivalent of the organic matter content of a sample that is susceptible to oxidation by a strong chemical oxidant and is less specific than BOD since it measures everything that can be chemically oxidized, rather than just levels of biologically oxidized organic matter. COD analyses were in fact undertaken for samples collected from the Milnerton Lagoon, but as the laboratory analyses used the potassium dichromate and photospectrometry method, in which chloride (present in high concentrations in seawater) interferes with the analysis and artificially increases the values reported, the results are not sufficiently accurate for use in this analysis.

6.3 River aeration

6.3.1 Theory of river aeration

The high BOD levels in the Milnerton Lagoon between Otto du Plessis bridge and Woodbridge are an indication of the zone where artificial aeration of the river could be most effective to alleviate the noxious odours emanating from the anaerobic microbial action. Artificial aeration is known to improve the quality of the water by pushing high volumes of air into the water. This assists in water circulation, stabilises pH balances, reduces alkalinity, and removes carbon dioxide from the water.

Figure 6-6 shows a simple oxygen mass balance and the process by which oxygen enters or leaves a segment of a stream (or reservoir), excluding other contributions from precipitation, photosynthesis and other chemical reactions.

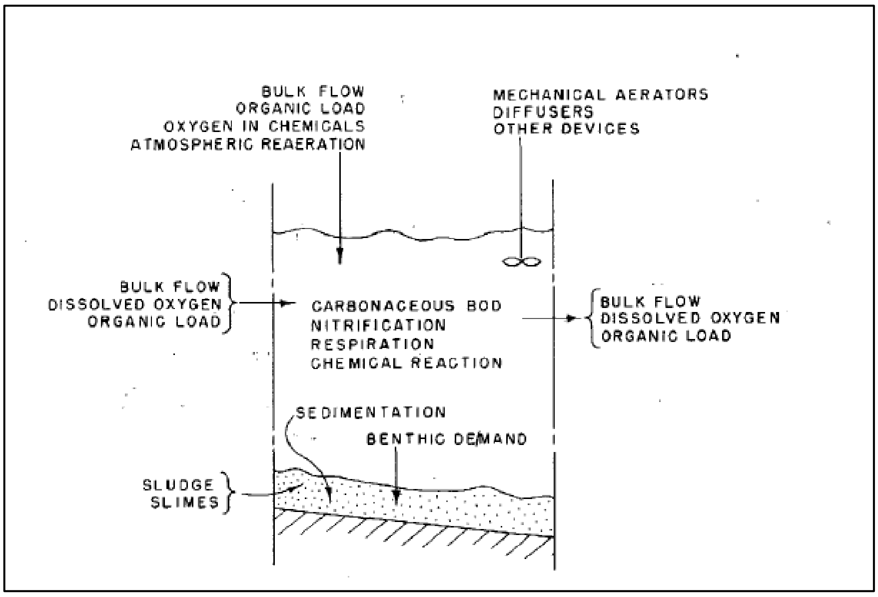


Figure 6-6: The oxygen mass balance in a finite element system (from King, 1970)

From the figure it is apparent that there are a few possible methods of improving the concentration of dissolved oxygen in the water column viz:

- **A.** Remove the dissolved oxygen organic load on the system (achievable only by improving discharge quality at the Potsdam WWTW).
- **B.** Dredge high BOD contaminated sediments from the system to remove the additional sediment oxygen demand (i.e. Remediation Option 1).
- **C.** Improve the atmospheric reaeration by means of weirs and waterfalls (not feasible where there is no significant vertical elevation, as in the case of Milnerton Lagoon)
- **D. Add additional mechanical/artificial aeration – the subject of this section.**

Option D is proposed as a short term, interim measure to improve the dissolved oxygen concentrations and return the microbial action to aerobic decomposition. River aeration is meant to provide supplemental treatment, on a part-time basis at the most critical times of the year which would be more cost effective than secondary or tertiary treatment facilities at the WWTW.

The most important natural addition of oxygen occurs through atmospheric aeration. Artificial aeration may be accomplished with air diffusers, mechanical aerators or other devices. The efficiency of reaeration devices is proportional to the oxygen deficit with an upper limit of 40% saturation for economical aeration. A surface aerator lifts liquid above the surface for exposure to the atmosphere, while a diffuser pumps air below the surface to introduce oxygen lower in the water column.

Aerator efficiencies are highest for two conditions viz:

- Large dissolved oxygen deficits (i.e. low DO in the water column)
- Higher volumetric flow rates in the river (for placement)

Oxygen transfer efficiencies using atmospheric air are poor relative to pure oxygen, as saturation is approached quickly. Aeration using air is limited to situations in which the DO deficit is large. In the case of Milnerton Lagoon, the sampled DO concentrations are anoxic throughout the tidal cycle, maximising the oxygen deficit. In addition, the tidal influence cools the water and thus decreases the oxygen saturation.

In addressing the oxygen deficit, King (1970) found that mechanical aerators had an efficiency of 2-5 lb/kWh of oxygen to the water body, whilst the oxygenation capacity of a floating aerator was quoted as 41 kg/h with an energy consumption of 1.52 kWh/kg O_2. Various manufacturers have their own efficiencies dependent on application and design, but there is a theoretical maximum that may be obtained using atmospheric air for a set water temperature and atmospheric pressure. Ideally, measurements of reaeration rates should be performed in-situ. Where it is not possible to do so, empirical methods must be selected, resulting in a level of uncertainty.

A review of the present state of the art of river and stream aeration (see section 6.5) indicates that the performance of the system is strongly related to the minimum DO level set as a standard. Mechanical surface aerators and diffusers are not efficient for maintaining DO levels above 4 mg/l.

6.4 Required outcomes

In a typical activated sludge WWTW, a dissolved oxygen concentration between 0.2 and 1.5 mg/L is desirable (or 2 mg/L in the aerobic reactor zone). This parameter ensures the oxygen uptake rates of bacteria oxidizing carbonaceous organics are not oxygen limited. For nitrification to proceed at optimum rates, dissolved oxygen values > 2.0 mg/L are required which can go as high as 4.0 mg/L during an organic shock load (Mueller et al., 2002). From an ecological perspective, a minimum of 4 mg/l of dissolved oxygen is required for living organisms.

For these reasons the target for the Milnerton Lagoon is a **4 mg/l final dissolved oxygen** concentration. 4 mg/l of dissolved oxygen is the most economical maximum concentration that may be reached using air, and any further increase would require the use of liquid oxygen.

The Standard Oxygen Transfer Rate of mechanical aerators (typically 0.9 – 1.8 kg O_2/kWh) is normally stated for maximum speed and is measured in clean water at sea level and 20℃. The actual oxygen transfer rate is the rate under field conditions which is influenced by a variety of factors (Metcalf & Eddy, 2003).

During low to zero river flows and the effluent discharge of Potsdam WWTW at an average of 1250 m^3/h (0.35 m^3/s), increasing the DO concentrations in the lagoon from 0 to 4 mg/l would require 5kgO_2/h for saturation which results in an aerator power of 10kW.

6.5 Technology options for surface aeration

Technology options for aeration are described briefly below, followed by an overview of expected costs and electrical supply requirements.

6.5.1 Low-speed surface aerator (Option 1)

In the wastewater treatment industry, the use of **low-speed vertical shaft aerators** is by far the most prevalent in the South African context, due mainly to lower maintenance requirements and greater ease of operation.

This type of aerator is generally mounted on a (fixed or moored) platform and aerates the liquid medium by creating small droplets that are propelled through the air to create the necessary liquid/gas contact area. The conventional low speed surface aerator comprises an electric motor, a flexible coupling connecting the motor to a step-down gearbox, a foundation plate bolted to the platform, a rigid coupling between gearbox and aerator shaft, and the aerator shaft and impeller themselves. In order to adjust aeration input in terms of actual oxygen demand required by the process, there are three possible ways to alter the power input and oxygen transfer of the aerator. These include changing aerator submergence, reversing rotation or changing speed.

In order not to interfere with the aerator's designed splash pattern (which is crucial for efficient aeration), a minimum freeboard of at least 1.2 m between top water level and soffit of the platform must be maintained. The splash area normally varies between 8 and 12 m in diameter. The tip speed of the aerator is normally limited to 6.5 m/s, which implies a rotational speed in the region of 40 – 60 revolutions per minute. This requires a multi-step gearbox arrangement to reduce the speed from 1450 rpm (for a 4-pole motor) to the target speed. A typical unit is shown in Figure 6-7.

The City of Cape Town has confirmed the availability for use in the Milnerton Lagoon of three 90 kW surface aerators with a vertical shaft design.

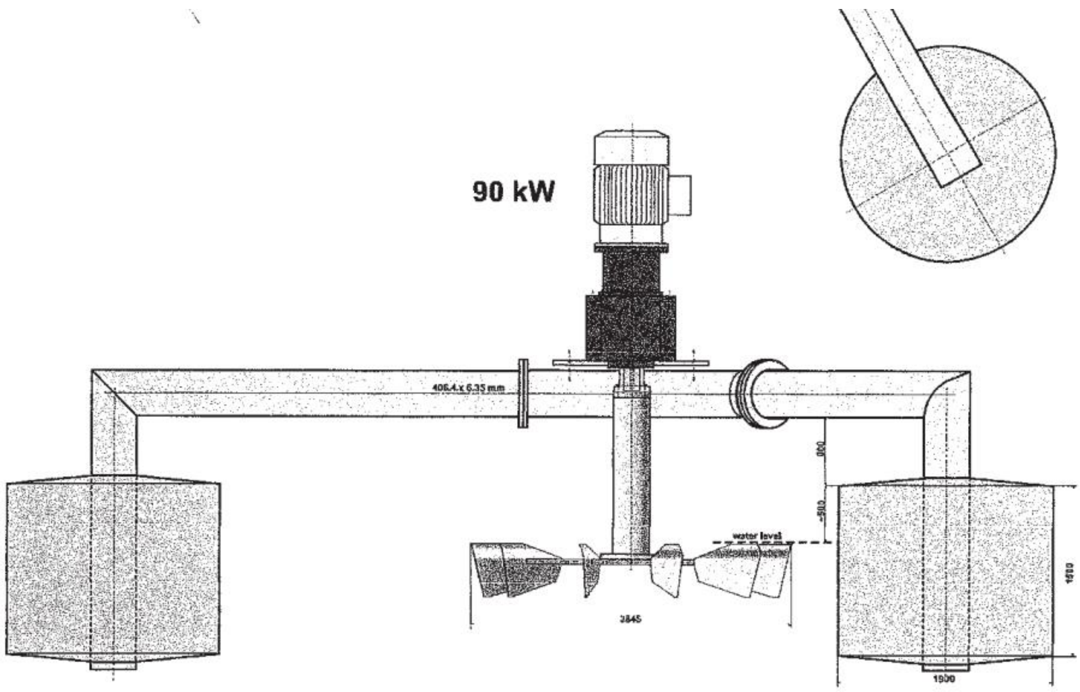


Figure 6-7: Floating vertical shaft low speed aerator available within the City of Cape Town for potential use

6.5.2 Aspirator-type surface aerator (Option 2)

An aspirator-type surface aerator utilises an angled axis with a hollow shaft and a spiral propeller to create a directional bubble plume below the surface by drawing atmospheric air through the hollow shaft and discharging it into the water stream created by the rotating propeller. Figure 6-8 shows the type of aspirator aerator proposed for Milnerton Lagoon. Indicative pricing for the units is shown in Table 6-4 for a 3-unit, 4.0 kW option including estimated costs of the equipment and electrical supply infrastructure.

The aerator is mounted on a float assembly comprising a frame and pontoons, and is approximately 1.6 metres square. The aerators would be moored crosswise to the stream and connected to each other with mooring ropes. Alternatively, a landside mounted pivot arm can be installed.

Figure 6-8: Floating aerators with mooring ropes and pivot arm

6.5.3 Jet aerator (Option 3)

Jet aeration (venturi jet aeration) uses the injection of air under pressure to aerate a waterbody. The jet aerator is a simple device consisting of a submersible pump coupled to one or more ejectors. The ejector system consists of air suction pipes which protrude above the water surface and a venturi like nozzle underwater. The unit (Figure 6-9) is installed at the bottom of the river channel. A pump generates the primary liquid flow and as the flow passes through the nozzle a zone of low pressure is created which draws air down the suction pipe into the ejector. In the mixing zone the liquid and air flows are combined into a liquid jet containing fine bubbles, which shoots through the ejector. Aerators are self-aspiring down to 7 m submergence and are typically constructed of stainless steel to be resistant to corrosion. Oxygen is transferred to the water by the bubbles formed in the pumped jet.

The oxygen transfer is a function of the bubble size, initial dissolved oxygen concentrations, placement, depth and flowrate. Bubbles are ejected transversally, which extends contact time with the water and improves efficiencies relative to a diffuser in which bubbles move only upward through the water column. Surface obstruction is minimal, and the bulk of the infrastructure is submerged, reducing the vulnerability to vandalism.

Units are installed with concrete weights / foundations to hold them in place. Installation requires an electrical supply. Indicative pricing for jet aerator units is shown in Table 6-4 for a 3-unit, 35 kW option including estimated costs of the equipment and electrical supply infrastructure.

6.5.4 Air diffuser

For an **air diffuser** system, air is piped to a distribution system where it is introduced directly to the water through porous ceramic heads, finely perforated tubing or jets. Various materials (ceramics, plastics and membranes) with varying permeabilities are used, to provide the highest efficiencies.

These perforated pipes or diffusers (Figure 6-10) are installed at the bottom of the river with compressed air blown through the pipes producing bubbles. The air bubbles come into close contact with the water with low dissolved oxygen, and oxygen is transferred to the water. The oxygen transfer is a function of the bubble size, initial dissolved oxygen concentrations, placement, depth and gas flowrate.

Figure 6-9: Typical jet aeration system

The greater the depth at which the diffuser is installed, the greater the contact time between gas bubbles and the water, thus increasing oxygen absorption. Greater depth also incurs greater hydrostatic pumping costs. Diffuser systems are not portable and cannot be used in a channel maintained by dredging. They are installed on the bottom of the watercourse and are less vulnerable to vandalism than other systems, though the compressor must be placed nearby and is typically noisy. Where a long distance exists between the compressor and the diffuser, a greater head losses and lower efficiencies are expected. Diffuser systems have a high susceptibility to clogging if not used on a continuous basis.

They have a very low efficiency in shallow systems like the Milnerton Lagoon (depth <1m), where contact time between the bubbles and water is low. An air diffuser system is only effective in systems deeper than 5 metres and where retention time is higher than an instream environment. This system is **not recommended** for the Milnerton Lagoon.

Figure 6-10: Typical diffuser aeration systems

6.6 Installation location

The most efficient location for diffusers or mechanical aerators is at the point of maximum oxygen deficit, and ideally as close as possible to the point at which the negative impacts of the anoxic conditions are most strongly felt. This implies a location between Woodbridge Island and Otto du Plessis – ideally within the zone indicated as Option A ('**downstream site**') in Figure 6-11. It is recommended that the units be placed in water of at least 2 metres in depth, although an absolute minimum of 1 metre is possible. They should be placed where the flow is the fastest and the water depth is sufficient along with the required electrical infrastructure (Figure 6-11). If at the chosen location the threat of vandalism is high, then alternatives closer to Woodbridge could be considered; this would not, however, be ideal as the water is very shallow and the channel much wider and a large proportion of the flow would bypass the aeration units. An alternative installation option is indicated as B ('**upstream site**'), where electrical supply may be simpler. Installation at the Woodbridge itself (C) is not recommended due to the relatively greater width and shallower profile of the lagoon at this point, which reduce the potential efficiencies significantly.

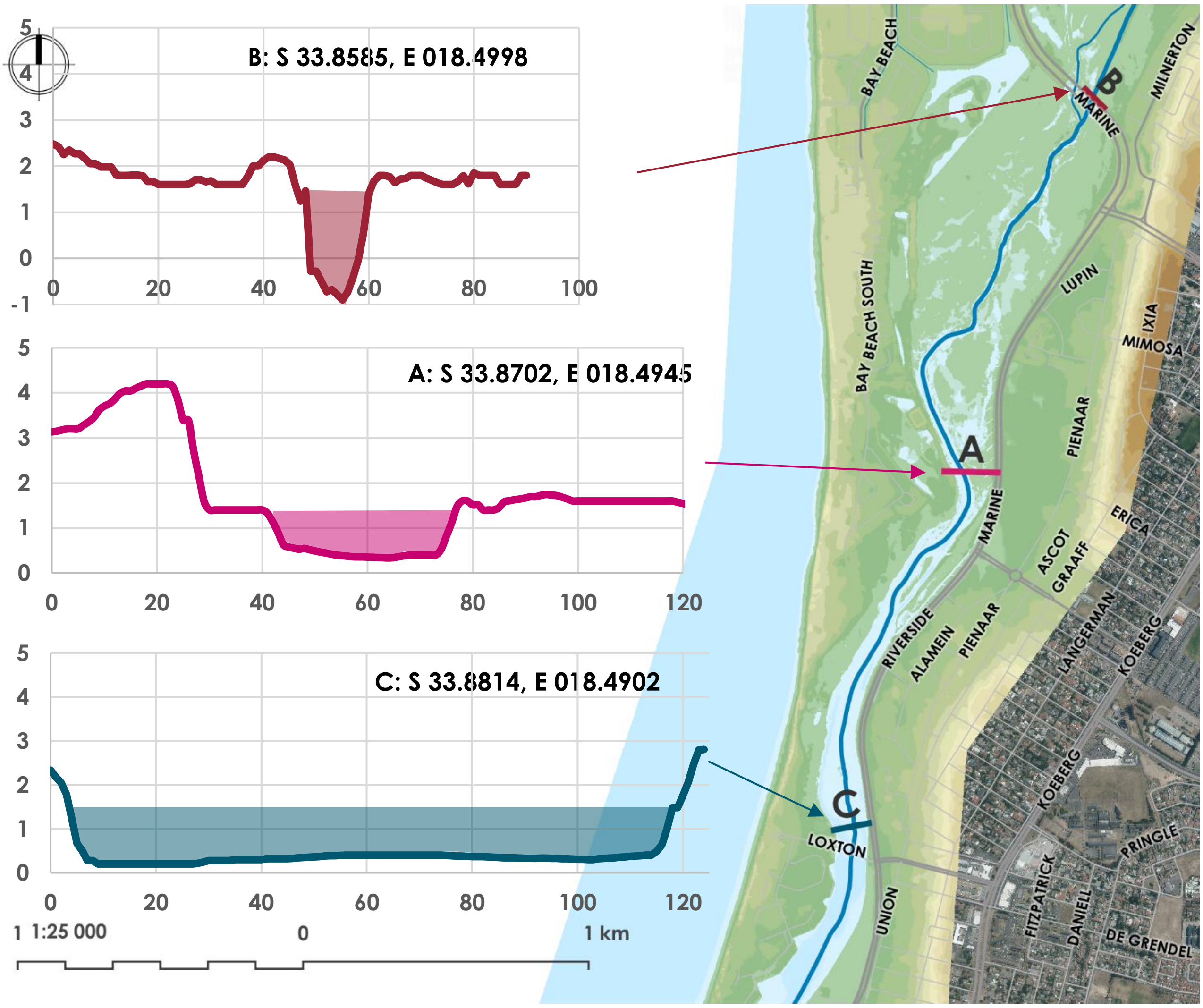


Figure 6-11: Location options for the aerators, with an indicative water level of 1.5 m above mean sea level shown on the elevation profiles at left (vertical and horizontal axes in metres)

6.7 Summary of options

Table 6-1 compares the aeration technology options for Milnerton Lagoon.

Table 6-1. Comparison of aerator options

Type	Availability	Size and installation	Noise
Low-speed surface aerator	CCT has aerators in stock	» Approximately 4 m x 4 m » Floating	High noise
Aspiration-type surface aerator	Available from local suppliers	» Approximately 1.6 m x 1.6 m » Floating, immersed to a depth of 0.50 metres » Portable and lightweight » No specific depth requirements – most units float	Low noise
Jet aerator	Available from local suppliers	» Approximately 0.6 m x 0.465 m x 2 m high » Low portability as unit weighs > 400 kg and requires concrete weights/ foundation. » Depth sufficient to submerge pump and outflow is required. Lower efficiencies in shallow water	Low noise

6.8 Electrical supply

All aeration options would require an electrical supply. Sites A and B were investigated by B2A Consulting Engineers in relation to the electrical installation requirements.

Both sites fall within the City of Cape Town's area of supply, and the Energy Directorate (Electricity Department) would provide the necessary electrical supply. Formal application will need to be made to the Electricity Department for the electricity supply, but discussion with the Department at this stage indicates that:

- The required electricity supply for the various options below is available from the existing electrical network.
- The Electricity Department will install all cabling and infrastructure required to make the required electricity supply available at the respective sites.
- The user department within the City (e.g. the Biodiversity Management Branch) would be required to provide at each site:
  - Meter panel to accommodate the Electricity Dept's metering equipment.
  - Space to accommodate a miniature substation if the larger aerator options are selected. A miniature substation will be supplied and installed by the Electricity Department.
- The user department will be required to pay a connection fee and bulk development levies in accordance with the Electricity Department's current tariff structure. The quotation for this will be provided on formal application of the required electricity supply.
- The user department would be required to supply and install:
  - Control panel housed within a lockable enclosure to control the aerators.
  - All cabling from the Electricity Department's meter panel and the control panel.
  - Electrical and control cables between the control panel and the aerators.
- Refer to Figure 6-12 and Figure 6-13 for approximate locations and routing of electrical infrastructure.

Figure 6-12. Electrical supply infrastructure routing (Downstream site)

Figure 6-13. Electrical supply infrastructure routing (Upstream site)

Table 6-2 overleaf summarises budget estimates for equipment purchase, power requirements and associated costs, and running costs for electrical supply. Should it be necessary – and feasible – to provide for backup power during load shedding, Table 6-3 summarises costs for generator backup power.

Table 6-2: Aerator capital and operating costs with mains power supply (costs exclude VAT)

Option	Equipment costs		Power supply requirements	Electrical supply cost item	Capital Cost Estimate	Total capital costs	Electrical supply costs per month (mains)
1 **3 x 90 kW floating low speed surface aerators**	**None** (CCT owns three units available for use in the Milnerton Lagoon)		330 kVA, 400V, 500A	Connection Fee & bulk development levy	R 1 700 000		
				Meter Panel	R 180 000		
				Control Panel	R 420 000		
				Cabling	R 220 000		
				Contingency	R 120 000		
				Subtotal	**R 2 640 000**	**R 2 640 000**	R 550 000 p.m.
2 **3 x 5.5 kW aspirator-type surface aerators**	Equipment purchase: 3 units at R 225 000 each	R 775 000	25 kVA, 400V, 37A	Connection Fee & bulk development levy	R 180 000		
	Estimate: Installation and supervision (15%)	R 115 000		Meter Panel	R 90 000		
				Control Panel	R 150 000		
				Cabling	R 80 000		
				Contingency	R 50 000		
	Subtotal	**R 890 000**		**Subtotal**	**R 632 500**	**R 1 522 500**	R 45 000 p.m.
3 **3 x 37kW submerged jet aerators**	Equipment purchase: 3 units at R 755 000 each	R 2 265 000	150 kVA, 400V, 225A	Connection Fee & bulk development levy	R 800 000		
				Meter Panel	R 150 000		
	Estimate: Installation and supervision (15%)	R 340 000		Control Panel	R 320 000		
				Cabling	R 170 000		
				Contingency	R 100 000		
	Subtotal	**R 2 605 000**		**Subtotal**	**R 1 540 000**	**R 4 145 000**	R 255 000 p.m.

Table 6-3: Electrical power supply costs for generator backup (costs include VAT)

Option	Generator size	Generator capital cost estimate	Alternative option: Generator rental costs per month	Generator running costs per month (4 hours per day)	Generator running costs per month (24 hours per day)
1	360 kVA	**R 1 060 000**	n/a	R 340 000 p.m.	R 2 040 000 p.m.
2	30 kVA	**R 420 000**	R 30 000 p.m.	R 40 000 p.m.	R 240 000 p.m.
3	180 kVA	**R 750 000**	n/a	R 185 000 p.m.	R 1 110 000 p.m.

6.9 Monitoring

If installation of aerators is to be performance tested in the lagoon, it is essential that their effectiveness be monitored, to determine whether to continue operating these interventions in the medium term. Suggested monitoring parameters, frequencies and indicators of success or failure are indicated in Table 6-4.

Table 6-4. Monitoring requirements and indicators of success

Parameter	Equipment	Sites	Frequency	Indicator of success
Dissolved oxygen in mg/L and % saturation	Portable DO probe	» 10m upstream of installed aerators » 10m downstream of installed aerators » At sites MLLG02-04 sampled in this assessment » Probe lowered to 30 cm above the sediment	» High tide and low tide » Two days per week » Additional sampling during high flow releases from Potsdam WWTW	Increase in high-tide and low-tide dissolved oxygen levels averaged over a two-week period and corrected for fluctuation in background levels

6.10 Legislative considerations

The proposed installation of aerators may constitute at least one listed activity in terms of the National Environmental Management Act, 1998 and the Environmental Impact Assessment Regulations, 2014 (as amended by GNR 324 and 327 of 2017):

» Activity 14 of Listing Notice 3: The development of **infrastructure or structures with a physical footprint of 10 square metres or more**; where such development occurs outside an urban area and —(a) within a watercourse; (b) in front of a development setback; or(c) if no development setback has been adopted, within 32 metres of a watercourse, measured from the edge of a watercourse; f. Western Cape i. Outside urban areas, or ii. Inside urban areas: (aa) Areas zoned for conservation use or equivalent zoning, on or after 02 August 2010; (bb) A protected area identified in terms of NEMPAA, excluding conservancies; or (cc) Sensitive areas as identified in an environmental management framework as contemplated in chapter 5 of the Act as adopted by the competent authority.

» Activity 19A of Listing Notice 1: The infilling or depositing of any material of more than 5 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 5 cubic metres from— (i) the seashore; (ii) the littoral active zone, an estuary or a distance of 100 metres inland of the highwater mark of the sea or an estuary, whichever distance is the greater; or (iii) the sea.

As the aerators themselves are floating, moveable machinery rather than 'infrastructure', **the key determinant is the footprint and excavation required for the required electrical supply cabling and infrastructure, including substations and control panels.** The smaller, aspirator-type aerators proposed for Option 2 may not exceed the 10 square metre threshold for an environmental authorisation and may therefore not require a lengthy Basic Assessment process, provided that the cabling can be installed without excavation greater than the 5 cubic metre threshold. The scale of the mini-substation required for Options 1 and 3 is likely to require environmental authorisation, however. Engagement with the competent authority is required for confirmation, and was in progress at the time of completion of this report.

It is probable that the aeration would also be considered a water use in terms of section 21 (i) of the National Water Act, 1998, viz:

» altering the bed, banks, course or characteristics of a watercourse

In terms of the above legislative instruments, an **environmental authorisation** and a **water use authorisation** may therefore be required from the Department of Environmental Affairs and Development Planning, and the Department of Water and Sanitation respectively. These would require an application process of approximately nine months at minimum.

Given the timeframes associated with such authorisations, and the intended purpose of the aeration in addressing the effects of pollution, it is recommended that the City approach the respective authorities to determine potential alternative mechanisms for the authorisation of these activities, which may include the adoption of an ad hoc development setback line and/or a maintenance management plan.

6.11 Potential ecological risks and benefits associated with aerating riverine flows

River aeration is likely to have positive ecological impacts, principally linked to increasing dissolved oxygen levels in the water, which will make the estuary more habitable for fish and invertebrates. Additionally, keeping DO concentrations above the 2 mg/l threshold required to sustain most aquatic life, should reduce the frequency of fish and invertebrate die-offs.

The most significant ecological risk posed by the proposed aeration methodologies is that of noise, with the underwater jet aerator potentially causing disruptions to fish and invertebrates due to elevated underwater noise levels. The mechanical aerator might cause noise disruptions to local waterbird species should the volume exceed 70dB, or if the noise is abrupt (Cutts *et al*. 2013). If deployed in the channel, it is possible that the rotating surface aerators may pose a collision risk to fish migrating upstream or downstream through the channel or cause behavioural disruption due to noise or visual disturbance.

These risks are minor in relation to the ecological benefits that will be introduced through aeration. The underwater noise concern due to jet aeration is low significance due to the virtual absence of fish living in the estuary due to the poor water quality. The higher dissolved oxygen concentrations linked to aeration should therefore have a substantial net positive impact for estuarine fish. Furthermore, the proposed aeration area is located a considerable distance upstream of much of the estuarine area, thus resulting in large areas which will not be affected by noise. Noise impacts on bird due to diffuser aeration are also anticipated to be low, as the noise is unlikely to exceed the ambient road noise, and the consistent nature of the pumping noise will likely lead to habituation by the local bird population (Cutts *et al*. 2013). Collision risks are also of low concern, and it is much more likely that fish will simply avoid the surface disturbance or swim beneath it.

In summary, river aeration is seen as having substantial ecological benefits, particularly for fish and invertebrate species due to increase DO concentrations, with only minor ecological risk.

6.12 Summary

- Aeration of the lagoon is expected to improve the anoxic conditions currently predominating.
- Two possible locations for the proposed aerators are described – namely the Otto du Plessis road bridge, and the channel west of the Milnerton High School. The latter is preferred as it is closest to the area of desired effect – i.e. the lower lagoon.
- Three technology options are presented:
    - **Low-speed, vertical axis surface aerators** of the type currently used at various wastewater treatment plants within the city, of which three 90 kW units are currently available for use. These devices are noisy and approximately 4 metres in diameter, operating by splashing water using spinning fins. The electrical supply to accommodate three of these units would be 330 kVA, requiring a new mini substation as well as cabling, a meter and control panel to be installed at an estimated cost of R 2.6 million. Electrical running costs are estimated at R 550 000 per month. Should a generator backup be needed, costs for a suitable generator are estimated at R 1.06 million, with fuel costs of R 340 000 per month for 4 hours per day or R 2 040 000 for full time generator supply.
    - **Aspirator-type surface aerators**, which float and operate by drawing air into the water column through a hollow shaft and angled spiral propeller. The electrical supply to accommodate three of these units would be 25 kVA, requiring cabling, a meter and control panel to be installed at an estimated cost of R 632 000 in addition to the purchase and installation costs of R 890 000. Electrical running costs are estimated at R 45 000 per month. Should a generator backup be needed, costs for a suitable generator are estimated at R 420 000, with fuel costs of R 40 000 per month for 4 hours per day or R 240 000 for full time generator supply. A generator of the required size may be available for rental, at an estimated cost of R 30 000 per month.
    - **Venturi jet aerators**, which are submerged and operate by drawing air into the water column through a snorkel. The electrical supply to accommodate three of these units would be 150 kVA, requiring cabling, a meter and control panel to be installed at an estimated cost of R 1.54 million in addition to the purchase and installation costs of R 2.6 million. Electrical running costs are estimated at R 255 000 per month. Should a generator backup be needed, costs for a suitable generator are estimated at R 750 000, with fuel costs of R 185 000 per month for 4 hours per day or R 1 110 000 for full time generator supply.
- The lowest-cost option is the aspirator-type aerator, which is also low-noise and relatively portable. These aerators would need to be procured via tender, which will add a significant lead time to the installation.
- An environmental authorisation is likely to be required for the electrical infrastructure to be constructed to supply the aerators. This has an estimated duration of 8 to 9 months.

A framework of the decisions involved in determining whether and how implement this option is included as Annexure H.

7 REMEDIATION OPTION 3: SEAWATER FLUSHING

7.1 Description of possible pumping schemes

The purpose of the option to pump seawater into the Milnerton Lagoon is to compensate for the poor quality of the water from the Diep River by dilution with seawater. It can be expected that clean seawater mixed with the contaminated effluent from the Diep River will improve the oxygen concentration in the mixture and possibly improve its water quality by supplementing the natural tidal flushing of the estuary. The scope of this Study includes hydrodynamic/water quality modelling to determine the degree of improvement of the lagoon's water quality and thereby assess this option's effectiveness and feasibility. The addition of seawater to the Lagoon will have to be done in a manner and at a location with due consideration to the natural cyclic tidal flow in and out of the lagoon. The location where seawater could be added will be tested at different locations and different cycles to establish where and at which cycle the natural processes will be impacted the least - e.g., considering the cyclic tidal salinity penetration into the lagoon shown in Figure 7-1.

The hydrodynamic/water quality modelling study considered and tested flow rates of 100 l/s, 200 l/s and 400 l/s and the effect that these flows had on conditions in the lagoon. This section looks at various pumping systems, cost and time required for flow rates of 100 l/s, 500 l/s and 1000 l/s. The current Potsdam WWTW's effluent discharge is approximately 400 l/s and the forecast effluent discharge by the end of 2027 will be approximately 1000 l/s.

Figure 7-1. Excerpt from Botes et al (2004) showing salinity penetration in the lagoon during spring tide

The following sources have been considered for seawater to be pumped into the Milnerton Lagoon:

1. Open ocean (– refer to Figure 7-2 for possible locations 1 and 2);
2. Cape Town harbour at a suitable location with acceptable water quality to be confirmed; and
3. Beach Wells

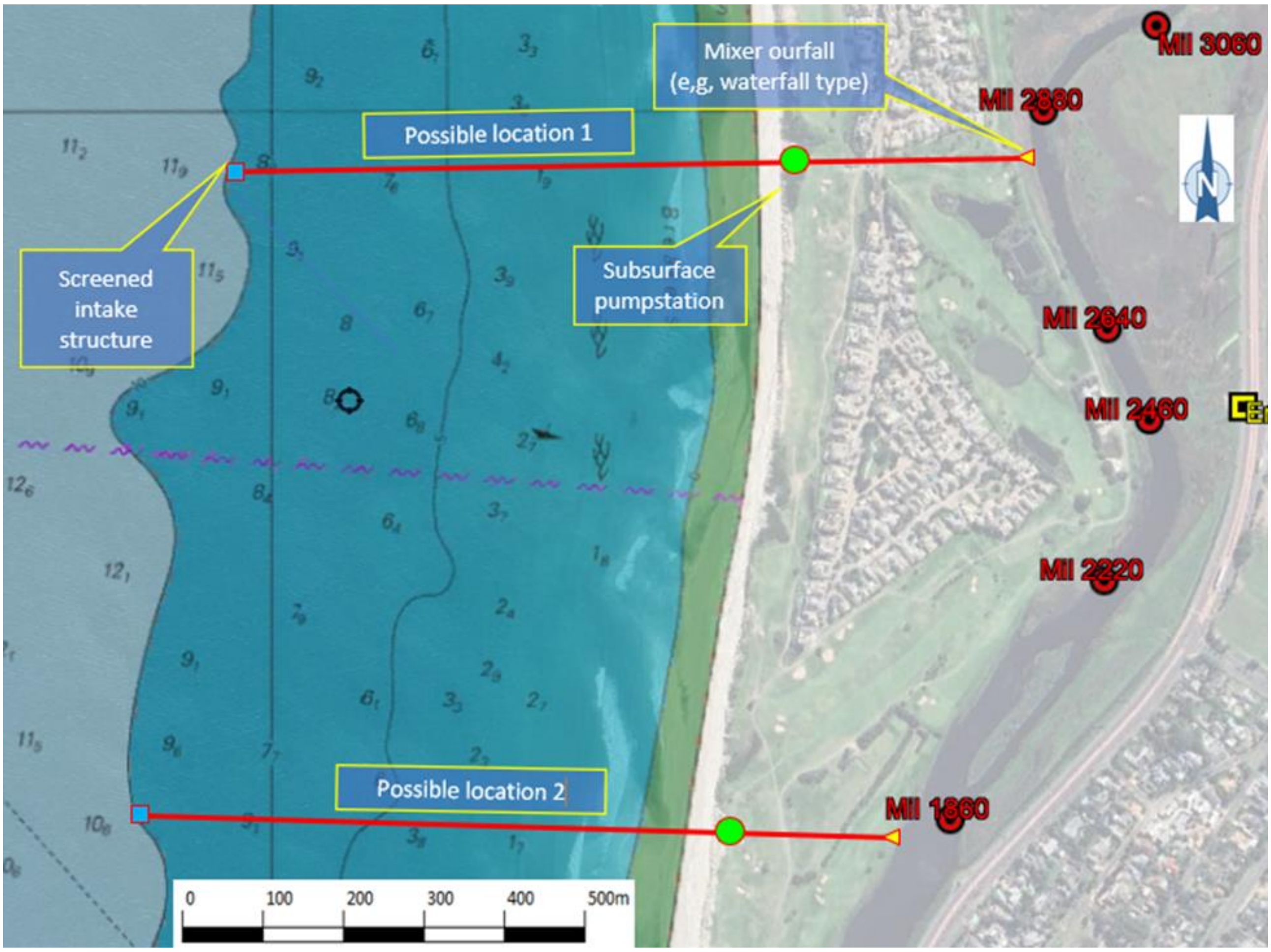


Figure 7-2. Schematic layout of two alternatives of an open sea intake with pumphouse on shore and discharge in the lagoon

7.1.1 Alternative 1 - Seawater abstraction from the open ocean

As shown in Figure 7-2, it is proposed that seawater will be abstracted via an intake structure installed on the seabed at -10m CD. The intake structure inlets will be raised above the seabed to reduce the entrainment of suspended seabed sediments and will be fitted with coarse screens to prevent marine organisms from entering. The intake structure will be marked by a navigation buoy and the intake screens will require regular maintenance to remove marine growth.

The total seawater intake pipeline length, from the point of abstraction to the onshore seawater pump station, would be approximately 750 m.

Based on a typical marine intake pipeline velocity in the range 1.0-1.5 m/s, flow rates of 100 l/s and 500 l/s will require a 500 mm and a 900 mm (outer diameter) HDPE pipe, respectively. Similarly, a flow rate of 1000 l/s will require a 1200 mm (outer diameter) HDPE pipe.

It is envisaged that the seawater intake pump station will be located close to the coastline, with the pumps installed below ground level, at a suitable depth, to ensure that the minimum required pump suction head is satisfied under all operating conditions.

The pumped discharge of seawater into the lagoon could be in the form of water cascading over a weir to enhance aeration of the seawater as well as to encourage proper mixing with the flow in the lagoon. The HDPE pipe required for the seawater discharge pipeline from the seawater pump station to the weir structure would be either 315 mm, 630 mm or 900 mm (all outer diameters), depending on the design flow rate (either 100 l/s, 500 l/s or 1000 l/s). A discharge pipeline length of 350 m has been assumed.

For a design flow rate of 100 l/s, the pump duty head is estimated to be 12 m. The associated power demand would be approximately 15 kW. For a design flow rate of 500 l/s, the pump duty head is estimated to be 10 m. The associated power demand would be approximately 70 kW. For a design flow rate of 1000 l/s, the pump duty head is estimated to be 9 m. The associated power demand would be approximately 120 kW.

Installation of the subsea pipeline will use a float and sink S-lay procedure providing sufficient concrete weight collar ballast along the pipeline to float while air-filled and to be stable on the seabed when flooded. The subsea pipeline will be buried through the shore crossing and the surf zone either by pre-trenching, protected by a sheet piled cofferdam, or by post-trenching, requiring schedule allowances for divers to operate in benign conditions. The assembly of the pipeline into strings for the float and sink can be carried out in an onshore stringing yard and launch-way directly landwards of the shore crossing. Alternatively, the strings can be fabricated and floated from a protected slipway or harbour, nearby, towed to site by workboats/tugs in manageable lengths and winched through the breaker zone from shore. If additional stabilisation measures are required to be installed after the float and sink, consideration could be given to post-placed anchors installed by divers.

Photo 7-1. Subsea pipe string assembled in an onshore stringing yard, fitted with concrete weight collars.

Photo 7-2. An example of construction of a sheet-piled cofferdam to protect the excavation of a pre-trenched shore crossing.

7.1.2 Alternative 2 - Seawater abstraction from the Cape Town harbour

The alternative of abstracting seawater from the Cape Town harbour, and then pumping the seawater to Milnerton Lagoon, is shown in Figure 7-3. At this stage, two potential abstraction locations have been indicated in Figure 7-3 (locations A and B). The suitability of these locations, in terms of water quality and land availability, will need to be assessed in future engineering stages, as well as other alternative locations that may be identified in the harbour.

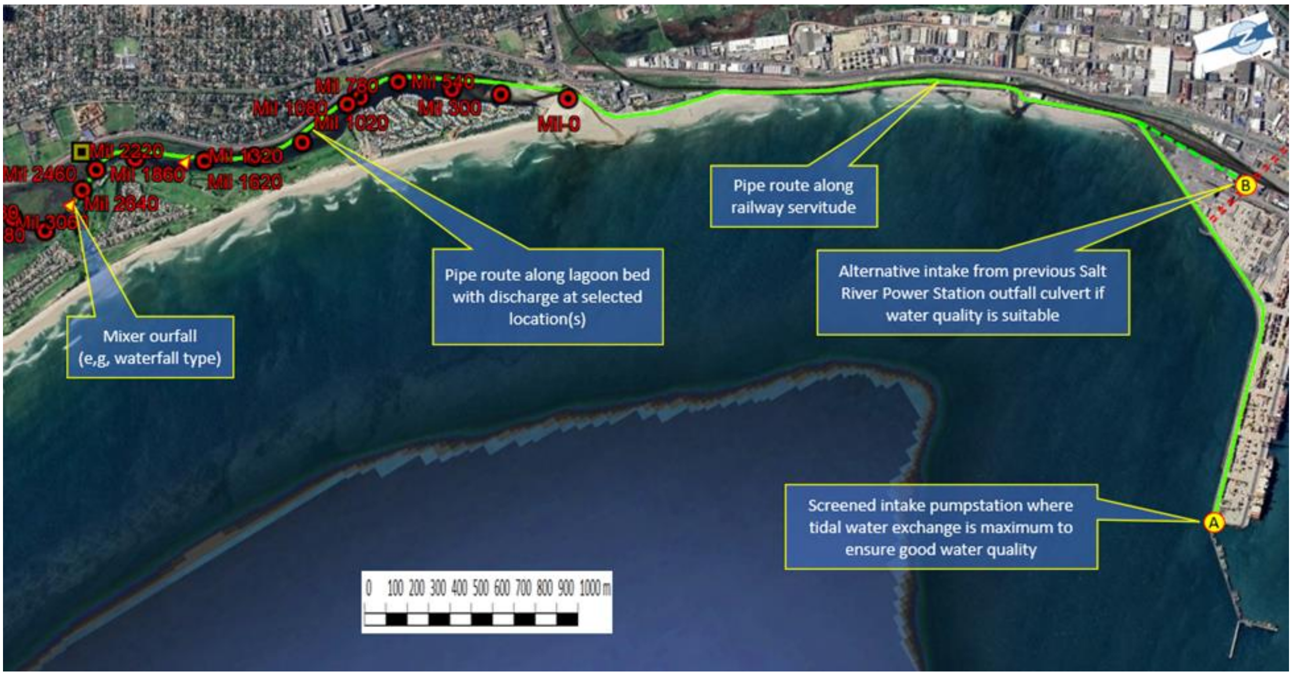


Figure 7-3. Two alternatives (A and B) for seawater intake from Cape Town harbour and pipe route along railway servitude and on lagoon bed.

It is assumed that the seawater discharge pipeline between the harbour and the lagoon could be acceptably routed along the existing railway servitude and on the bed of the lagoon. The length of the discharge pipeline for alternative A, along the assumed route is approximately 8000 m.

Based on a discharge velocity in the range 1.5 - 2.0 m/s, the HDPE pipe required for the seawater discharge pipeline from the seawater pumps at the harbour to the weir structure at the lagoon is estimated to be 710 mm OD for a design flow rate of 500 l/s, and 900 mm OD for a design flow rate of 1000 l/s.

For a design flow rate of 500 l/s, the pump duty head is estimated to be 47 m. The associated power demand would be approximately 320 kW. For a design flow rate of 1000 l/s, the pump duty head is estimated to be 47 m. The associated power demand would be approximately 625 kW.

7.1.3 Alternative 3 - Seawater abstraction beach wells

Beach wells are not suitable for the higher flow rates, but they do become an option to consider for the lower flow rate of 100 L/s. Figure 7-4 shows a potential beach well layout. Figure 7-5 shows an example of a typical beach well unit. Beach wells will be established by jetting well casings into the sand along the back of the beach. These wells will need to be spaced at approximately 20 m apart. Each well will have a submersible pump installed in the well and below the water table. Each well will typically produce between 5 L/s to 10 L/s of water. Therefore 15 to 20 beach wells will be required to supply 100 L/s.

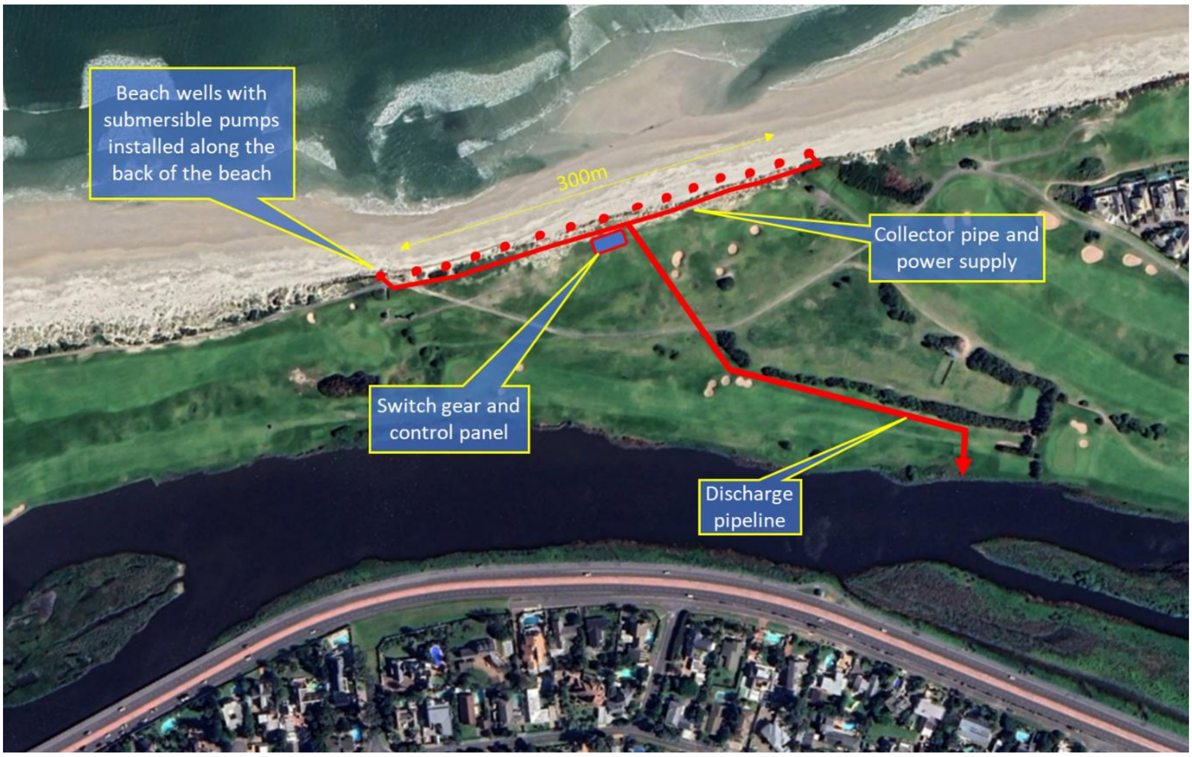


Figure 7-4. Typical layout of beach wells along the back of beach for a 100l/s seawater supply.

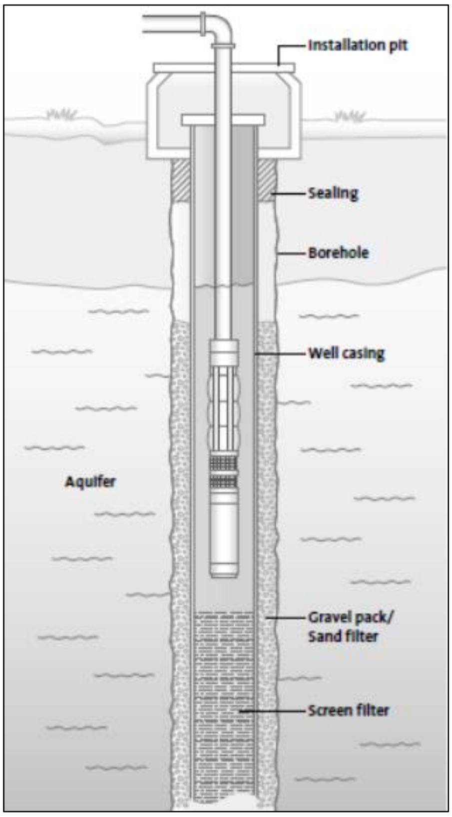


Figure 7-5. Typical beach well with submersible pump (Grundfos SP engineering manual).

7.2 Estimated cost of alternatives

Abstraction of seawater from the open ocean is estimated to cost approximately R42 million, R70 million and R90 million (excluding VAT) for the design flow rates of 100 L/s, 500 L/s and 1000 L/s respectively as shown in Table 7-1 and Table 7-2

Table 7-1. Rough estimate cost breakdown for 100 L/s seawater flushing open ocean alternative.

Element	Total Costs excl. VAT (100 L/s) Ocean
Subsea pipelines	R 20 000 000
Onshore pipelines	R 3 000 000
Seawater intake structure	R 7 000 000
Pump station	R 11 000 000
Aids to Navigation	R 1 000 000
Total	**R 42 000 000**

Table 7-2. Rough estimate cost breakdown for 500 L/s seawater flushing open ocean alternative.

Element	Total Costs excl. VAT (500 L/s) Ocean
Subsea pipelines	R 35 000 000
Onshore pipelines	R 7 000 000
Seawater intake structure	R 11 000 000
Pump station	R 16 000 000
Aids to Navigation	R 1 000 000
Total	**R 70 000 000**

Abstraction of seawater from the Cape Town harbour is estimated to cost approximately R180 million and R215 million (excluding VAT) for the design flow rates of 500 L/s and 1000 L/s respectively and as shown in Table 7-3.

Table 7-3. Rough estimate cost breakdown for 500 l/s seawater flushing harbour intake alternative.

Element	Total Costs excl. VAT (500 L/s from harbour)
Onshore pipeline	R 169 000 000
Pump station	R 11 000 000
Total	**R 180 000 000**

Abstraction of seawater from beach wells is estimated to cost approximately R10 million (excluding VAT) for the design flow rates of 100 L/s as shown in table Table 7-4.

Table 7-4. Rough estimate cost breakdown for 100 L/s seawater flushing from the beach well alternative.

Element	Total Costs excl. VAT (100 L/s) Beach Wells
Site investigations and testing	R 500 000
Beach well installation	R 4 000 000
Delivery pipework	R 3 000 000
Electrical supply and switch gear	R 2 500 000
Total	**R 10 000 000**

7.3 Estimated time for implementation

The following implementation durations are envisaged (these exclude permitting and approval processes):

- Site investigations and preliminary design — 6 months
- Detail design — 3 months
- Tender process — 3 months
- Construction — 8 -18 months

Total estimated duration: 20 months for the beach well installation and up to 30 months for an open ocean water intake, excluding permitting and approval processes.

7.4 Feasibility of seawater addition to Milnerton Lagoon

The feasibility of this option will be determined by:

- Its degree of efficiency to improve the Milnerton water quality, which is determined by the outcome of the hydrodynamic/water quality study – refer to section 13.7.
- The cost and time required for implementation as summarised above.

7.5 Requirements to advance the design of a seawater flushing system

For a more detailed technical feasibility assessment of this option, the following scope of work is required:

- Site investigations:
- Bathymetric survey;
- Topographical survey;
- Beach jet probes;
- Offshore vibrocores;
- Onshore boreholes;
- Preliminary engineering
- Preparation of Request for Quotation documentation including technical specifications and drawings
- Obtain RFQ submissions from one or more experienced contractors.
- Prepare feasibility report

7.6 Legislative considerations

The National Environmental Management Act (NEMA, Act 107 of 1998) requires an environmental authorisation to be obtained prior to the commencement of certain activities which have been identified as being potentially harmful to the environment. These activities are listed in three listing notices published under the NEMA Environmental Impact Assessment Regulations, 2024 (GNR 326 of 2027, as amended). Activities listed in Listing Notice 1 and Listing Notice 3 require a basic assessment as part of the environmental authorisation application process and activities listed in Listing Notice 2 require a scoping and environmental impact assessment as part of the environmental authorisation application process.

Listed activities potentially applicable to the proposed seawater flushing of the lagoon, and requiring a **scoping and environmental impact assessment**, would include:

- **Activity 14 of Listing Notice 2:** The development and related operation of any other structure or infrastructure on, below, or along the seabed.
- **12 of Listing Notice 1:** The development of weirs, where the weir, including infrastructure and water surface area, exceeds 100 square metres in size, or the development of infrastructure or structures with a physical footprint of 100 square metres or more, where such development occurs within a watercourse or, if no development setback exists, within 32 metres of a watercourse, measured from the edge of a watercourse.
- **15 of Listing Notice 1:** The development of structures in the coastal public property where the development footprint is bigger than 50 square meters.
- **19 of Listing Notice 1:** The infilling or depositing of any material of more than 10 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from a watercourse
- **19A of Listing Notice 1:** The infilling or depositing of any material of more than 5 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from … an estuary or a distance of 100 metres inland of the highwater mark of the sea or an estuary, whichever distance is the greater
- **14 of Listing Notice 3:** The development of infrastructure or structures with a physical footprint of 10 square metres or more; where such development occurs—(a) within a watercourse; (b)

in front of a development setback; or(c) if no development setback has been adopted, within 32 metres of a watercourse, measured from the edge of a watercourse; f. Western Cape i. Outside urban areas, or ii. Inside urban areas: (aa) Areas zoned for conservation use or equivalent zoning, on or after 02 August 2010; (bb) A protected area identified in terms of NEMPAA, excluding conservancies; or (cc) Sensitive areas as identified in an environmental management framework as contemplated in chapter 5 of the Act as adopted by the competent authority.

It is probable that the seawater flushing will also be considered a water use in terms of section 21(c) and (i) of the National Water Act, 1998, which requires a water use licence for the water uses of "impeding or diverting the flow of water in a watercourse" and "altering the bed, banks, course, or characteristics of a watercourse" respectively.

7.7 Potential ecological risks and benefits of seawater flushing

The option of flushing the estuary with seawater is likely to have a number of benefits, including increasing dissolved oxygen and salinity levels in the estuary, and reducing nutrient concentrations through dilution. This would therefore make the estuary more habitable for fish and invertebrate species, and reduce the likelihood of oxygen levels dropping below the 2 mg/l critically limit where fish kills are common. Historically the estuary was more saline than it is currently, thus increased salinity levels will increase the suitability of the estuary for marine fish species, and increase its nursery value. High salinities are also toxic for most faecal bacteria, which should reduce risks associated with waterborne pathogens in the estuary.

Ecological risks associated with seawater flushing include construction concerns, such as disturbance of estuarine vegetation and fauna when the pipelines and pump station are installed, as well as operational risks. Construction processes may result in the disturbance of bird species and cause them to temporarily leave the immediate area. Furthermore, installation of the pipeline in the estuary may disturb fish species and disrupt movement up and down the estuary. The principal ecological concern associated with pumping seawater upstream in an estuary is that it will generate a reverse salinity gradient, in which salinities are higher in the upper reaches of the estuary than near the mouth potentially disorienting fish that are seeking to migrate into and out of, and longitudinally within, the estuary. Depending on the location of the pumpstation, noise may also be of concern, and may negatively affect bird species.

Direct impacts on the ocean itself are expected to be minor throughout all stages of the seawater pumping process. Some direct impacts on marine organisms are expected during the construction phase of the open-sea intake option, particularly through disturbance and destruction of benthic communities in the construction footprint, however, due to the shallow, sandy, nature of the coastline and relatively small extent of the construction, these impacts should be minor. The degree of physical disturbance to benthic communities will depend on the methodologies used to deploy the pipeline, with pre-trenching using a sheet pile cofferdam leading to more substantial impacts than the post-trenching option using divers. Additionally, due to the pipeline resting on the seafloor and being buried across the beach and surf zone, the baseline benthic communities should rapidly revert back to their original condition. More substantial (although still minor) impacts would be expected in areas where the pipeline is unburied (from the surf zone to the intake), as this will act as artificial hard substrate which may become fouled with organisms and may form a type of artificial reef, which may also become colonised by opportunistic alien species. However, due to the active nature of the coastline and sandy substrate, the unburied areas of the pipeline will also likely be buried rapidly, minimising this fouling risk. Operational phase impacts near the inlet are expected to be negligible, assuming that the intake has sufficient filters and low-enough suction force to prevent marine organisms from being drawn into the pipe. Given the small volumes of water pumped from the sea, the loss of this volume of water from Table Bay is inconsequential, and the water will ultimately return to the bay via the mouth of the Milnerton Lagoon. The waterfront option is likely to have lower marine impacts than the open-ocean option, however, this small benefit is not sufficient to offset the substantially greater cost and logistical challenges.

Impacts on the Atlantic Ocean as a result of seawater flushing out of the mouth of the Milnerton Lagoon are expected to be negligible. The methodology is unlikely to introduce additional contaminants into the local surf zone, as the low total volumes of seawater will likely be insufficient to mobilise additional sediment into the channel. Rather, it is predicted that the introduction of seawater will lead to greater dilution of the contaminants already in the water column. This dilution will likely have a positive effect on the local surf zone near the estuary mouth, as it will lead to reduced contaminant concentrations.

Relative to the positive impacts associated with pumping seawater, these risks are fairly minor, with the construction phase impacts (which are the most significant) being short-term. The two potential discharge points in the estuary are located within the estuarine channel at positions not known to be densely populated by waterbird species, with most birds inhabiting the sandbanks near the mouth or upstream in the Rietvlei area. The proposed pipeline locations will also traverse the Milnerton Golf Course, which is not seen as ecologically significant estuarine habitat, despite falling within the EFZ. The formation of a reverse salinity gradient is also not seen as a serious risk as the current conditions are almost completely inhospitable to estuarine species, thus a possible degree of disorientation is not seen as a major concern in relation to the substantial positive ecological impacts associated with increased salinities, DO, reduced nutrients, etc. Operational phase noise impacts are likely also insignificant due to the proposed pump station locations close to the shore. Furthermore, due to the consistent nature of the pumping noise, there will likely be habituation by bird species, therefore reducing disturbance over time (Cutts *et al*. 2013).**The combination of low ecological risks and potential significant ecological benefits means that seawater pumping should be considered for a trial in the Diep Estuary if feasible.**

8 REMEDIATION OPTION 4: MARINE OUTFALL

8.1 Description of marine outfall

The City of Cape Town currently operates the following three existing marine outfalls discharging treated effluent offshore:

Table 8-1. Existing marine outfalls discharging treated effluent

Marine outfall	Date commissioned	Length (m)	Diameter (mm)	Design capacity (Ml/day)	DWS discharge Licence (Ml/d)	Depth (m)
Green Point	1993	1 676	800	40.0	44.0	28
Hout Bay	1993	2 162	450	9.8	10.78	39
Camps Bay	1977	1 497	550	5.5	5.5	23

The development of an offshore wastewater discharge proposal includes plume dispersion modelling to verify acceptable dilution and dispersion at the proposed flow rates, location and for all metocean conditions. However, for the purposes of this concept study, based on the existing outfalls, a marine outfall length of 1 800 m has been assumed, which reaches an approximate depth of -18 mCD.

The following marine outfall alternatives were initially identified:

1. Marine outfall pipeline with the shortest direct route to the ocean;
2. Outfall pipeline along the bed of the estuary to the lagoon mouth with either:
    - Mixing of the effluent at the lagoon mouth with the normal outflow; or
    - Extending the pipeline offshore to a water depth of about 18 m (-18 m CD).

Both alternatives include the requirement for a pump station at Potsdam WWTW.

For the second alternative (i.e. including an outfall pipeline along the estuary bed) discharging of wastewater would be restricted to periods of tidal outflow/ebb at the mouth and would result in a doubling of the concentration of the mixture at this location. The extension of the pipeline offshore to a water depth of about 18 m would interfere with the Port of Cape Town's existing designated anchorage area. Therefore, the second alternative was not considered further and only the first alternative with the shortest direct route to the ocean, is evaluated.

The layout and components of a marine outfall for the treated effluent from Potsdam WWTW are presented in Figure 8-1. At this stage it is assumed that the design condition for this outfall should be such that it could handle the full 2027 capacity of Potsdam effluent i.e. 100 Ml/d or about 1.2 m^3/s.

The pipeline length from Potsdam WWTW to the seaward end of the marine outfall at approximately -18 m CD is approximately 5 000 m (inclusive of the marine outfall length of approximately 1 800 m). At the seaward end of the outfall a multi-port diffuser will be provided to ensure that the minimum regulatory effluent dilution in the near and far field is maintained. A "duck bill" type valve will be required at each diffuser port to prevent sediment ingress into the diffuser as well as to enhance dilution of the discharged effluent.

To maintain a minimum pipeline cleaning velocity of 2 m/s an HDPE pipe (outer) diameter of 900 mm is required. The required pump duty head is approximately 16 m. The associated power demand would be approximately 250 kW. Appropriate pumps (such as submersible solids handling sewage pumps)) should be installed with 100% standby capacity. Pumping can be intermittent if the effluent is less than 1.2 m^3/s in order to maintain the minimum velocity in the outfall pipeline.

Figure 8-1. Possible layout of a marine outfall for discharging Potsdam WWTW effluent in the ocean indicating the locations of the pipe route pump station in Potsdam WWTW and diffuser in the ocean are shown.

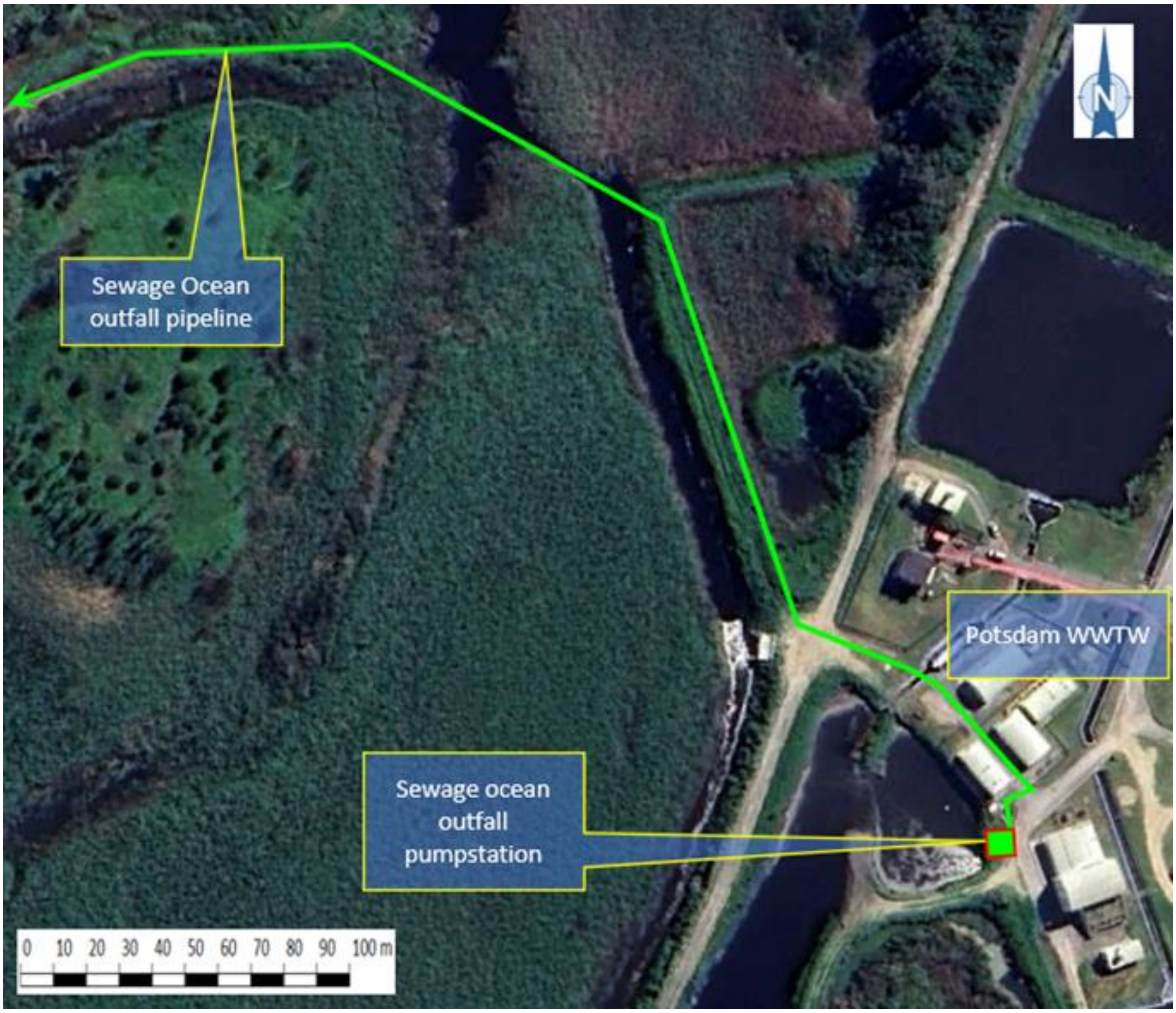


Figure 8-2. Location of ocean outfall pumpstation inside the Potsdam WWTW

8.2 Estimated cost of ocean outfall

The implementation of the marine outfall option including onshore pump station at Potsdam WWTW 3 200m long onshore pipeline and 1 800 m marine outfall pipeline is estimated to cost in the order of R190 million excluding VAT (Table 8-2).

Table 8-2. Rough estimate cost breakdown for wastewater marine outfall.

Element	Total Costs excl. VAT
Subsea pipelines	R 89 000 000
Onshore pipelines	R 84 000 000
Pump station	R 16 000 000
Aids to Navigation	R 1 000 000
Total	**R 190 000 000**

8.3 Estimated time for implementation

The following implementation durations are envisaged (these exclude permitting and approval processes):

- Site investigations and preliminary design — 6 months
- Permitting and approval processes — 12 months
- Detail design — 3 months
- Tender process — 3 months
- Construction — 12-18 months

Total estimated duration: 24 to 30 months excluding permitting and approval processes.

8.4 Feasibility of a marine outfall from Potsdam WWTW

The feasibility of this option will be determined by:

- Its degree of efficiency to improve the Milnerton water quality. This is determined by the outcome of the hydrodynamic/water quality study – refer to section 13.7.
- The cost and time required for implementation.

8.5 Requirements to advance the design of an outfall system

For a more detailed technical feasibility assessment of this option, the following scope of work is required:

- Site investigations:
- Bathymetric survey;
- Topographical survey;
- Beach jet probes;
- Offshore vibrocores;
- Onshore boreholes;
- Preliminary engineering
- Preparation of Request for Quotation documentation including technical specifications and drawings
- Obtain RFQ submissions from one or more experienced contractors.
- Prepare feasibility report

8.6 Legislative considerations

The proposed marine outfall may constitute various listed activities in terms of the National Environmental Management Act, 1998, and the Environmental Impact Assessment Regulations, 2014 (as amended by GNR 324 and 327 of 2017). Listed activities potentially applicable to the proposed marine outfall, and requiring a **scoping and environmental impact assessment**, would include:

- **Activity 6 of Listing Notice 2:** The development of facilities or infrastructure for any process or activity which requires a permit or licence or an amended permit or licence in terms of national or provincial legislation governing the generation or release of emissions, pollution, or effluent.
- **14 of Listing Notice 2:** The development and replated operation of any other structure or infrastructure on, below, or along the seabed.
- **10 of Listing Notice 1:** The development and related operation of infrastructure exceeding 1 000 metres in length for the bulk transportation of sewage, effluent, process water, wastewater, return water, industrial discharge or slimes, with an internal diameter of 0,36 metres or more or with a peak throughput of 120 litres per second or more.
- **12 of Listing Notice 1:** The development of infrastructure or structures with a physical footprint of 100 square metres or more, where such development occurs within a watercourse or, if no development setback exists, within 32 metres of a watercourse, measured from the edge of a watercourse.
- **15 of Listing Notice 1:** The development of structures in the coastal public property where the development footprint is bigger than 50 square meters.
- **19 of Listing Notice 1:** The infilling or depositing of any material of more than 10 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from a watercourse.
- **19A of Listing Notice 1:** The infilling or depositing of any material of more than 5 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 10 cubic metres from … an estuary or a distance of 100 metres inland of the highwater mark of the sea or an estuary, whichever distance is the greater
- **14 of Listing Notice 3:** The development of infrastructure or structures with a physical footprint of 10 square metres or more; where such development occurs—(a) within a watercourse; (b) in front of a development setback; or(c) if no development setback has been adopted, within 32 metres of a watercourse, measured from the edge of a watercourse; f. Western Cape i. Outside urban areas, or ii. Inside urban areas: (aa) Areas zoned for conservation use or equivalent zoning, on or after 02 August 2010; (bb) A protected area identified in terms of NEMPAA, excluding conservancies; or (cc) Sensitive areas as identified in an environmental management framework as contemplated in chapter 5 of the Act as adopted by the competent authority.

A water use licence in terms of section 21 of the National Water Act is required for marine outfalls, as well as a **coastal waters discharge permit** in terms of GNR 382 of 2019, as contemplated in section 69(1) of the Integrated Coastal Management Act no 24 of 2008.

8.7 Potential ecological risks and benefits associated with building and operating a marine outfall

Diversion of effluent from Potsdam to a marine outfall would result in the removal of the single greatest contaminant source in the system. The removal of the highly nutrient dense, contaminated, continual water source would lead to an improvement in water quality flowing downstream, including increased DO, lower nutrients etc., and dramatically slow down the rate of future organic sludge accumulation in the estuary. However, this option would also seriously alter the hydrodynamic functioning of the system. Firstly, since the catchment is so highly modified, much of the natural mean annual runoff (MAR) has been diverted, with the return flows bringing total "freshwater" inflows to approximately 95% of natural condition, as opposed to ~61% without Potsdam (Clark *et al*. 2018). This is most impactful during the summer months when natural flows are very low, meaning that most of the flows in the estuary are derived from this treated effluent. Besides being of poor quality, these summer flows perform an important function in keeping the estuary mouth open, which enables tidal flushing and allows passage of marine species into and out of the system. Without these flows, the estuary mouth would be closed most of the time, certainly during the summer months and evaporation might lead to hyper-salinity. As the winter rains began, the estuary would fill, and the water levels would rise until a sufficient hydraulic head was generated to break open the berm, causing a large amount of sediment scour and transporting sediment out to sea, as well as opening the estuary to the sea again. Reverting to a predominantly closed mouth state for this system is problematic for a number of reasons. Firstly, the water entering the estuary from the catchment as well as stormwater drains and other sources besides Potsdam is also generally very poor quality, largely consisting of grey and black water. Despite not contributing to the same level of loading as Potsdam, should the mouth be closed in summer, there will be a long residence time of this polluted water in the estuary. Coupled with the absence of any seawater flushing and introduction of dissolved oxygen from the Atlantic Ocean, the water quality in the estuary may decrease further making it even more toxic than it is at the moment. Additionally, the closed mouth would block recruitment of fish into the system and prevent fish from escaping to the sea during low oxygen or high ammonia events, and would likely result in further fish kills. A further threat posed by summer mouth closures is that it presents a flooding risk to the nearby areas during the winter months, as the water levels will rise to much higher levels than the current maxima if the mouth is not opened artificially on a timeous basis.

It might be possible to achieve the ecological benefits of diverting the Potsdam flows whilst not having such long residence times and negative consequences, however, this would require very active mouth management similar to what is currently being conducted at the Zandvlei Estuary Mouth in Muizenberg, with the mouth being opened artificially in the summer months to allow for flushing of the contaminated water and entrainment of clean seawater. There is no guarantee that this would work, however, as the reduced freshwater inflows to the estuary (<60% of natural), coupled with this artificial mouth opening may lead to the accumulation of marine sand in the estuary, which would reduce water depth. These marine sands may not be flushed during the winter months since the mouth would be opened before a large head of water could develop in order to prevent flooding. This type of active mouth management is also resource intensive, and would require an established mouth management plan and coordination by management authorities to prevent serious negative ecological and human consequences.

The construction phase for this marine outfall option may also have a number of short-term negative consequences, including causing noise and other disturbances to the local bird population, particularly since the proposed pipeline will traverse through the centre of Rietvlei. There is also a risk of short-term pollution during the construction of the pipeline, and the formation of turbidity plumes in the vlei, should this not be properly managed with appropriate measures such as silt curtains. Furthermore, the construction of the undersea pipeline will also have negative impacts on the benthic marine species living in proximity to the offshore pipeline and construction area.

Additional operational risks besides those mentioned above include the possible risk of pipeline leaks into Rietvlei, which is currently separated from the Potsdam effluent by the diversion channel. Since the new pipeline will traverse directly between North and South vlei, any leaks from the pipeline will directly impact upon this waterbody. Additionally, depending on the location of the pumpstation, there may be localised noise impacts on the local bird community. Furthermore, should this contaminated effluent be pumped offshore, there will likely be some negative pollution impacts around the outfall, and potentially on the nearby beach area. A potential risk of discharging effluent offshore is that this would be much less visible, therefore reducing social pressure to improve effluent quality.

In summation, the construction phase impacts associated with a marine outfall are likely not of serious concern due to the short-term duration of the project, and they can easily be mitigated. Of much greater concern are the operational phase impacts, with the potential alteration of the mouth state being of central concern. Without active mouth management, the loss of freshwater inflows in summer could have serious negative ecological and human health impact, rendering the operation counterproductive. Of lesser concern are the risks of leaks into Flamingo Vlei, and offshore pollution from the outfall, which should be unlikely in the first case, and of vastly reduced significance due to dilution effects in the second case. Operational noise impacts on local waterbirds due to pumping are also likely not of serious concern due to the high ambient noise levels in the area and habituation effects due to the near-continual nature of the pumping.

Diversion of Potsdam's effluent to a marine outfall is therefore not recommended due to the potential serious alteration of the mouth state during the summer months, and the associated negative ecological and human health consequences that may arise without intensive mouth management.

9 REMEDIATION OPTION 5: SUPPLEMENTARY TREATMENT OF RUNOFF USING BIOFILTRATION

9.1 Scope and purpose

Contaminated surface water runoff from areas which have informally densified beyond the capacity of their sanitation infrastructure – such as Joe Slovo – and informal settlements – such as Du Noon – comprises grey- and black water with elevated nutrient loadings, drugs and pharmaceuticals, and high bacterial and coliform counts, especially during conditions of low flow. Untreated water is often directed into existing stormwater conduits and discharged into the receiving environment. Biofiltration systems (biofilters) are investigated as a sustainable and low-cost option for treating polluted runoff from small streams near informal settlements, without the addition of electrical energy, chemical treatment, or significant capital and maintenance costs. There is limited research on the use of biofilters in or downstream of informal settlements and the fate and risk of pollutants in the environment after treatment is relatively unknown. The work presented in this section draws from the research and experience of researchers at the Water Hub research and innovation site in Franschhoek (Winter et al., 2023). The research results show significant improvements in water quality whereby water of varying quality is retained in large scale filter systems that are packed with different media and released every 5 to 7 days. The site and design configuration at the Water Hub is different to what is being proposed in Milnerton Lagoon Rehabilitation Plan, but the treatment process is similar. The most important difference at the Water Hub is that water is abstracted from a river that flows past an informal settlement and is constrained by the energy required to pump water into the biofiltration beds, whereas the proposal in this report is to create longer gravity-fed offtake channels that are packed with different filtration media and can treat water during low flow before discharging back into the channel and into the Diep River system. It proposed that a pilot project is introduced in one or more of the channels downstream of one of the informal settlements in the catchment to analyse the effectiveness of biofilters in improving the water quality drawing on the experience at the Water Hub, but more importantly, to understand the social acceptance of the treatment system, the operational and risk management in understanding the fate of these pollutants in the environment after remediation.

Biofiltration cells at the Water Hub are designed to operate as a treatment train. Six cells each 16m x 3.5m x 0.7m are paired with the same media in each cell. One of the pairs is vegetated and the other acts as a control to compare the potential advantage of phytoremediation in the uptake of contaminants (Photo 9-1). A comparison between vegetated and non-vegetated treatment suggests that plants play a relatively small role in a horizontally fed system. Rather, significant treatment occurs in an anoxic environment found below 30cm in the cell column which harbours anaerobic bacteria and microbes that are fed by constant flow and retention of water in the treatment process. The dissolved O_2 concentration effluent at the point of discharge is between 2 -3 mg/l, and the release of the treated water passes through a swale with multiple weirs that aerate water before it is released back into the river (Photo 9-2).

Photo 9-1. Biofiltration cells at the Water Hub. Each cell can be controlled separately or managed as a treatment train. The supply is abstracted from the river and pumped into two 10Kl tanks that feed the cells.

Photo 9-2. A vegetated swale structure post treatment at the Water Hub provides a final polishing phase with low v-notch weirs that are designed to aerate the water before it is discharged into the river alongside (DO 5 -6 mg/l at point of discharge).

Table 9-1 displays findings from studies conducted at the Water Hub, showing the average reduction of orthophosphate and ammonia-nitrogen. Stone aggregate data represent studies from monthly sampling over 2.5 years, while results from the biochar filter media are from more recent studies conducted over 3 months. The results from the latter are promising. Of interest is that the biochar is produced from alien vegetation biomass as the feedstock.

Table 9-1. Water quality ranges measured at the point of irrigation or effluent following biofiltration treatment (Water Hub 2018 – 2020).

	Large stone aggregate (19 to 22mm)	Biochar	Small stone aggregate (7-9mm)
PO_4^{3-}	74%	72%	16%
NH_3	72%	97%	75%

Water quality is being treated to comply with South African and World Health Organisation guidelines for irrigation. Table 9-2 shows ranges of different water quality parameters at the point of irrigation at the Water Hub. E.coli is removed close to 100% following a 5 to 7 day retention period. Water quality is most cases meets average discharge guidelines to streams and water bodies, for example,

- Ammonia nitrogen – 5 mg N/L
- Nitrate and nitrate nitrogen ("oxidized nitrogen") – 10 mg N/L;
- Total phosphorus - 2 mg P/L;

Table 9-2. Water quality ranges measured at the point of irrigation or effluent following biofiltration treatment (Water Hub 2018 – 2020).

Water quality ranges	pH	EC	Na	K	Ca	Mg	Fe	Cl	PO_4^{3-}	TIN[1]	B	Cu	*E. coli*
Irrigation Tank		uS/cm	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	cfu/100 ml
Low	6.5	250	30.4	1.9	12.8	<1	0.05	34	0.01	0.02	<0.08	<0.02	<1
High	7.1	310	38.7	7.8	23.6	3.7	0.18	38.5	4.25*	29.2**	<0.08	<0.02	8

[1] Total inorganic nitrogen (NH_3 + NO_2 + NO_3) *Mean = 1.04 **Mean = 5.27

9.2 Limitations

The results from the Water Hub analyses were achieved under experimental conditions in which the site infrastructure could be repurposed to support a batch-controlled treatment process that is controlled. In-situ trials of this technology have not yet been made, nor have gravity-fed flowthrough systems been tested. Implementation in the Milnerton Lagoon context would therefore be experimental, and is not a core remediation measure for the purposes of this plan.

The design of biofiltration systems is necessarily site-specific and careful attention would need to be given to the site and surrounding conditions. Access and safety, including road access for solid waste removal are needed, along with regular maintenance and ongoing management. A regular sampling and monitoring programme is also necessary, to provide a long-term record. The biofilters can take three to four months to achieve optimal performance, depending on circumstances, to develop the biofilm and habitat for microbes and bacteria.

9.3 Potential sites for biofiltration pilot

A total of 11 sites were identified from desktop observations and site visits. Consideration was given to access to sites, safety and security, low flow channels, potential risk of flooding, and a reasonable ability to treat water along a river stretch where biofilters could span a distance of at least 50m. The design primarily aims at the degradation and removal of elevated concentrations of TN, PO_4^{3-} and bacteria. The proposed sites should accommodate large biofiltration cells that are installed as a treatment train in the form of a low flow diversion channels, but is capable of bypassing the intervention during peak flow. A total of 11 sites were examined, with three of them (Figure 9-1) identified as possible sites for installing a low flow channel.

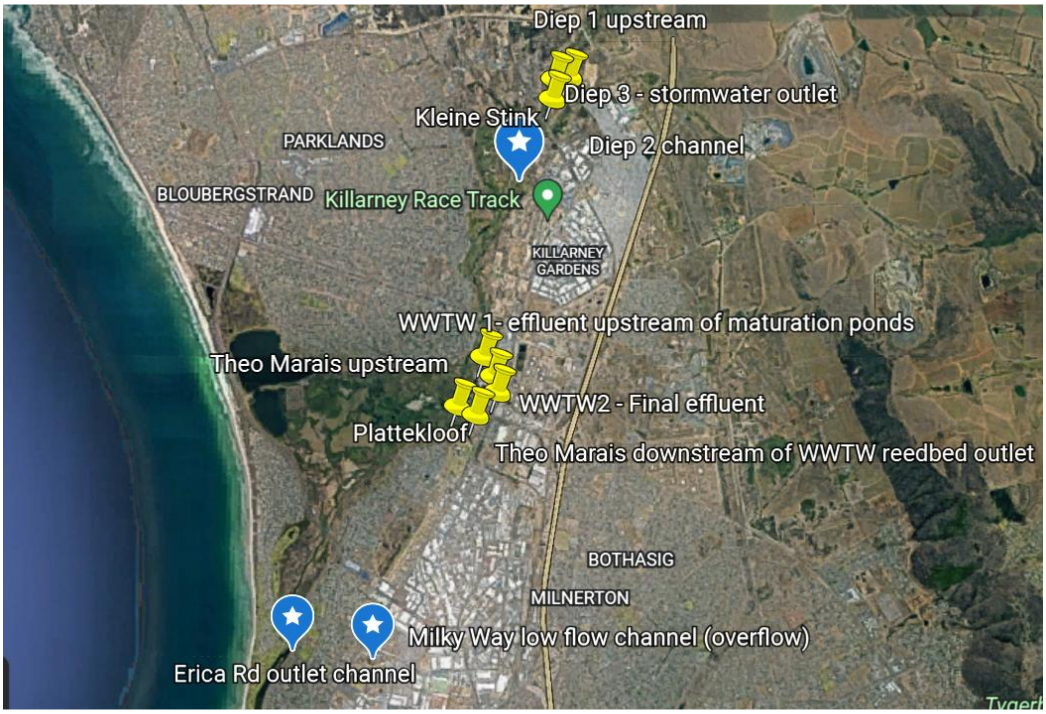


Figure 9-1. Three potential sites for intervention are indicated with blue pins.

9.3.1 Erica Road outfall

Erica Road outlet is the most accessible and manageable, but the volume of water is far too high for effective treatment of all flows; flows estimated at 5 – 7 Ml/day are already diverted to sewer.

Photo 9-2. Outlet channel near Erica Road which is currently overpumping contaminated water from Joe Slovo and other sources to the sewerage system on the opposite side of Otto du Plessis Drive. Arrow indicates a possible route to an existing low flow channel alongside the road.

A series of biofilters could be placed in a low flow diversion channel running parallel to the Otto du Plessis Drive (Figure 9-2). A swale would be excavated between the Erica Road outfall and the existing swale running along Otto du Plessis Drive to the south. The new swale would be 3m wide and 2.5m deep, with its dimensions, grade and elevation to be based on a required flowthrough rate of no more than 1m/s. The channel would be filled with large stone aggregates (25 mm diameter) for the first 25 m and the remaining 50m with biochar, to a depth of 1.5m. A weir at the downstream end would allow for aeration before discharging into the existing roadside channel.

Figure 9-2. Proposed biofilter diversion channel to link with existing swale alongside Otto du Plessis Drive.

Photo 9-3. Proposed site of excavation channel

9.3.2 Milky Way

A recently installed diversion channel in Phoenix, commissioned by the City, diverts contaminated water from Joe Slovo to sewer. Observations from two site visits indicated that the diversion is not fully functional, possibly because of load shedding issues or malfunction of pumps. There is potential to install another low diversion channel alongside the existing stretch of channel as shown in Figure 9-4.

There may be potential for improvement of low-flow polishing by constructing a second channel parallel with the existing one. A pilot test is recommended comprising a series of biofilters that are placed in a low flow diversion channel running parallel to the existing channel. The channel can be packed with a large stone aggregate (19-25mm) and biochar followed by a series of low weirs to offer some aeration before it enters the culvert downstream. The minimum channel size should be between 70 and 100 cm in depth and 3 metres in width. Since there is high volume of flow, and therefore limited retention time, it recommended the filters should be at least 50m in length based on the research data from the Water Hub (Winter, et al. 2023). A litter trap and constructed weir would also be required. The use of robust polywood should be considered at this site as a way of reducing cost and reducing the risk of theft and vandalism.

The installation costs of the proposed biofiltration media are roughly estimated at R400 000 which includes transport cost, limited excavation, materials, labour and project supervision. The construction is relatively quick to implement and a temporary weir and link to a diversion channel would be cost effective. Construction should take no more than 10 working days depending on any site-specific challenges.

Photo 9-4. Milky Way channel transporting contaminated flow from Phoenix and Joe Slovo settlements

Figure 9-3. Aerial view of the Milky Way channel showing the diversion channel and sewer pump, and overflow downstream of Milky Way Drive. Yellow line indicating potential position to install low flow biofilter channel

9.3.3 Kleine Stink, Du Noon

Potential the most challenging site for an intervention and to achieve a short-term gain because of numerous small channels from that are being used as conduits for the discharge of unwanted water and sewage. Site access, monitoring, management, litter removal and safety are issues that need to be considered. Nevertheless, if participation process and incentives could be achieved through facilitation and willingness on the part of residents, then the Kleine Stink could be a significant intervention and ideal pilot site to test the efficacy of biofiltration using a series of low flow channel diversions. Public involvement in this project is likely to be long term, ongoing and will come with significant challenges. Despite the challenge, this part of the catchment is in significant need of interventions targeting water quality improvements.

Figure 9-4. Intervention sites in the Kleine Stink streams from Du Noon informal settlement

9.4 Potential conflicts in assignment of land for pilot study

All land that is envisaged for these pilot schemes is public open space, but this will need further investigation once a pilot study has been selected.

9.5 Limitations on the use of biofiltration for water quality

Retention time is crucial for bacteria to degrade ammonia to nitrite and nitrate, and ultimately release nitrogen to the atmosphere. Similarly, although less well understood, the development of a biofilm within and on the surfaces of filtration media is necessary to capture orthophosphate and other compounds. To compensate for the lack of retention in a low flow design, it will be necessary to increase the length and depth of the filter media designs relative to existing prototypes implemented by Winter et al. (2023). A pilot study will be necessary to confirm these assumptions.

9.6 Coarse estimates of biofilter water quality amelioration capacity

A rough estimate of the cost of each proposed interventions is between R250 000 and R400 000 depending on the site selection. It is recommended that only one site is selected as a pilot study.

9.7 Time frames

Once agreement is met with respect to the site and situation, including land ownership, legal compliance, and public participation/involvement in the project, the intervention is relatively quick to implement. The performance of the filtration beds is expected to increase after three to four months, especially if there is an opportunity to allow the free-flowing water, post-treatment, to aerate sufficiently before entering the Diep River system.

9.8 Legislation

The proposed treatment trains may constitute at least one listed activity in terms of the National Environmental Management Act, 1998 and the Environmental Impact Assessment Regulations, 2014 (as amended by GNR 324 and 327 of 2017) **if they occur within a watercourse or estuary, such as at the Erica Road outfall**:

- **Activity 19A of Listing Notice 1**: The infilling or depositing of any material of more than 5 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles or rock of more than 5 cubic metres from— (i) the seashore; (ii) the littoral active zone, an estuary or a distance of 100 metres inland of the highwater mark of the sea or an estuary, whichever distance is the greater; or (iii) the sea.

It is probable that the aeration would also be considered a water use in terms of section 21 (i) of the National Water Act, 1998, viz:

» altering the bed, banks, course or characteristics of a watercourse

In terms of the above legislative instruments, an **environmental authorisation** and a **water use licence** may therefore be required from the Department of Environmental Affairs and Development Planning, and the Department of Water and Sanitation respectively. These would require an application process of approximately nine months at minimum.

The Milky Way site is not expected to require environmental authorisation, and may be preferred for this reason.

9.9 Potential ecological risks and benefits associated with biofiltration

The development of a biofiltration train for the treatment of raw sewage at the Erica Road outfall would come at the cost of permanent alteration in the reedbed and estuarine margins abutting Otto Du Plessis Drive in that area. These margins, which act as a buffer between the road and the estuary, would be converted into permanent water quality treatment works, of untested efficacy, in the face of an ongoing stream of highly polluted outflows. Diversion to sewer of the bulk of these flows, both close to source (Milky Way area) and upstream of Erica Road, would be a more efficient and appropriate means of treating these outflows. An important risk of putting in place a biofiltration system at Erica Road would be that it may give the impression of having addressed the issue, thus reducing the motivation to fully address an untenable situation, without easing actual water quality impacts. Furthermore, the system would require high levels of maintenance, which would be difficult to monitor and is unlikely to be implemented.

In the Du Noon outfall area, where low-flow diversion opportunities are currently limited, the construction of a biofiltration train might alleviate a small degree of pollution. However, a significant risk would be further loss of floodplain wetland to constructed biofiltration tanks – the wetlands are in surprisingly good condition in the Du Noon area where they are too wet to have been constructed on, and these areas would be vulnerable to further loss and development encroachment if they were targeted for biofiltration works.

9.10 Conclusions and recommendations

In summary, nature-based treatment processes hold some promise in polishing runoff with low flow. In this case they mimic processes that are found in young wetland systems, but are dependent on in-situ conditions and variability in water quality and retention time to achieve the best performance.

A pilot project alongside the Milky Way channel would offer the best opportunity to test the efficacy through a systematic process of monitoring, analysis and incremental management without requiring the transformation of existing floodplain wetland. This is not expected to contribute significantly to improvements in the Milnerton Lagoon water quality, and should instead be viewed as an opportunity to trial a technology for potential future upscaling.

10 REMEDIATION OPTION 6: TREATMENT WETLANDS

10.1 Purpose of treatment wetlands

The use of wetlands for the upstream treatment of polluted water currently entering Milnerton Lagoon has been proposed by concerned members of the public and has been included in the terms of reference for this project team to consider. The purpose of treatment wetlands in the current context would be to effect an immediate improvement in the quality of water entering the lagoon.

10.2 Assumptions

It is assumed that the treatment wetlands, if implemented, would be surface flow (SF) wetlands, also referred to as free water surface systems (Fonder and Headley 2013) and that they would comprise artificial, constructed wetlands. The use of natural wetlands within the existing Rietvlei and Milnerton Lagoon systems would not be advocated, given the fact that these ecosystems have already been substantially degraded by a history of receipt of poor water quality. Their further use for water quality treatment is thus not considered further here.

10.3 Potential sites for treatment wetlands

10.3.1 Identification of potential treatment wetland sites

Figure 10-1 illustrates five areas where there is availability of space outside of the Diep River floodplain and within the vicinity of significant polluted inflows into Milnerton Lagoon. These sites have been identified on the basis of available open spaces, not currently used for public amenity and with little if any ecological value or sensitivity.

Five sites have been identified, with a total area of 7.3 ha. The smallest of these comprise two areas around the Milky Way stormwater channel downstream of Jo Slovo Park (0.49 and 0.76 ha each). The largest areas comprise two existing reedbeds (No. 2 and No. 3) south of the WWTW, of area 2.26 and 1.90 ha. A nominal fifth area is also indicated in the figure, immediately east of these wetlands, within the degraded open space area abutting the Theo Marais channel. The delineated area is a 1.89 ha area.

Figure 10-1. Identified open space areas that could be considered for treatment wetland establishment and/or remediation

10.3.2 Potential conflicts in assignment of land for treatment wetlands

The creation of wetlands for the treatment of runoff requires dedication of significant areas of land for these purposes that might otherwise be used for housing; recreational benefit; or other purposes. The two wetlands indicated around the existing channel in the Milky Way area would be created at potentially high social cost, as they would result in loss of scarce play areas (eastern potential wetland) while the establishment of dense reedbeds required to achieve a measure of purification through filtration and settlement would be associated with potentially increased security risks, as they would create areas with poor surveillance for passers-by. The Theo Marais site is also in use for stockpiling associated with bulk sewer and pump station upgrades, and may be required for dredge dewatering. Plans are reportedly in place to rehabilitate the Milky Way ponds with alternative treatment options.

Treatment wetlands moreover require ongoing maintenance, in order to retain their desired functionality and to prevent breakdown in the wetland's capacity to effect the retention of nutrients and other pollutants. Maintenance measures would include regular cutting back of reedbeds, to ensure their active growth and root function and prevent senescence; as well as periodic removal of contaminated wetland soils in which nutrient-build-up threatens ongoing wetland function.

10.4 Limitations on the use of wetlands for water quality amelioration

SF wetlands have been used widely world-wide for the amelioration of water quality impacted by various sources of pollution, including treated effluent from municipal waste sites. Kadlec (2020) notes, on the basis of > 20 years of data from different treatment wetland sites, that these wetlands are effective in <u>polishing</u> highly treated domestic wastewater before it is discharged into natural water bodies, but their efficacy is subject to the following upper water quality limits:

- Biological Oxygen Demand (BOD) – 15 mg/L;
- Ammonia nitrogen – 5 mg N/L
- Nitrate and nitrate nitrogen ("oxidized nitrogen") – 10 mg N/L;
- Total phosphorus - 2 mg P/L;
- TSS – 50 mg/L.

The Kadlec (2020) review notes that if water quality exceeds the above limits, wetland treatment will be impacted by the build-up of sediments; excessive eutrophication and/or the creation of anaerobic conditions that do not promote decomposition and nutrient uptake. **The use of wetlands for the treatment of runoff with high levels of sewage and associated nutrients and organic sediments is not advocated as an effective use of scarce space in an urban context**. The transfer of such effluent into an effectively managed WWTW for treatment is more effective.

Water quality from the point-source runoff of most concern in this study (i.e. the WWTW; the Erica Road outlet; the Joe Slovo channel through Milky Way; and the Du Noon outlets into the Kleine Stink and Diep Rivers) all exceeds the above parameters – in some cases by an order of magnitude.

10.5 Coarse estimates of wetland water quality amelioration capacity

Where treatment wetlands are utilised for polishing of urban stormwater that does not include black or grey water inflows, Kadlec (2020) suggests that a method of presumptive sizing of a potential treatment SF wetland is by calculation of wetland to watershed area. This method suggests that wetland size should be a specified fraction of the contributing watershed –in the range of 1-5%. As an example, Cerfonteyn and Day (2009) cite the size of the catchment area of the Theo Marais channel as some 1200 ha. Assuming that flows from this channel were not contaminated at times with raw sewage, a treatment wetland of 12 - 60 ha would thus be required to treat stormwater runoff from this catchment. This method of sizing of treatment wetlands does not however specify actual nutrient or sediment removal rates, and is rather used for broad planning purposes and in some municipal best-practice approaches for stormwater management in urban planning. With regard to sediment removal, many SF wetlands receiving normal urban stormwater inflows (i.e. not contaminated with poorly treated or raw sewage effluent) achieve high removal rates (50-60%) in wetlands sized at between 0.03 and 0.07% of the watershed area (in this case, 3.6 to 8.4 ha).

10.6 Time frames

Kadlec (2020) notes that treatment marshes typically take a period of one or two years to outgrow the start-up period. Experience in the Cape Town context supports these data, noting that at least one growing season is required for the establishment of wetland plants at levels where they are able to play a role in essential functions such as sediment trapping through filtration and sedimentation.

10.7 Legislation

There is a possibility that both the National Environmental Management Act and the National Water Act would apply to the creation of treatment wetlands. This would be dependent on whether the wetlands are created in a terrestrial area outside of the 1:100 year flood line (which would require no authorisations) or within the 1:100 year flood line (which would require authorisations is terms of both Acts). The important distinction between the two is whether the wetland is created within the "extent of a watercourse" which is defined in GN 509 of 2016 as " (a) the outer edge in the 1 in 100 year flood line and/or delineated riparian habitat, whichever is the greatest distance, measured from the middle of a watercourse of river, spring, natural channel, lake, or dam and (b) the delineated boundary (outer temporary zone) of any wetland or pan". Wetland development within a watercourse would require the abovementioned authorisations, which wetland development outside of a watercourse would not.

If the wetlands are developed within the extent of an existing watercourse, the following listed activities would be applicable, and a **basic assessment** would be required.

- **19 of Listing Notice 1:** The infilling of depositing of any material of more than 10 cubic metres into, or the dredging, excavation, removal or moving of soil, sand, shells, shell grit, pebbles, or rock of more than 10 cubic metres from a watercourse.

It should also be noted that a general authorisation for water use (in terms of the NWA) would be required, as sections 21(c) and 21(i) would be triggered by the wetland creations.

10.8 Ecological risks associated with the development of artificial treatment wetlands

Reedbed remediation (that is, removal of contaminated soils and vegetation and re-establishment of reedbed vegetation, potentially with earthworks designed to improve water quality polishing capacity, would be a positive ecological outcome compared to the present situation. Ecological risks include the likelihood of reedbed re-contamination from WWTW overflows – this is a serious consideration, but it could be remediated to some extent by the proposed division of the wetland into sections, with those closest to the WWTW designed rather as temporary pollutant holding areas in the event of overflows from the works.

Other ecological risks could include the passage of sediment into the bypass channel or Theo Marais channel – these systems are already exposed to poor water quality and high levels of TSS and their fauna and flora would be unlikely to show sensitivity to additional short term inputs of high TSS. Dewatering of the reedbeds during remediation could however pass noxious plugs of water into the above systems, with potential impacts on the lagoon downstream, particularly if it was in a state of rehabilitation following aeration and dredging. Any dewatered waste would therefore need to be directed into the WWTW for treatment.

Finally, it is possible that reedbed remediation earthworks could disturb nesting birds and other fauna in the reedbeds – these reedbeds are however highly contaminated and do not appear to support large numbers of nesting birds, although there is birdlife along their margins, abutting Theo Marais channel. Risk of bird impacts would be least if remediation was carried out in mid- to late summer, after spring nesting – this would in any case be recommended as the reedbeds would be driest.

The ecological advantages of reedbed remediation and the development of artificial treatment wetlands in the same area outweigh the ecological risks, which can all be mitigated. The creation of additional treatment wetlands in an effort to mitigate water quality effects of inflows of raw and /or poorly treated sewage into the lagoon is not recommended – it is likely to be ineffectual and have high space and management requirements that could be better focused on other activities, land uses and interventions at the WWTW.

10.9 Conclusions and recommendations

In the present situation, treatment wetlands might be used for polishing of stormwater runoff from relatively "clean" catchments – that is, catchments that are not exposed to essentially grey and black water inflows. The Killarney sub-catchment and runoff from parts of the Montague Gardens area could lend themselves to passage through a treatment wetland, provided that this was adequately designed to prevent hydraulic short-circuiting and ensure efficient utilisation of available space and adequate retention times (typically seven days or longer - Kadlec and Knight 1996 and Kadlec 2020).

If the existing reedbed area south of the WWTW were used for these purposes, they would first require extensive remediation, including removal of contaminated sediments and the re-establishment of reedbed vegetation. This is in any case a strongly recommended intervention, as the reedbeds have been contaminated by a history of receipt of raw sewage overflows as well as stormwater and are assumed to play a role in actively contributing to pollution loads into Milnerton Lagoon. In this case, it is recommended that the wetlands should be designed to include capacity for the isolation of WWTW overflows within contained areas of the wetland, rather than allowing them to spread over into and contaminate the entire wetland. Stormwater flows from the refinery area might reasonably be treated within this wetland, which would require ongoing management.

While the above measures are recommended as essential contamination remediation measures, they are not short-term measures; they require legal authorisation, and they would not in themselves be likely to bring about any measurable change in the current condition of the Milnerton Lagoon.

Additional treatment wetlands might also be created in the area coarsely indicated east of the existing WWTW reedbeds. The actual benefit that would derive from such a measure, compared to the land costs and subsequent maintenance requirements for the wetland are however questionable and water quality would arguably be better served with investment in urgent interventions at the WWTW and in at-source measures such as the provision of back-up generators and other mechanisms to improve pump station efficacy under load shedding conditions; and the diversion of contaminated stormwater flows to sewers.

11 REMEDIATION OPTION 7: MICROBIAL OR ENZYMATIC INOCULATION

11.1 Scope and purpose

'Bioremediation' is a catch-all term for the use of naturally occurring or introduced organisms to break down environmental pollutants. Microbial bioremediation is the use of microorganisms to degrade and remove pollutants from contaminated environments, such as sewage-polluted waters. This approach relies on the metabolic activity of microorganisms such as bacteria, fungi, and algae, which break down pollutants through various biochemical pathways, producing carbon dioxide, water, and other by-products. The effectiveness of microbial bioremediation depends on various factors, including:

- the type and concentration of pollutants;
- the desired end products;
- environmental conditions, such as salinity, temperature, oxygen levels, pH, nutrient availability, and light;
- retention time, or the length of time microbes have to act on the pollutants; and
- the diversity and abundance of microbial communities.

A number of specific products have been recommended to the City of Cape Town by commercial producers and suppliers as potential solutions to the build up of sludge-like organic sediment in the Milnerton Lagoon. These products are in many cases repurposed from their intended use in closed systems such as septic tanks, pit toilets, or wastewater treatment plants. Their employment in a natural system with flowing water, varying salinity and temperature, and generally dynamic conditions requires detailed consideration. Although some microorganisms can persist in extreme environments, optimal growth (and therefore effectiveness) may be difficult to achieve under in situ conditions.

11.2 Method of assessment

To explore the potential for application of commercially available products in the Milnerton Lagoon, the City of Cape Town's Scientific Services Branch issued a formal Request for Information (RFI024/2022/23). See **Annexure E** for a copy of the RFI.

Because of the proprietary nature of, and scarcity of detailed technical information regarding commercial preparations of microbes and enzymes used for bioremediation, this study could not rely on published literature to recommend products for implementation. Instead, the onus was placed on suppliers to document the efficacy of the products proposed, and where no evidence was provided this study did not make a finding on whether a given product could be utilised in the Milnerton Lagoon,

Prospective suppliers were asked to submit information to assist in determining which products are commercially available that will assist with the breakdown of organic matter in sediments into non-toxic substances that can easily be flushed out of the water body with natural flow. Products were required to have a proven track record of application with relevant supporting data and information. Suppliers were requested to confirm dosing concentration; application method and frequency; costs; and time required for each intervention to work effectively. They were also asked to confirm whether the proposed products complied with the South African Water Quality Guidelines for Aquatic Ecosystems (as per DWAF 1996) when diluted at supplier-recommended dosing rates. Onekilogramme samples, supporting documentation, and a standardised set of information was to be supplied to the City of Cape Town without any obligation to use such products or proceed with procurement.

Where available, supporting documentation in the form of product brochures or marketing material; material safety datasheets; scientific or technical reports on case studies; product case studies with or without accompanying primary water quality or sediment quality data; and/or peer-reviewed or published reports on case studies was requested.

The RFI closed on 3 March 2023, by which time a total of 12 suppliers had submitted products with varying levels of supporting information ranging from material safety datasheets only to technical reports.

Documentary submissions were reviewed in detail by the professional team, with a focus on evidence for the following aspects:

» **Nature of evidence provided**, with a preference for scientific or published reports, primary data or technical reports over photographs or generalised statements of efficacy.
» **Compliance with the SA Water Quality Guidelines for Aquatic Ecosystems**, to reduce the risk of potential harms. Note that suppliers were asked to confirm this; of the 14 submissions, nine claimed compliance, two indicated non-compliance, and three did not respond. Testing was not conducted by the project team.
» **Evidence of the product's efficacy in a saline or brackish environment**. The Milnerton Lagoon is significantly more saline than a typical septic tank or wastewater maturation pond, and suppliers were advised in the RFI of the wide salinity ranges in which a product would need to be effective.
» **Indication of the retention time required in order to achieve effective outcomes using the product.** As indicated above, many commercial products proposed are repurposed from their primary application in standing water or septic tank systems, while the Milnerton Lagoon has significantly lower retention time with a constant freshwater inflow from the catchment and WWTW.
» **Dosing rates** indicated by the manufacturer.

Table 11-1 below lists the submissions received, in an anonymised form, with a summary of evidence for the above aspects and a recommendation by the professional team.

Table 11-1. RFI Responses

ID	Tested in saline environment	WQ guideline compliance claimed	Retention time	Dosing rates	Recommendation
A	No evidence provided	Yes	No data provided - references are to closed systems	10 kg per megalitre of liquid	Insufficient information / evidence of efficacy
B	No evidence provided	Yes	No data provided	Not provided	Insufficient information / evidence of efficacy
C	No evidence provided	Yes	Case studies: 10-14 days standing (no controls). Rutter Pond with limited exchange.	1kg proposed for Milnerton Lagoon	Insufficient information / evidence of efficacy
D (1)	Case study provided indicating efficacy in salinity of 10 to 17 ppt	Yes	Case study provided indicating efficacy with 7 hours retention and 26 000 m^3 flow per day	60 litres proposed per 10 megalitres of flow	**Some evidence of efficacy – an *in situ* test is recommended**

ID	Tested in saline environment	WQ guideline compliance claimed	Retention time	Dosing rates	Recommendation
E (2)	No evidence provided	Unknown	Case studies are largely in WWTW or ponds. Retention time not specified	Variable (different products) rates per cubic metre proposed.	**Some evidence of efficacy – an *in situ* test in a saline environment is recommended**
F	No evidence provided	No	Retention time not described and no case studies of flowing systems provided	40 tons proposed for Milnerton Lagoon	Insufficient information / evidence of efficacy
G	No evidence provided	Yes	One case study in WWTW pond - no flushing.	Provided – one block proposed per 0.5 megalitres per day	Insufficient information / evidence of efficacy
H	No evidence provided	No	No data provided	Unclear	Insufficient information / evidence of efficacy
I	No evidence provided	Yes	No indication - no case studies provided	Rate pf 1:1000 proposed	Insufficient information / evidence of efficacy
J (3)	Case study provided of an estuarine system	Yes	Case study presented of a flow-through system in an estuary in addition to a closed freshwater pond	Rate of 50 to 100 parts per million proposed	**Some evidence of efficacy – an *in situ* test is recommended**
K	No evidence provided	Yes	No in situ trials or case studies - efficacy demonstrated in controlled, lab conditions with long retention times including a 96 hour closed system trial	Rate of 5 kg per megalitre proposed	Insufficient information / evidence of efficacy
L	No evidence provided	No	Testimonials are pit toilets and small-scale with no quantification	Rate of 125 g per m^3 proposed	Insufficient information / evidence of efficacy

11.3 Ecotoxicity testing

Products identified D, E and J (anonymised) have been selected based on the documentary evidence provided by the suppliers as showing some promise of efficacy in an *in situ* application.

These three products were analysed by the City of Cape Town's Scientific Services Branch to determine the potential ecotoxicity in conditions resembling the Milnerton Lagoon. They are identified below as **1** (Product D), **2**, (Product E), and **3** (Product J).

11.3.1.1 Aims

The aim of laboratory testing of products is to determine whether any of the products result in toxic conditions likely to impact on the health of aquatic organisms exposed to its use as a bioremediation treatment. Given that the three products all include enzymes and or bacteria in various media, designed to inactivate living material until released into environmental waters, such toxicity would be unlikely. However, best practice and the Precautionary Principle indicate that at least coarse-level testing of potential toxicity impacts should be investigated prior to any field-based trials or at-scale interventions. The aim of the laboratory testing was not to test product efficacy. The ecotoxicity analyses are listed in Table 11-2. In addition to these tests, various in situ and laboratory water quality measurements were also carried out on water samples collected at different stages in the laboratory trials.

Table 11-2. Ecotoxicity analyses carried out on the products

Analysis	Test Method
Vibrio fischeri	*Vibrio fischeri* toxicity test is the BIOTOX™ kit
Selenastrum	*Selenastrum capricornutum* (renamed to *Pseudokirchneriella subcapitata*) toxicity test is the ALGALTOXKIT F™
Daphnia magna	*Daphnia magna* toxicity tests is the DAPHTOXKIT F™ from MICROBIOTEST(Inc)

11.3.1.2 Test location

Water from the Milnerton Lagoon was collected (Longitude 18°29'25.348"E; Latitude 33°52'55.141"S). The laboratory experiment was conducted at Scientific Services Branch (SSB) in the Research and Development (R&D) laboratory.

11.3.1.3 Sample collection

Water for use in the laboratory tests was collected from Milnerton Lagoon at 6:30 a.m. on the 12th of May with an expected high tide at 8:41 a.m. and a swell height of 1.34 m. Water was collected using 10 L galvanised steel buckets and then decanted into two 220 L plastic containers until 100 L of water was sampled. This water was transported to Scientific Services Branch (SSB) laboratory immediately after collection. All containers used in the collection and subsequent laboratory experimentation were washed with methanol, and rinsed three times with reverse osmosis (RO) water and once with Milnerton Lagoon water before use.

11.3.1.4 Experimental method

- The experimental design allowed for four independent replicate samples for each experimental treatment and control sample;
- Experimental treatments comprised each of the three (anonymous) numbered submissions (4,5 and 10); Follow up communication with the respective suppliers was initiated by the City's Scientific Services Branch in which the optimum dosage for 8 L Milnerton Lagoon samples was requested and whether their products required once-off or multiple repeat dosages over a 5-day (120 h) experimental period and the dosage for the treatment was set accordingly;
- The two 220 L containers filled with 100 L of Milnerton Lagoon water were gently stirred with a 5 L measuring beaker before each water sample was aliquoted into each experimental bucket. From the one 100 L sample 5 L water was aliquoted in each of the 16 experimental buckets, 3 L from the other 100 L sample was then also aliquoted into each experimental bucket to a total volume of 8 L in each of the 16 buckets. The three products were then added to each experimental bucket according to the supplier instructions. After product addition, each bucket was gently mixed for 3 min with sterile plastic pipettes. The control samples received no addition of product and were also gently mixed for 3 min with sterile plastic pipettes.
- The sixteen, 13 L experimental buckets were arranged on the laboratory bench with respect to the window as outlined in Figure 11-1Figure 11-1.
- The lights in the laboratory were switched off for the entire duration of the experiment to ensure that the buckets were exposed to the natural diurnal cycle. After gentle mixing, the samples were left stagnant for 60 min. After 60 min various *in situ* measurements were taken using a YSI-multimeter. Turbidity was measured by sampling 10 mL from each bucket with a syringe and readings were taken using a Hach TU5200 turbidimeter.
- A 1 L sample was then siphoned from the top aqueous layer and submitted for chemistry and eco-toxicity analyses.

» *In situ* measurements and turbidity readings were recorded at the following time intervals: 24, 48, 72, 96, and 120 h. After 120 h another 1 L sample was siphoned from the top aqueous layer of each bucket and submitted for chemical and eco-toxicity analyses. After the 1 L sample was taken at 120 h the remaining samples were vigorously homogenised for 1 min after which a 2 L sample was taken and submitted for sediment analysis. The time intervals and determinants tested are outlined in Table 11-3.

Window

Bio-enzyme 1:C	Bio-enzyme 2:C	Bio-enzyme 3:C	Control: C	Bio-enzyme 3:D	Bio-enzyme 2:D	Bio-enzyme 1:D	Control:D
Control: A	Bio-enzyme 1:A	Bio-enzyme 2:A	Bio-enzyme 3:A	Bio-enzyme 1: B	Bio-enzyme 2:B	Bio-enzyme 3:B	Control: B

Figure 11-1. Sample layout in the laboratory

Table 11-3. Outline of when *in-situ* measurements were done and samples taken and submitted for various analyses.

Time	60 min	24 h	48 h	72 h	96 h	120 h
Date	**12 May**	**13 May**	**14 May**	**15 May**	**16 May**	**17 May**
***In situ* measurements**						
pH	x	x	x	x	x	x
Dissolved oxygen (DO)	x	x	x	x	x	x
Electrical conductivity (EC)	x	x	x	x	x	x
Temperature	x	x	x	x	x	x
Salinity	x	x	x	x	x	x
Turbidity	x	x	x	x	x	x
Chemistry determinants						
Ammonia	x					x
Nitrate + Nitrite as Nitrogen	x					x
Un-ionised ammonia	x					x
Ortho-phosphate (OP)	x					x
Inorganic nitrogen	x					x
Chlorophyll-A	x					x
Eco-toxicity determinants						
Vibrio fischeri	x					x
Selenastrum	x					x
Daphnia magna	x					x
Sediment analysis						
Total solids (TS)						x
Volatile solids (VS)						x
Total suspended solids (TSS)						x
Settable Solids (SS)						x

Table 11-4. Interpretation of toxicity hazard (after Persoone et al., 2003)

Effect Percentage	Hazard		Symbol	Explanation
< 20%	Class I	No Acute Hazard	☺	None of the tests show a toxic effect (i.e. an effect value that is significantly higher than that in the control)
20% < EP < 50%	Class II	Slight Acute Hazard	☹	A statistically significant EP is reached in at least one test. The effect level is below 50%
50% < EP <100%	Class III	Acute Hazard	☠	The EP50 is reached or exceeded in at least one test but the effect level is below 100%
EP = 100% in at least one test	Class IV	High Acute Hazard	☠ ☠	The EP100 is reached in at least one test
EP = 100% in all tests	Class V	Very High Acute Hazard	☠ ☠ ☠	The EP100 is reached in all tests

11.3.1.5 <u>Data analysis</u>

The results of the laboratory testing are illustrated in Figure 11-2 to Figure 11-4 for key water quality and ecotoxicity variables. Variables that were measured only at the start and end of the experiment (i.e. phaeophytin, chlorophyll-a, orthophosphate, volatile solids and free ammonia (NH_3)) were analysed by difference – with positive values indicating an increase from starting concentrations and negative values indicating a decrease (Figure 11-2). The other variables, measured at intervals throughout the experiment, are presented as mean data over time, with standard deviations between replicates. Standard deviation around the mean was generally very low, and in many cases thus not clearly visible in the graphics (Figure 11-4).

These data indicate the following broad patterns:

- Treatments 1 and 3 were generally similar in terms of their impact on lagoon water quality, and statistically similar with the control treatment (Treatment 4);
- The only instance where the above did not hold true was for phaeophytin, where Treatment 1 resulted in a significantly smaller increase than Treatment 2, 3 and the control;
- Treatment 2 resulted in significantly increased TSS, chlorophyll-*a* and volatile solids (i.e. organic sediments), and significantly lower orthophosphate phosphorus (PO4-P) than the control or other two treatments – the increased chlorophyll-*a* levels in Treatment 2 suggest increased phytoplankton growth and algal uptake of orthophosphate might also explain the marked reduction in orthophosphate for this treatment;
- Free ammonia (NH_3) increased in all treatments including the control between hour 1 and hour 120. At hour 1, NH_3 concentrations varied between 0.199 and 0.178 mg/L, and at hour 120, concentrations ranged between 0.63 and 0.61 mg/L for Treatments 1, 3 and the control, but rose only to 0.23 mg/L for treatment 2. It should however be noted that all of these concentrations lay well within the range for acute ammonia toxicity (> 0.1 mg/L, as per DWAF 1996 and DWAF 2008);
- EC values increased in all three treatments and the control, with no statistical difference between them, and it is assumed that this was due to a level of evapoconcentration over the experimental period;
- pH values dropped in Treatment 2 after throughout the experiment after hour 1, and this explains the lower concentrations of NH_3 in this treatment, since larger proportions of total

ammonia are in the potentially toxic NH_3 form at elevated pH, particularly at pH > 8 – mean pH approached pH 8 in all three of the other treatments;

- Temperature increased in all three treatments over time, at near identical levels, suggesting similar laboratory conditions that were perhaps more stable and warmer than lagoon or transport conditions;
- Turbidity was markedly higher in Treatment 2 than in the others from the start of the experiment, in all treatment replicates. Although it reduced thereafter, it remained significantly higher than in the other treatments and control;
- Dissolved oxygen concentrations generally decreased over time in all three treatments and the control, from a relatively high level (>8.0 mg/L). Hour 1 DO concentrations are assumed to have been influenced (elevated) by stirring and transportation prior to the experiment, across all treatments and the control, and were markedly higher than measured lagoon water DO;
- DO concentrations for Treatment 2 dropped markedly, to well below acute oxygen deprivation levels, and remained in this range throughout the experiment, albeit increasing slightly by Hour 120.

The above discussion both highlights **the existing toxicity of the Milnerton Lagoon** even without added products (ammonia toxicity) and suggests that **Product 2 (RFI submission E) would be problematic** in the context of the lagoon with its already depleted oxygen concentrations, because of its impact on dissolved oxygen.

The results of the actual ecotoxicity tests undertaken by the SSB (Figure 11-3) indicate that:

- *Daphnia magna* were markedly affected in all treatments in Hour 1, with the greatest effects (into the "acute hazard" range in two of four replicates) in both Treatment 1 and 2, but with three of four replicates in the control site also falling within the "slight acute hazard". *Daphnia magna* occurs in fresh and brackish waters but is not a marine species, and the toxicity tests might reflect generally poor habitat for these crustaceans rather than induced toxicity. It is however interesting and not explainable that much lower mortalities consistently occurred in water collected at Hour 120 across all treatments and the control, unless an element of aeration occurred during water extraction;
- *Selenastrum capricornatum* generally responded positively to all four treatments, indicating no significant toxicity to this algal species as a result of either lagoon water quality or treatment additives. Low oxygen concentrations would presumably not be problematic to this algae, as photosynthesis produces oxygen. Only Treatment 1 resulted in even slight increases in orthophosphate (see Figure 11-2) and algal growth is assumed to reflect a natural response to nutrient availability in the water, rather than suggesting that the treatments themselves might exacerbate algal growth – this is supported by similar levels of growth in the controls.
- Treatment impacts on *Vibrio fischeri* varied, with generally enhanced growth in water from the 1 Hour samples for Treatments 1 and 2, and generally increased mortalities in the 120 Hour samples, and for most replicates in the 1 Hour samples for Treatment 3 and the control – mortality levels were below or just within the "slight acute hazard" range.

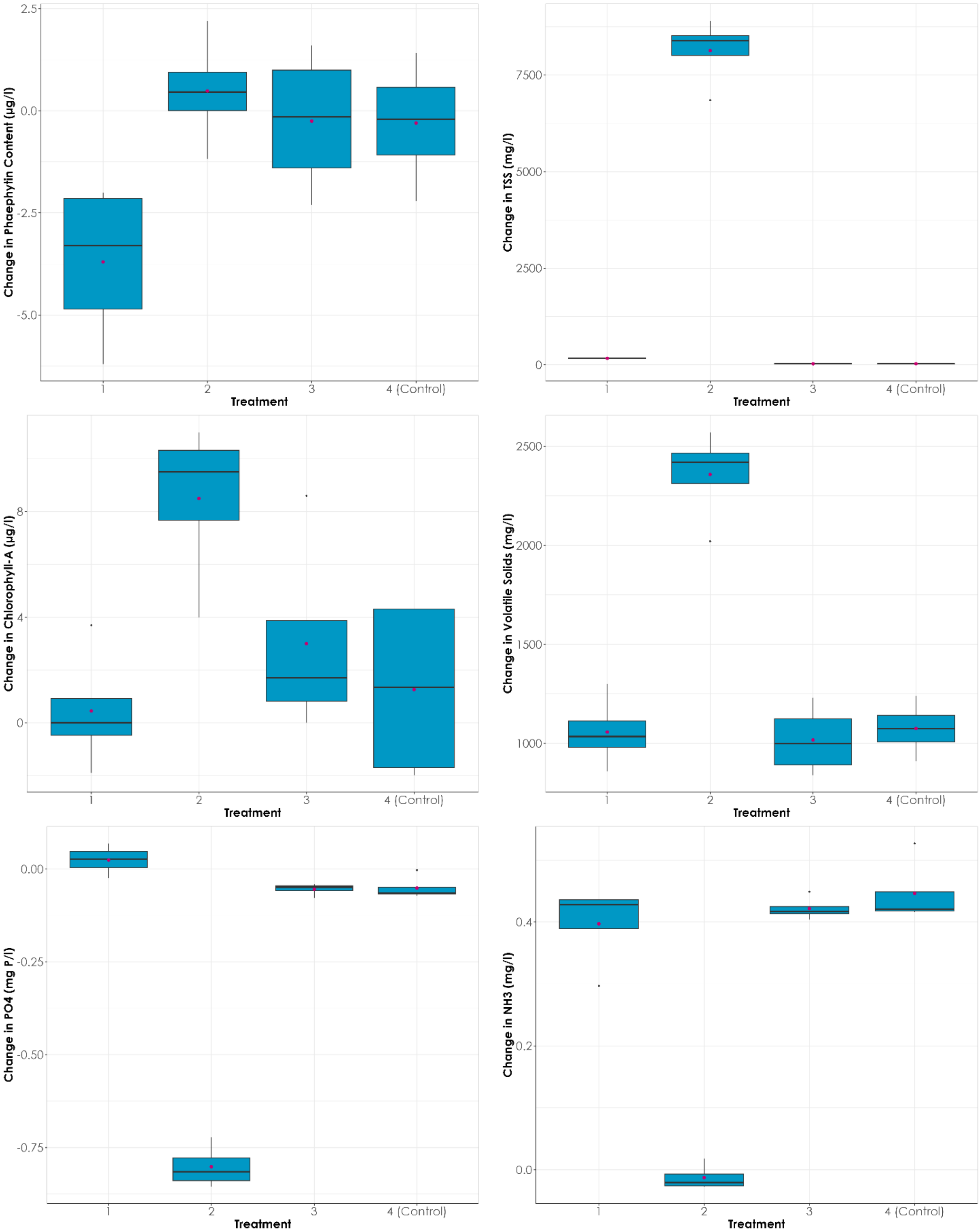


Figure 11-2. Data indicating change in concentration in variables analysed one hour and 120 hours after commencement of the laboratory experiments. Box plots show mean (red dot), median (line), inter-quartile range (blue box) and total range.

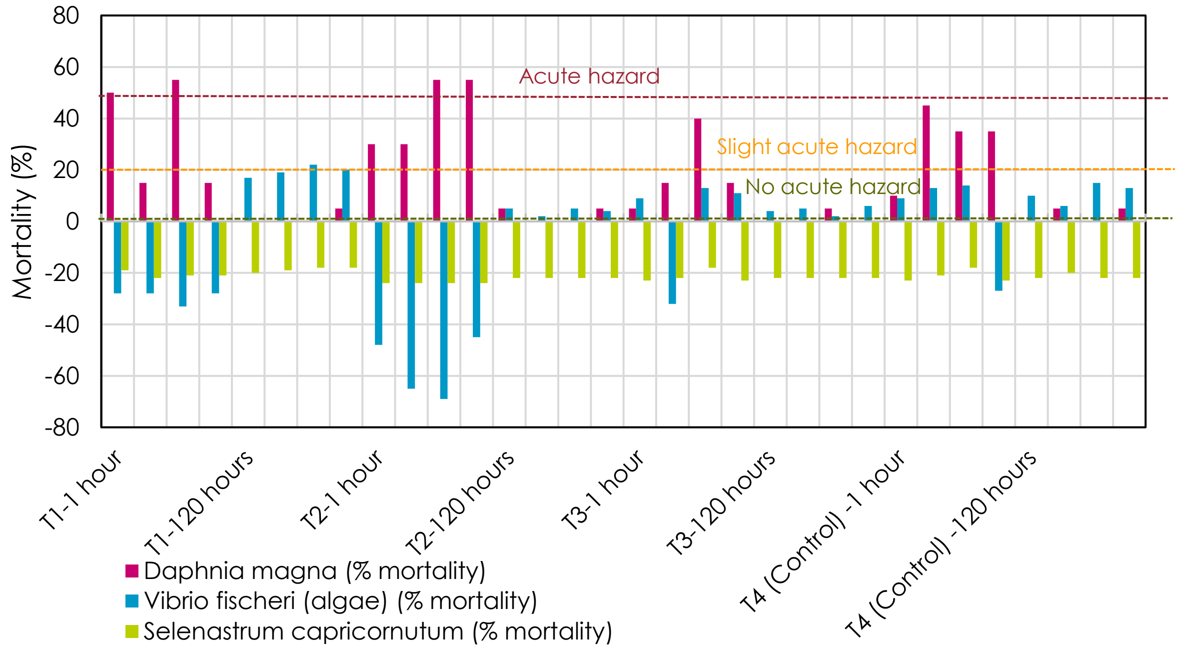


Figure 11-3. Ecotoxicity results, with interpretative thresholds based on Table 11.2. "T" prefix indicates Treatment number (i.e. product or control). Negative values indicate a positive impact. Positive values indicate mortalities.

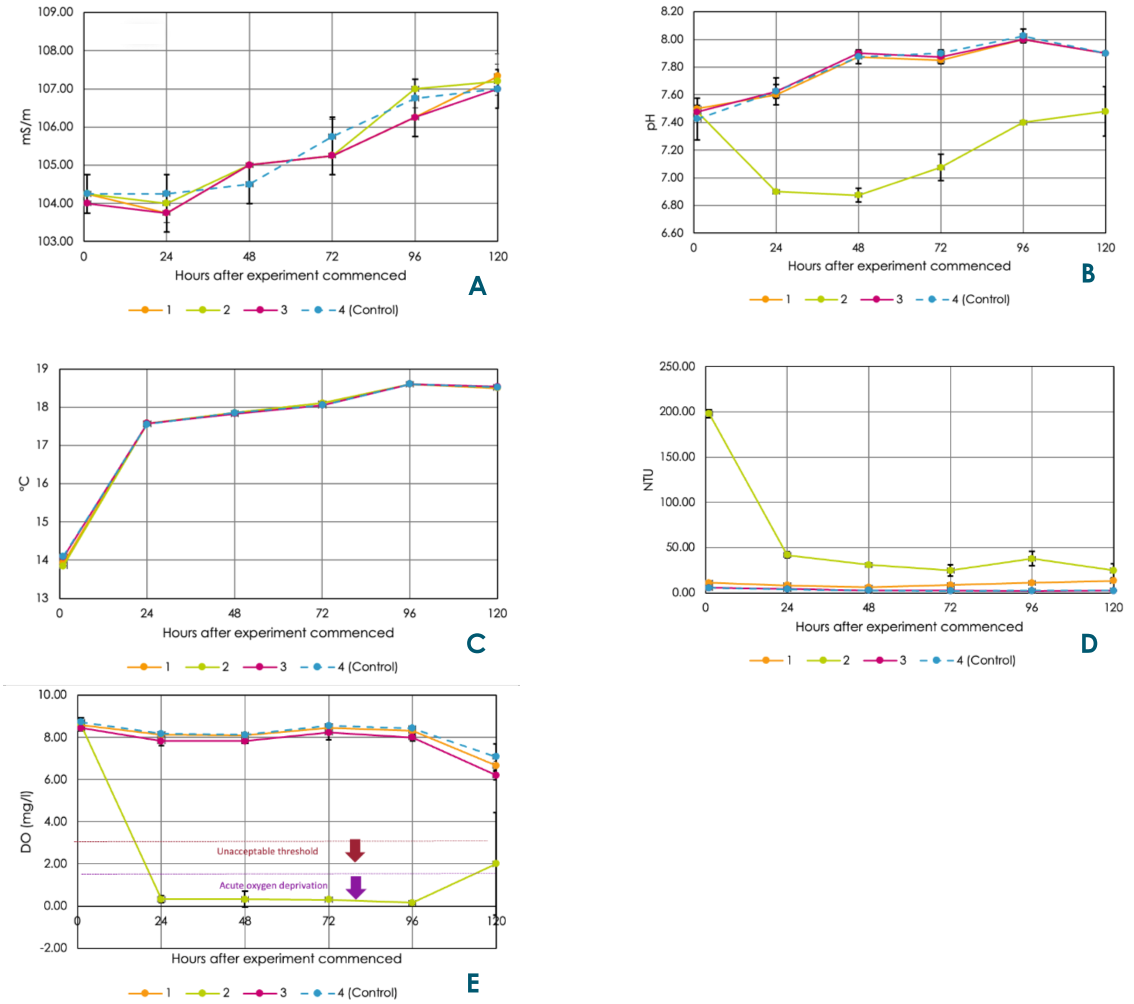


Figure 11-4. Analyses of water quality changes during laboratory testing of products 1-3 and control (4). (A) Electrical conductivity (B) pH (C) Temperature (D) Turbidity (E) Dissolved Oxygen

11.3.1.6 Outcomes

None of the products tested showed strong ecotoxicity, although the consistent drop in DO for Treatment 2 (Product E) is of concern, particularly in Milnerton Lagoon.

> On this basis, it is recommended that at least **Products D and J** (Treatments 1 and 3) should be taken forward for trial in lagoon-based mesocosms (as set out in the following section). It is suggested that this trialling proceed with urgency, and final decisions around the extent of dredging versus the extent of bioremediation should be made on the basis of the efficacy of these trials.

11.4 Recommended approach for field-based mesocosm-type efficacy tests

An experimental design for field-based mesocosm efficacy tests is presented below.

Two pivotal requirements are applicable:

1. **Replication**: Multiple trials must be conducted using the same product/technique and results achieved should be evaluated for the consistency or reliability. Only if the same or similar results are achieved in all tests can the product be considered reliable;
2. **Use of control or reference samples**: The experimental design must incorporate control or reference samples which are treated in a manner that is as similar as possible to the treated samples except that no treatment is applied, that can be used to benchmark the efficacy of the treatment being considered such that the possible influences of the experimental approach can be ruled out. This is especially important for laboratory-based and mesocosm experiments where the experimental conditions (e.g. any kind of enclosure introduced to contain the experiment) can influence the outcomes.

A mesocosm (meso- or 'medium' and -cosm 'world') is any outdoor experimental system that examines the natural environment under controlled conditions (Wikipedia 2023). They tend to be medium-sized to large (e.g., 1 - 10,000 litres) and typically contain multiple trophic levels of interacting organisms. In contrast to laboratory experiments, mesocosm studies are normally conducted outdoors in order to incorporate natural variation (e.g., diel cycles). Mesocosm studies may be conducted in either an enclosure that is small enough that key variables can be brought under control or by field-collecting key components of the natural environment for further experimentation. Mesocosm studies provide a link between field surveys and highly controlled laboratory experiments.

For the purposes of this study we recommended condition mesocosm-type trials with three of the product samples submitted by suppliers to the CoCT that were identified as showing some promise based on information provided by the suppliers.

Given the limited amount of product available from each supplier, and taking account of the requirements outline above, it is recommended that mesocosm experiments be set up in the most severely afflicted portion of the Milnerton Lagoon (i.e. immediately above or below the Wooden bridge) comprising of suite of steel or plastic sided cylindrical structures between 0.5 and 1.0 m in diameter that can be pushed into the sediment to a depth of around 30 cm. These structures must be open at the top and bottom and the height of the structure should be such that it extends up to the mid tide level at the point of deployment. Water depth at the point of deployment should be approximately 1 m ±0.1 m depth at mid tide (the vertical height of the cylinder will thus need to be approximately 1.3 m ±0.1 m). This will enable water from the estuary to overtop and flow into the cylinder at high tide but the contents of the cylinder will be isolated from the surrounding water at low tide. This will allow a period of time when the water in the cylinder will be isolated from the surrounding environment for a period of time (2-3 hours) any treatment applied will be able to exert its influence to maximum advantage. The cylinders may need to be anchored with ropes on the sides to ensure that they are not carried away by the prevailing currents. We recommended a minimum of three such structures per treatment (to allow for adequate replication) as well as a corresponding number of cylinders that can act as reference or control sites (i.e. will include any influence that the structures may exert on the substratum but to which no active treatment has been applied). The structures should be arranged in three groups of four (three treatment and one control in each group) and numbers (1-4) assigned to each cylinder in a randomised fashion. Care should be taken not to disturb the sediment surface inside of the cylinder area during installation, and should probably undertaken by boat. First application of the product to the various cylinders (the same product going into the cylinder with the same number in each case) should only be undertaken 72 hours after the cylinders have been installed and it is clear that the cylinders are stable and are not at risk of toppling over or being carried away by the prevailing currents. Application of the product should be undertaken at the same time for each product (immediately after the tide level has dropped below the top of the cylinder) and the means of application (including dosage and frequency of application) should be undertaken in accordance with the manufactures' specifications. In most cases, this will require repeated application once per day after the tide level has dropped below the height of the top of the cylinder.

Samples of water (1 l, collected from approximately 0.5 m below the water surface) and samples of sediment (0.5 l, collected using a grab sampler) should be collected from each cylinder (including the control cylinders) at the time of low tide on each day at the following intervals:

1. 48 h after each cylinder has been installed in the estuary but prior to application of any product to any of the cylinders;
2. 24 h, 48, 72, 96, and 120 h after first application of the various products. The timing of the measurements may have to be adjusted slightly to correspond with low tide on each day. Samples should be collected as close in time to one another as possible.

Sediment samples should be analysed for the following parameters:

- Particle size distribution (PSD);
- Total Organic Carbon (TOC); and
- Total Organic Nitrogen (TON)

Water samples should be analysed for the following parameters:

- Nitrate (NO_3-N)
- Nitrite (NO_2-N)
- Ammonia (NH_4-N)
- Orthophosphate (PO_4-P)

In situ measurements should be made of the following water quality parameters at the same time that the water samples are collected as well as between 30-60 minutes after application of the product in each compartment:

- Temperature
- Conductivity/salinity
- pH
- Dissolved oxygen (mg/l and % saturation)

A comparative statistical assessment should be undertaken of all water and sediment quality data collected for each treatment to ascertain if there are any statistically significant differences between the treatment and controls samples.

11.5 Ecological risks associated with the deployment of microbial or enzymatic bioremediation measures

It is possible that deployment of the suggested microbial or enzymatic bioremediation measures discussed in this section could result in toxicity to lagoon fauna and flora, which would be a significant issue if it was prolonged. The likelihood of this is however very low, as the products recommended for further testing all include microbial components that would also be vulnerable to acute toxicity effects. One product is however stored in a hydrogen sulphide matrix, used to keep the bacteria dormant until treatment commences, and deployment might result in short-term localised increases in hydrogen sulphide in the lagoon. The lagoon environment is however already prone to the development of hydrogen sulphide and this impact is considered of low significance.

The proposed laboratory tests should eliminate any products with a real risk of creating toxic conditions in the lagoon, and it should be noted that at least at present, the lagoon environment is already highly toxic to aquatic life.

Other possible ecological risks associated with some products would be the creation of extensive flocs on the lagoon bed, as a result of floccing agents, or of effective infilling of treatment areas, where large volumes of product are required, nested in fine clay or other inorganic matrices. These products have not however been selected for further testing in this study.

12 HYDRODYNAMIC AND SEDIMENT TRANSPORT MODELLING OF DREDGING OPTIONS

Hydrodynamic modelling with sediment transport was used to evaluate the dredging options (i.e. **Option 1**) to determine the effect dredging would have on flow patterns, deposition rate of sediment and areas that the contaminated sediment may settle out after dredging. Table 12-1 shows the scenarios that have been identified to evaluate dredging as a remediation option to improve the water quality of Milnerton Lagoon.

Table 12-1. Hydrodynamic modelling scenarios

Scenario	Description	Simulation type	Simulation period	Inflow model boundary	Inflow sediment load (t)	Bathymetry
1	Calibration	Hydrodynamic only	2014/03/11 – 2014/03/26	SHETRAN + Potsdam observed flows	0	2023 survey modified to 2014 mouth conditions
2	Two-week spring tide	Hydrodynamic with sediment transport	2022/03/27 – 2022/04/10	SHETRAN + Potsdam flows	114 (8.16 t/d)	2023 survey
3	2-year Annual Recurrence Interval (ARI) flood hydrograph	Hydrodynamic with sediment transport	79 hours (3d 7h)	Flood hydrograph	1345 (409.6 t/d)	2023 survey
4	10-year ARI flood hydrograph	Hydrodynamic with sediment transport	79 hours (3d 7h)	Flood hydrograph	26245 (7993.3t/d)	2023 survey
5	Phase 1 dredged	Hydrodynamic with sediment transport	2021/01/01 – 2022/01/31	SHETRAN + Potsdam flows	21170 (58.0 t/d)	2023 survey dredged to Phase 1
6	Phase 1&2 dredged	Hydrodynamic with sediment transport	2021/01/01 – 2022/01/31	SHETRAN + Potsdam flows	21170 (58.0 t/d)	2023 survey dredged to Phases 1 & 2
7	2-year ARI flood hydrograph	Hydrodynamic with sediment transport	79 hours (3d 7h)	Flood hydrograph	1345 (409.6 t/d)	2023 survey dredged to Phases 1 & 2
8	10-year ARI flood hydrograph	Hydrodynamic with sediment transport	79 hours (3d 7h)	Flood hydrograph	26245 (7993.3t/d)	2023 survey dredged to Phases 1 & 2

Scenario 1: A calibration against the observed water level logger data at Otto du Plessis bridge was carried out for a 2-week spring tide period in 2014. The purpose of the calibration scenario was to ensure that the model is set up accurately to the observed conditions in the Milnerton Lagoon. This calibrated model could then be used to model a more recent time period using a more recent survey and a period when Potsdam effluent flow data was available.

Scenario 2: After calibration of the hydrodynamic model, a 2-week spring tide period was simulated to evaluate the hydrodynamic model, with a sediment transport module, against the observed inflow and outflow boundary conditions. The portion of the lagoon bed that was determined to have significant settled sludge-like sediments was made erodible to evaluate the stability of the sediment transport module for normal tidal cycles and outflows. This ensured that the following scenarios were able to accurately model the transport of the sludge material from the bottom of the lagoon and from the inflow boundary.

Scenarios 3, 4, 7 and 8: A frequent flood event and a larger flood were selected to simulate the effects of sediment transport for a flood event to determine the potential of flushing the accumulated organic sediments from the bed of the lagoon. Scenarios 3 and 4 focused on the current conditions to determine whether the sediments could be flushed during a flood. Scenarios 7 and 8 evaluated whether the incoming sludge and sediment would be sluiced through the dredged areas and/or where deposition would occur.

Scenarios 5 and 6: Phases 1 and 2 of the proposed dredging operations described in Figure 5-13 were simulated as these two phases will see the largest impact on the bathymetry of the lagoon. Phase 1 was simulated separately from 2 to determine the potential for sludge from the Phase 2 area to deposit within the dredged area of Phase 1. These two scenarios were also used to evaluate where future deposition of sludge and sediment would occur once dredging has been carried out.

The 8 scenarios described above were used to better understand the effects dredging would have on the lagoon in terms of the potential for future deposition, water levels and flow velocities. The models were also used to identify areas where future deposition may take place and where future maintenance dredging may be required. It is important to note that only the sludge portion of the bed was made erodible and not the sandy substrate below the sludge layer, as this is the target material for dredging to improve water quality in the lagoon. The implication of this method of simulating the sludge is that one cannot predict the changes at the mouth or the bed level change of the sandy sections within the lagoon. The simulations were set-up in this way to be able to specify the sludge layer properties and to accurately evaluate the effect of different flow conditions on the sludge layer only.

12.1 Hydrodynamic model setup and boundary conditions

To model the flow patterns and sedimentation of the estuary a 2DH hydrodynamic model was used. Mike 21C with a coupled sediment transport module by the DHI group, Denmark, was used to model the hydrodynamics and sediment transport in the Milnerton Lagoon. The bathymetric survey (Tritan Survey, 2023) was extended using the CoCT LiDAR survey of the surrounding areas to set up the domain of the numerical model. The hydrodynamic model was set up with a land-side boundary at the Otto du Plessis bridge and a sea-side boundary approximately 300 m into the ocean. The sea floor was approximated with a 1:80 (V:H) bed slope. Figure 12-1 shows the bathymetry as set up for the hydrodynamic modelling.

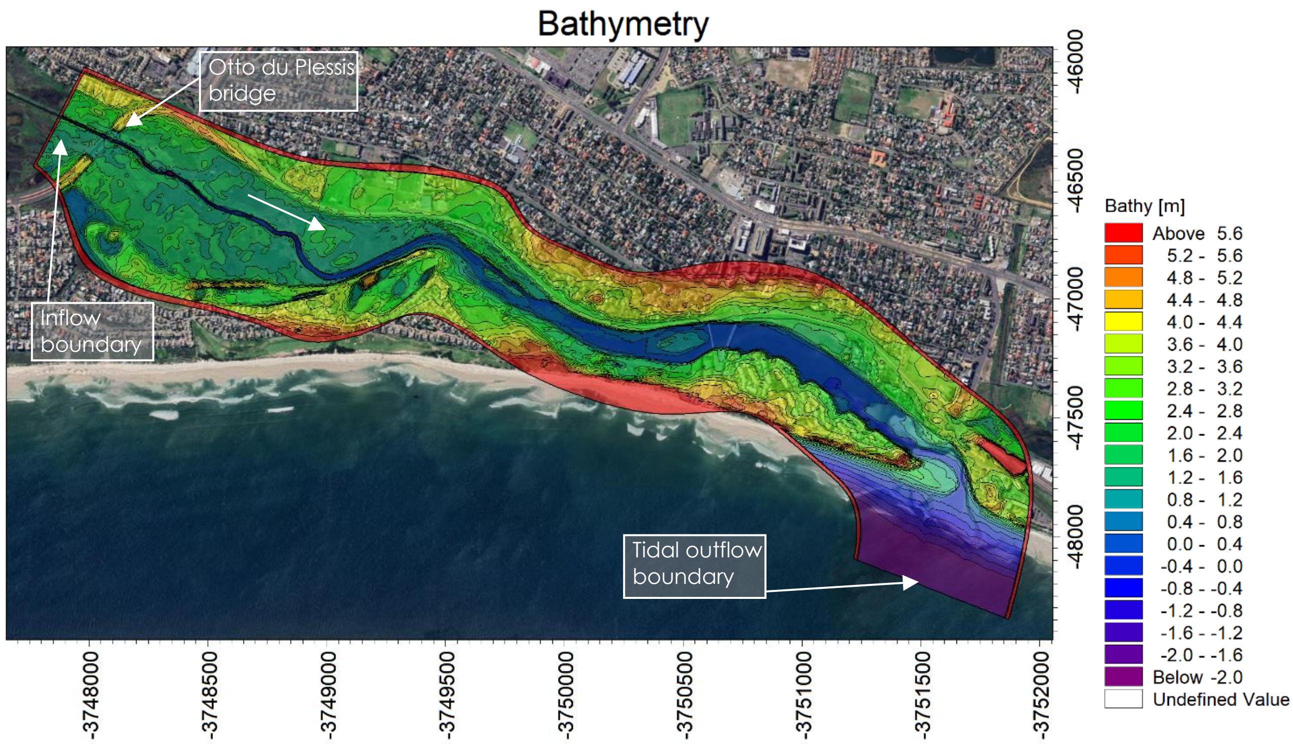


Figure 12-1. Bathymetry of Milnerton Lagoon as set-up for the hydrodynamic model

For the purposes of modelling the transport of the sludge material, only the surveyed sludge thicknesses were used to determine the erodible portion of the bed. Three samples of the sludge material were taken to determine the sediment grading of the sludge layer, and these samples were combined to determine the bed grading. Figure 5-11 shows the three samples and Table 12-2 shows the sediment properties as used in the numerical model. The sediment gradings of each sample are shown in **Annexure C**. For the sediment concentration at the inflow boundary at Otto du Plessis bridge, the hydrological model of the catchment (SHETRAN) generated daily flow record with sediment concentrations for three sediment fractions were combined with the weekly Total Suspended Sediment (TSS) concentrations from the Potsdam WWTW.

Table 12-2. Sediment properties as used in the numerical model

Fraction	Grain size (mm)	Cohesive/non-cohesive	SG	Bed fraction (%)	Inflow fraction (%)
1	0.01	Cohesive	2.63	15	97.3%
2	0.32	Non-cohesive	2.63	35	2.7 %
3	1.15	Non-cohesive	2.63	50	≈0 %

SHETRAN hydrological model

SHETRAN is a physically-based hydrological model (SHETRAN, 2013) that uses observed rainfall and evaporation data in combination with land-use, land cover, soil type and topography to determine a daily flow record with sediment loads from the catchment. Figure 12-2 shows the generated SHETRAN Diep River catchment with the river network developed for the 2022 assessment (Infinity Environmental, 2022). The Diep River daily flow record and sediment concentrations were obtained from the 2022 study.

Figure 12-3 shows the daily flow record for the modelling period of 2021/01/01 to 2022/02/01. The observed average Potsdam effluent discharge for this period was 0.318 m³/s, or 27.5 ML/day.

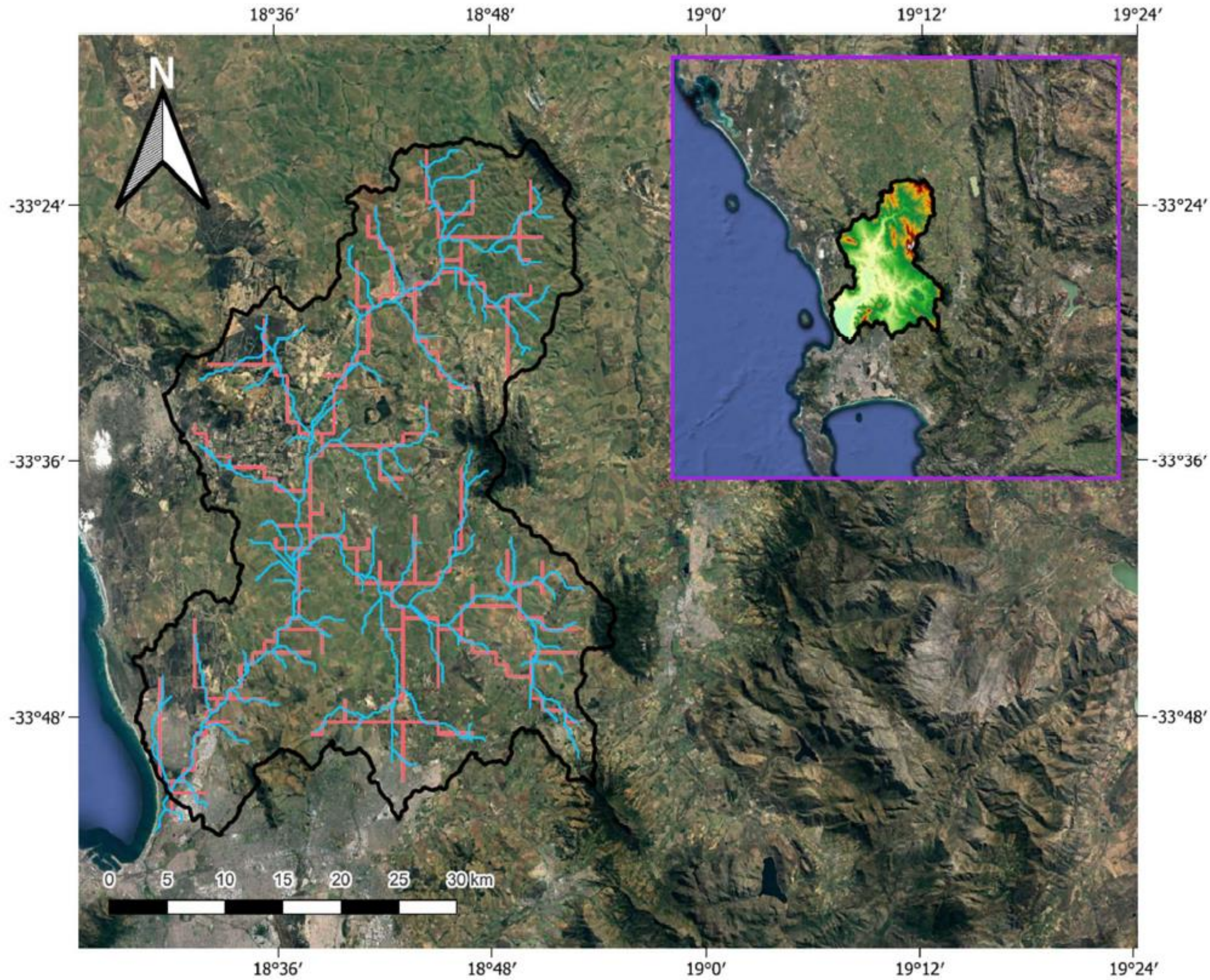


Figure 12-2. SHETRAN generated river network compared to the SRTM DEM of the Diep River catchment

The bathymetry shown in Figure 12-1 consists of a curvilinear nested grid with a grid spacing varying from 2.5 m (perpendicular to the flow direction) x 6 m (in the flow direction) near the main channel inflow boundary to 10 m (perpendicular to the flow direction) x 6 m (in the flow direction) on the flood plains on either side of the main channel and lagoon area. A nested grid model was used to accurately model the narrow main channel near the inflow boundary, while still allowing for a large area to be modelled, the modelling area of the setup was 5.1 km in length by 0.59 km in width. A bed roughness of Manning's n = 0.04 was used for the main channel and a Manning's n = 0.06 for the flood plains. All the simulated scenarios (except the longer term dredging scenarios 5 and 6) were simulated as fully-hydrodynamic since it was observed that the tidal cycles play a role in the morphology of Milnerton Lagoon.

Figure 12-4 shows the layer thickness as set up in the numerical model for the current scenario set-ups. The volume of sludge in the main body of the lagoon is 210 628 m^3, similar to the Tritan Survey volume, as set up in the model based on the survey data. The model area shown in Figure 12-1 was used to calculate the total rate of erosion and deposition for all scenarios modelled in this section. A cohesive sediment porosity of 0.5 and a porosity of 0.35 for the non-cohesive sediment were used to represent the sludge layer material. A bed shear stress for erosion of 1.0 N/m^2 was used for the cohesive fraction. In raw river water the critical condition for re-entrainment of these fine deposited cohesive sediments was found to be in the order of 0.2 Pa to 0.5 Pa and are the default values used in the Reference

Manual of the DHI Group Software (2013) for natural cohesive sediments and raw water (not sewage). ASCE (2007) carried out hydraulic tests on biofilm in sewers and recommended for PVC pipes coated with biofilm, the minimum shear stress to achieve self-cleansing is around 1.4 Pa. This value is considerably higher than that for natural sediments and for this study the critical bed shear stress of 1.0 Pa is considered more realistic to use for the deposited consolidated cohesive sediment considering field data of reservoir sediment management studies in South Africa with similar sediment characteristics (de Villiers and Basson, 2006).

The results for all hydrodynamic modelling scenarios are reproduced in **Annexure F**, while a summary of the main findings is presented below.

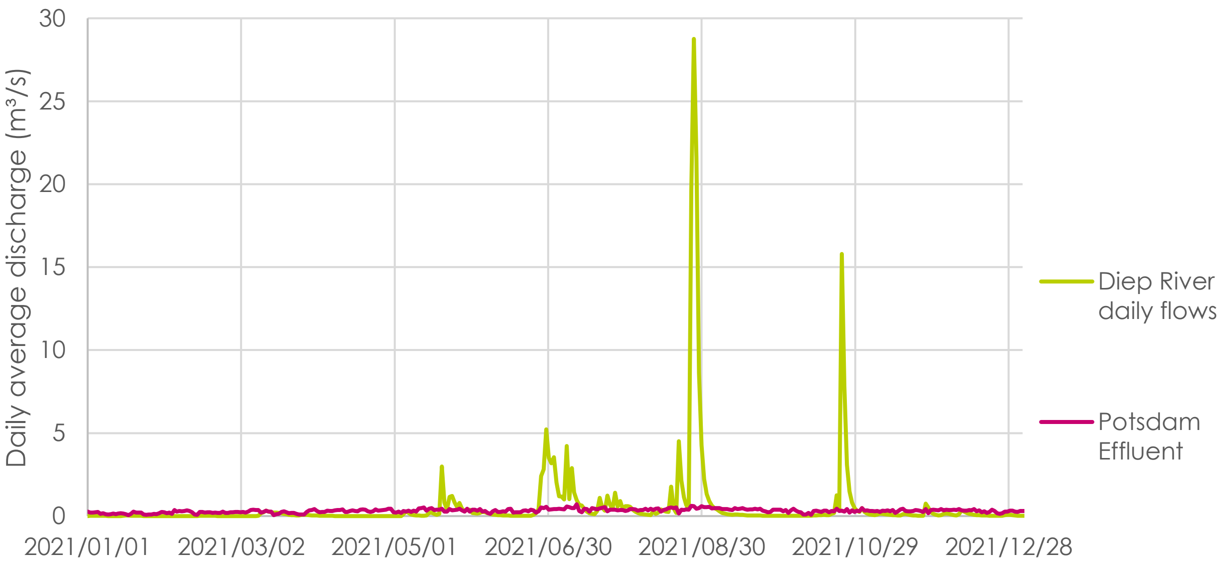


Figure 12-3. Observed daily Potsdam WWTW effluent discharge and simulated Diep River flows

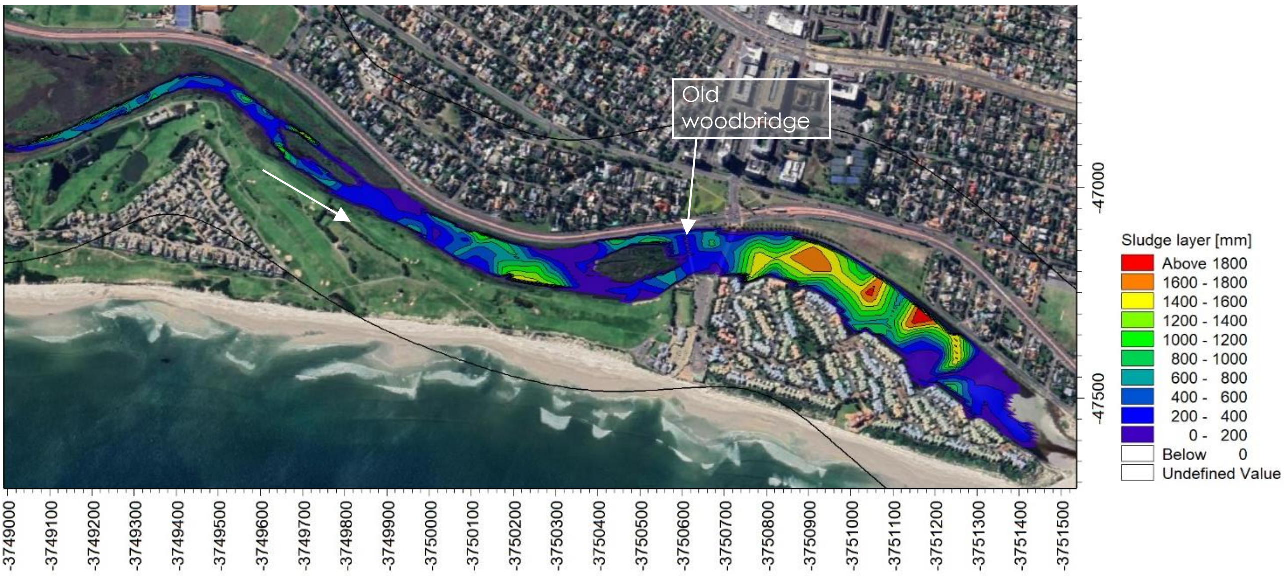


Figure 12-4. Layer thickness as set up in the numerical model as surveyed (Tritan, 2023)

12.2 Scenario 1: Calibration against water levels

The hydrodynamic model was calibrated against the Otto du Plessis bridge water level logger to ensure that the numerical model is set up accurately. The selected calibration period was a 2-week period from 2014/03/12 to 2014/03/26 where water level logger data was available. There were a few factors to consider when comparing the water level logger data to the simulated data:

- The bathymetry was surveyed at the end of 2022 and there may be notable differences between the 2014 and 2022 bathymetry which could alter the simulated results. In this case the mouth was opened to match 2014 aerial photography, shown in Figure 12-1, and the minimum bed level was reduced to 0 masl in the main channel to allow the water to flow out of the lagoon during low tide.
- The datum level of the observed water levels is unknown and therefore only the shape and lag time of the water level logger data could be used as part of the calibration.
- The generated river flows are not gauged flows and there were no effluent data available for the Potsdam WWTW at the time of the simulation. Figure 12-6 shows the Potsdam flows and generated Diep River flows used for the inflow boundary.

Figure 12-5. 2014 mouth conditions (left) compared to the 2022 mouth conditions (right)

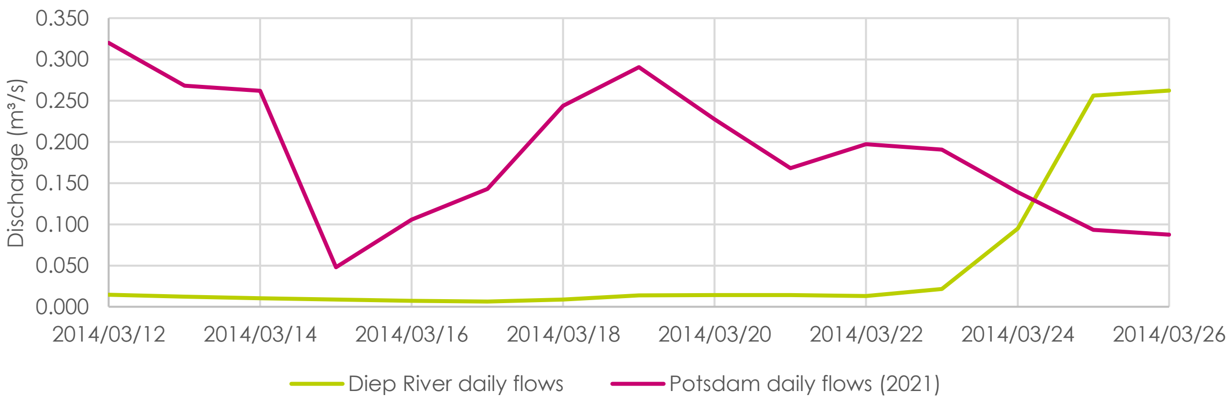


Figure 12-6. Diep River and Potsdam daily flows for the 2014 calibration period

From the simulated water levels, it was found that the hydrodynamic model could simulate the tidal peaks at Otto du Plessis bridge relatively accurately, considering the unknown datum level of the gauge. The tidal influence on Milnerton lagoon can be seen in Figure 12-6, considering that the Otto du Plessis bridge logger is located at the land-side boundary of the numerical model. The difference in lag times between the observed and simulated logger levels could be attributed to the accuracy of the inflow boundary data and the higher bathymetry causing a reduced rate of exchange between the lagoon and sea water during tidal cycles. The inflow boundary discharge also plays a role in the water level achieved at the location of the water level logger, and the SHETRAN generated catchment runoff is daily-averaged flows which could impact the simulated water levels at any given time.

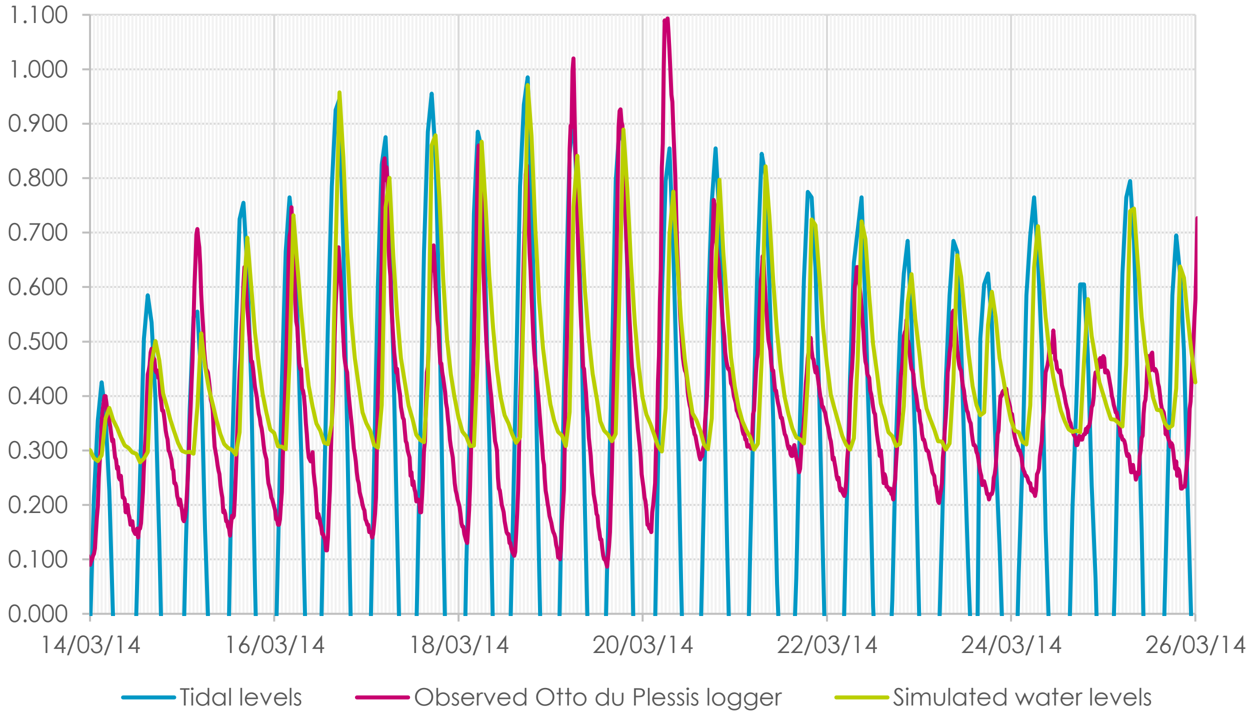


Figure 12-7. Comparison between the simulated and observed water levels at the Otto du Plessis bridge

12.3 Scenario 2: Two-week spring tide

The two-week spring tide period was simulated between 2022/03/27 and 2022/04/10 in order to determine whether an equilibrium between the current sludge layer and the outflow has been reached. Figure 12-8 shows the inflow sediment concentration and discharge at the inflow boundary of the model. Figure 12-9 shows the resulting bed level change after the two-week simulated period. Whilst there were some scour and deposition areas upstream (it is normal for adjacent cells to show very similar rates of erosion and deposition, especially near the water's edge as the bathymetry used in the model finds an equilibrium), the most notable section of sludge movement was near the mouth where the effect of the tidal cycles were most notable. The effect of the tidal cycles is not seen deeper into the estuary where the flow velocities are very low, as can be seen from Figure 12-10 which shows the maximum flow velocities for the simulation period. The two-week spring tide scenario was simulated using the current bathymetry and shows that **a spring tide cycle cannot disturb the sludge layer within the lagoon**.

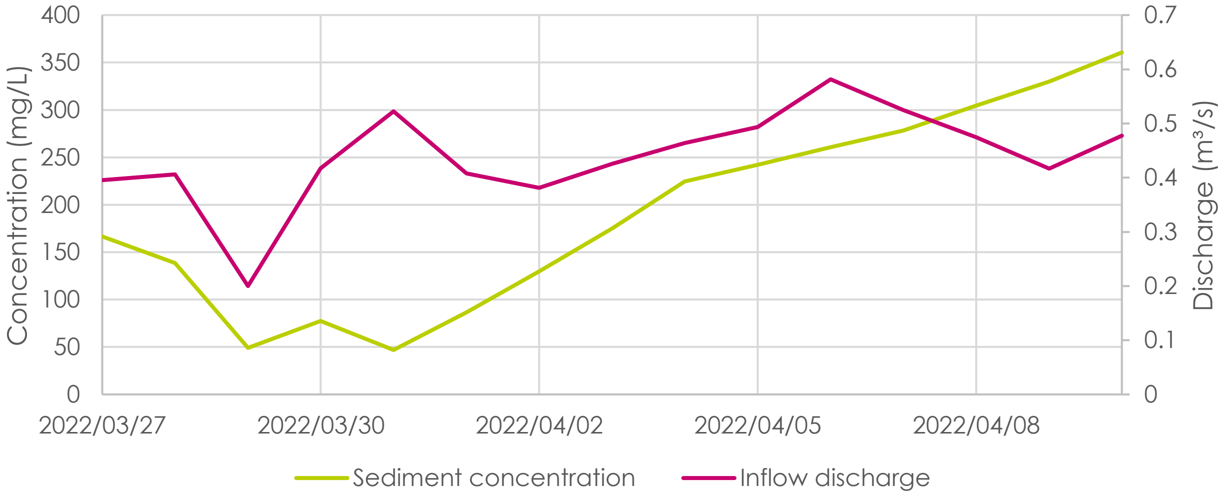


Figure 12-8. Sediment inflow concentration and inflow discharge for the two-week spring tide period

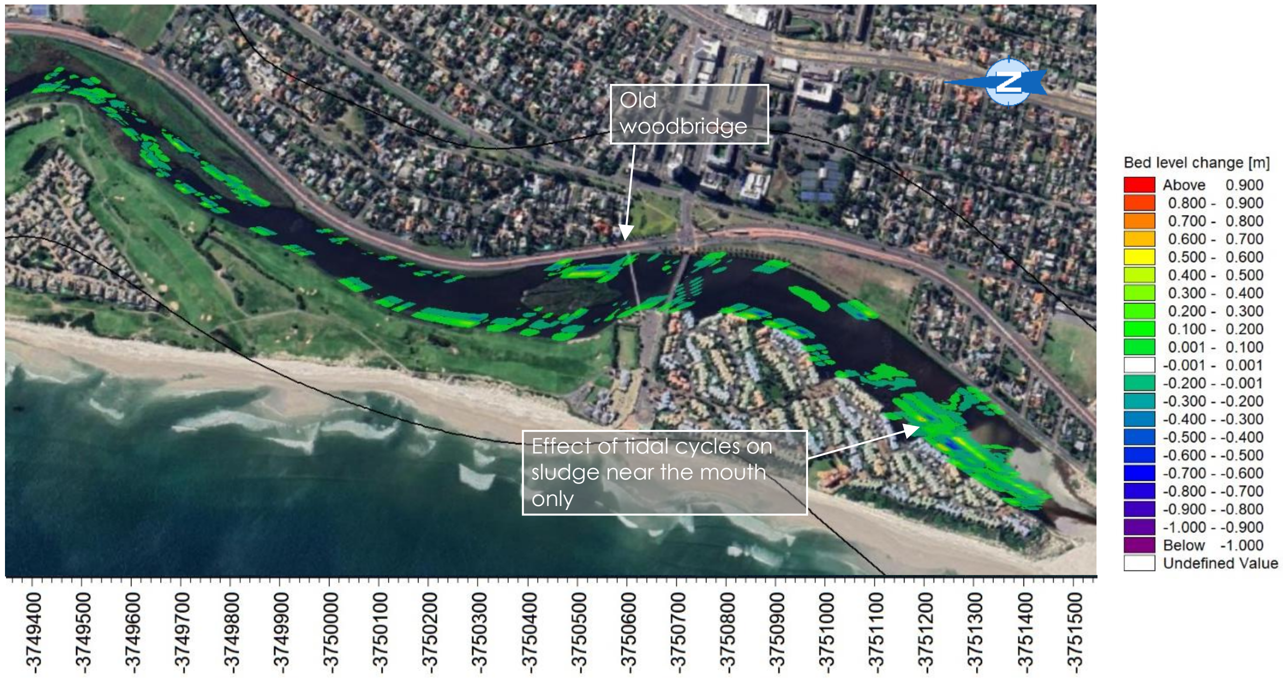


Figure 12-9. Simulated bed level change after a two-week spring tide period

12.4 Scenario 3: 2-year flood hydrograph for the current scenario

The peak routed discharge for the Q2-year flood is 72.6 m³/s. The routed flood hydrograph and inflow sediment concentrations are shown in Figure 12-11. The Q2-year flood is considered to be a more frequent flood event. Shown in Figure 12-12, the simulated bed level change after the flood event shows very little erosion or deposition in the main body of the lagoon. The main areas of scour are in narrower areas where the flow velocity can be higher. Deposition zones are identified near obstacles in the main channel and downstream of narrow areas where the flow velocity is reduced. This pattern of erosion and deposition is mainly shown near the upstream island in the main channel. Near the mouth where the velocity is again higher more erosion can be observed. **A Q2-year flood is not capable of flushing the poor-quality sludge from the main body of the lagoon**. At the peak of the Q2 year flood the deposited volume of sediment in the main body of the lagoon was 109 m³ and at the end of the flood event the deposited sediment was 204 m³.

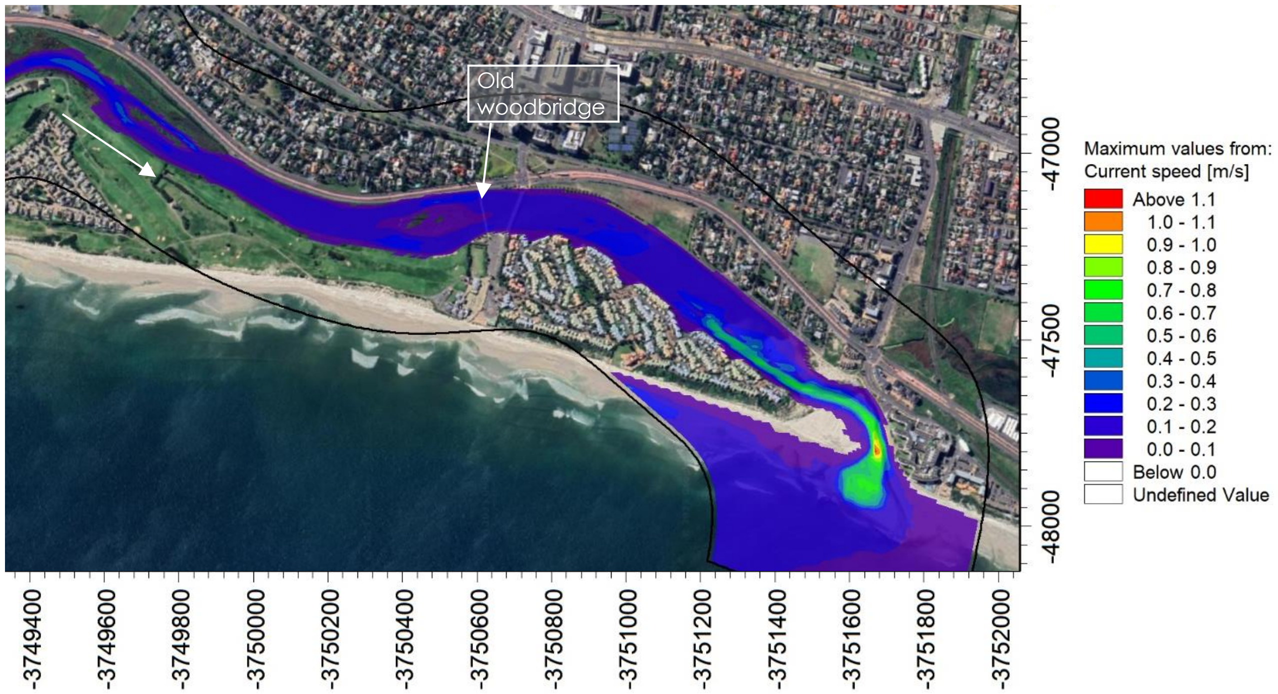


Figure 12-10. Simulated maximum velocities for the two-week spring tide period

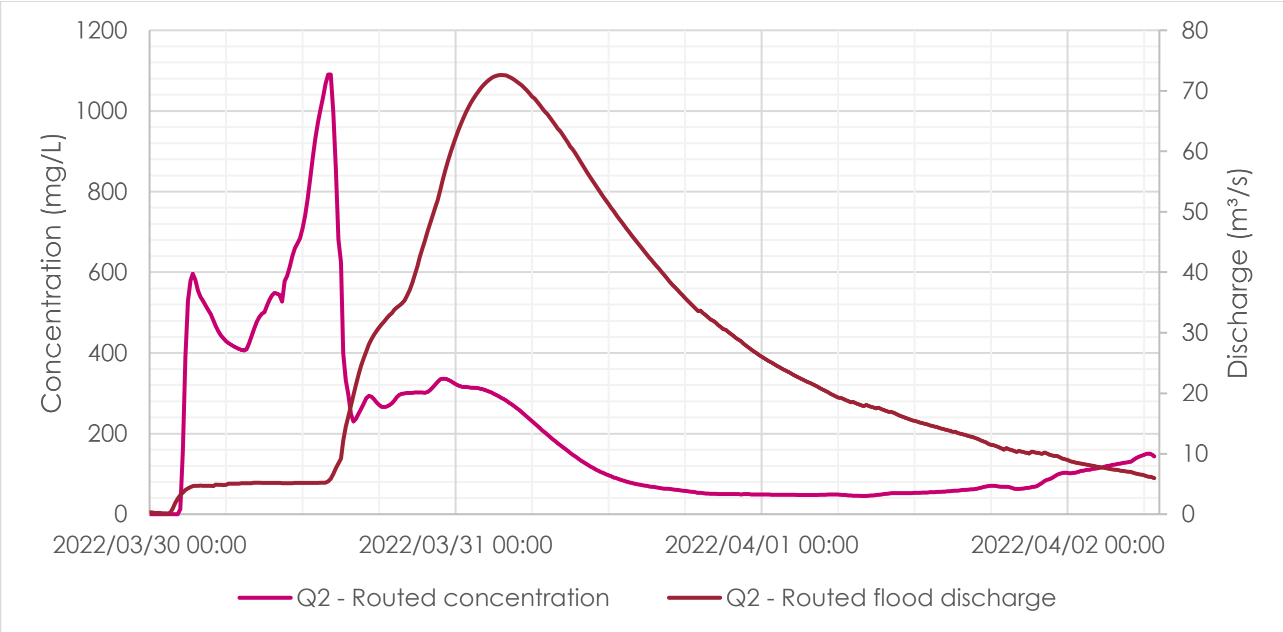


Figure 12-11. Flood hydrograph and sediment concentrations routed through Rietvlei

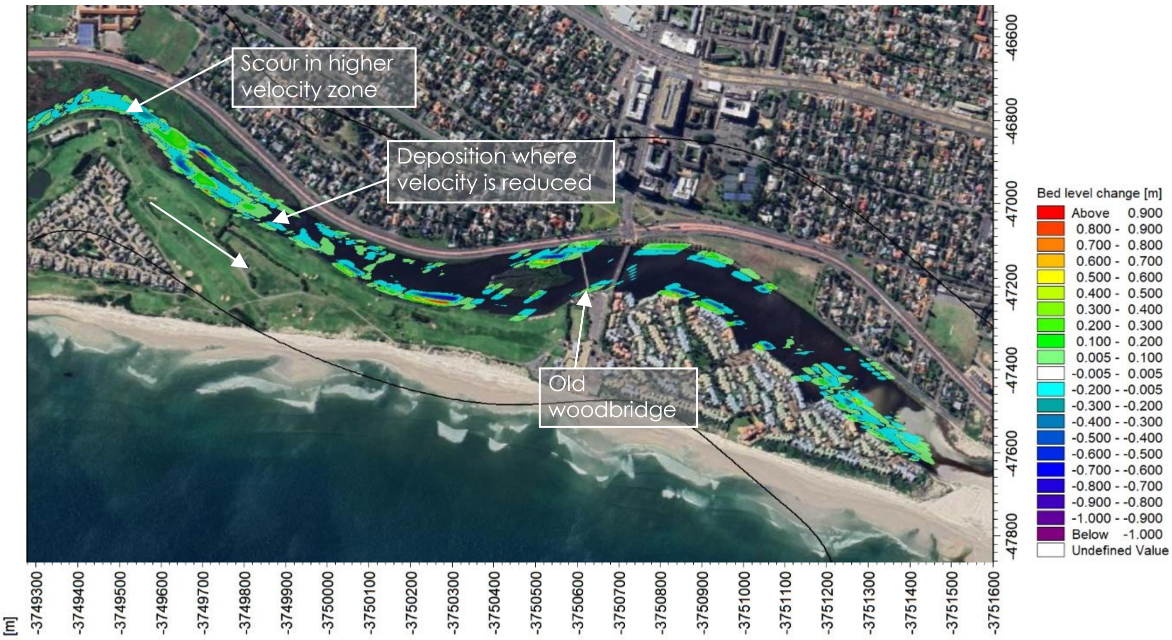


Figure 12-12. Simulated bed level change for the Q2-year flood hydrograph (72.6 m³/s)

Figure 12-13 shows the simulated maximum flow velocities during the Q2 year flood. The areas of erosion in Figure 12-12 correspond with the areas of flow velocities above 0.6 m/s. The velocities in the lagoon are influenced by the narrow outflow channel since only the sludge material was simulated as erodible, but this effect would only be localised since the lagoon area is very flat and the tidal levels influence the water levels in the lagoon. A limitation of the numerical model is that the grading of the bed has to be uniform throughout the model, and a distinction between sand and sludge could therefore not be made. The simulations focused on the fine sediment/sludge in the lagoon, which was set up a with cohesive particle size of 10 microns. The sandy bed of the lagoon was made non-erodible. The floods were also routed through Rietvlei before being simulated in the lagoon area. Therefore the limited deposition only indicates that the sludge portion of the bed material would not deposit within the current lagoon.

Figure 12-13. Q2-year flood maximum flow velocities

12.5 Scenario 4: 10-year flood hydrograph for the current scenario

The routed flood peak for the Q10-year flood is 358 m^3/s. The hydrograph for the routed flood is shown in Figure 12-14, along with the sediment concentrations at the inflow boundary. The bed level change for the lagoon shown in Figure 12-15 shows areas of erosion and deposition. The net bed level change, for the lagoon area as identified in Figure 12-4, was simulated as -251 m^3 at the peak of the flood and -563 m^3 at the end of the flood event, meaning that there was a net erosion of sediment in the lagoon area. Deeper scour was simulated around the outer edges of the bends. The deposition depths upstream of the old Wood bridge were typically in the order of 0.1m to 0.2 m and downstream of the bridge the deposition was generally found to be less than 0.1m depth.

Figure 12-16 shows that the flow velocities in the main channel remained low, between 0.5 m/s and 0.75 m/s and that the velocities only increased near the mouth to above 3 m/s.

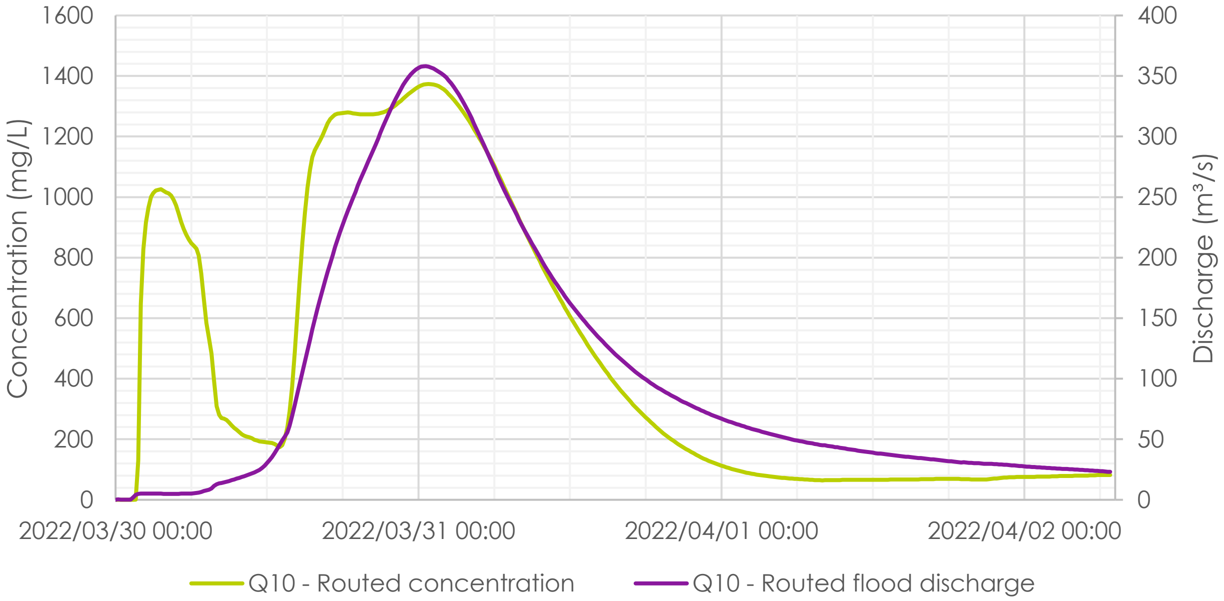


Figure 12-14. Routed Q10 year flood and inflow sediment concentrations

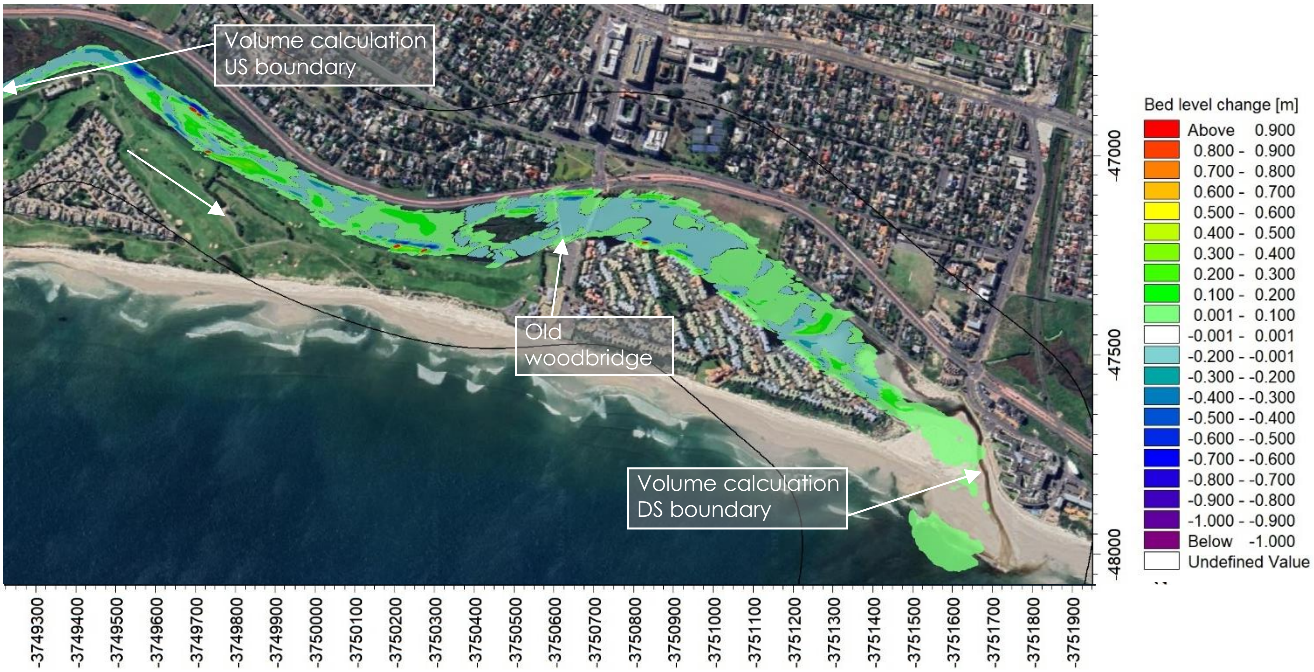


Figure 12-15. Simulated bed level change after the Q10-year flood hydrograph (399 m³/s)

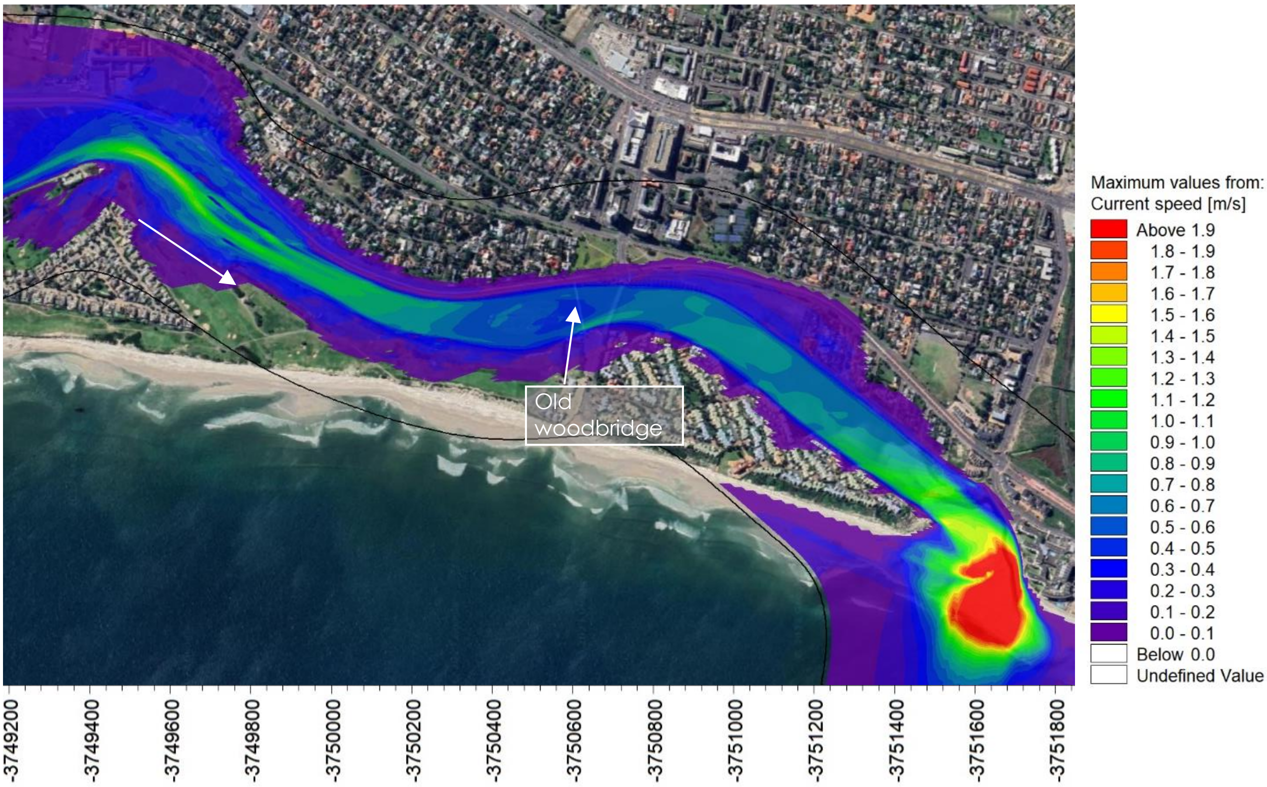


Figure 12-16. Simulated maximum flow velocities for the Q10-year flood (399 m³/s)

12.6 Scenario 5: 1-year simulation of Phase 1 dredging

A one-year period from January 2021 to December 2021 was simulated for the long-term dredging scenarios. During this period the Potsdam TSS-data was available and incorporated into the cohesive fraction of the inflow concentration. Phase 1 dredging as described in Section 5.3 was simulated with the remaining sludge layer upstream remaining erodible. Figure 12-17 shows the bed level change after the one year simulation of river flows, Potsdam effluent discharges and tidal cycles. The dredged portion has affected the upstream un-dredged section by causing the area upstream of the Old Wood bridge to erode downward to form a new equilibrium by depositing the sludge into the upstream area of the dredged bed. For the Phase 1 dredging it would mean that maintenance dredging would be required between the Old Wood bridge and the Loxton Road bridge, and the maintenance dredging would be required on a periodic basis until a new equilibrium in the upper section has been found, or until all the fine material has been transported downstream. The simulated deposition volume and mean sediment depth after one year within the Phase 1 dredged area are 9 227 m^3 and 67 mm respectively. It is therefore concluded that **only dredging Phase 1 (the lagoon downstream of the Woodbridge) would not result in a favourable outcome as the sediment from the upper reaches would settle out in this area over time**. Sedimentation is expected to fill the phase 1 dredging area to current levels within approximately 10 years following the dredging, if the levels of inflowing sediment from the WWTW remain the same as at present.

Figure 12-17. Simulated bed level change for the Phase 1 dredging scenario after 1 year

12.7 Scenario 6: 1-year simulation of Phase 1 and 2 dredging

Phase 1 and 2 dredging (as described in Section 5.3) were simulated with no upstream moveable bed area, meaning all the deposition that occurred was due to the inflow concentrations from the Potsdam effluent and the Diep River. The resulting deposition after a year, as shown in Figure 12-18, mainly took place within the area adjacent to the golf course upstream of the Woodbridge, and to a lesser extent in the area towards the mouth. The simulated deposition volume and mean sediment depth after one year within the Phase 1 dredged area are 7 263 m^3 and 17 mm respectively. For scenario 6 most of the sediment deposition occurs upstream of the Woodbridge and therefore the average deposition depth over the total dredged zone after one year is small. It is important to note that sediment deposition from the ocean into the lagoon through tidal cycles and wave action was not modelled and the resulting deposition downstream could increase due to marine sediment penetrating the estuary through the mouth. The focus is however on the accumulation of the sludge-like sediments contributing to poor water quality and odour. Sedimentation is expected to fill the phase 1 and phase 2 dredging areas to current levels within approximately 20 years following the dredging, if the levels of inflowing sediment from the WWTW remain the same as at present.

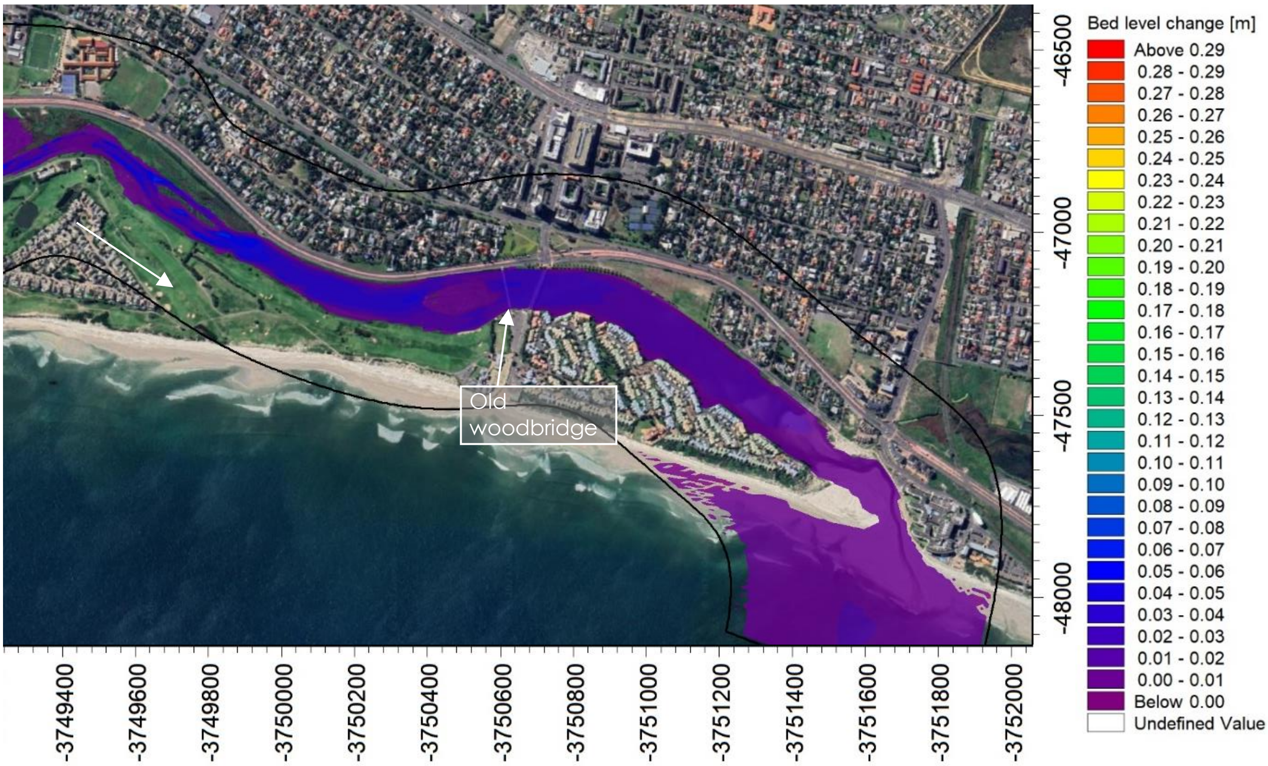


Figure 12-18. Long term results of the phase 1 and 2 dredging scenarios

12.8 Scenario 7: 2-year flood with the dredged lagoon

A two-year flood showed little to no deposition in the main channel of the lagoon, but a limited amount of deposition was observed near the island upstream of the Woodbridge as shown in Figure 12-19. At the peak of the flood the simulated deposition was 151 m³ (undredged scenario 109 m³) and at the end of the flood event the deposition was 275 m³ (undredged scenario 204 m³). Figure 12-20 shows the maximum simulated flow velocities for the Q2 year flood.

Figure 12-19. Bed level change after the Q2-flood for the Phases 1 and 2 dredged bathymetry

Figure 12-20. Simulated maximum flow velocities for the Q2 year flood event for the dredged scenario

12.9 Scenario 8: 10-year flood with the Phases 1 and 2 dredged lagoon

Similarly to the Q2-year flood, the Q10-year flood did not show any deposition in the main body of the lagoon, apart from around the island upstream of the Old Wood bridge. Figure 12-21 shows the simulated bed level change after the routed Q10-year flood. There was however some deposition on the flood plains, especially in the upper reaches, and some deposition in the pier areas on Woodbridge Island. At the peak of the flood, the deposition in the lagoon area was 421 m³ and at the end of the flood event the deposition was 748 m³ (current scenario had erosion of 251 m³ at the peak of the flood and even more erosion of 563 m³ at the end of the flood). This rate of deposition was substantially more than the rate of erosion simulated for the Q10 current scenario. This difference in erosion patterns is attributed to the dredged lagoon with slower flow velocities and the current scenario having a movable bed made up of the sludge layer that could be eroded, compared to the dredged scenarios having only an inflow concentration to supply the lagoon with sludge. The simulated maximum flow velocities for the Q10 flood are shown in Figure 12-22.

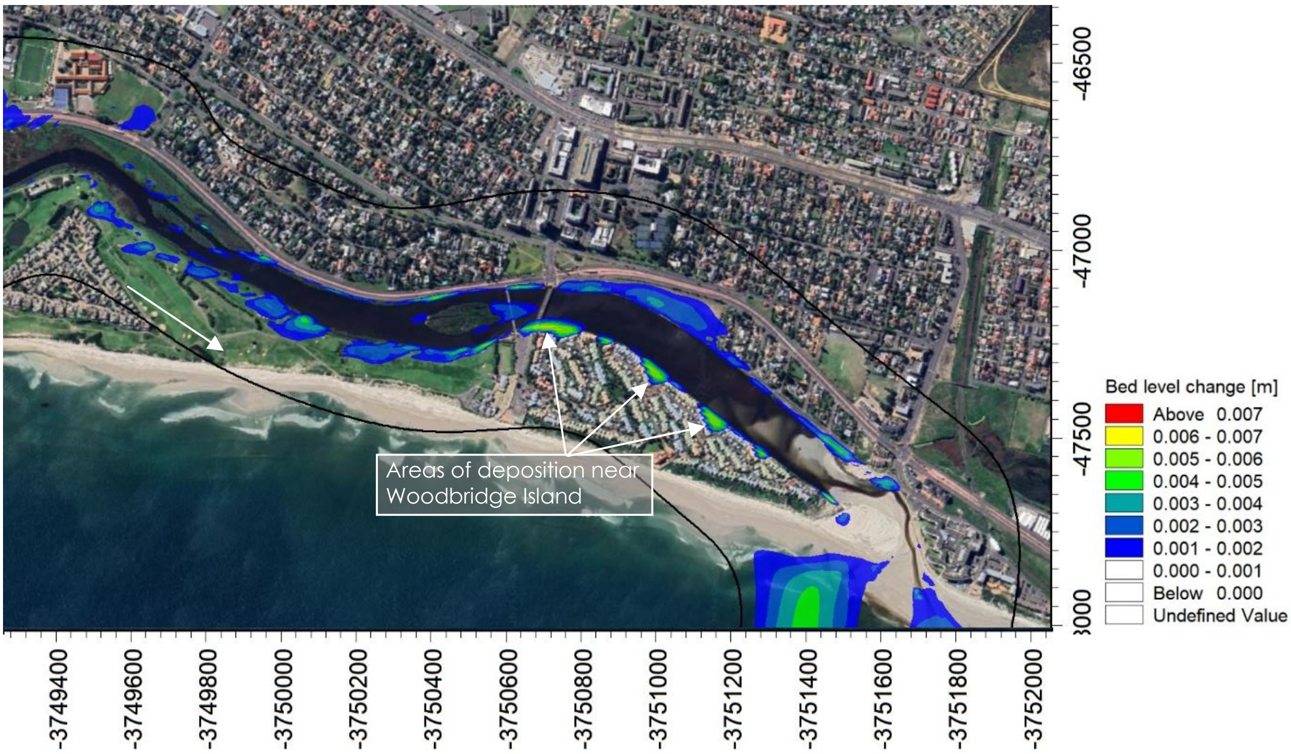


Figure 12-21. Bed level change after the Q10-flood for the Phases 1 and 2 dredged bathymetry

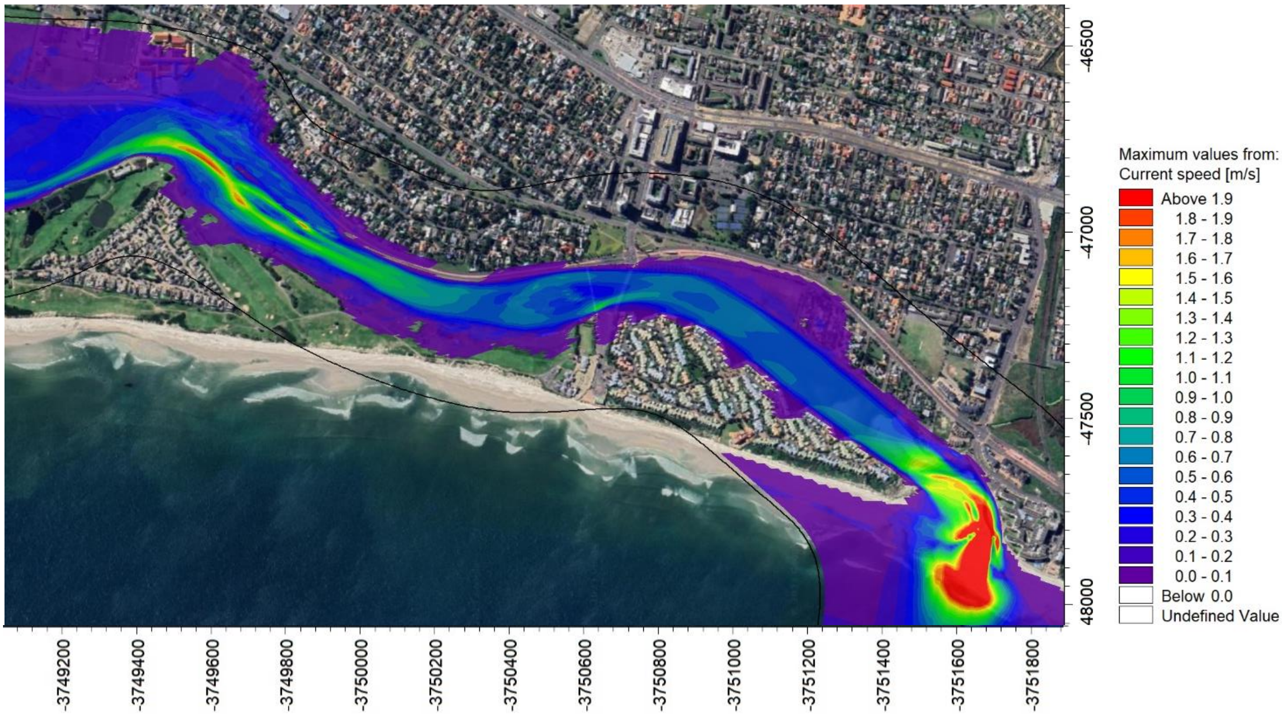


Figure 12-22. Simulated maximum flow velocities for the Q10 year flood event for the dredged scenario

12.10Summary of the hydrodynamic modelling of the dredging options

The main findings for the hydrodynamic modelling of the sludge layer in the lagoon and the phase 1 and 2 dredging options of the Milnerton Lagoon are presented below:

- Frequent flood events < Q2 flood are not capable of flushing the sludge from the lagoon due to the low flow velocities simulated in the wider areas of the lagoon. The narrow cross-section of the main outflow channel reduces the flow velocities in the lagoon preventing re-entrainment and the relatively low flow velocities promote deposition.
- The Q10-year flood was able to scour some of the sludge from the main body of the lagoon, with a total volume of 563 m³ of sludge scoured for the current scenario. However, compared to the volume of sludge in the lagoon, this is not an effective means of flushing sludge from the lagoon, considering that a Q10 year flood is a larger, less frequent, flood event.
- Dredging could be carried out in phases, if absolutely necessary for budgetary reasons, but the consequence of a phased approach means that some form of maintenance dredging would be required to prevent the upstream sludge from spreading into the dredged areas. It would be much more effective to dredge the system as a whole.
- The simulated deposition volume and mean sediment depth after one year within the Phase 1 dredged area are 7 263 m³ and 17 mm respectively, with most of the sediment depositing upstream of the old Wood Bridge.
- If inflows of organic sediment remain constant, the dredged Phase 1 area will fill to current levels within 10 years after dredging, while the dredged Phase 1 and Phase 2 areas would fill to current levels within 20 years after dredging, should both be dredged.
- The results from the small flood scenarios and long term scenarios indicate that it is not flood events that cause the sediment/sludge deposition in the lagoon, but rather the daily flows; it is therefore important to ensure that the daily flows are of good quality to prolong the effect of dredging the lagoon.

13 WATER QUALITY MODEL-BASED EVALUATION OF THE REMEDIATION OPTIONS

The purpose of this study was to understand the water quality dynamics of the system and to model management scenarios aimed at improving the water quality of the estuary. The methodology involved firstly the conception of a water quality model that reproduced the current water quality scenario in the estuary, thereby establishing a baseline to which all further scenarios could be compared. The water quality model provides insights into the water temperatures, dissolved oxygen concentrations, salinity, nutrient and algal dynamics in the water column. The approach adopted was to model the current estuary and then subject it to proposed management scenarios and compare the resultant water quality. The scenarios modelled include the following:

- Operating Potsdam WWTW at its licensed effluent discharge quality and current quantity
- Diverting the Potsdam WWTW discharge to the marine environment
- Operating Potsdam WWTW at 100Ml/day with (a) current quality and (b) licensed effluent quality
- The addition of seawater at various locations and rates.
- Dredging the estuary to remove the contaminated sediments.

13.1 The HEC-RAS model

The Hydrologic Engineers Centre River Analysis System (HEC-RAS) water modelling suite was used as the system is a very shallow and narrow river, and a 1-dimensional model is sufficient. HEC-RAS which was first released in 1995 is software that performs unsteady flow river hydraulic calculations, sediment transport, water temperature analysis and generalised water quality modelling (nutrient fate and transport). The water quality component can perform detailed temperature analyses and transport of algae, dissolved oxygen, carbonaceous biological oxygen demand, dissolved ortho-phosphate, dissolved organic phosphorus, dissolved ammonium nitrate, dissolved nitrite nitrogen, dissolved nitrate nitrogen, dissolved organic nitrogen as well as user defined conservative constituents and non-conservative constituents. The HEC-RAS model is not restricted, and individuals may use the program without any charges. The HEC-RAS model is a deterministic model which calculates events exactly, without the involvement of randomness.

The surveyed geometric data (Tritan, 2023) was used to generate cross sections as bathymetry for the software with the SHETRAN river flows, Potsdam WWTW discharge daily volume, sampled water quality and tidal elevations used as boundary conditions, whilst incorporating the meteorological data to calculate surface water elevations, temperatures and other water quality constituents within the system.

13.2 Model cross sections

An example of the generated cross section and estuarine longitudinal profile is shown in Figure 13-1 as used by HEC-RAS with the mouth at station 0 on the LHS.

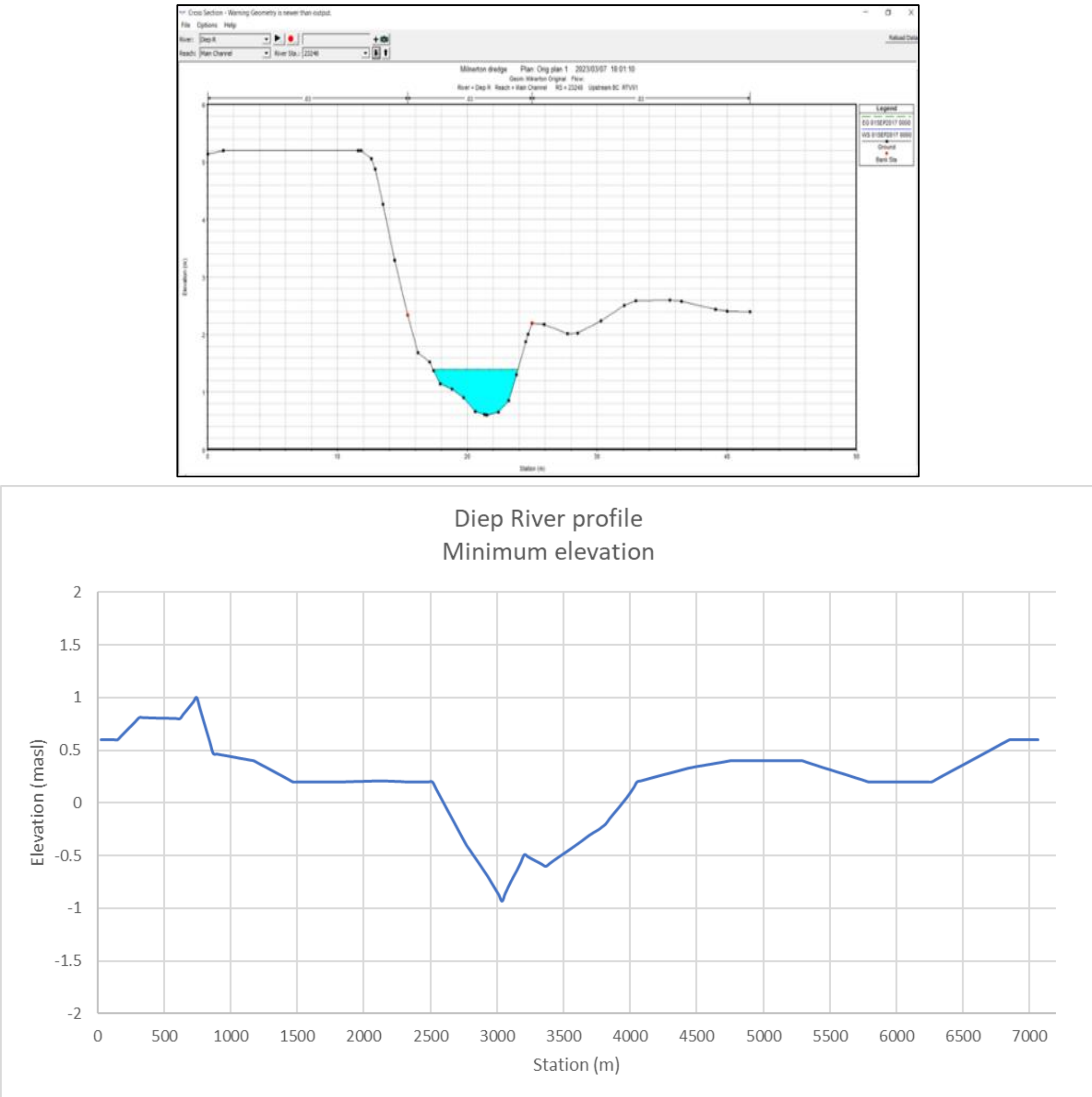


Figure 13-1. A longitudinal plot, x-y-z perspective and river profile. The blue line is the bottom elevation of the cross section and 0 is the mouth

Figure 13-2 shows a 3-dimensional HEC-RAS visualisation of the main river channel and terrain as well as model segmentation.

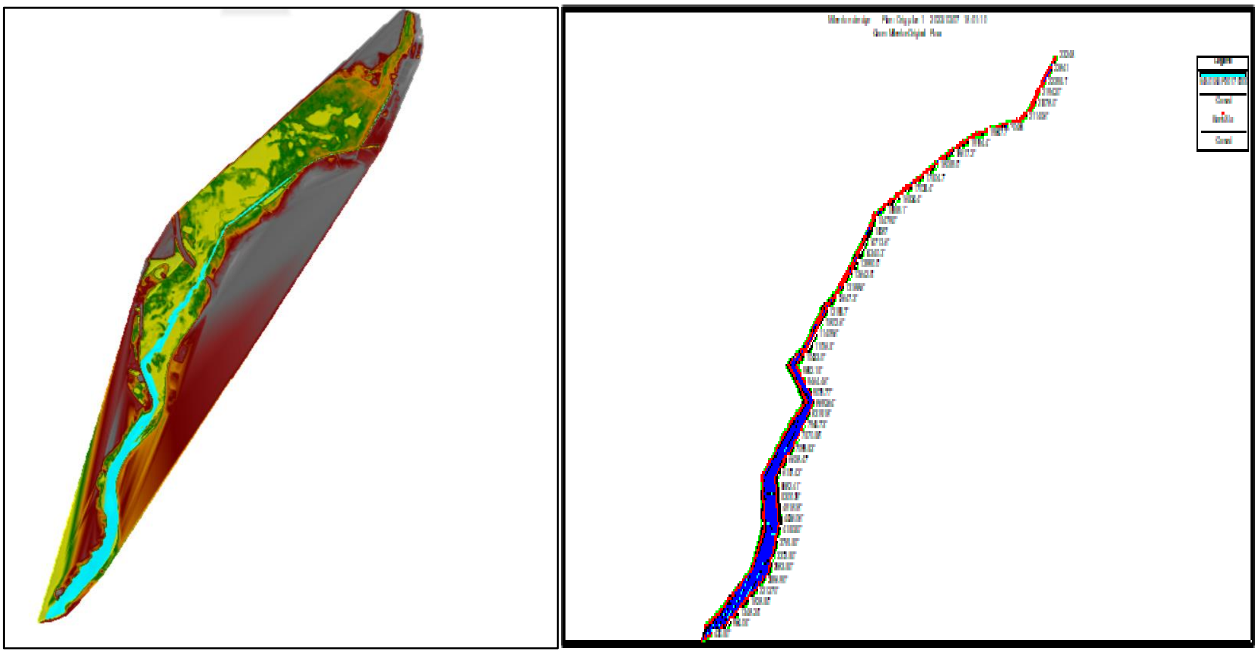


Figure 13-2. The x-y-z- perspective and terrain plot of the estuary

From Figure 13-1 it is apparent that the system has hardly any slope and has a higher elevation (suggesting deposition) at the mouth of the estuary. For modelling purposes the entire estuary was modelled but for the purposes of this report only cross sections that correspond to sampling positions (RTV11 etc. as indicated in Table 13-1) were expounded upon; these are shown in the following table and along with their corresponding HEC-RAS cross sections.

Table 13-1. The sampling sites in the system

Sampling site	HEC-RAS cross section	Comment
RTV01	23310	Diep River head sampling point
RTV11	21159.7	In the bypass canal
RTV05	14067	Otto Du Plessis bridge
RTV18	8294.06	Opposite Broad Road
RTV09	4313	Woodbridge
RTV10	989.89	Mouth area

13.3 Model inputs and assumptions

After the geometric data was incorporated, the specific boundary condition data used to run the model is shown in Table 13-2.

Table 13-2. The required modelling boundary conditions

Variable	Source	Specific component	Requirement
Diep River flows	SHETRAN modelling	Volumetric flows (m^3/s)	Boundary condition
River water quality	City sampling data	Constituent (mg/l)	Boundary condition
Potsdam WWTW discharge	Potsdam WWTW data	Volumetric flows (m^3/s)	Boundary condition
Potsdam WWTW quality	Potsdam WWTW data	Constituent (mg/l)	Boundary condition
Tidal elevations	Recorded data SA Hydrographic office	Elevations (masl)	Boundary condition
Water elevations	City recorded logger	Elevations (masl)	Calibration and validation

Variable	Source	Specific component	Requirement
In river water quality	City sampling data	Constituent (mg/l)	Calibration and validation
Meteorological data	ECMWF interim data from www.ecmwf.int	Air temperature, air pressure, windspeed and direction, solar radiation, cloud cover	Conservation of energy and heat balance

No groundwater interactions were modelled as there is no data for flows or water quality or for any diffuse sources or stormwater. Figure 13-3 shows the time period that could be modelled based on data overlap. The figure shows that any period between 1 July 2017 and 31 August 2019 may be modelled as this is the only period that has an overlap with the required meteorological data set.

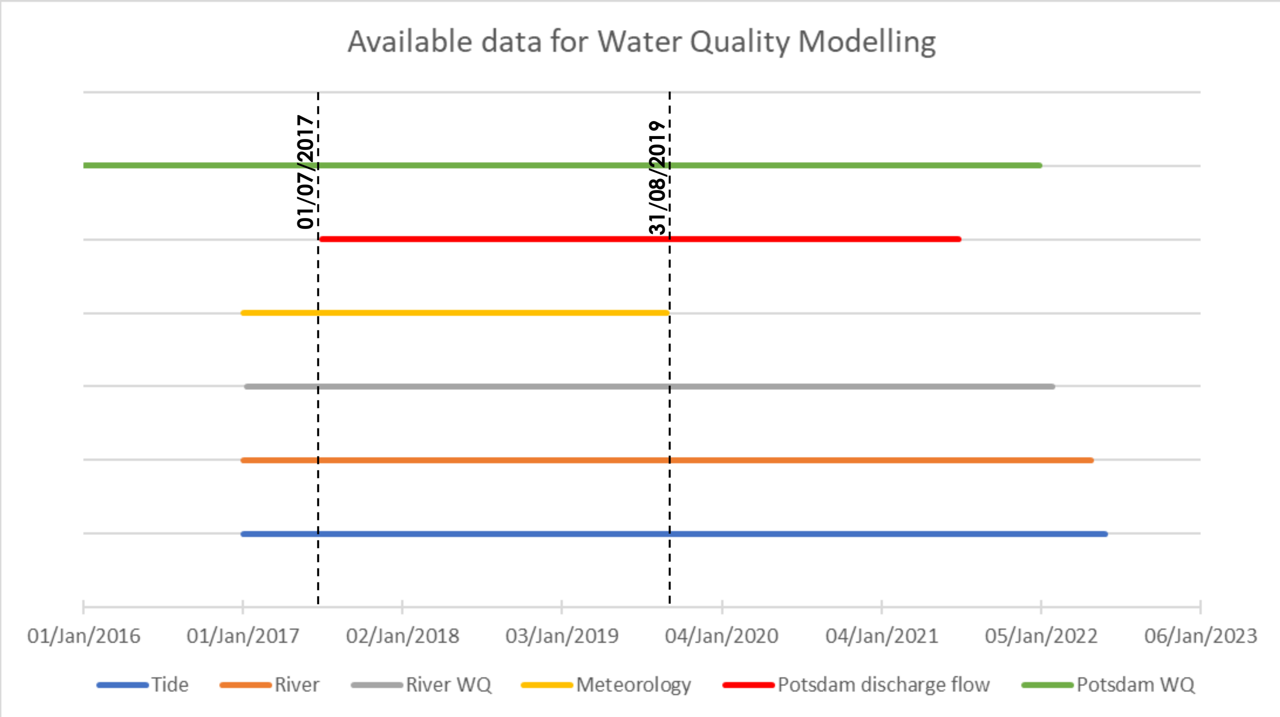


Figure 13-3. The data period available for the water quality modelling

Analyses of the City-supplied sampled water quality data for this period found that chlorophyll-a data was only sampled onwards from February 2020. This would affect predicted algal concentrations and any variance shown between the various scenarios from the baseline would be considered relative changes. All calibration and validation of the model were thus performed on the available data until an acceptable best fit solution with sampled data was obtained. Only after this process was the model deemed calibrated and validated and used for the proposed scenarios.

The various modelling scenarios that were compared to the baseline scenario and used to evaluate the final water quality dynamics included:

- Operating Potsdam WWTW at its licensed effluent discharge quality (i.e. no specific remediation interventions but a significant improvement in WWTW effluent quality)
- Evaluating Potsdam WWTW operating at 100Ml and licensed discharge quality, **or** discharging effluent at its current quality (i.e. the baseline scenario for 2025 onward but without improvement in effluent quality)
- Dredging the estuary to remove the contaminated sediments (i.e. **Option 1**)
- Aerating the estuary artificially (i.e. **Option 2**)

» The addition of seawater at various locations and rates (i.e. **Option 3**)
» Diverting the Potsdam WWTW discharge to the sea via an outfall (i.e. **Option 4**)

Options 5, 6 and 7 are not capable of inclusion in the model without more information than currently available on their design parameters.

The time period modelled for the baseline scenario was the hydraulic year 2017/2018 i.e. 1 October 2017 to 1 October 2018.

The assumptions made for this modelling exercise were:

» That the ocean salinity is constant for the simulation period and its temperature is the historical average monthly temperatures.
» No transport of sediment is modelled and thus the bathymetry is static for the simulations.

HEC-RAS does not resolve lateral or profile variations of any of the constituents in the waterbody. The system is well mixed by the incoming tides and river flows. It is a narrow river with shallow depth, and a 1-dimensional model with the available data will adequately perform long term water quality simulations. The model outputs water quality for the entire estuary but only the City's regular sampling locations (Figure 2-9) were chosen for comparison.

The constituents modelled for water quality were:

» Water temperature
» Dissolved oxygen (insufficient data exists to model chemical and biological oxygen demand)
» Algal concentrations and nutrients (note that not all nutrients for algal growth are routinely sampled)
» Salinity

The following figure shows the hydrodynamic boundary conditions for the modelling period.

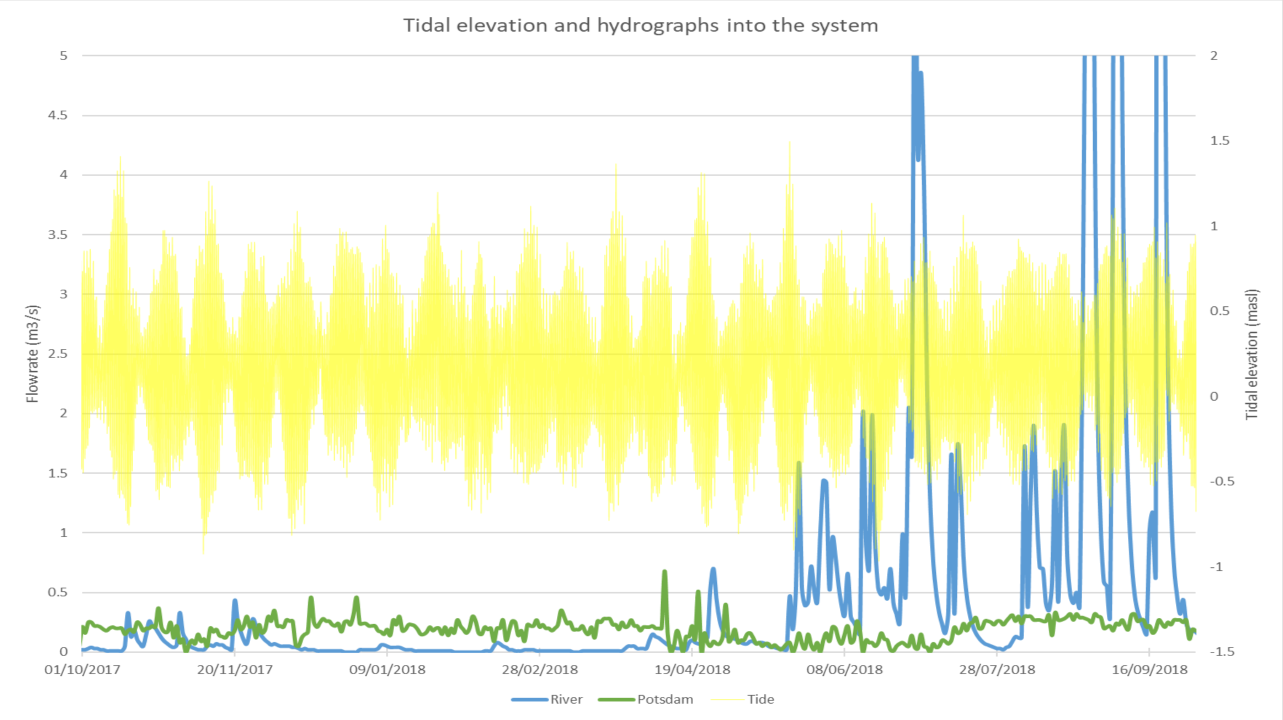


Figure 13-4. The tidal range and flow hydrographs for the Diep River (Tide on RHS axis)

From Figure 13-4 it is apparent that during the summer, the Potsdam discharge dominated the river flow, whilst in winter the flow of the river exceeded that of the Potsdam discharge. The river flow was obtained from the SHETRAN modelling component as there are no active flow gauges on the river near the estuary.

All meteorological data was obtained from the European Centre for Medium-Range Weather Forecast (ECMWF at www.ecmwf.int). The Centre provides climate reanalyses and specific datasets via the web, point-to-point dissemination. For this specific set there was no solar radiation data; only cloud cover, thus solar radiation was reproduced for HEC-RAS on the basis of global location and time of the day. The following figure (Figure 13-5) shows the seasonal and the diurnal air temperature over the system for the modelling period with a minimum of 5.8C and maximum of 29.5C.

Figure 13-6 shows the sampled water temperatures in the system at the various sampling sites shown in Table 13-1.

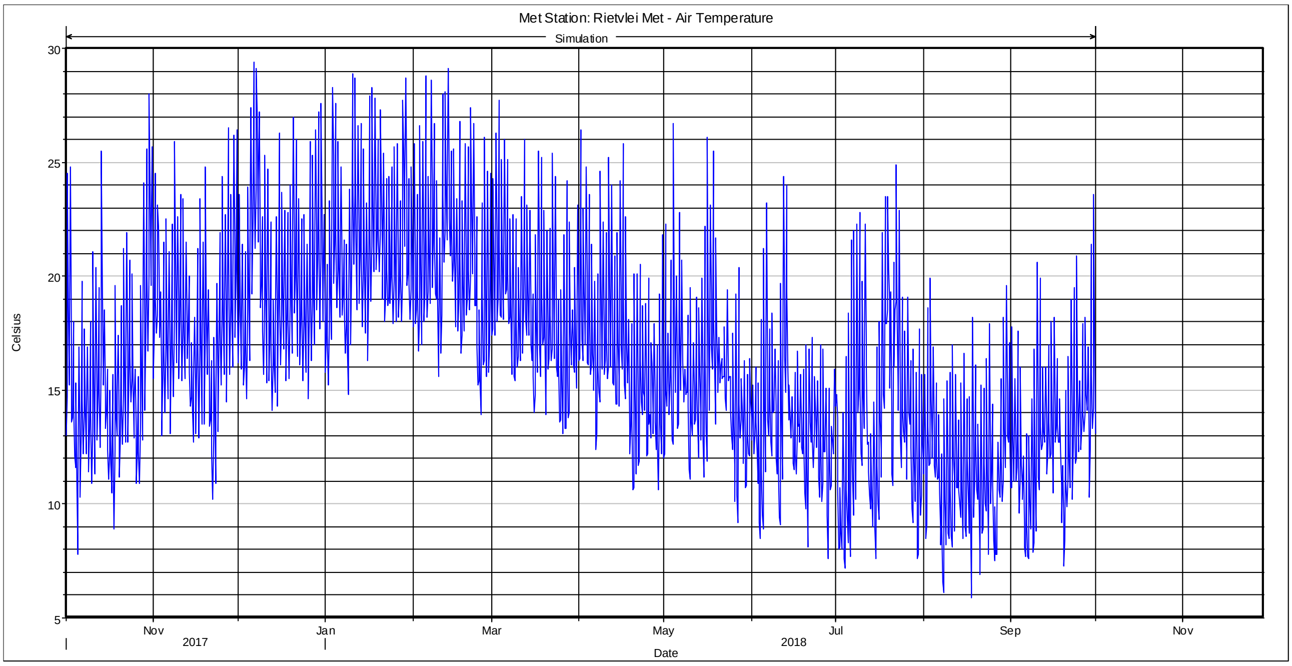


Figure 13-5. The air temperature at Table Bay Nature Reserve

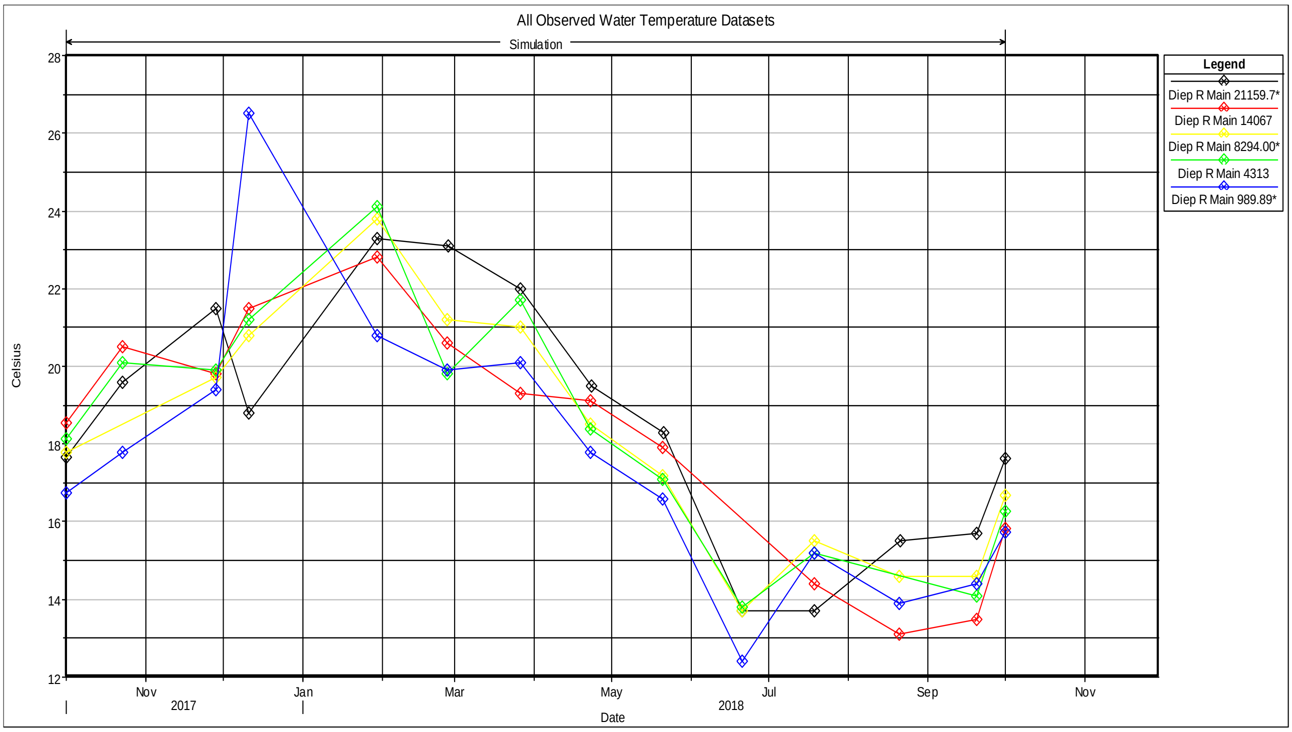


Figure 13-6. The sampled water temperature for the time period

The figures show that the water temperature followed the same pattern as the air temperatures and that closer to the mouth the water temperatures are cooler and further upstream it was warmer. The following figure shows the sampled dissolved oxygen concentrations in the system for the modelling period.

The highest dissolved oxygen concentrations were sampled at the mouth (Figure 13-7), whilst at Otto Du Plessis bridge (red line) the dissolved oxygen concentrations are on average at a minimum.

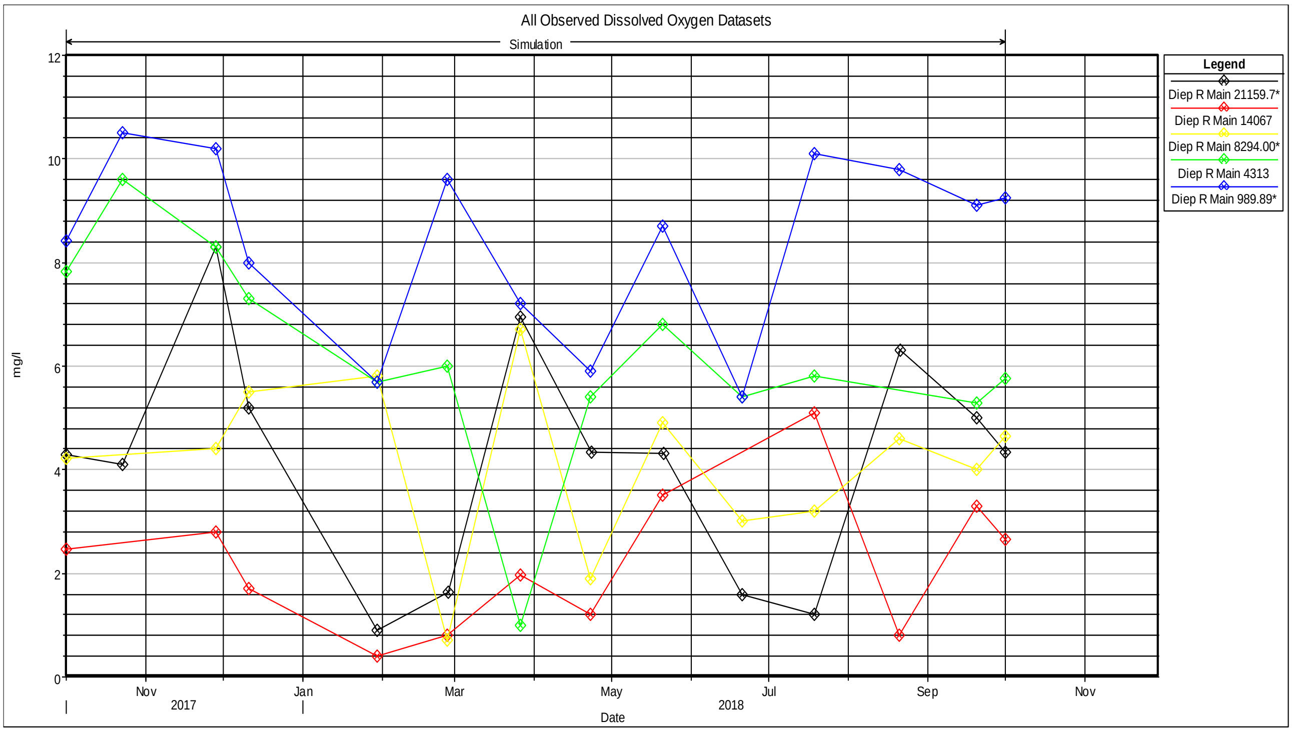


Figure 13-7. The sampled dissolved oxygen concentrations for the time period

Figure 13-8 shows the sampled nutrients in the system for the time period modelled.

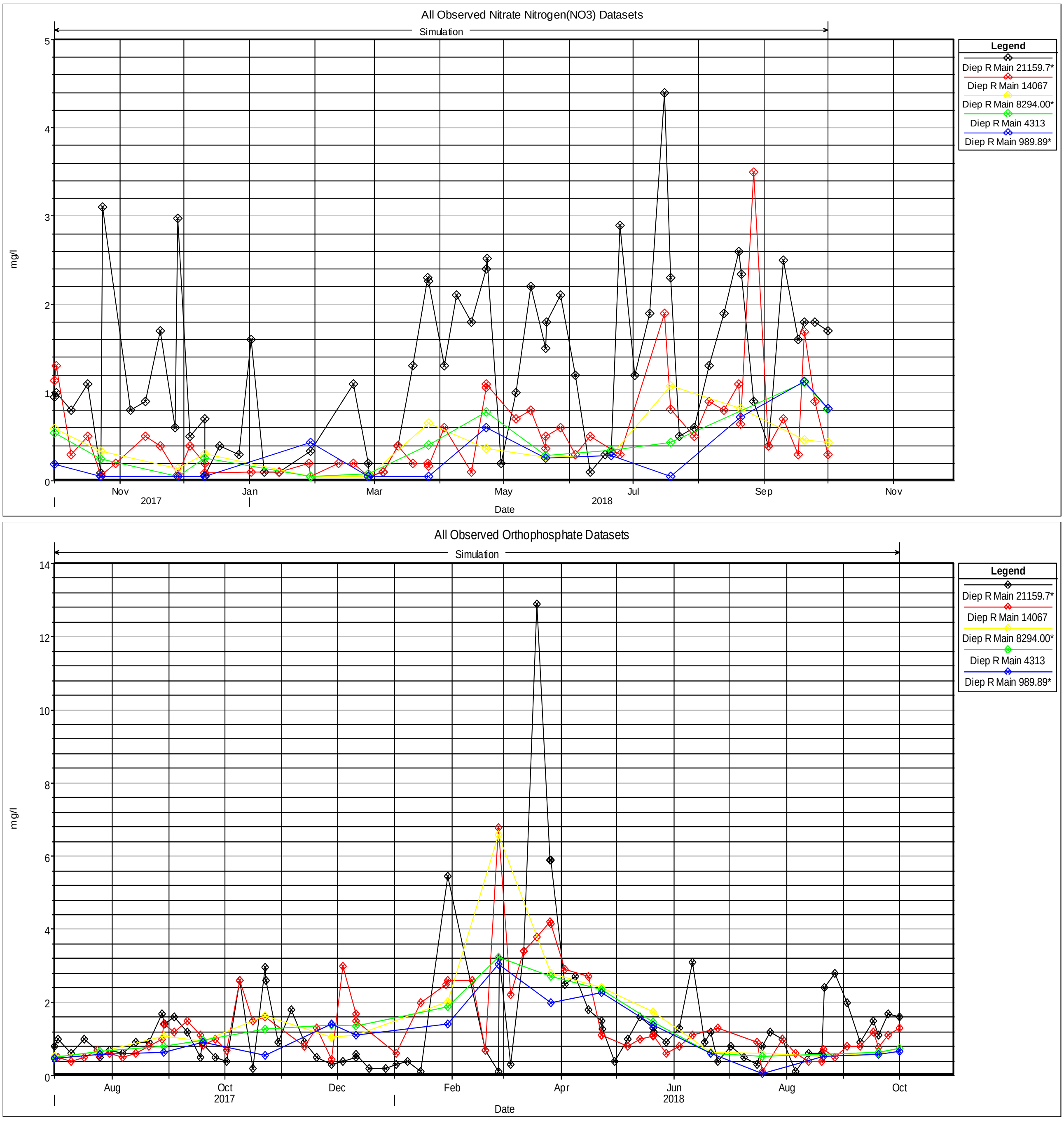


Figure 13-8. The sampled nutrients in the system (nitrate nitrogen top, orthophosphate bottom)

These figures show that for the modelling period the system was eutrophic and the nutrient concentrations were higher upstream in the system than closer to the mouth. Higher concentrations of nitrogen were evident upstream of the Potsdam discharge site, likely a consequence of fertilizer runoff in the catchment which coincided with increased river flows (See Figure 13-4 for February and April 2018 increases over base flows).

Not all the sampling data have an accurate time stamp to them, thus a direct comparison to the model output was not always possible as the model outputs every hour and there are 2 tidal cycles daily. Even with this shortcoming, the model would still predict the long-term trends and water quality. The sampled data was assigned as 10 am with an accuracy duration of 6 hours.

The modelling of any system is driven by its boundary conditions and for the time period there is no data for algal concentrations in the river, thus if the model matched nutrient and dissolved oxygen concentrations then its algal concentrations were deemed correct.

There was no sampling of salinity and dissolved oxygen concentrations of the ocean and in its absence, constant values of 35 PSU (35 000 mg/l) and 7 mg/l DO were used for the simulations. Ocean temperatures were assumed to be their historical monthly average for the time period modelled.

Salinity in the estuary is dependent on dispersion which in turn depends on the river discharge. Improving the modelled salinities would require more accurate longitudinal dispersion coefficients which need to be measured in-situ for the varying flowrates and mouth operating conditions.

13.4 The model calibration and validation

The initial model was set up and run for calibration, and the modelled output compared to sampled data at the beforementioned locations along the Diep River. Calibration was iterative and the model results were compared to actual system behaviour and modified until model accuracy was judged to be acceptable. The first variable that was calibrated was the surface water elevations which would indicate an erroneous inflow or outflow from the system. Once the model had produced an accurate enough surface water elevation, the next important variable calibrated was the water temperatures as these control the metabolic and growth rates of algae and affect the dissolved oxygen concentrations. The initial modelling period chosen for the calibration and validation was the hydraulic year **1 October 2017 to 1 October 2018**. Once the model had successfully reproduced the sampled data for this period it was considered as a validated model and was used to predict the water quality of the Milnerton Lagoon for the various management scenarios envisaged.

The relatively long timespans between empirical water quality data resulted in a decision to use all available data for model calibration without reserving some period of data for model verification. The first month of the calibration simulation was considered as a start-up period wherein the model was adjusting to initial condition specifications and was excluded from the calibration period. This method of calibration and validation was proposed by model developers for long term water quality modelling with limited input data (Cole and Wells, 2018).

13.4.1 The calibrated surface water elevations

The model was run with the SHETRAN river flows, Potsdam discharge volumes and tidal elevations. The surface water elevations at Otto Du Plessis Bridge were compared to the City's logger data as shown in the following figure.

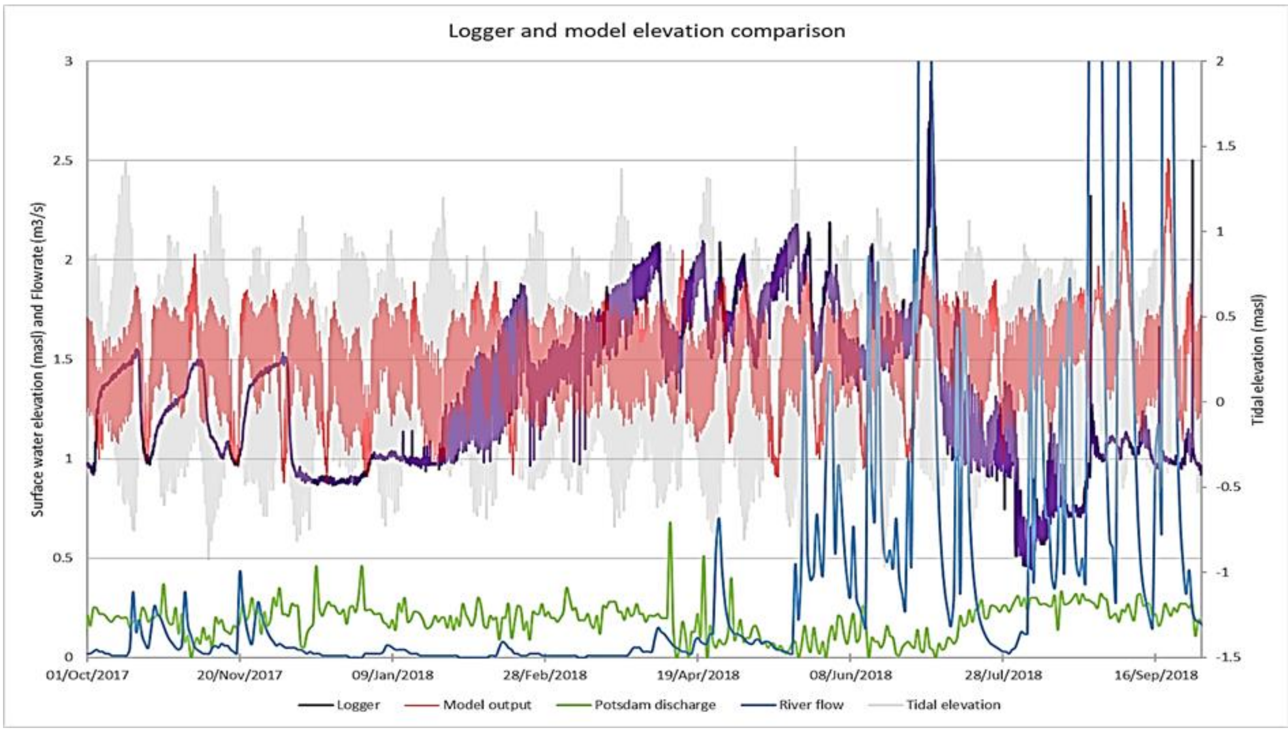


Figure 13-9. The logged surface water elevation and model comparison at Otto du Plessis Bridge

The modelled elevations show a relationship to tidal influence, river flows and Potsdam discharges.

The differences between the modelled elevations and logger were attributed to the following:

- The model does not account for groundwater interactions
- The model cannot account for stormwater addition to the system
- For this simulation period the logger does not show a relationship with tidal influences thus was considered inaccurate

13.4.2 The modelled water temperature

Cole and Wells (2018) recommend the use of temperature as a first step in hydrodynamic calibration followed by examination of dissolved oxygen (DO) concentrations. The water temperature controls the metabolic and growth rates and affects the dissolved oxygen concentrations in the water column. The following figure is the water temperatures at the boundary of the model.

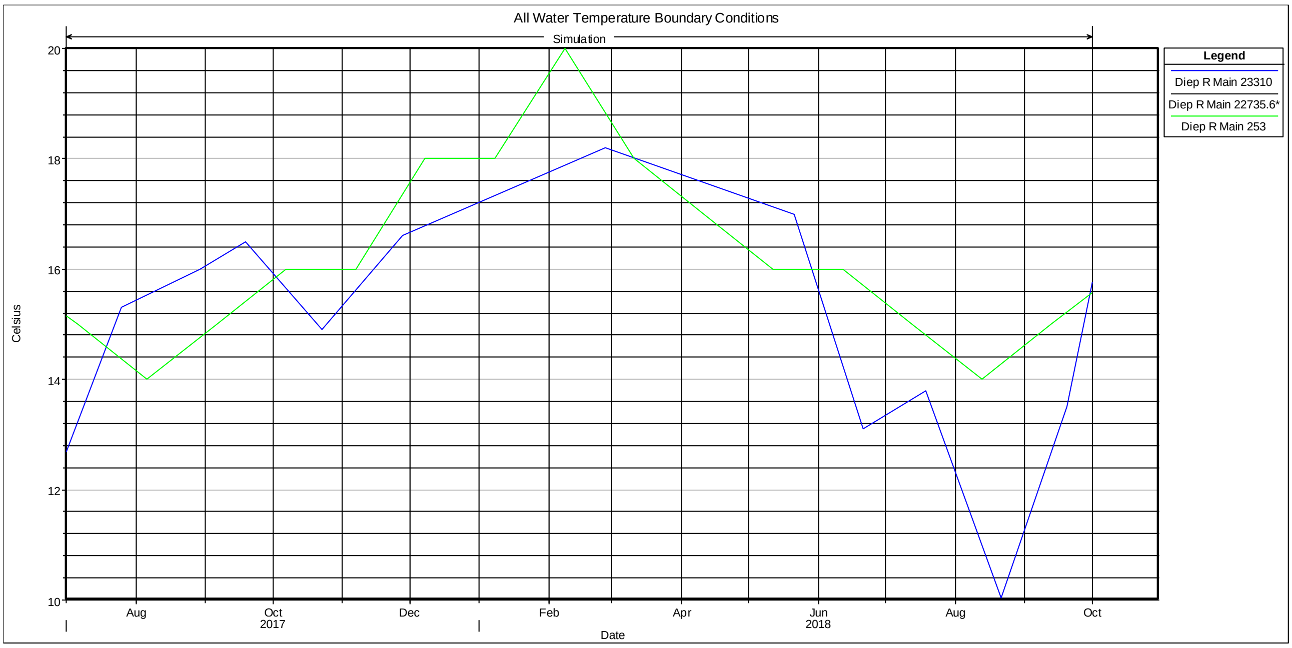


Figure 13-10. The water temperatures at the model boundary

These boundary conditions are not predicted by the model and must be specified and they ultimately converge the numerical solution of the model irrespective of the initial conditions, thus for any modelling the boundary conditions are more important than initialisation and need to be comparably frequent to the desired output. Figure 13-10 shows the sampled river temperature (blue), the monthly ocean temperature (green) and Potsdam discharge at 20°C (assumed as there is no sampled data) for the time period modelled. It is noted that the river temperature and ocean water temperatures follow a similar pattern in that the temperature is higher in summer and lower in winter. Figure 13-11 and Figure 13-12 show the modelled water temperatures at two points; additional outputs are included in **Annexure G**.

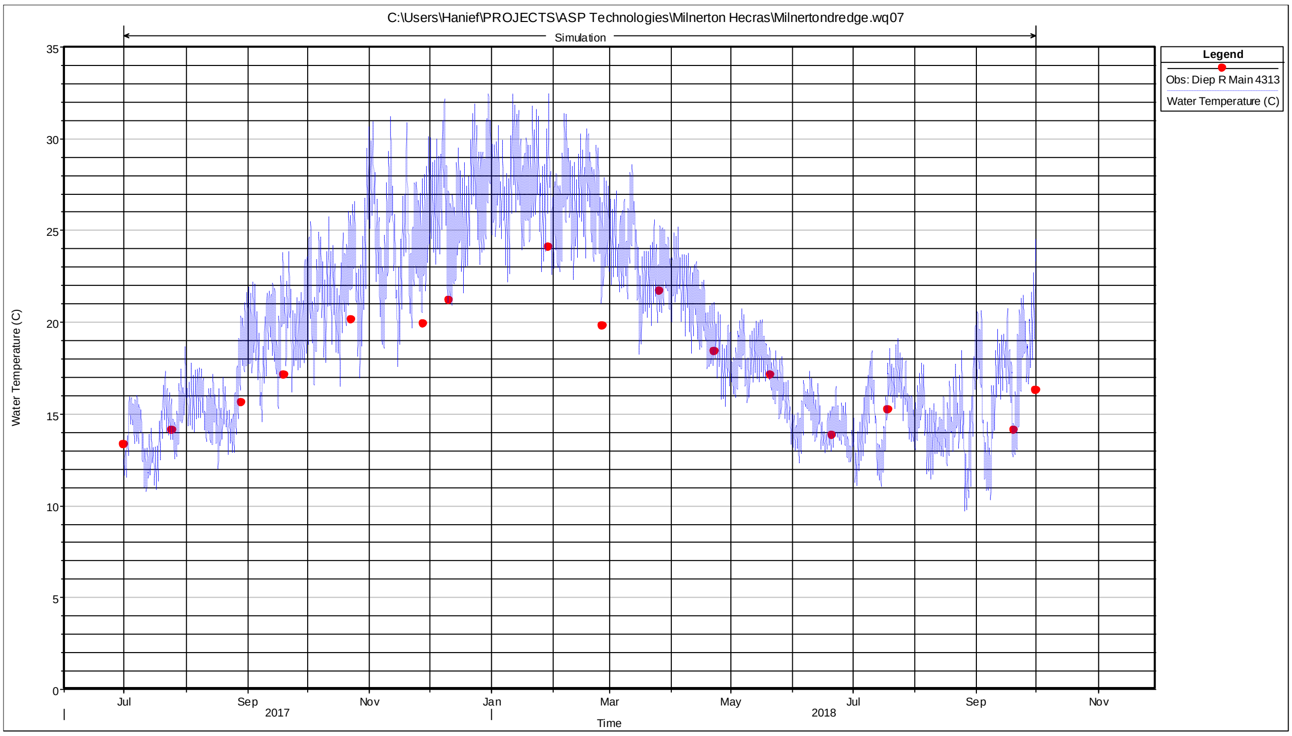


Figure 13-11. The modelled water temperatures for the time period at RTV09 (Woodbridge)

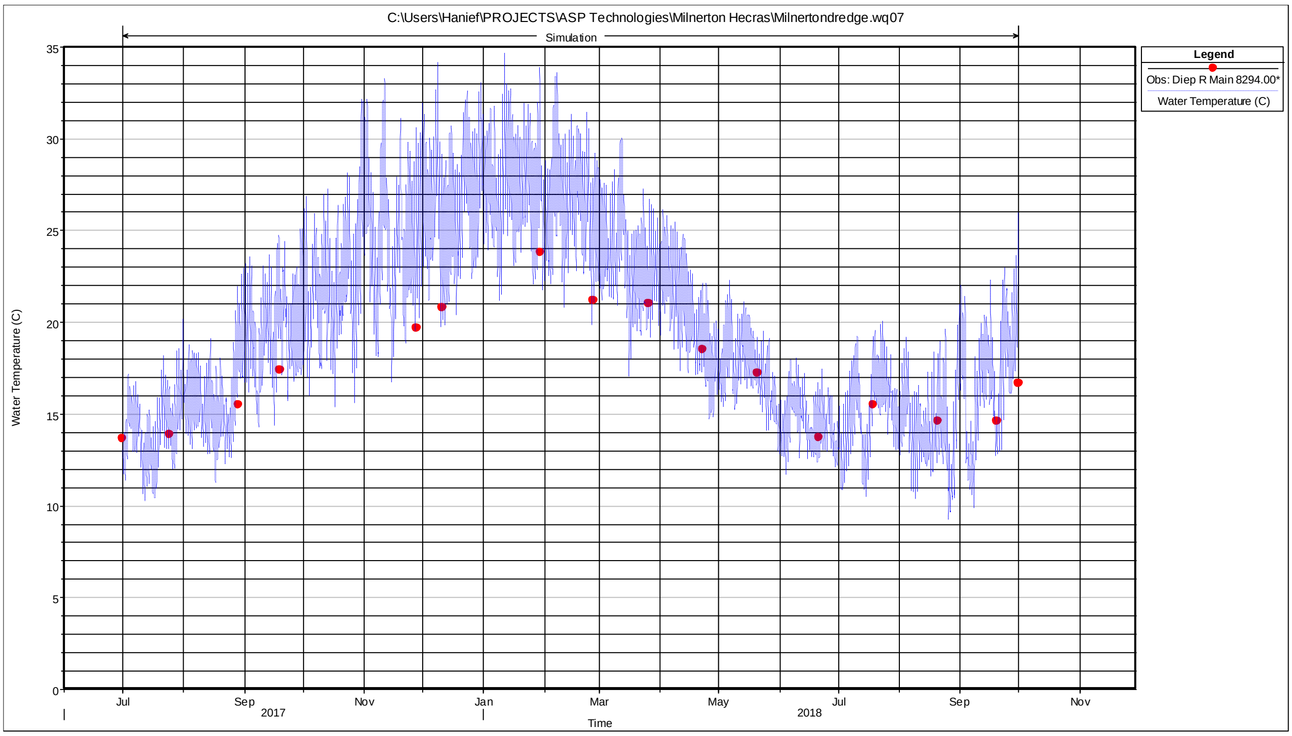


Figure 13-12. The modelled water temperatures for the time period at RTV18 (Broad Road)

The modelled water temperatures (1 October 2017 – 1 October 2018) are relatively accurate and reproduced the annual cyclic temperature variation as well as diurnal variations driven by the meteorological input to the system. The main drivers (boundary conditions) for water temperature in the system are:

- Meteorological inputs (6 hourly data from ECMWF reanalysis)
- Diep river water temperature (sampled)
- Potsdam effluent discharge temperature (no data)
- Sea water temperatures (historical monthly averages)
- Solar radiation extinction coefficients for detritus matter and algae in the water

To improve the modelled water temperatures would require sampled seawater temperatures, the Potsdam effluent temperature data, measurement of the extinction coefficients, as well as weather station data at the site. In the absence of this data no further improvement in water temperatures was possible and the model was deemed calibrated and validated for water temperature.

13.4.3 The modelled dissolved oxygen concentrations

Oxygen is one of the most important elements in aquatic ecosystems. It is essential for higher forms of life, controls many chemical reactions through oxidation, and is a surrogate variable indicating the general health of aquatic systems. If a single variable were to be measured in aquatic systems that would provide maximum information about the system state, it would be dissolved oxygen (Cole and Wells, 2018). The interactions modelled between dissolved oxygen and various other user-quantified water quality parameters are shown in Figure 13-13.

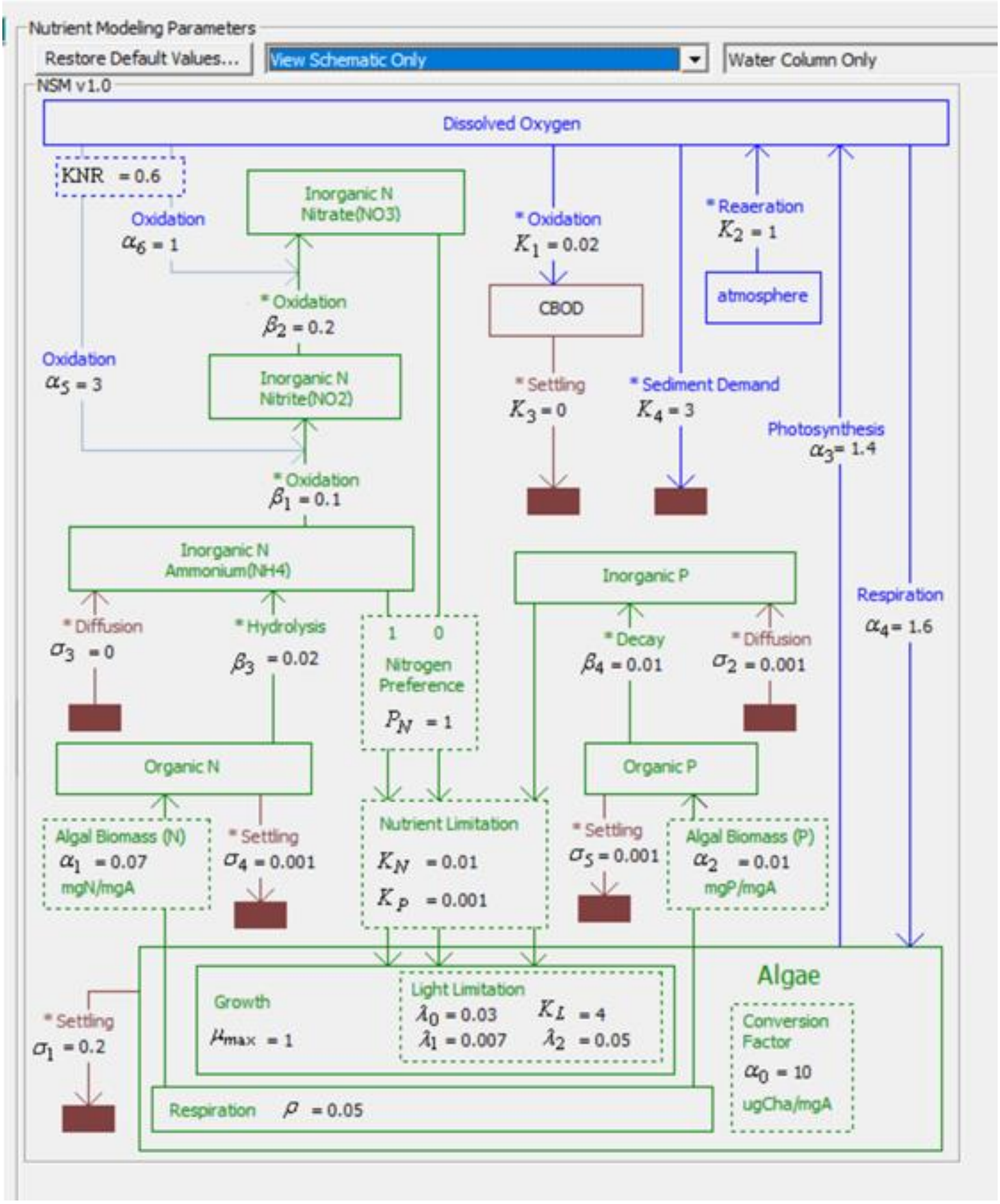


Figure 13-13. The internal flux of dissolved oxygen within the model

Figure 13-13 demonstrates that the accurate modelling of dissolved oxygen concentrations involves all the interactions in the water column as well as literature-determined or sampled in-water constants to replicate the observed concentrations. There was no sampled algal concentration for the time period at the boundary of the model (i.e. in the Diep River, the ocean or Potsdam discharge) and many of these parameters are not measured in the system. Most of the parameters are temperature dependent, emphasising the need to first calibrate the model for best fit for water temperatures before attempting any constituents.

As there is no data for sediment oxygen demand (SOD) from the Diep River, representative SOD rates were used. Literature quoted values for SOD range from 0.3 to 5.8 g O2 m^{-2} day^{-1} and may be even greater (Cole and Wells, 2018). The SOD process is quite complex and is not yet well defined in each intermediate step; thus, modelling efforts to obtain a quantitative and predictive assessment of SOD values are subjected to continuous evolution with improvements in modelling software. Comparative in-situ SOD values for polluted and unpolluted sediments reported by various researchers are shown in Table 13-3.

Table 13-3. Comparative literature SOD values (Butts, 1974)

Sediment conditions		Temperature (C)	SOD g/m²/day
Unpolluted	Polluted		
X		8 – 9	1.2 – 2.2
X		20	0.72 – 2.40
	X	22	0.55 – 5.17
	X	20	3.41 – 6.17
	X	20	3.36 – 19.20

Butts (1974) characterised SOD in terms of sediment composition and found that greater SOD levels associated with silts and clays at the same sampling sites. The initial SOD chosen was based on the water temperature and used as calibration for the dissolved oxygen dataset at 3 g/m²/day. The final dredged SOD was set at 1 g/m²/day, the upper and lower limits for polluted and unpolluted sediments as shown in the above table. Thus, literature quoted values were used for the simulations and any improvements in the modelling would require in-situ measurements of the SOD in Diep River. The dissolved oxygen boundary conditions for the modelled time period are shown in the following figure.

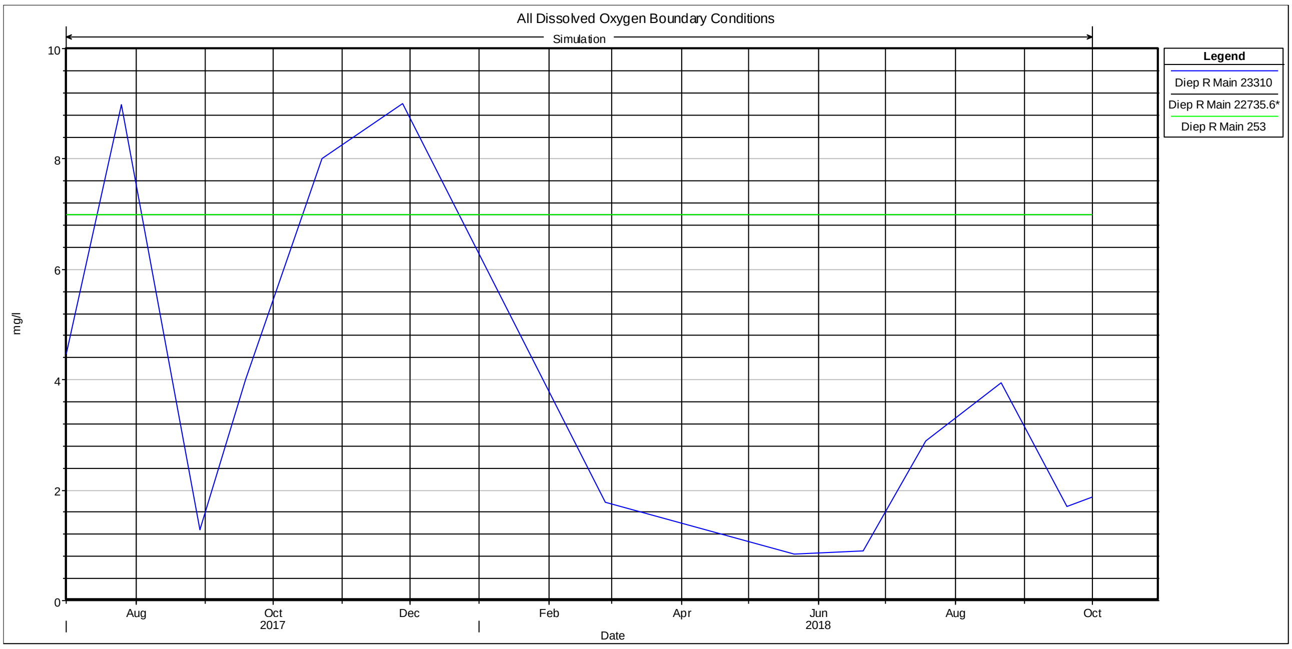


Figure 13-14. The dissolved oxygen concentrations at the model boundary

In Figure 13-14, only the river dissolved oxygen concentrations show variation whilst the ocean and Potsdam discharge have been fixed as constant for the time period. The modelled output of dissolved oxygen is shown in Figure 13-14 and **Annexure G**.

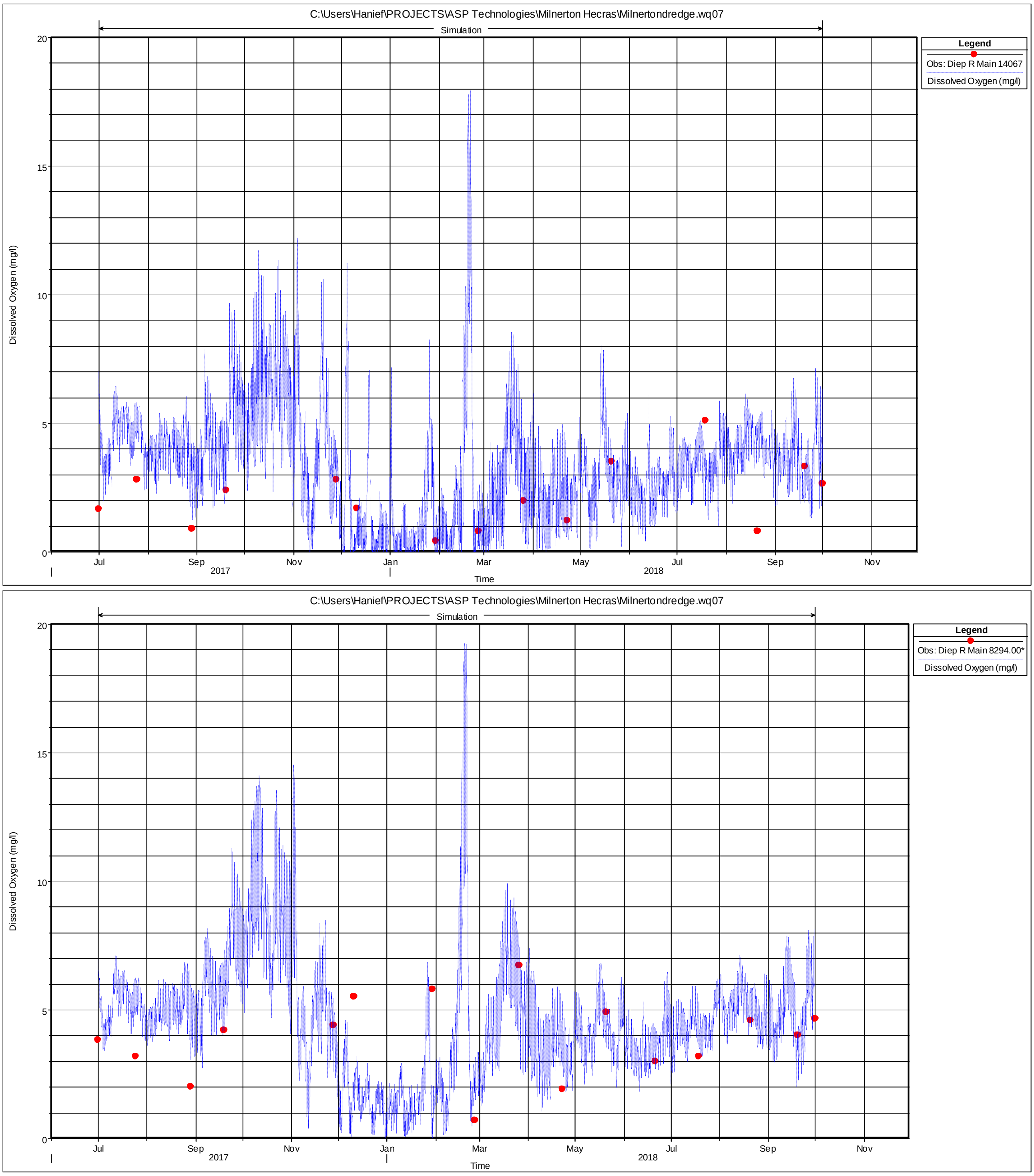


Figure 13-15. The modelled dissolved oxygen concentrations for the time period

For the time period the modelled output compares well to the sampled data, whilst highlighting diurnal fluctuations and supersaturation in February 2018, attributed to algal blooms as shown in the ensuing sections. Improvement of the model outputs would require sampled data for Potsdam effluent and seawater dissolved oxygen concentrations.

13.4.4 The modelled ortho-phosphate concentrations

For the purposes of the modelling, phosphorus was assumed to be completely available as ortho-phosphate (PO_4) for uptake by phytoplankton (algae). Measurements of soluble reactive phosphorus are closest to the form used in the model and the interactions are shown in Figure 13-13. The boundary ortho-phosphate concentration is shown in the following figure.

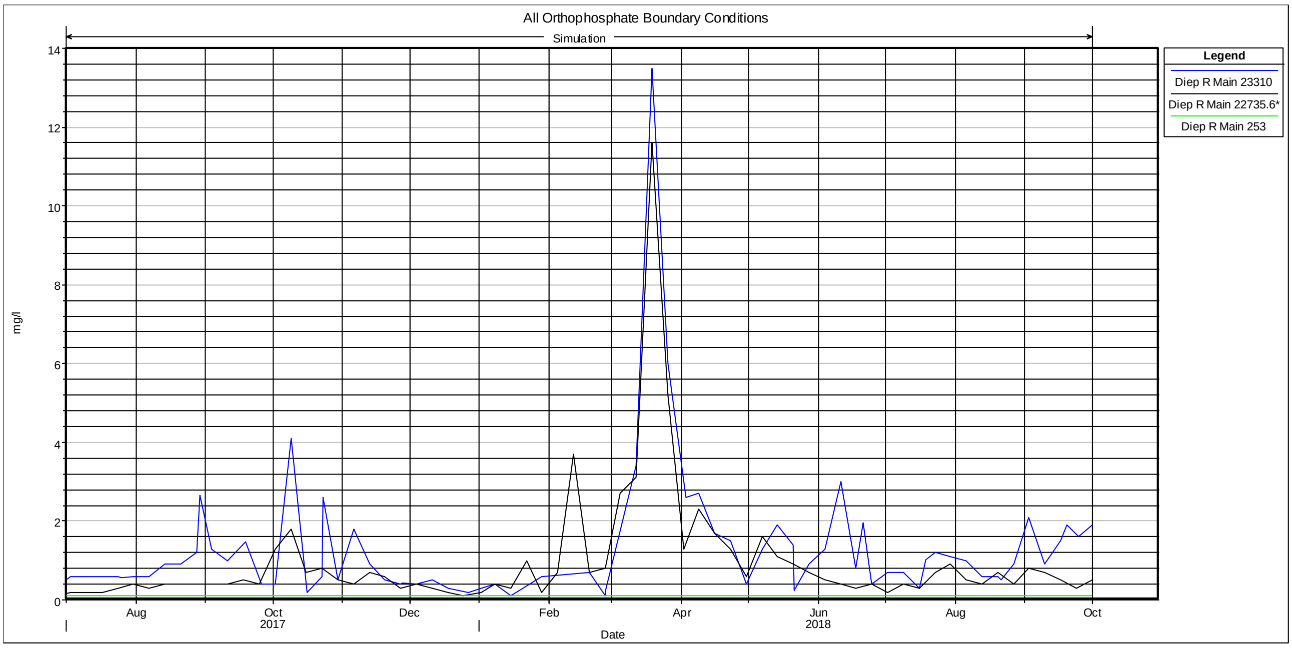


Figure 13-16. The ortho-phosphate concentrations at the model boundary

This figure shows that there was a significant inflow of ortho-phosphates into the system from both the Diep River and Potsdam discharge on the 19th of March 2018. The results of the modelling are shown in the following figures and in **Annexure G.**

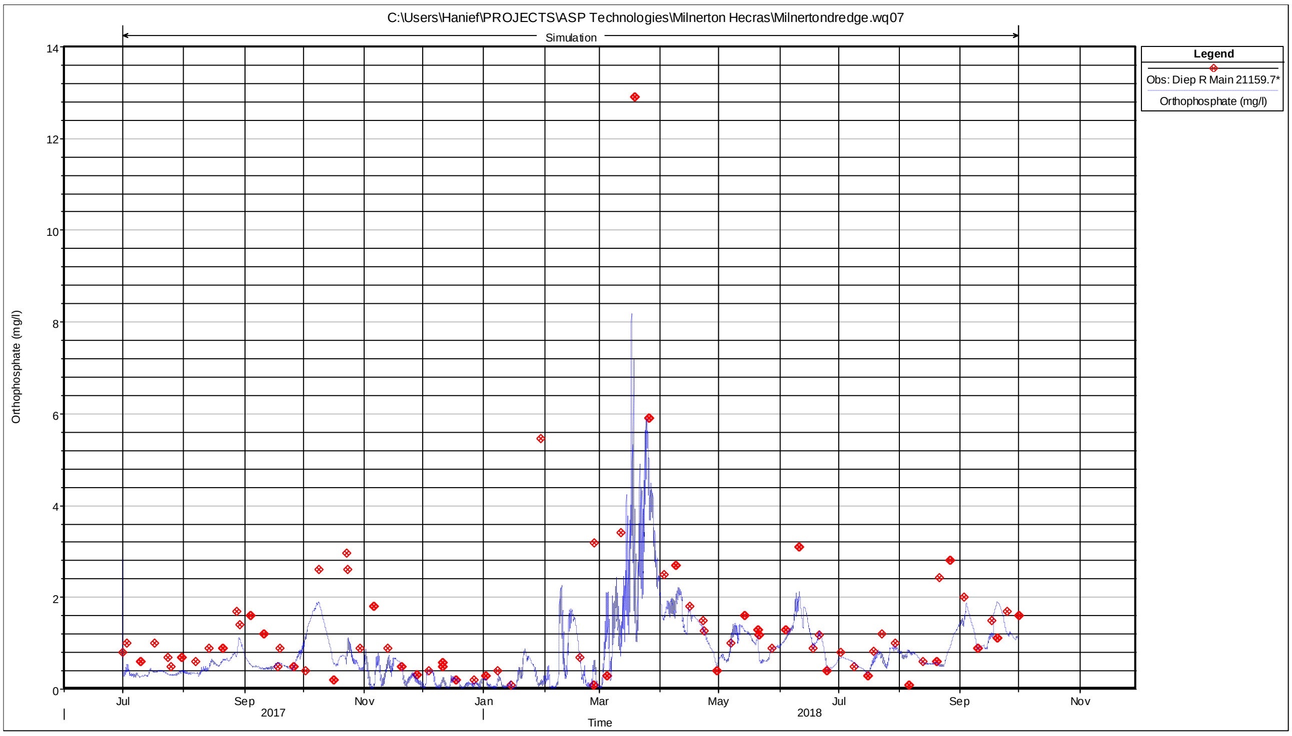


Figure 13-17. The modelled ortho-phosphates for the time period

The phosphate contributions are from the water column as well as for organic sediments when algae die, settle and nutrients are released by decay back to the water column. Figure 13-17 shows that the model predicts the ortho-phosphate concentrations for the time period which suggests it would predict algal concentrations well in areas where phosphorus is the limiting nutrient. Any improvements would require additional sampled ortho-phosphate concentration for the ocean as well as accounting for diffuse sources.

13.4.5 The modelled nitrogen concentrations

Nitrogen is an important plant nutrient and in the model nitrate-nitrite is affected by ammonium concentrations, denitrification and photosynthesis. All forms of nitrogen are accounted for, including inorganic nitrogen, nitrate (NO_3), nitrite (NO_2), ammonium (NH_4) and organic nitrogen. Nitrite is an intermediate product in nitrification between ammonium and nitrate. Nitrate is used as a source of nitrogen for algae during photosynthesis. The nitrogen concentrations for the boundary of the model are shown in Figure 13-18.

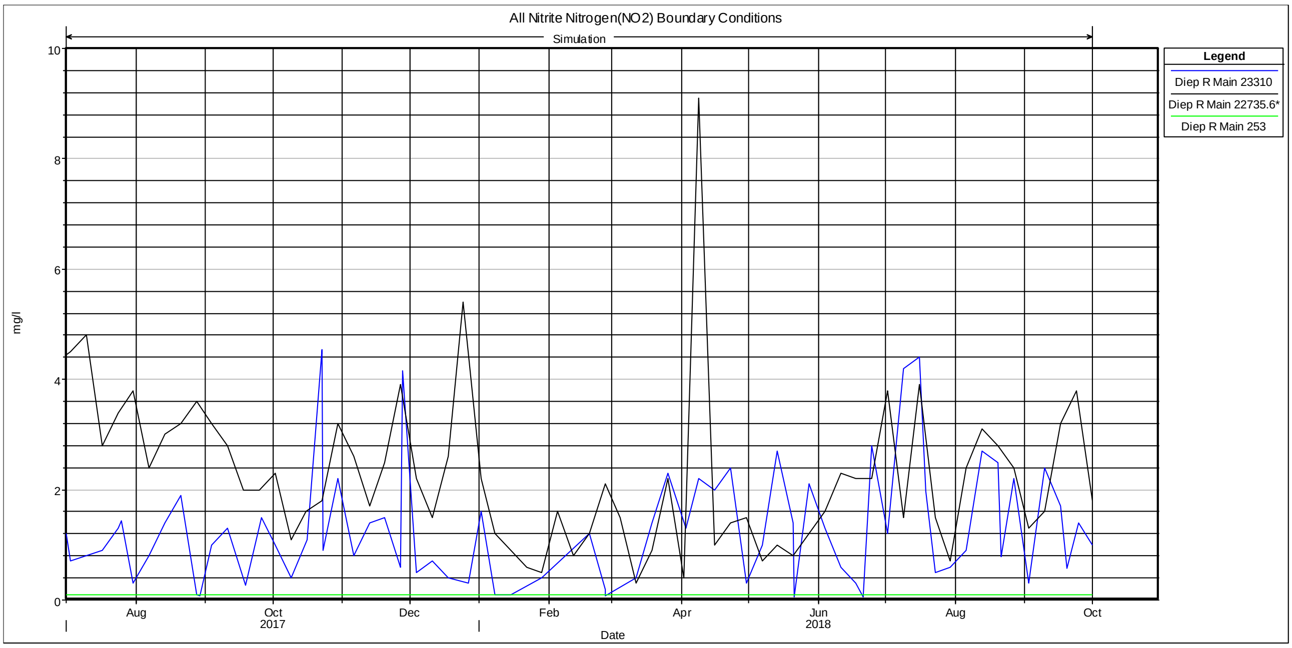


Figure 13-18. The nitrogen concentrations at the model boundary

From the figure it is apparent that the nitrogen concentration discharged by Potsdam is on average greater than the concentration in the Diep River. In the absence of sampled data, the nitrogen concentration of the ocean was fixed for the duration of the simulation. Figure 13-19 shows the modelled nitrogen concentrations for the time period with additional figures in **Annexure G**.

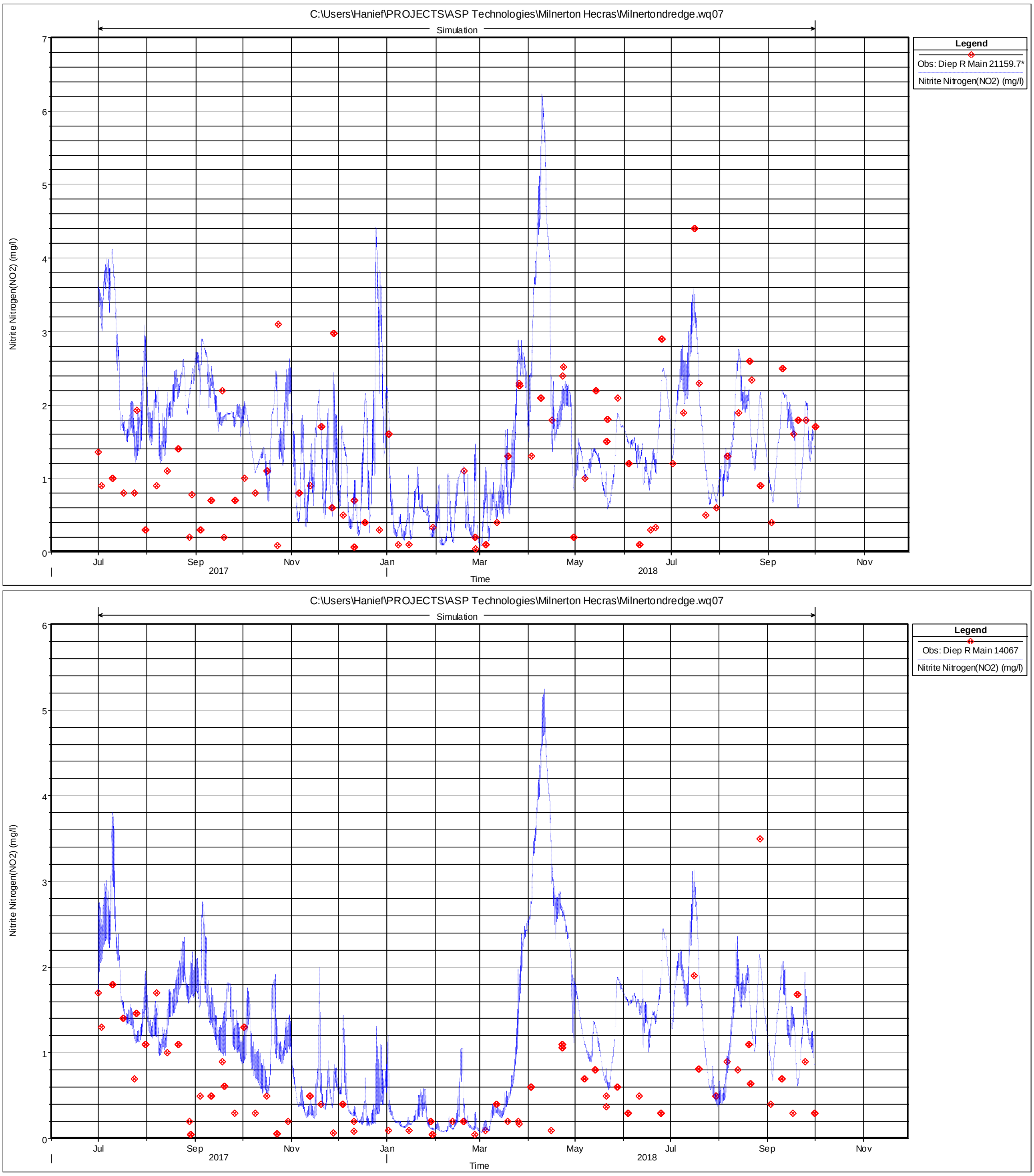


Figure 13-19. The modelled and observed nitrogen concentrations for the time period

The figures show that the spike in nitrogen during April 2018 was a result of high discharge from Potsdam and very low river flow which prevented dilution of the nitrogen as observed in the boundary conditions.

13.4.6 The modelled ammonium concentrations

Ammonium is oxidised to nitrites and nitrates as well as being assimilated by algae during photosynthesis to form proteins. The interactions between ammonium modelled and other water quality constituents include losses from the system via nitrification to nitrate-nitrites and photosynthesis, whilst sediment release and algal respiration account for the increases in ammonium. Figure 13-20 shows the ammonium concentrations at the boundary of the model.

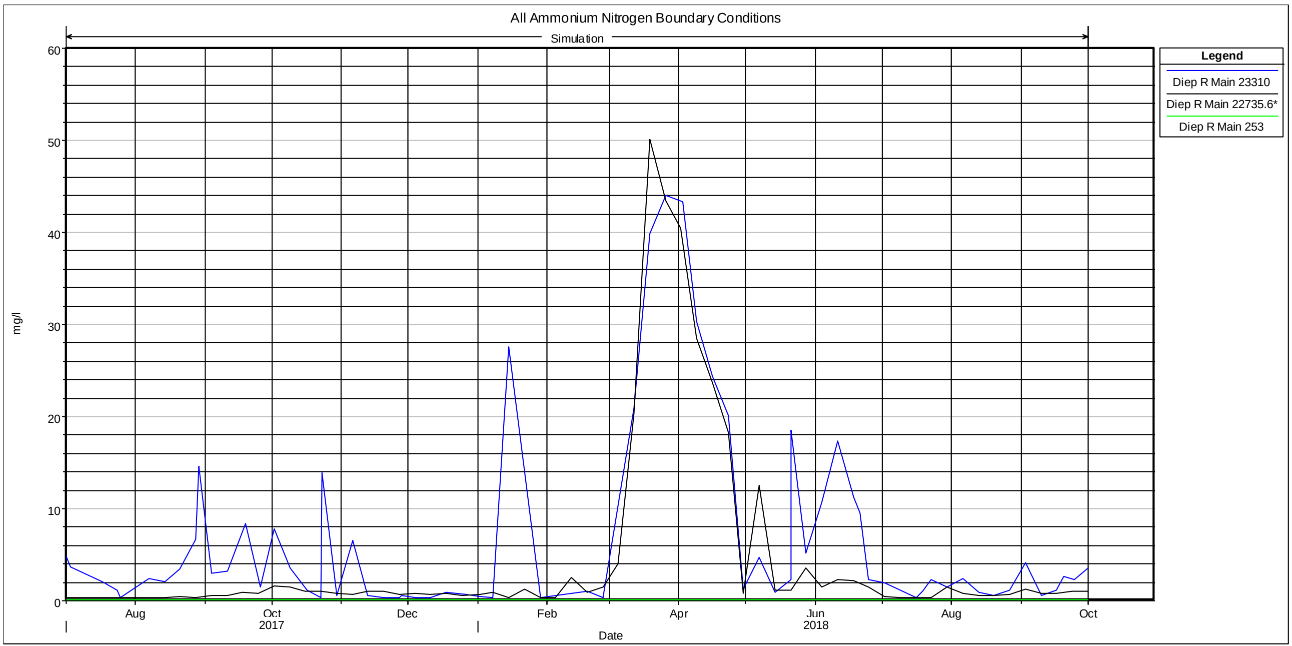


Figure 13-20. The ammonium concentrations at the model boundary

Figure 13-20 shows that the maximum ammonium concentration is similar for both the Diep River and Potsdam effluent, but on average the concentration is higher in the Diep River. The ocean ammonium concentration was fixed for the simulation in the absence of sampled data. Figure 13-22 and Figure 13-22 show the ammonium concentrations for the period modelled.

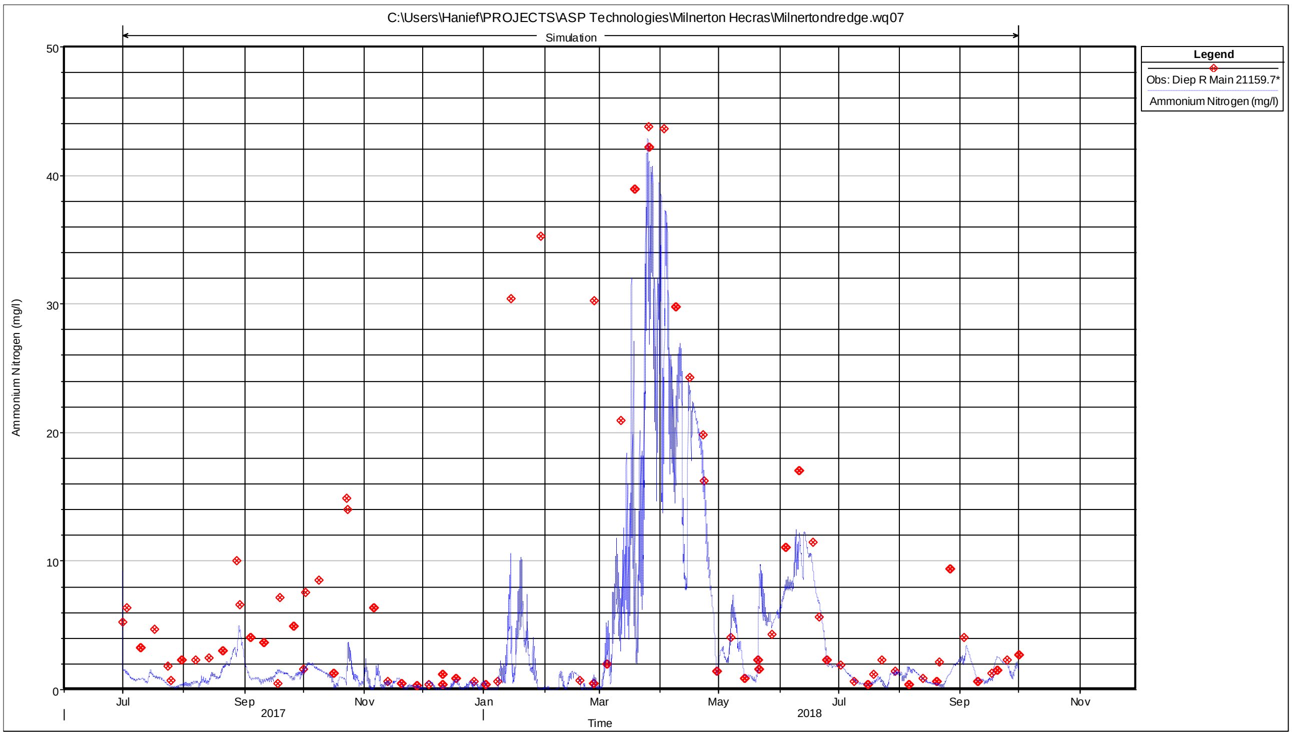


Figure 13-21. The modelled ammonium concentrations for the time period

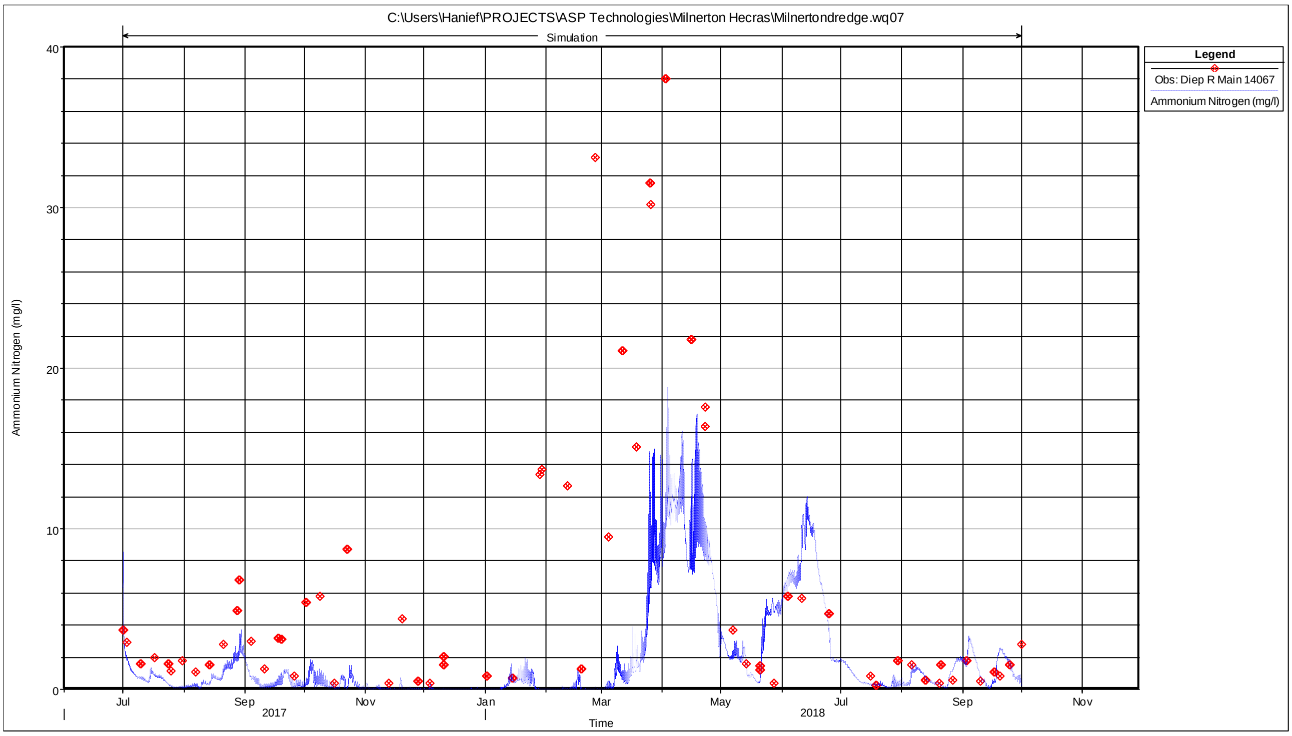
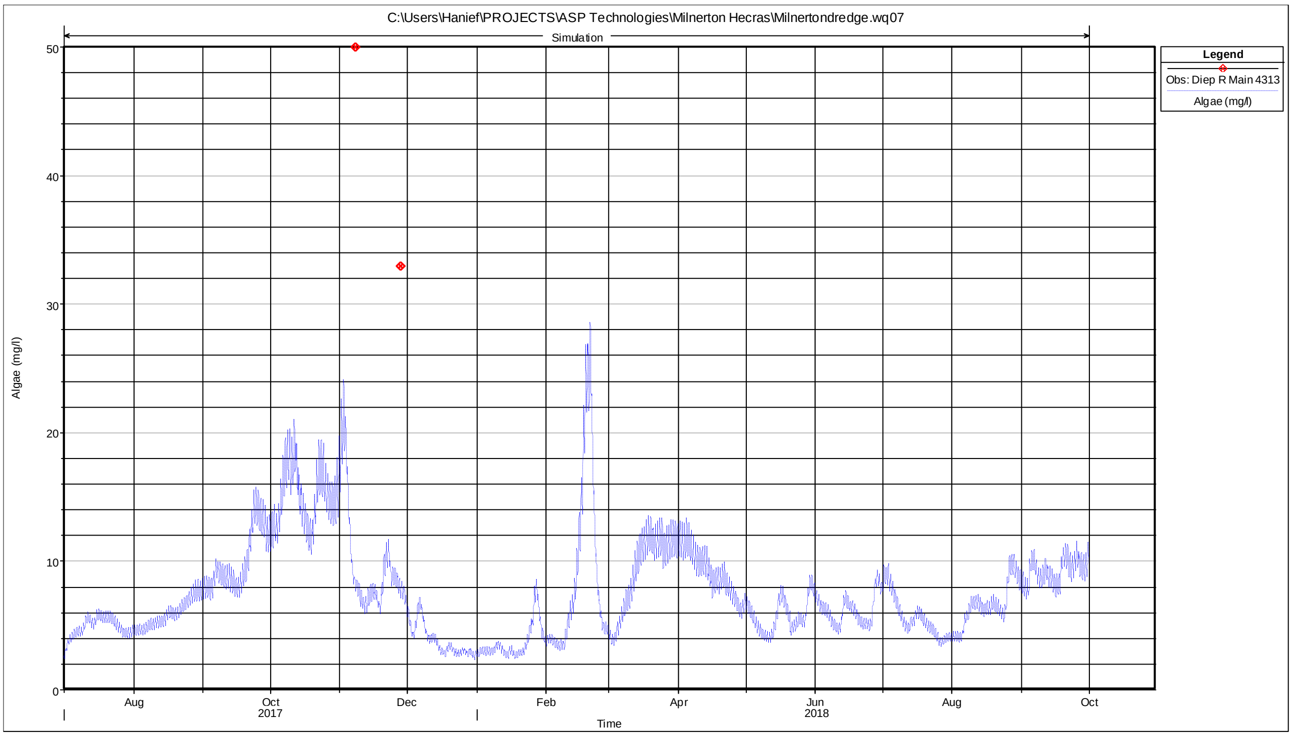


Figure 13-22. The modelled ammonium concentrations for the time period

Any improvements in modelled ammonium would require oceanic sampling for ammonium and in-water sampling for the various water quality parameters.

13.4.7 The modelled algal concentrations

Accurate modelling of algae demands the inclusion of all related water quality variables. Algal growth rates are computed at each iteration by modifying the maximum growth rate as affected by temperature, light and nutrient availability as well as rate multipliers limiting maximum algal growth due to nutrient limitations using the Monod relationship. Algal growth does not occur in the absence of light. Algal growth was not allowed to exceed the limit imposed by nutrient supply over a given timestep. Algal excretion is not allowed to exceed algal growth rates and algae passing to the sediments accumulate within the sediment compartment. For the time period there is a dearth of algal concentration sampling with only 2 data points. Algal sampling has however become more frequent since 2020.



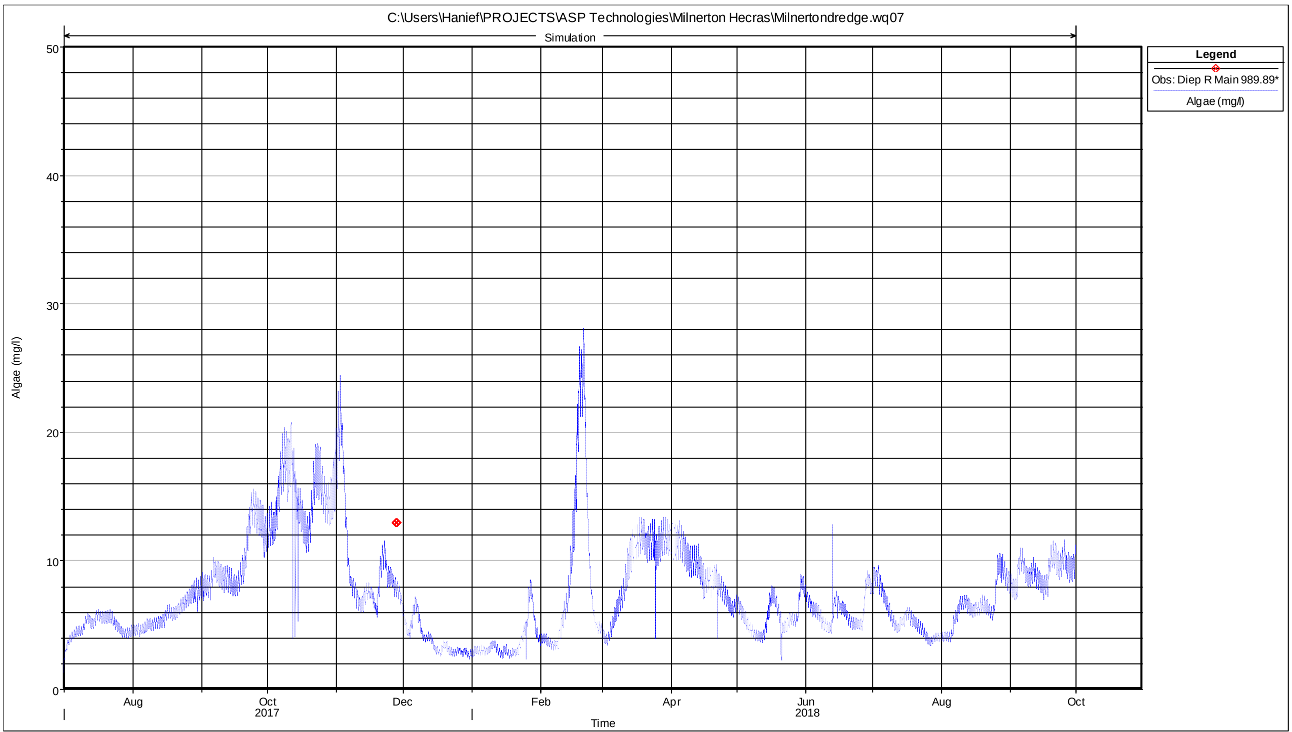


Figure 13-23. The modelled algal concentrations for the time period

The algal concentrations showed increases with increased nutrients and water temperature. The February supersaturation of dissolved oxygen is here seen to be the result of an algal bloom as a result of increased water temperatures and eutrophication. All algal concentrations at the boundary would need to be sampled including the Diep River (currently performed), Potsdam discharge and the ocean in order to improve the modelling predictions.

13.4.8 The modelled salinity

Conductivity is a measure of the ability of water to pass an electrical current. Because dissolved salts and other inorganic chemicals conduct electrical current, conductivity increases as salinity increases. Conductivity is also affected by temperature: the warmer the water, the higher the conductivity. Salinity is a measure of the amount of salts in the water and as dissolved ions increase salinity as well as conductivity, the two measures are related.

Conductivity in the estuary is a function of the oceanic inputs, Potsdam discharges and river conductivity. In the case of the Diep River, the Potsdam effluent discharge dominates the summer flows. Sampling of conductivity (mS/m) was performed as part of the routine water quality sampling for the city and this was converted to salinity as mg/l. This conversion correlates well but varies with temperature and air pressure. Figure 13-24 and Figure 13-25 show the salinity in the estuary at various locations.

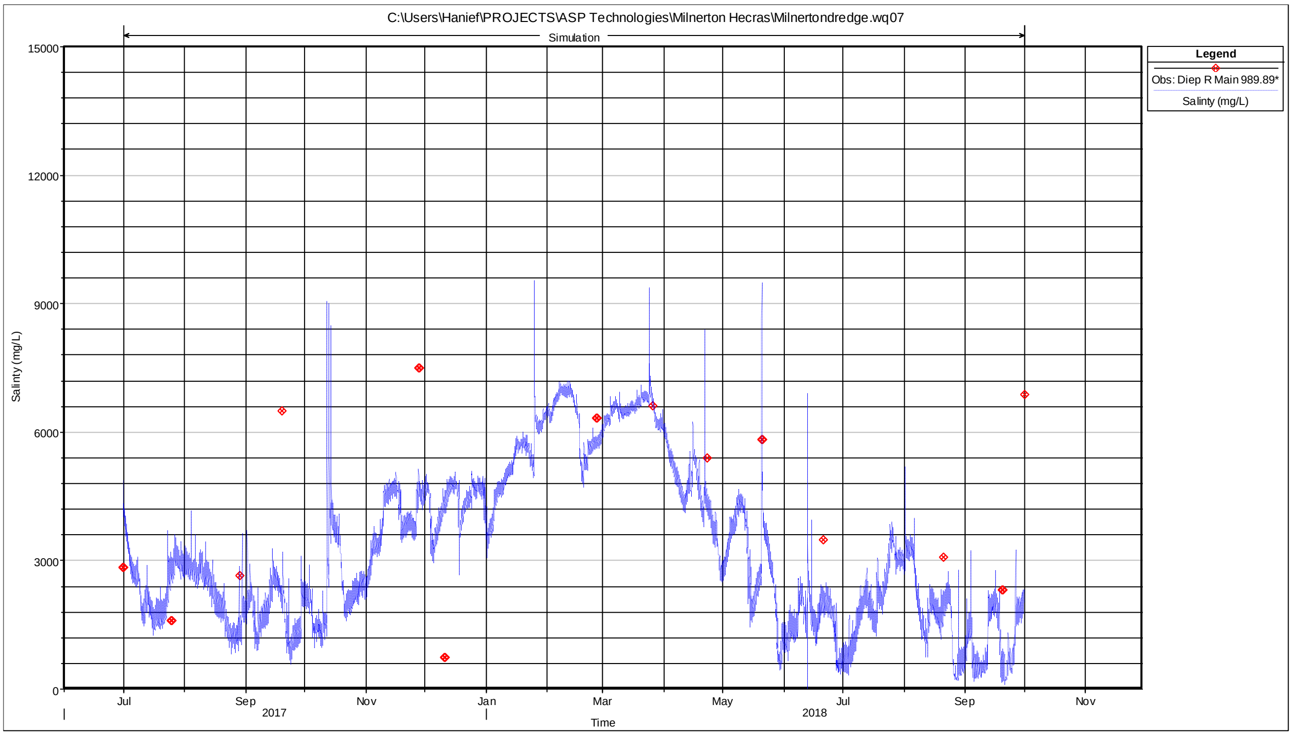


Figure 13-24. The modelled salinity for the time period

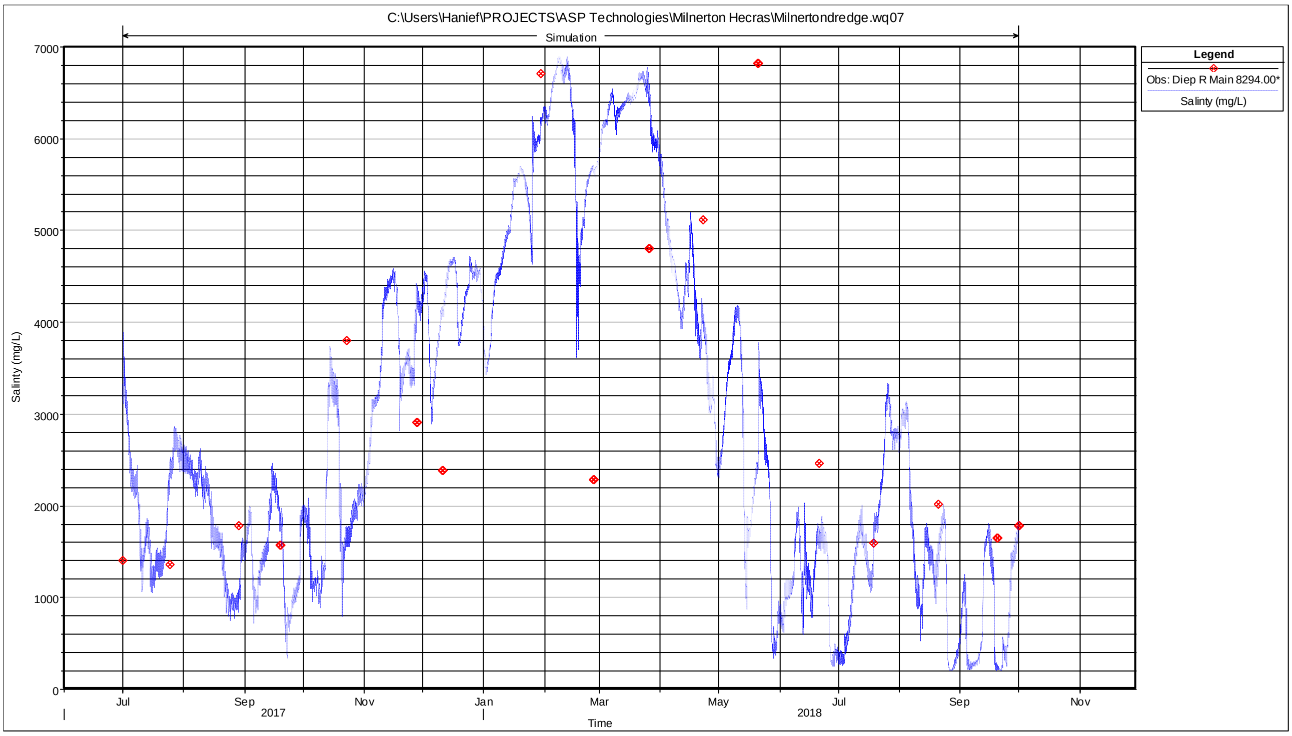


Figure 13-25. The modelled salinity for the time period

The salinity of the ocean was fixed for the simulations. From these figures it is evident that the model predicts the salinity (as mg/l where 1000 mg/l = 1 PSU) fairly well and shows an increasing trend during summer which is expected as the both the Diep River flows and the discharge from Potsdam decrease. In order to reconcile the differences, the dispersion coefficients should be measured in-situ.

13.4.9 Calibration and validation

With the available data, the model was now considered calibrated and validated for the modelling period of 1 October 2017 – 1 October 2018 i.e. one hydraulic year.

13.5 The new baseline modelling scenario

The model, having been calibrated, could then be used to predict the water quality of the system for the various management scenarios envisaged. These were used to test the water quality dynamics of the following scenarios:

1. The current Potsdam effluent at licenced constituent concentrations (i.e. the current scenario but with significant improvements in Potsdam effluent discharge quality)
2. Potsdam effluent discharge volume increased to 100Ml/day at licensed discharge quality and at the current water quality (i.e. the anticipated future maximum discharge volume but without significant quality improvements)
3. Dredging the lagoon and river bed to remove contaminated sediments (i.e. **Option 1**).
4. River aeration (i.e. **Option 2**)
5. The addition of seawater into the system (0.1, 0.2 and equal to Potsdam discharge in m^3/s) (i.e. **Option 3**)
6. The diversion of the Potsdam discharge to the ocean (i.e **Option 4**).

The water quality model outputs data for the entire bathymetry but for the purposes of brevity, the output was constrained to segments that include RTV05 (Otto Du Plessis), RTV09 (Woodbridge) and RTV10 (mouth). The output from these locations is compared to the model output as the new baseline simulation. The new modelling period is chosen as **1 December 2018 – 31 August 2019** as the Potsdam effluent data shows that was when the incidents of poor-quality effluent from Potsdam increased and 31 August is the last meteorological data point. The Potsdam effluent data is shown in Figure 13-26.

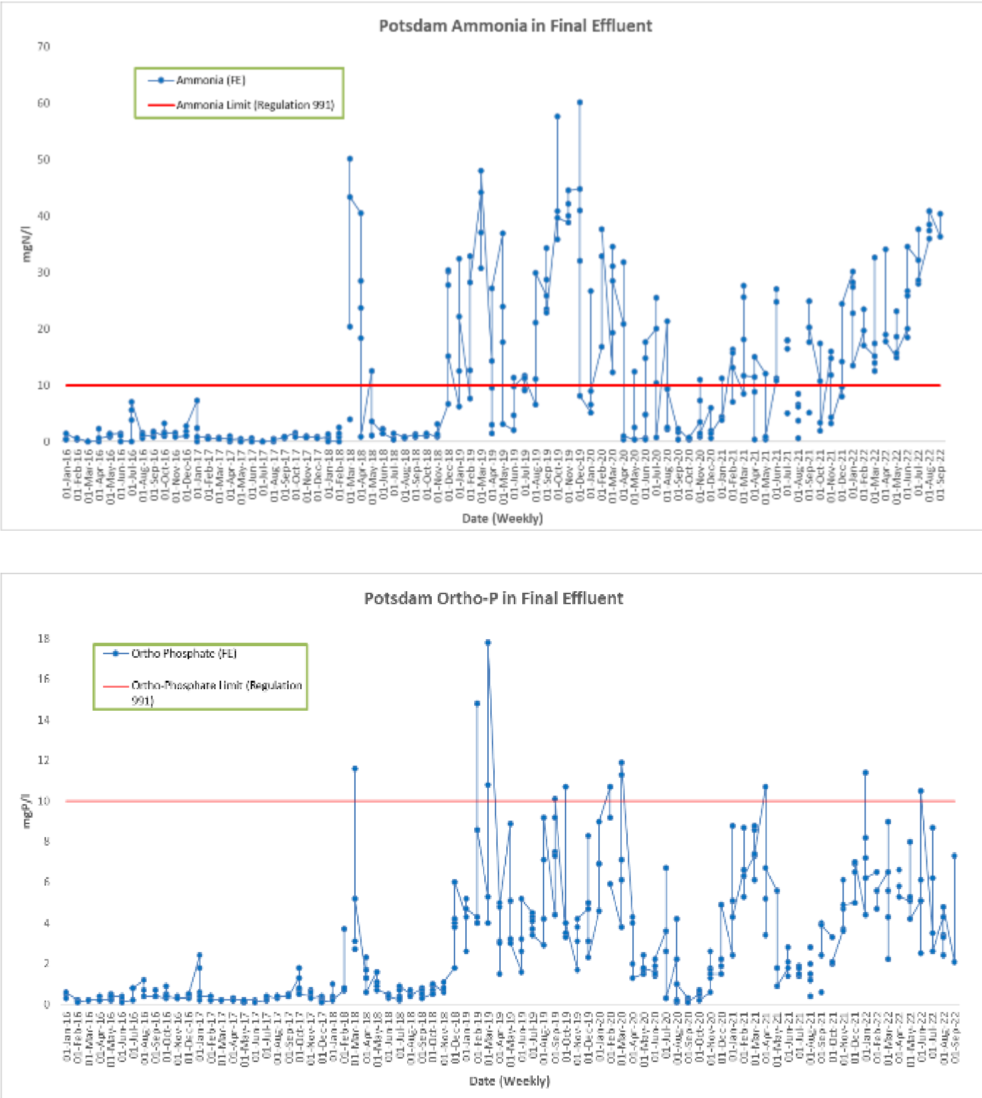
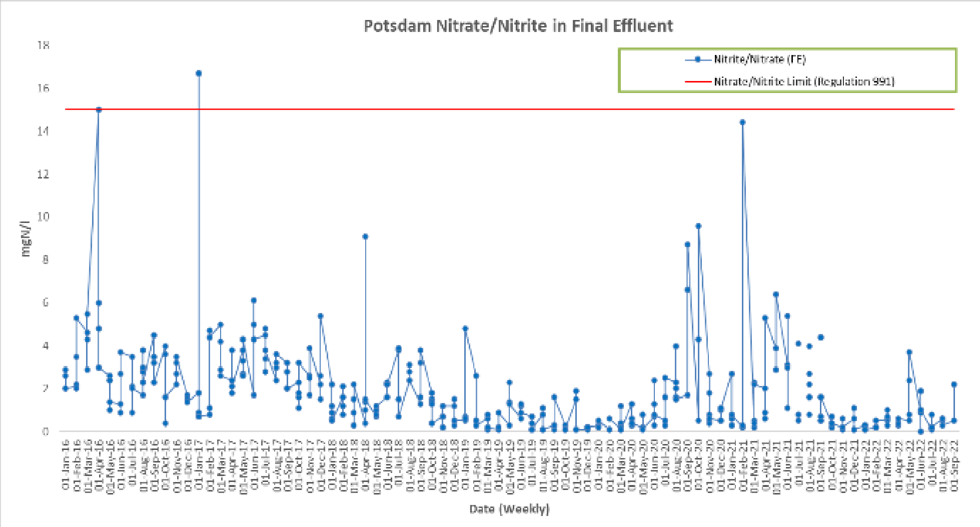


Figure 13-26. Potsdam final effluent regulation limits



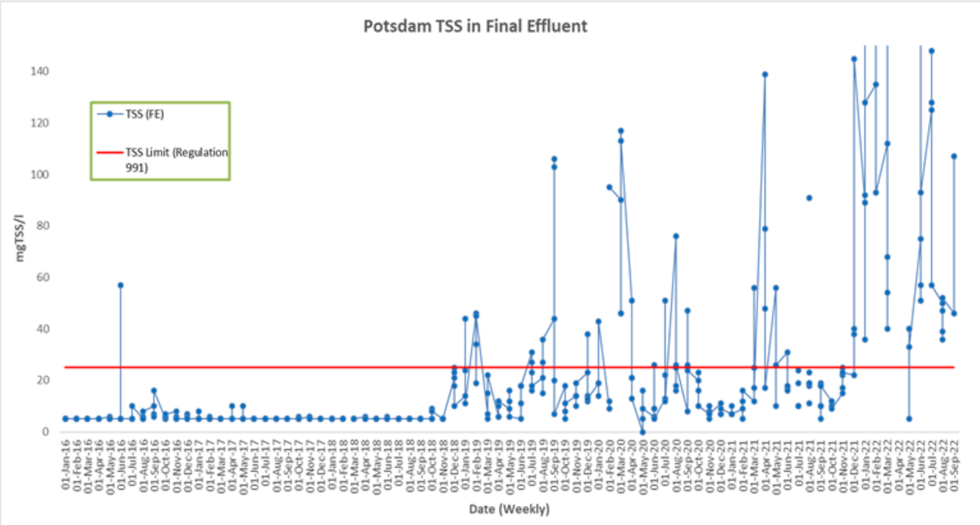


Figure 13-26 (cont.). Potsdam final effluent regulation limits

A new baseline scenario against which all other proposed scenarios would be compared was formulated by rerunning the model from 1 December 2018 to 31 August 2019, using the 1st month as a warmup period for the modelling. The following figure shows the tidal data and flows used for the simulation period.

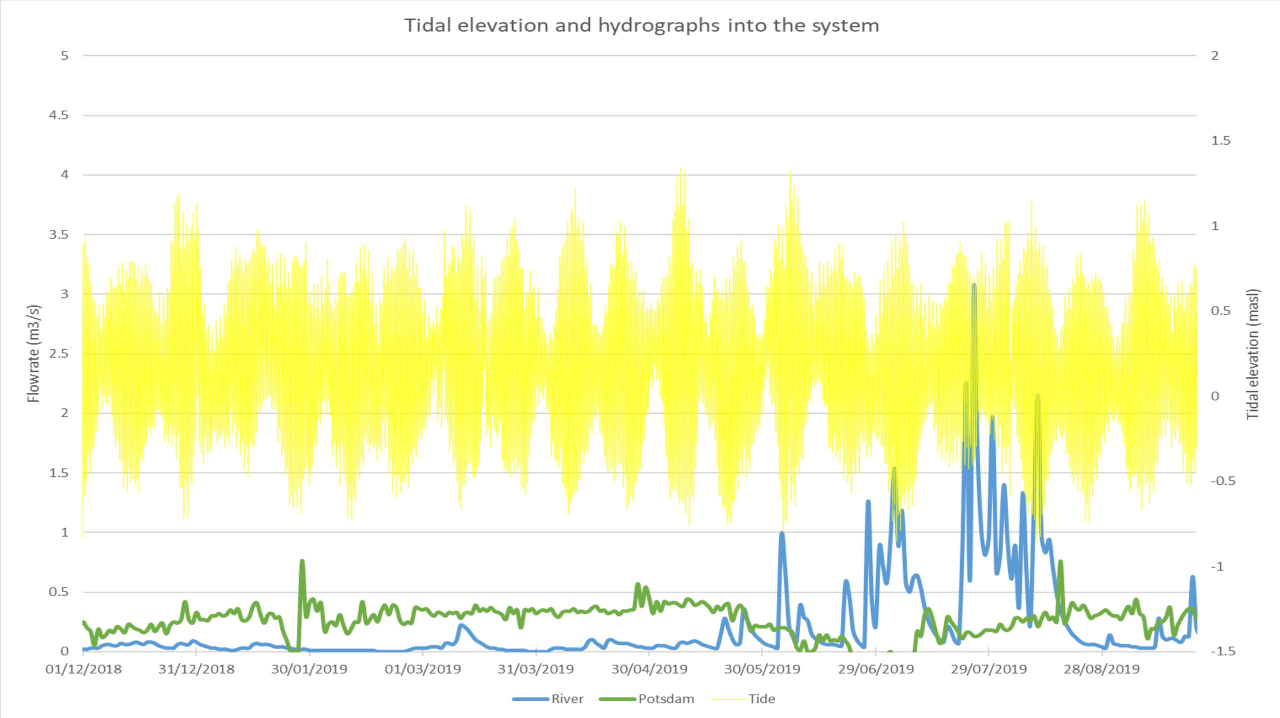


Figure 13-27. Tidal elevation and flows for the simulation period

The following figures show the water quality output for RTV05, with RTV09 and RTV10 in **Annexure G** for the new initial run.

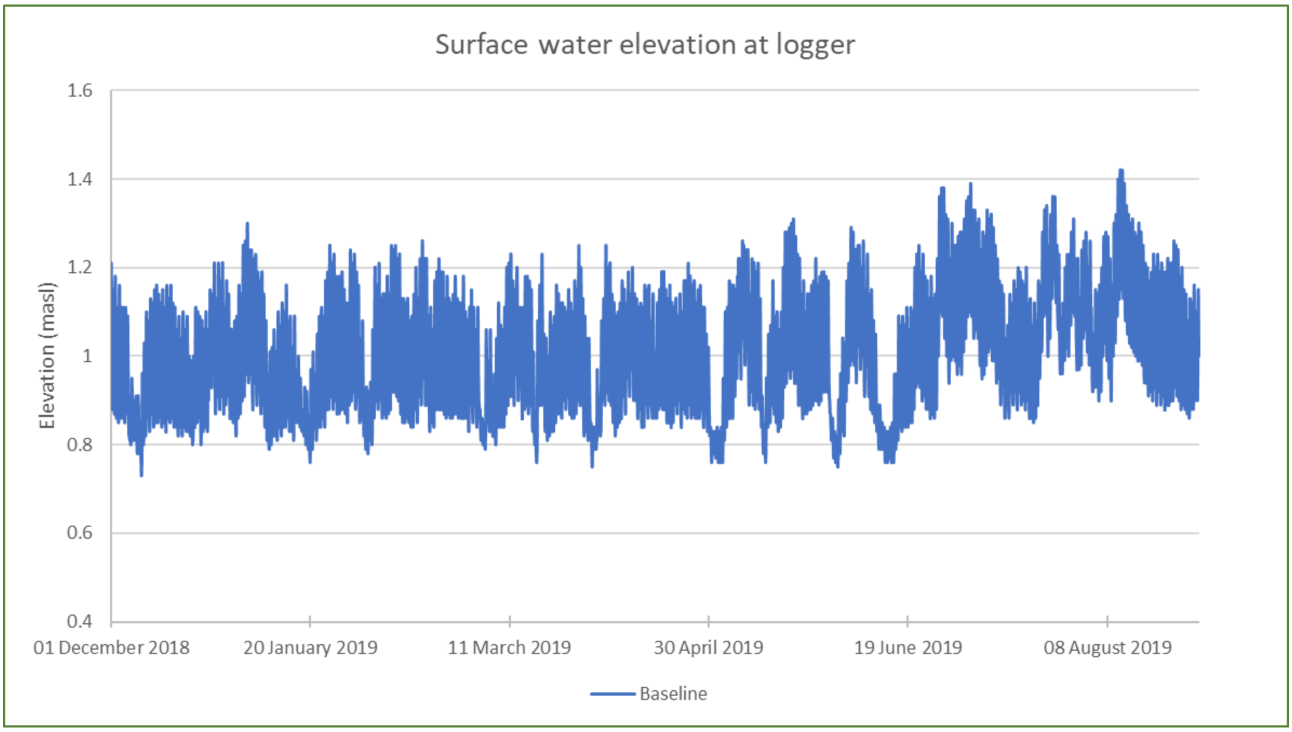


Figure 13-28. The simulated surface water elevation at the logger

Figure 13-29 shows good correlation between tides and river flows in that the elevation was influenced by higher tides and increased during periods of higher river flow. The following figure shows the water temperature for the time period.

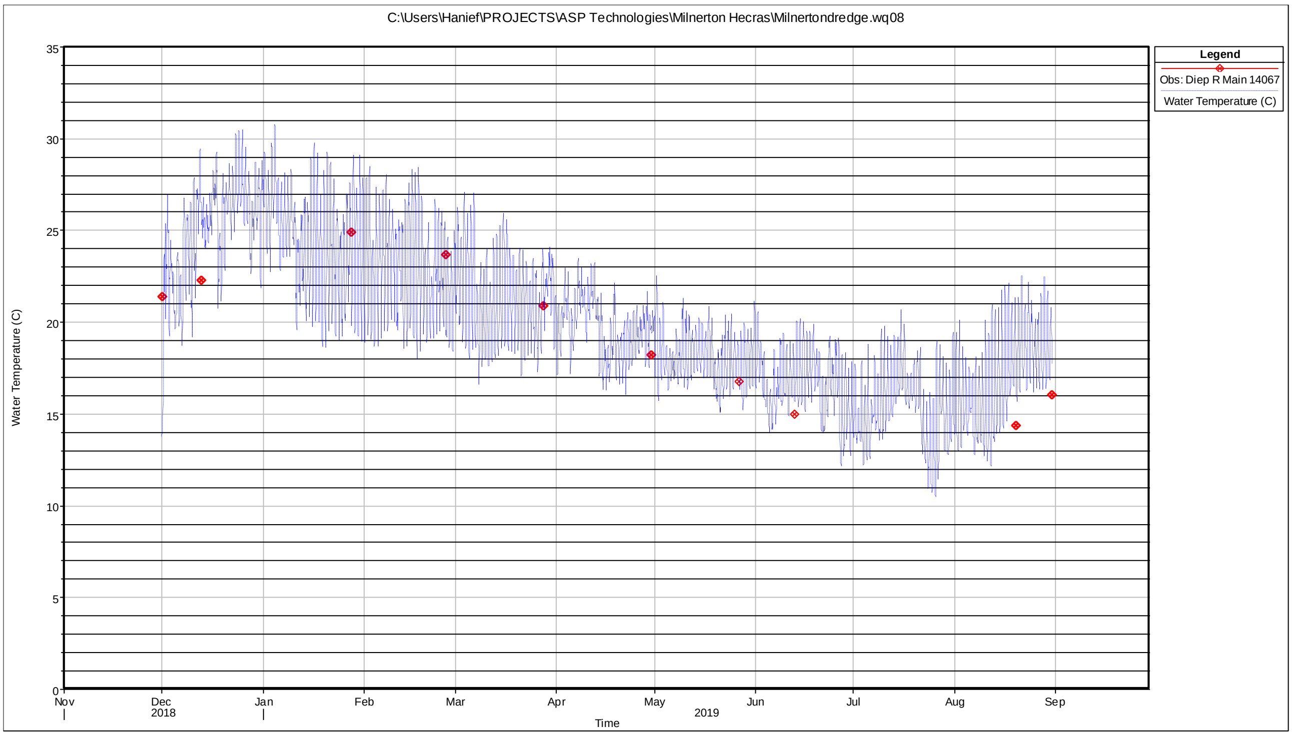


Figure 13-29. The water temperature basis for comparison

Figure 13-29 shows that the water temperature predictions spanned the sampled data for the time period modelled. The following figure shows the dissolved oxygen boundary conditions in the river for the simulation period.

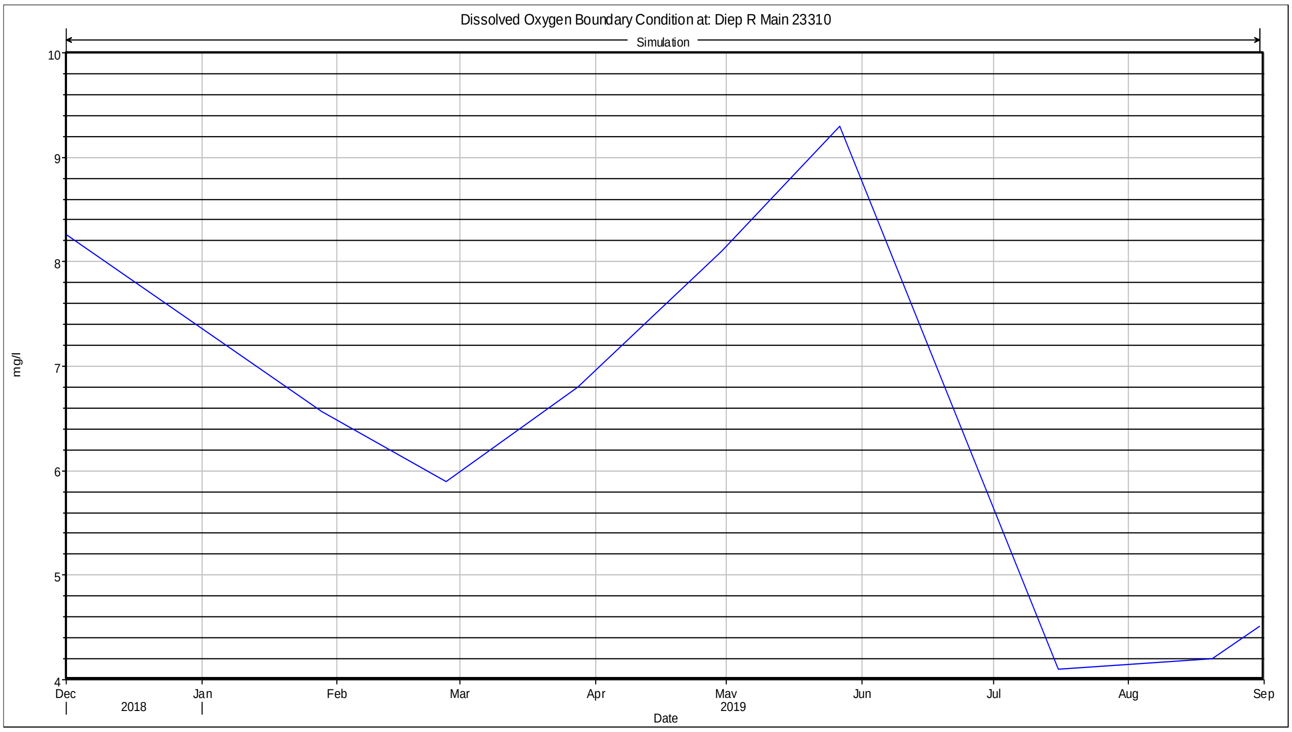


Figure 13-30. The dissolved oxygen concentrations for the baseline simulation

Figure 13-31 shows the dissolved oxygen concentrations of the Diep River, whilst that of the ocean and Potsdam discharge have been fixed due to a lack of sampled data. The following figure shows the baseline dissolved oxygen concentrations for the period.

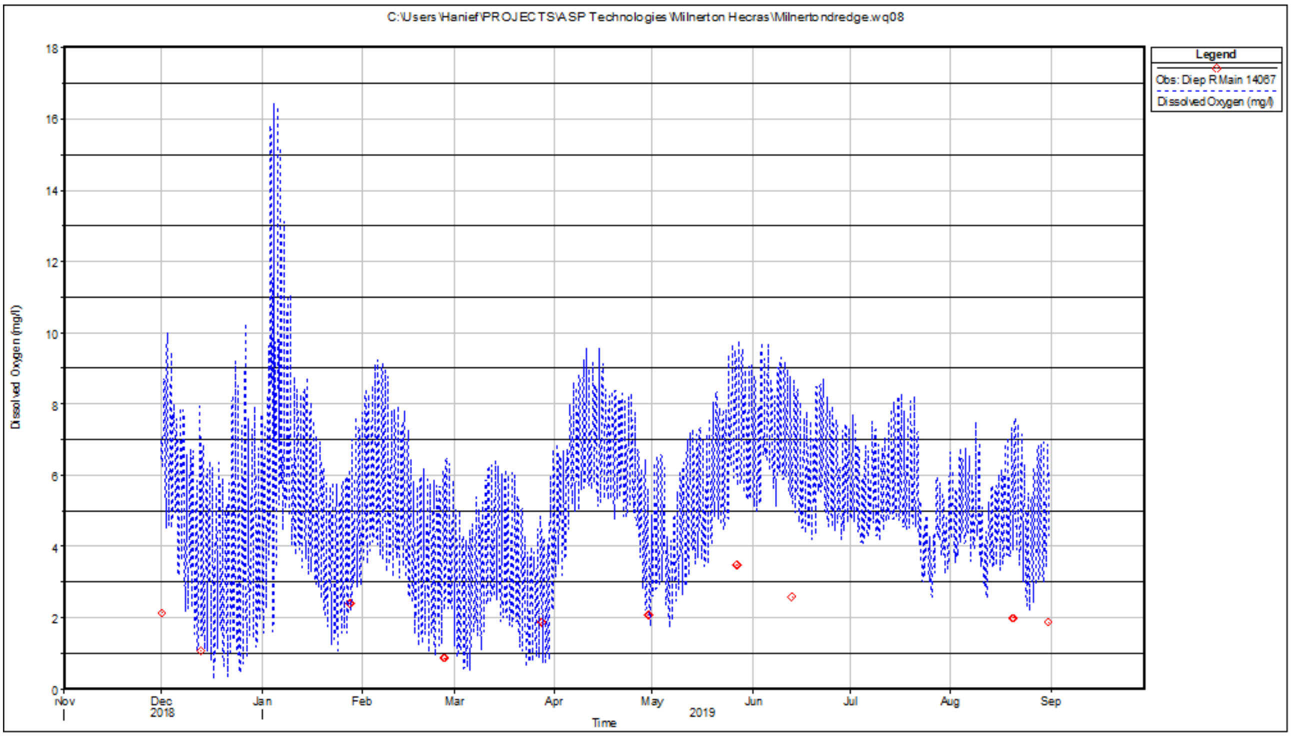


Figure 13-31. The dissolved oxygen basis for comparison

The model predicts the dissolved oxygen concentrations as well as a period of high saturation, a result of an algal bloom as shown in Figure 13-38. The model seems to overpredict the dissolved oxygen concentration in the latter period of the simulation but still followed the trend of the sampled data; this variance was attributed to possible changes in the SOD during the simulation period which the model does not capture. The Potsdam effluent data shows that effluent exceeding the licence limits was discharged from January 2019 onwards. These suspended solids would settle and contribute to increasing the SOD, resulting in lower real dissolved oxygen concentrations than the model can predict. The model used a single value for SOD for the entire modelling period which would elucidate the difference between the sampled data and the modelled dissolved oxygen concentrations. This overestimation is not considered a deficiency as all scenarios would show changes relative to the baseline.

The following figures show the nutrient boundary conditions and concentrations for the simulation period.

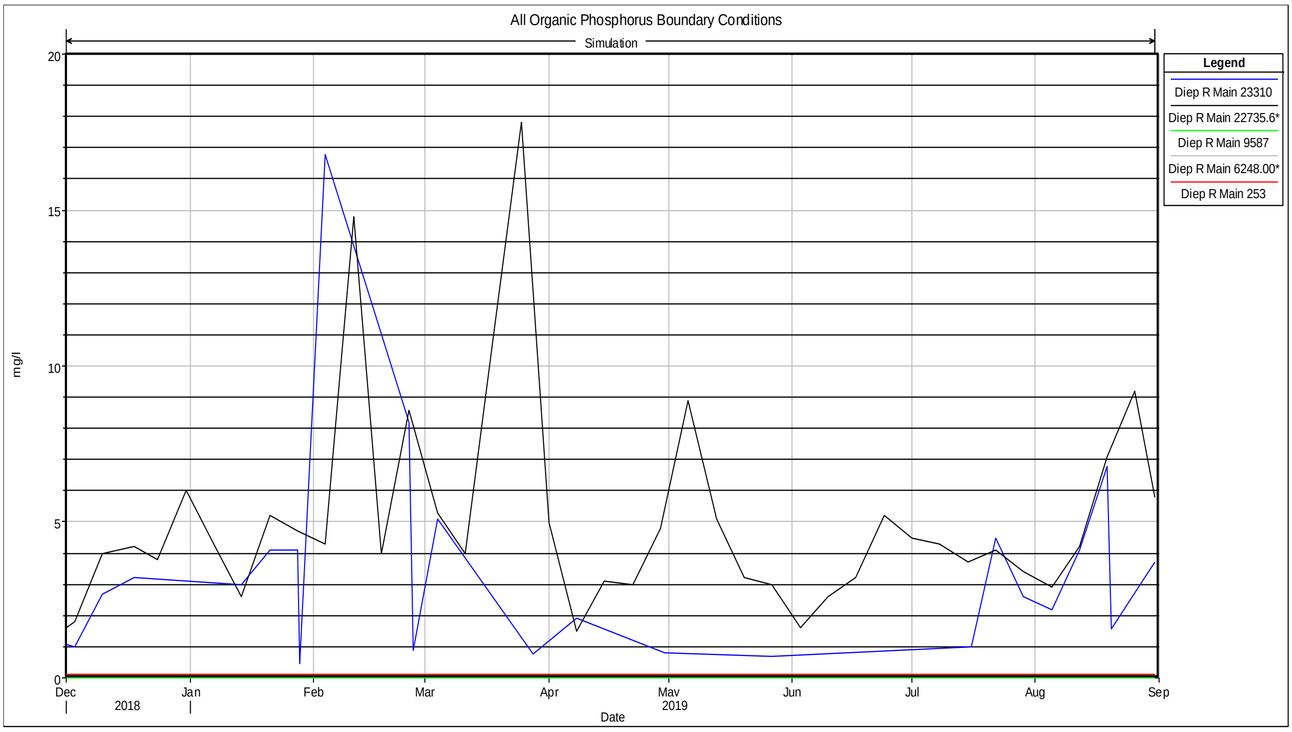


Figure 13-32. The ortho-phosphate boundary conditions

Figure 13-32 shows that on average the concentration of phosphates was higher in the effluent discharge than in the river. Figure 13-33 shows the modelled output.

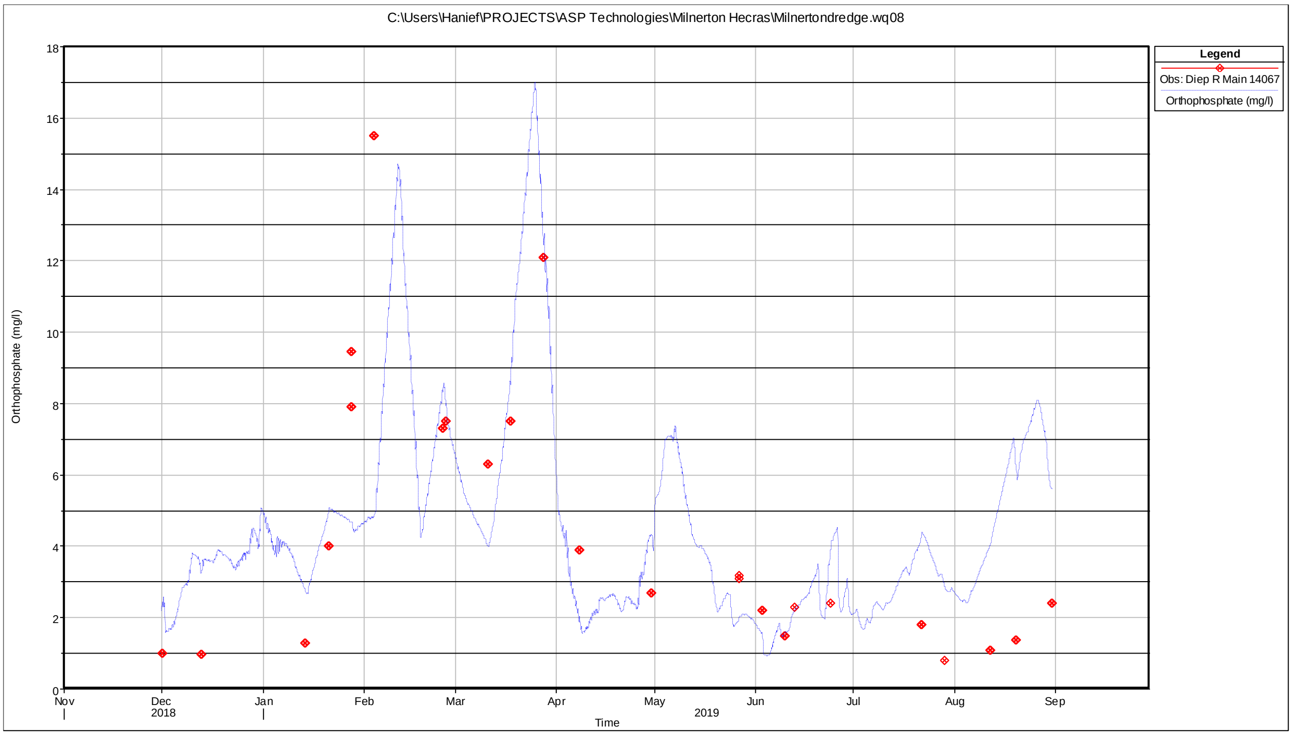


Figure 13-33. The ortho-phosphate concentrations basis for comparison

Figure 13-34 shows the ammonium boundary conditions for the time period. The figure shows that Potsdam discharged a higher concentration of ammonium than what was sampled in the river. Figure 13-35 shows the modelled ammonium concentration for the period.

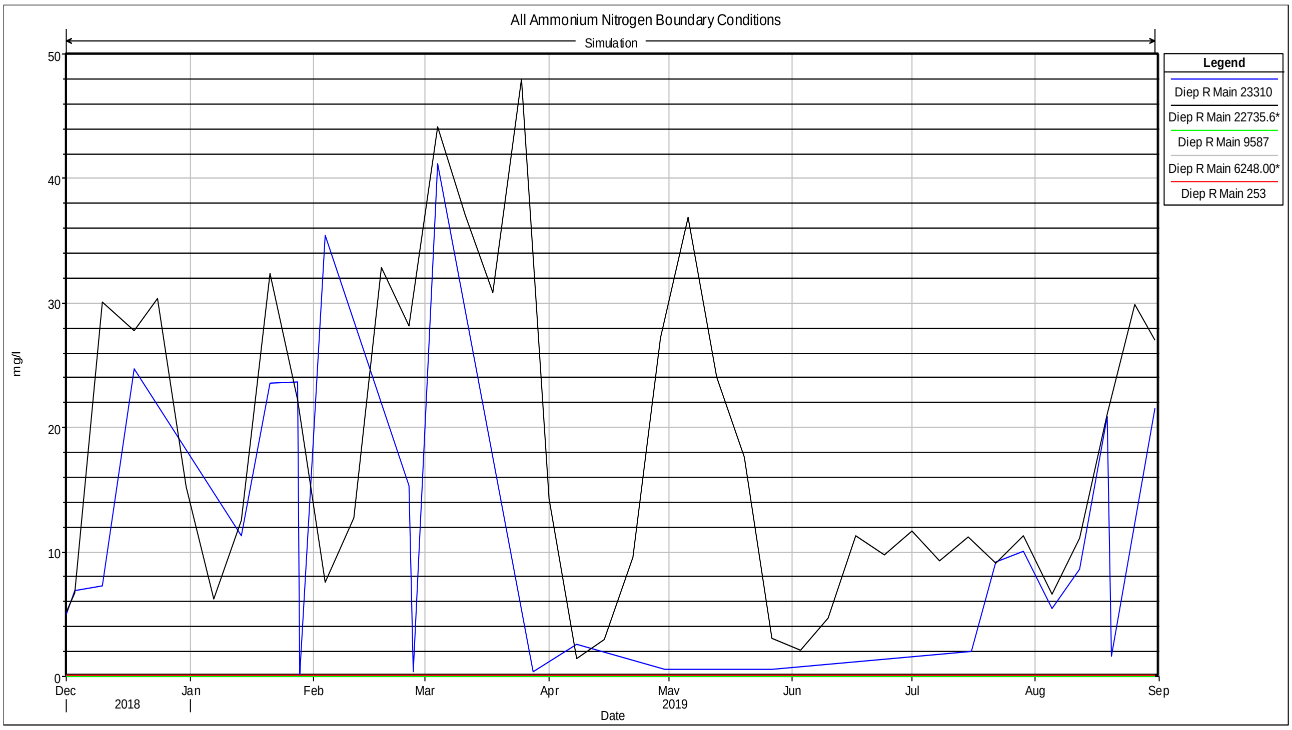


Figure 13-34. The ammonium boundary conditions

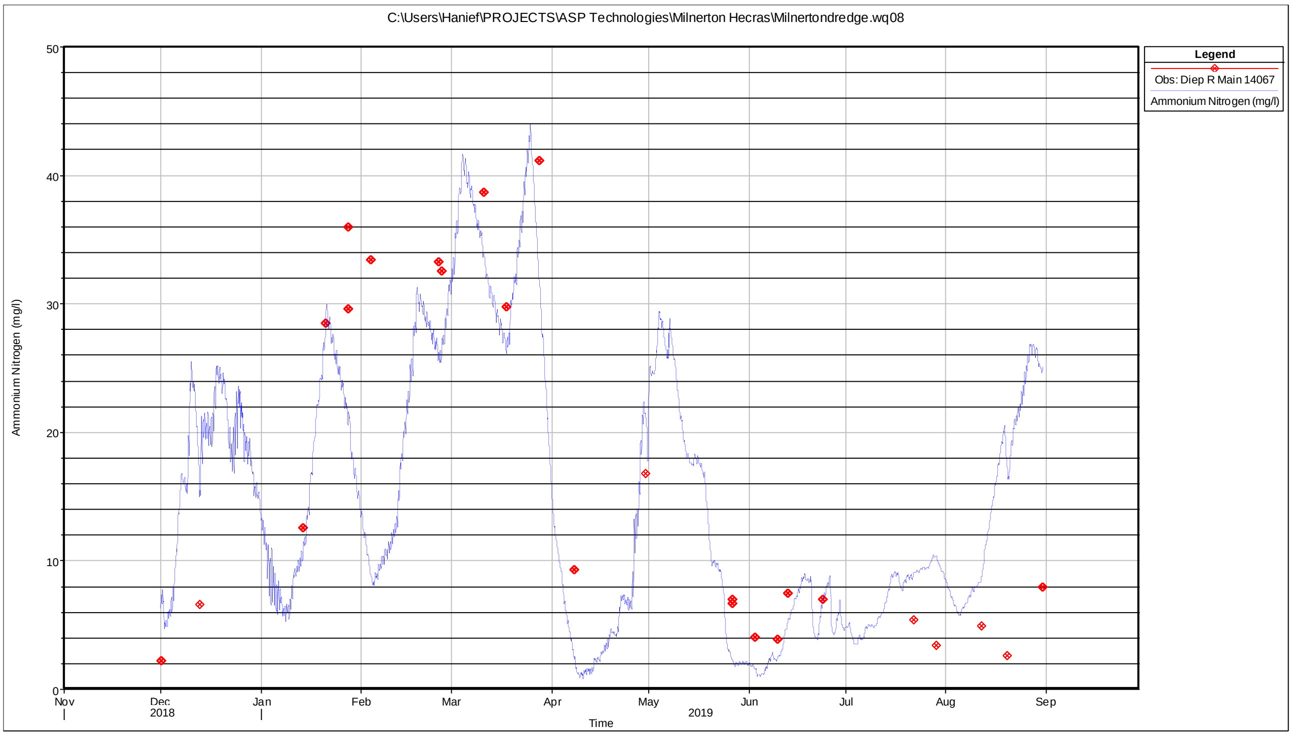


Figure 13-35. The ammonium concentration basis for comparison

Figure 13-36 shows the nitrogen concentration boundary conditions for the modelling period.

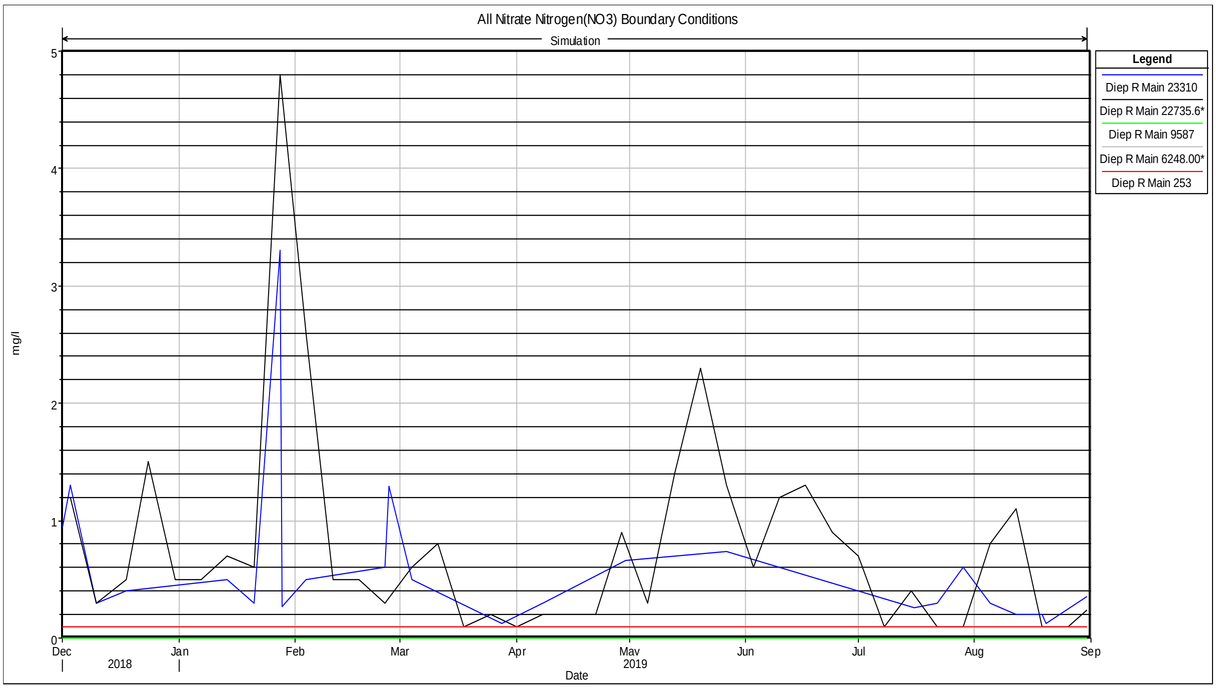


Figure 13-36. The nitrogen boundary conditions

Potsdam was the major contributor of nitrogen as it discharged a higher concentration of nitrogen than what was sampled in the river upstream. Figure 13-37 shows the nitrogen concentration for the new baseline modelling period.

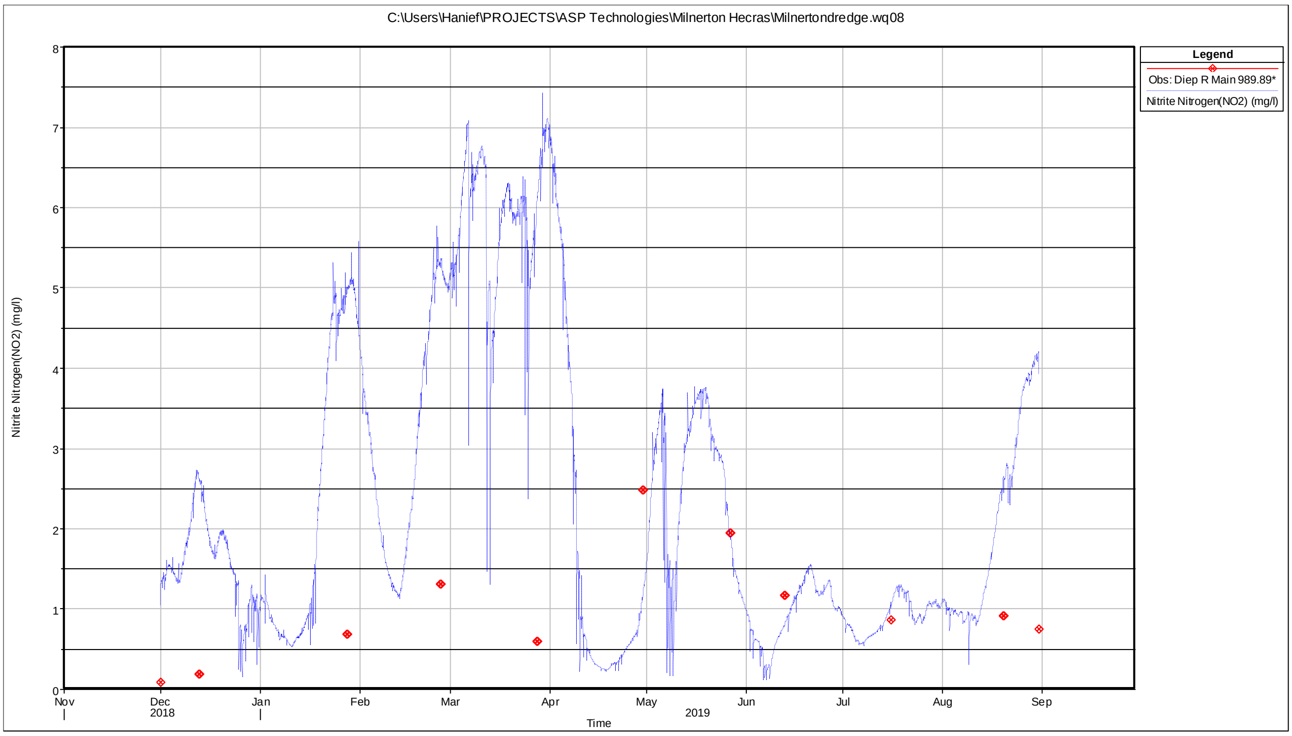


Figure 13-37. The nitrogen concentration basis for comparison

The nutrient concentration was well represented by the new baseline model. Figure 13-38 is the modelled algal concentration for the time period.

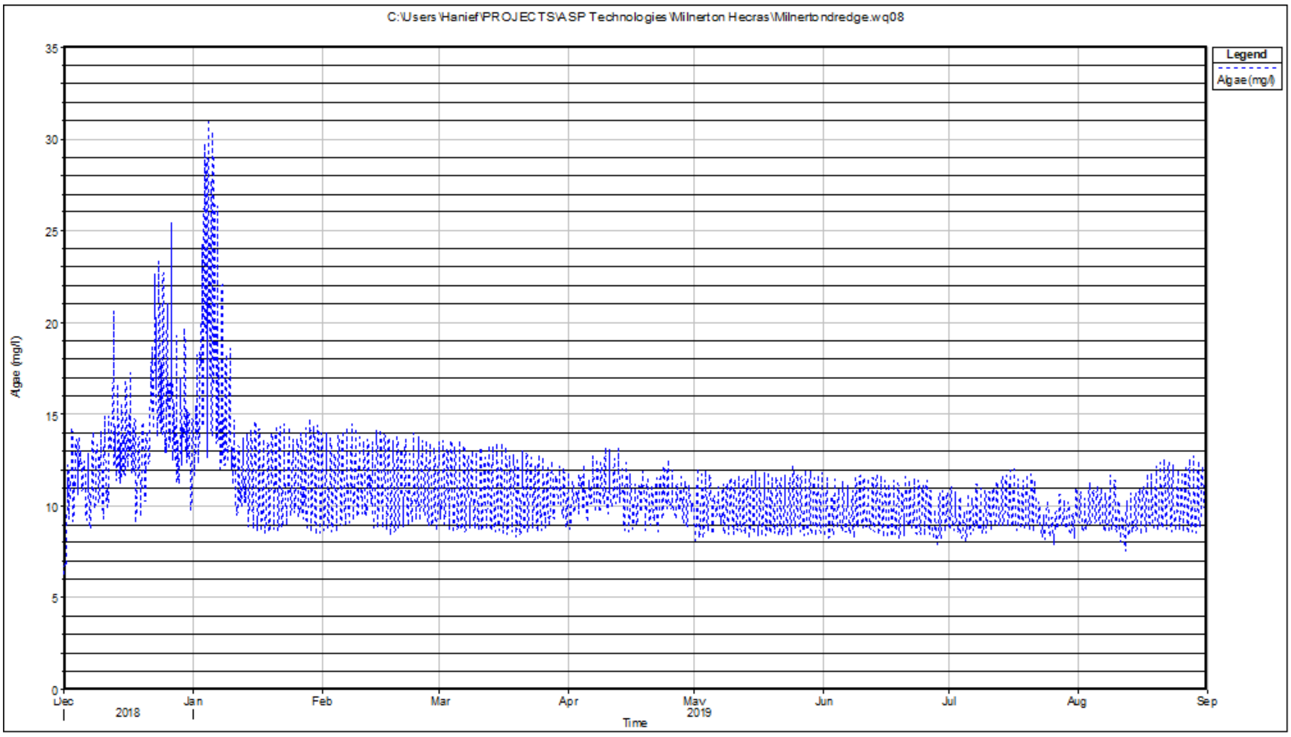


Figure 13-38. The algal concentration basis for comparison

The modelled algal bloom of January 2019 was the cause of the increased dissolved oxygen concentrations during this period. Without any sampled algal data the model cannot be adjusted for any greater accuracy and this parameter also affects all water quality constituents as shown in Figure 13-13.

Figure 13-39 shows the salinity for the time period.

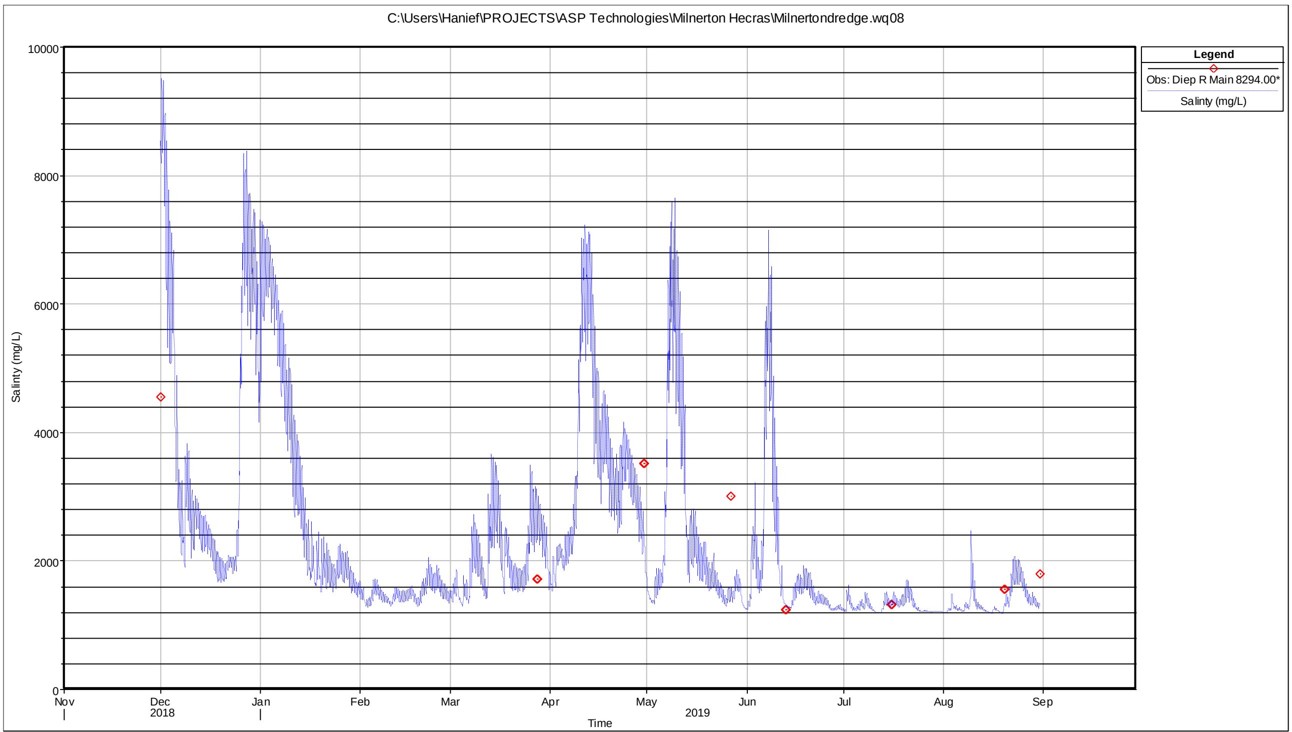


Figure 13-39. The salinity basis for comparison

The figure shows that salinity was low for the time period and decreased during periods of high river flows and increased Potsdam discharge as a result of conductivity loading.

The model was now considered to have predicted a baseline against which all proposed scenarios were evaluated. Any variances between sampled data and modelled output are transferred to the scenarios; these, when compared to the baseline scenario, provide **relative differences** allowing for assessment of their effects.

13.6 Modelling scenarios

The following scenarios to improve the water quality over the baseline were proposed. **Any modelled scenario compared to the original must be considered a relative change and not an absolute.**

13.6.1 Potsdam effluent at 40Ml licensed quality (the current scenario but with significant improvements in Potsdam effluent discharge quality)

This scenario proposed altering the Potsdam effluent discharge water quality to that of its license agreement and reporting on the subsequent water quality in the system. It was expected that this scenario would alter the nutrients in the system and subsequent algal production as well as dissolved oxygen concentrations but not affect the surface water elevations as the flowrates were to be unchanged from the baseline scenario. These limits are shown in the table below.

Table 13-4. The licensed agreement limits of water quality discharged from Potsdam WWTW

Parameter	Units	Limit
TSS Limit (Regulation 991)	mg/l	25
COD Limit (Regulation 991)	mg/l	75
COD Filtered Limit (Regulation 991)	mg/l	75
Ammonia Limit (Regulation 991)	mg N/l	10
Nitrate/Nitrite Limit (Regulation 991)	mg N/l	15
Ortho-Phosphate Limit (Regulation 991)	mg P/l	10
Ph- lower limit (Regulation 991)		5.5
Ph- upper limit (Regulation 991)		9.5
Conductivity Limit (Regulation 991)	mS/m	250
Chloride Limit (Regulation 991)	mg/l	200
Alkalinity Limit (Regulation 991)	mg CaCO3/l	-
Residual Chlorine Limit (Regulation 991)	mg/l	0.1
E.coli Limit (Regulation 991)	per 100ml	1000

13.6.2 Potsdam effluent at 100Ml licensed quality (the future upgrade current scenario and discharged at licensed agreement)

This scenario proposed altering Potsdam to 100Ml capacity and the effluent discharge water quality to that of its license agreement and reporting on the subsequent water quality in the system. It was expected that this scenario would lower salinity and increase water elevations.

13.6.3 Potsdam effluent at 100Ml current quality (the anticipated future maximum discharge volume but without significant quality improvements)

This scenario proposed increasing the Potsdam WWTW treatment capacity to 100Ml per day whilst discharging effluent at the current poor water quality and reporting on the subsequent water quality in the system. It is expected that the surface water elevations would increase as well as deteriorating water quality over the baseline scenario.

13.6.4 Modelling the effect of the proposed dredging (Option 1)

This scenario proposed the dredging of poor-quality sediments in the river bed and reporting on the subsequent water quality in the system. The two proposed phased dredge areas are shown in Figure 13-41. The proposed dredging is to be phased with the red area as phase 1 and the blue area as phase 2. Sediment oxygen demand (SOD) is linked to the decomposition of organic matter in the sediments, which produces methane, carbon dioxide and hydrogen sulphide resulting in an increased consumption of dissolved oxygen present in the overlying water column. The removal of poor-quality sediments would be expected to reduce the sediment oxygen demand, improve dissolved oxygen concentrations and remove any contaminants associated with the dredged sediments. The proposed dredging of the current bathymetry is shown in Figure 13-40 (bottom) down to -1 masl.

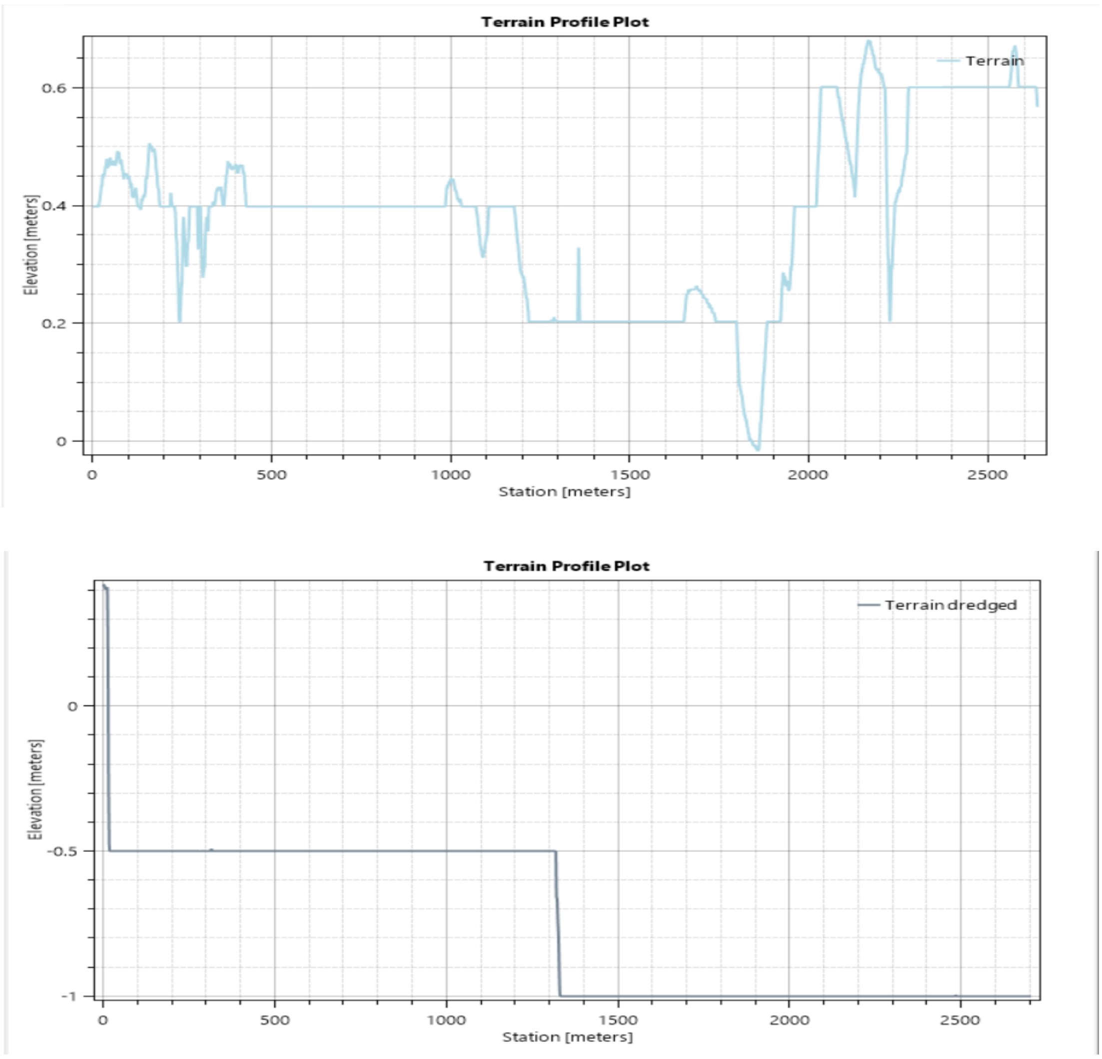


Figure 13-40. Comparison between the baseline (top) and dredged (bottom) bathymetric profiles

The dredging would remove the poor-quality sediment as well as decrease the SOD on the water column, thereby improving the dissolved oxygen concentration. Rong et al (2016) found that even when reducing all sources of pollution and improving water quality but not removing the contaminated sediments, the SOD greatly affected the dissolved oxygen concentrations in the water column.

13.6.5 Modelling additional aeration (Option 2)

This scenario proposes aerating the system to add dissolved oxygen to the water at the theoretical maximum rate of 5kg/h as outlined in section 6, in the vicinity of location A. It is expected that dissolved oxygen concentrations in the lagoon would improve above anoxic conditions.

13.6.6 Modelling seawater flushing (Option 3)

This scenario proposed adding seawater into the system to improve the dissolved oxygen concentrations as well as increase the salinity. Three scenarios for each intervention were modelled for 2 locations viz:

1. Seawater injection at 0.1 m^3/s
2. Seawater injection at 0.2 m^3/s
3. Seawater injection at Potsdam discharge rates

The proposed locations are at -33.86737, 18.49308 (A in Figure 13-42) and -33.87709, 18.48977 (B in in Figure 13-42, downstream of A) and it was expected that the system would experience increased surface water elevations, increased salinity and dissolved oxygen concentrations as well as decreased nutrients and subsequent algal growth.

13.6.7 Modelling Potsdam discharge directly to ocean (Option 4)

This scenario proposed redirecting the Potsdam effluent discharge to a marine outfall and reporting on the final water quality in the lower reach of the river. The elimination of this poor quality water from the system should improve the overall conditions in the system. See **Annexure G** for figure of comparison of flows.

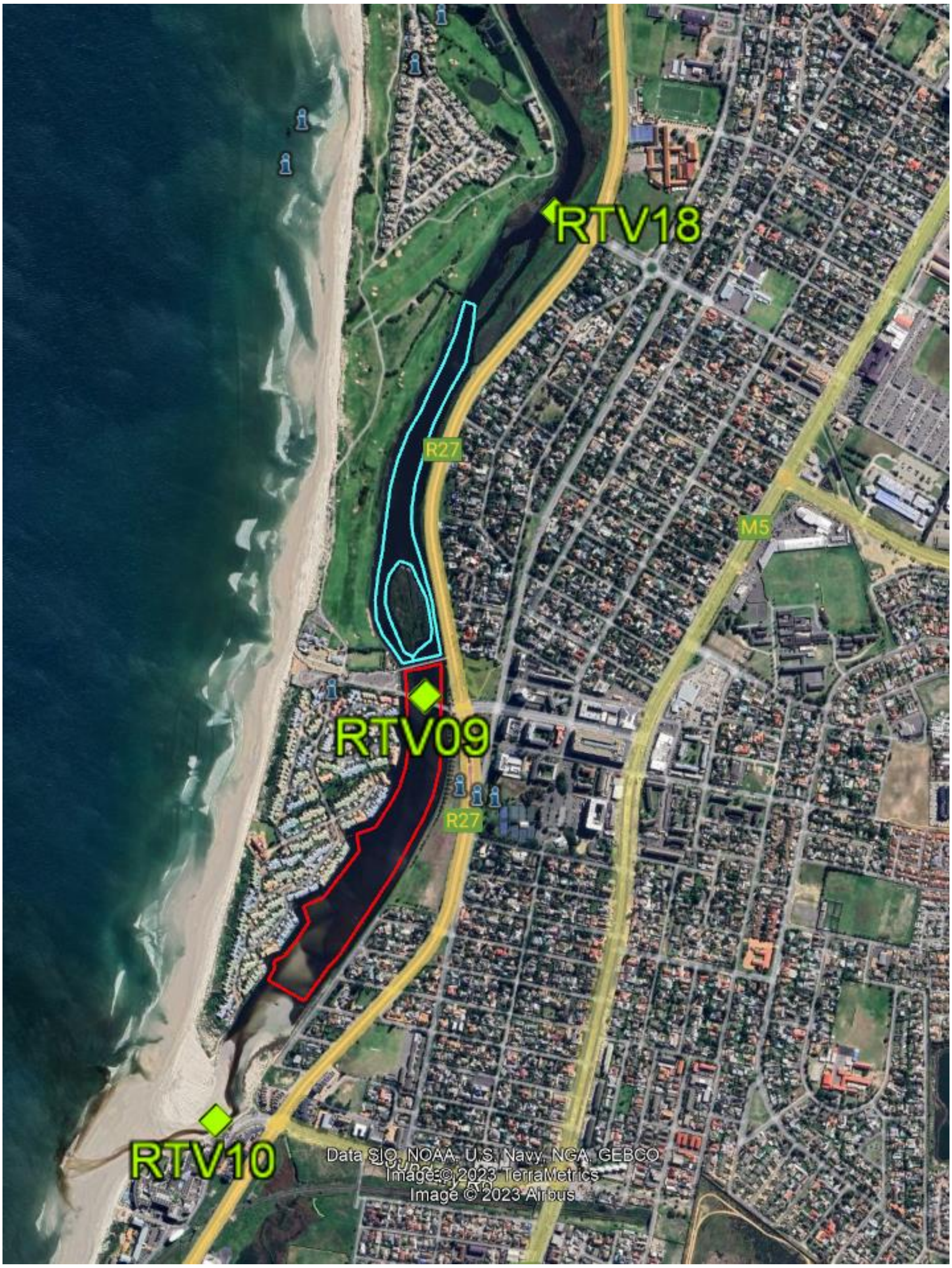


Figure 13-41. The proposed dredging at the mouth area

Figure 13-42. The proposed seawater injection sites

13.7 Results of the modelling scenarios

13.7.1 The water surface elevations

The surface water elevations for all scenarios are all recorded at RTV05 which is analogous to the logger and shown in Figure 13-43 and the statistics in Table 13-5.

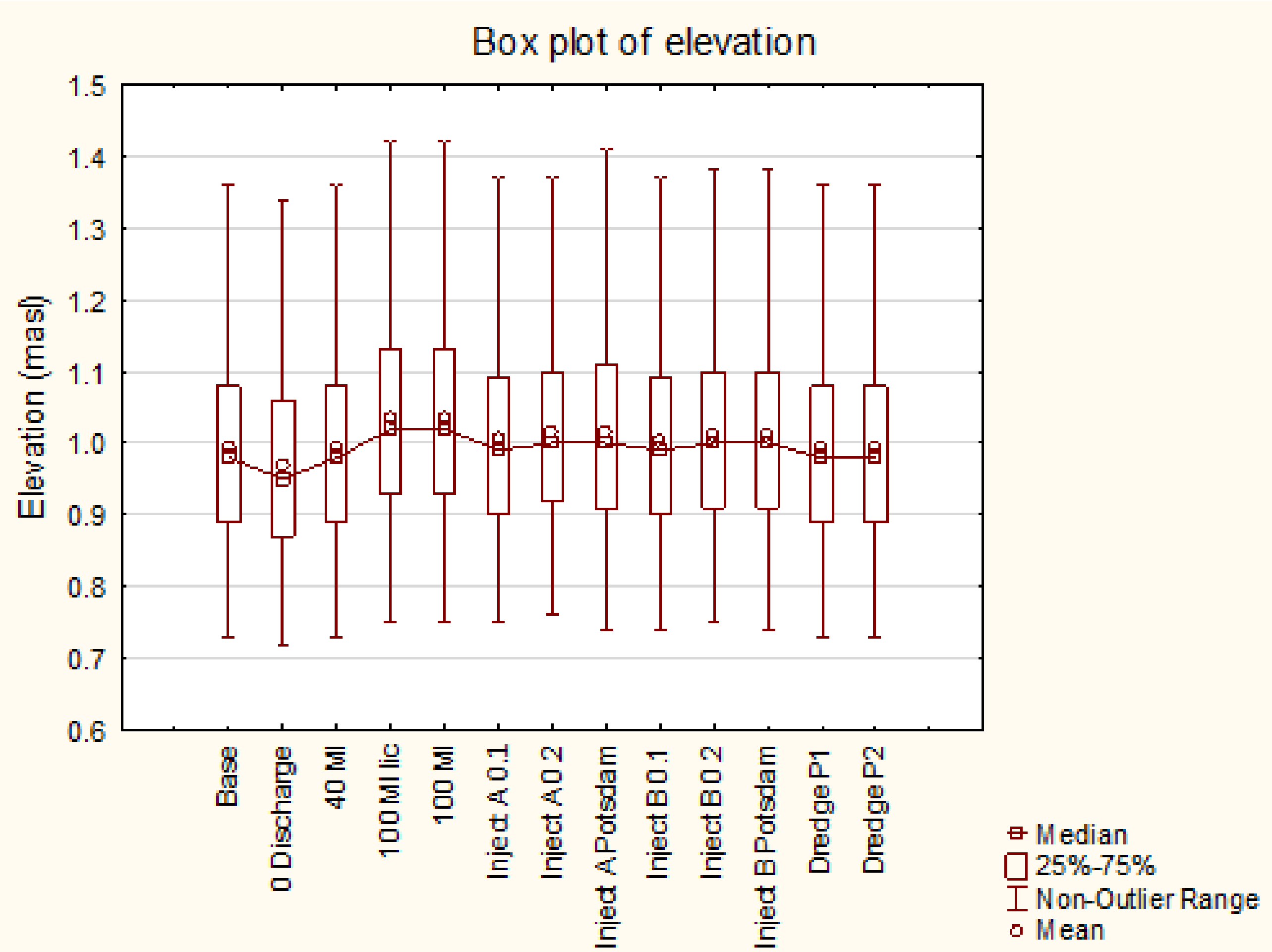


Figure 13-43. Box and whisker plot of elevations for all the scenarios

Table 13-5. Descriptive statistics for water elevation

Scenario	Descriptive Statistics							
	N	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	8161	0.99	0.98	0.73	1.42	0.89	1.08	0.12
Option 4: Zero discharge	8161	0.97	0.95	0.72	1.40	0.87	1.06	0.12
40 Ml discharge at licence quality	8161	0.99	0.98	0.73	1.42	0.89	1.08	0.12
100 Ml discharge at licence quality	8161	1.03	1.02	0.75	1.47	0.93	1.13	0.13
100 Ml discharge at current poor quality	8161	1.03	1.02	0.75	1.47	0.93	1.13	0.13
Option 3: Seawater addition 0.1 m³/s at Point A	8161	1.01	0.99	0.75	1.43	0.90	1.09	0.12
Option 3: Seawater addition 0.2 m³/s at Point A	8161	1.02	1.00	0.76	1.44	0.92	1.10	0.12

Scenario	Descriptive Statistics							
	N	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Option 3: Seawater addition matching Potsdam discharge at Point A	8161	1.02	1.00	0.74	1.44	0.91	1.11	0.13
Option 3: Seawater addition 0.1 m^3/s at Point B	8161	1.00	0.99	0.74	1.43	0.90	1.09	0.12
Option 3: Seawater addition 0.2 m^3/s at Point B	8161	1.01	1.00	0.75	1.44	0.91	1.10	0.12
Option 3: Seawater addition matching Potsdam discharge at Point B	8161	1.01	1.00	0.74	1.44	0.91	1.10	0.13
Option 1: Dredging (below Woodbridge only)	8161	0.99	0.98	0.73	1.42	0.89	1.08	0.12
Option 1: Dredging (whole lagoon)	8161	0.99	0.98	0.73	1.42	0.89	1.08	0.12

Inject = injecting sea water at 0.1 m^3/s or 0.2 m^3/s or at Potsdam discharge rates

The figure and table show that any additional inflows to the system over the baseline resulted in increased water elevations on the base scenario, whilst diversions decreased the surface water elevations. The dredging of the system did not influence the water elevations as the mouth area acts as a weir that retains the water elevations.

The model outputs water quality for the entire bathymetry but for the following sections the output is presented only for RTV10, the main study area. The average depth at this point is approximately 1 m.

13.7.2 The effect on water temperatures

The model was rerun for all the scenarios and the data collated. Figure 13-44 and Table 13-6 show the changes in the modelled water temperatures from the base scenario.

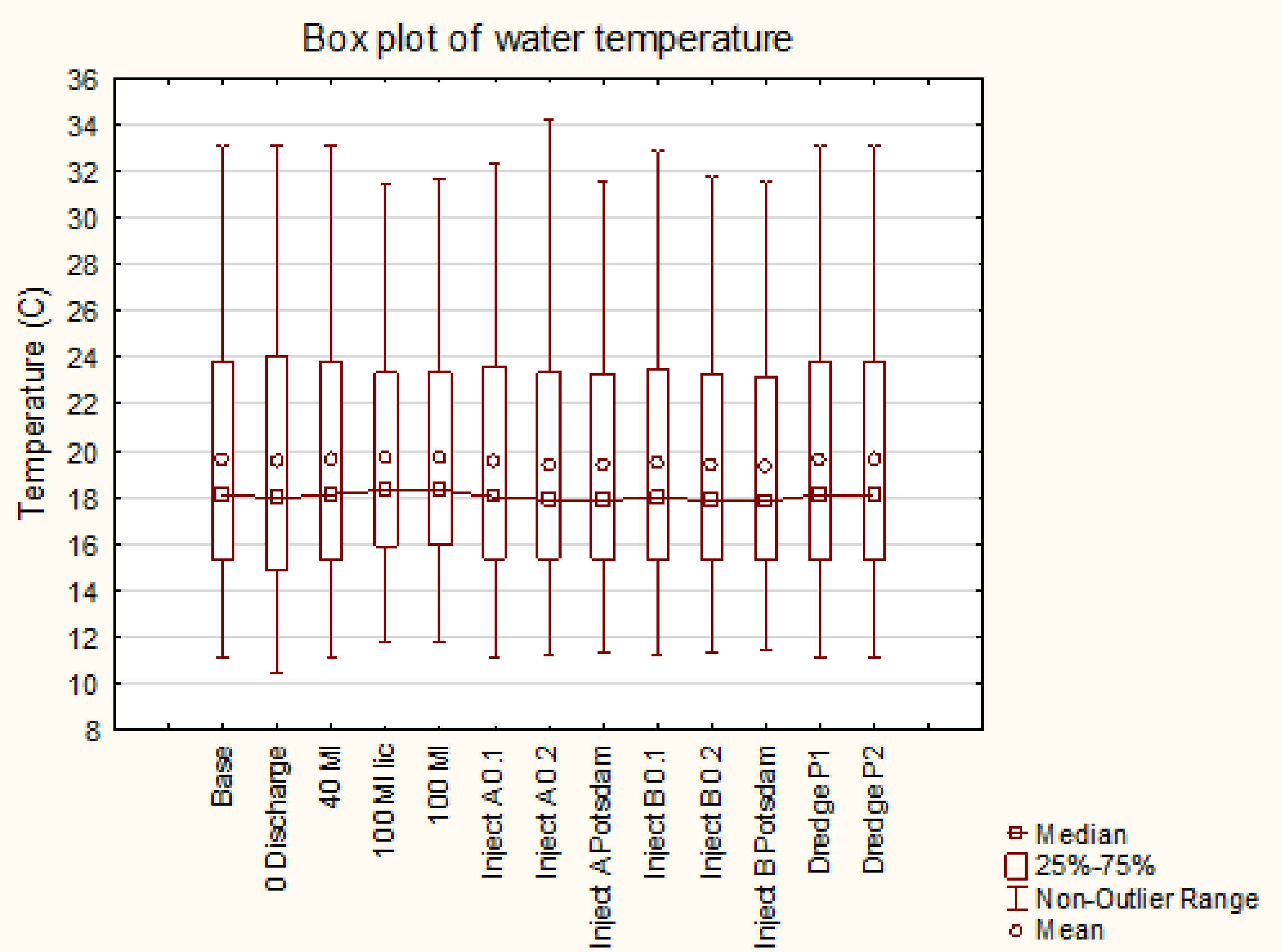


Figure 13-44. The box and whisker plot of water temperature

Table 13-6. Descriptive statistics for water temperature

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	19.6	18.1	11.1	33.1	15.3	23.8	5.2
Option 4: Zero discharge	19.5	18.0	10.4	33.1	14.9	24.0	5.4
40 Ml discharge at licence quality	19.6	18.1	11.1	33.1	15.3	23.8	5.2
100 Ml discharge at licence quality	19.7	18.3	11.8	31.5	15.9	23.3	4.6
100 Ml discharge at current poor quality	19.7	18.3	11.8	31.7	15.9	23.4	4.6
Option 3: Seawater addition 0.1 m³/s at Point A	19.5	18.0	11.1	32.4	15.4	23.6	5.0
Option 3: Seawater addition 0.2 m³/s at Point A	19.4	17.9	11.3	34.2	15.4	23.4	4.9
Option 3: Seawater addition matching Potsdam discharge at Point A	19.4	17.9	11.3	31.6	15.3	23.3	4.9
Option 3: Seawater addition 0.1 m³/s at Point B	19.5	18.0	11.2	32.9	15.4	23.5	5.0

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Option 3: Seawater addition 0.2 m³/s at Point B	19.4	17.9	11.3	31.8	15.4	23.2	4.8
Option 3: Seawater addition matching Potsdam discharge at Point B	19.3	17.8	11.4	31.9	15.3	23.2	4.8
Option 1: Dredging (below Woodbridge only)	19.6	18.1	11.1	33.1	15.3	23.8	5.2
Option 1: Dredging (whole lagoon)	19.6	18.1	11.1	33.1	15.3	23.8	5.2

Inject = injecting sea water at 0.1 m³/s or 0.2 m³/s or at Potsdam discharge rates

These results showed that scenarios did not alter the water temperatures significantly but that with injecting seawater into the system there was a cooling effect on water temperatures. The scenario where Potsdam discharged directly to the ocean allowed the area to reach its minimum temperature as there was maximum penetration of cooler oceanic waters.

13.7.3 Dissolved oxygen concentrations

Figure 13-45 and Table 13-7 show the modelled changes in dissolved oxygen concentrations from the baseline scenario.

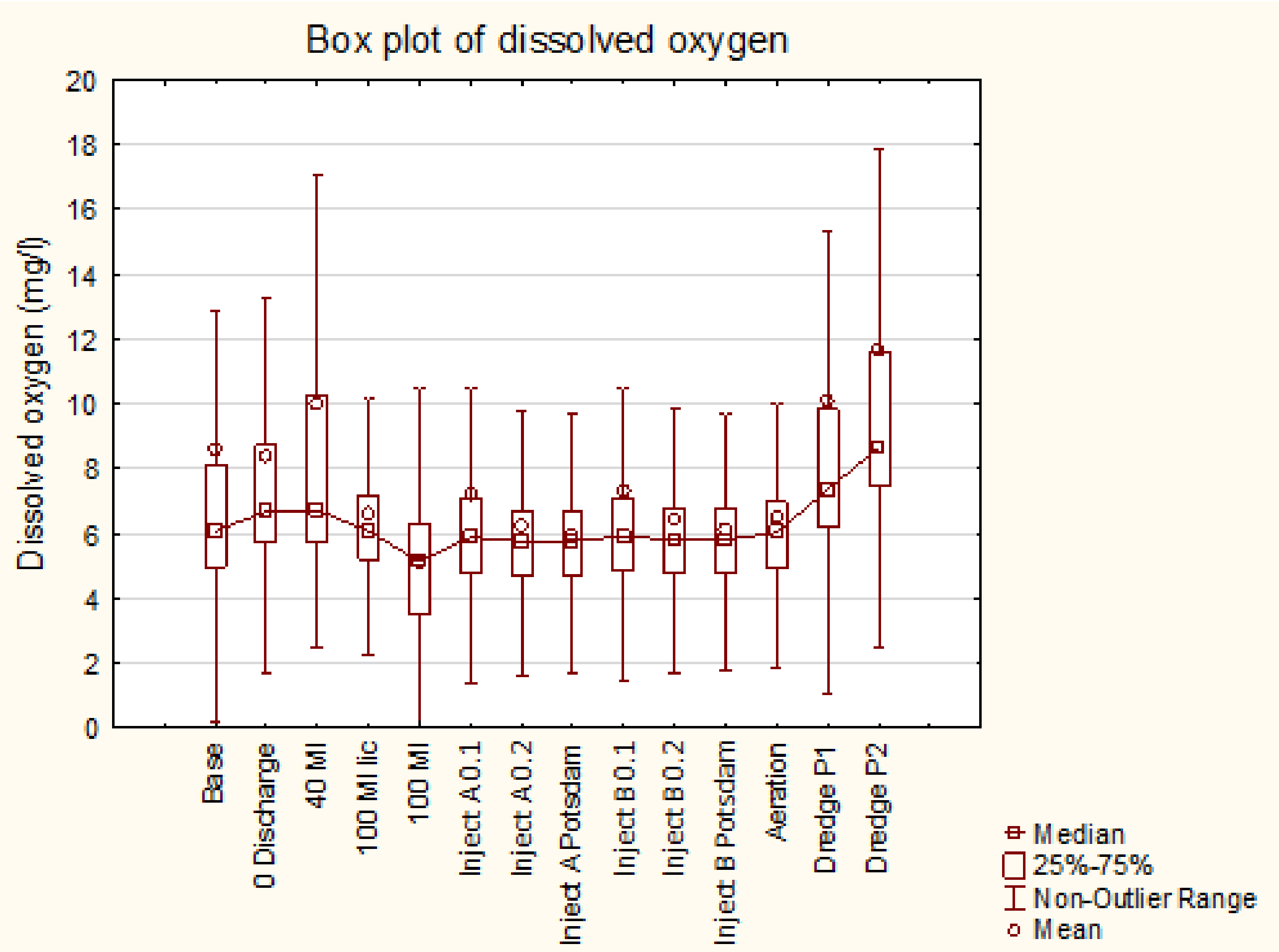


Figure 13-45. The box and whisker plot for dissolved oxygen concentrations

Table 13-7. Descriptive statistics for dissolved oxygen concentrations (mg/l)

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	8.6	6.1	0	23.6	5	8.1	6.8
Option 4: Zero discharge	8.4	6.7	0.8	22.0	5.7	8.7	4.6
40 Ml discharge at licence quality	10.0	6.7	2.5	24.7	5.7	10.3	7.2
100 Ml discharge at licence quality	6.6	6.1	2	17.2	5.2	7.2	2.3
100 Ml discharge at current poor quality	5.1	5.1	0	11.1	3.6	6.3	2.5
Option 3: Seawater addition 0.1 m³/s at Point A	7.2	5.9	0	17.7	4.8	7.1	4.6
Option 3: Seawater addition 0.2 m³/s at Point A	6.2	5.8	0.3	13.3	4.7	6.7	3.0
Option 3: Seawater addition matching Potsdam discharge at Point A	6.0	5.8	0.2	12.5	4.7	6.7	2.4
Option 3: Seawater addition 0.1 m³/s at Point B	7.3	5.9	0	18.4	4.8	7.1	4.8
Option 3: Seawater addition 0.2 m³/s at Point B	6.4	5.8	0	13.9	4.8	6.8	3.2
Option 3: Seawater addition matching Potsdam discharge at Point B	6.1	5.8	0.3	11.6	4.8	6.7	2.6
Option 2: Aeration of lagoon	7.3	6.0	0.1	17.8	4.9	7.2	4.6
Option 1: Dredging (below Woodbridge only)	10.1	7.4	1.1	25.2	6.2	9.8	7.1
Option 1: Dredging (whole lagoon)	11.6	8.7	2.5	26.8	7.4	11.6	7.4

A minimum of 0 mg/l of dissolved oxygen was modelled for the baseline scenario as well as for the modelled discharge of 100Ml of poor-quality effluent from Potsdam. The injection of seawater at location A yielded better dissolved oxygen concentrations than at B with flowrates equal to that of Potsdam discharge outperforming the lower flowrates of 0.1 and 0.2 m³/s. Aeration of the river increased the minimum dissolved oxygen concentration above the baseline scenario but in general dredging showed the greatest improvements to dissolved oxygen.

The modelling showed that for dissolved oxygen the most beneficial intervention was dredging, followed by ensuring good quality discharge effluent from Potsdam and then diversion of effluent to a marine outfall to seawater. Ranking the improvements in terms of minimum values and lower quartile over the baseline scenario for the RTV10 location resulted in the following:

1. Phase 2 dredging
2. 40 Ml good quality effluent
3. 100 Ml good quality effluent
4. Phase 1 dredging only
5. Potsdam diversion to marine outfall

6. Inject seawater at B at Potsdam discharge rates
7. Inject seawater at A at Potsdam discharge rates
8. Inject seawater at A at 0.2 m^3/s
9. Aeration (only improved the minimum dissolved oxygen concentration)
10. Inject seawater at B at 0.2 $m^3/$
11. Injecting seawater at A or B at 0.1 m^3/s
12. 100 Ml of poor effluent discharge from Potsdam resulted in the lowest overall dissolved oxygen concentrations.

These results showed that dredging was most beneficial for improving the dissolved oxygen concentrations. It should be noted that these results are for RTV10.

13.7.4 Ortho-phosphates

Figure 13-46 and Table 13-8 are representative of the changes in ortho-phosphate concentrations from the base scenario.

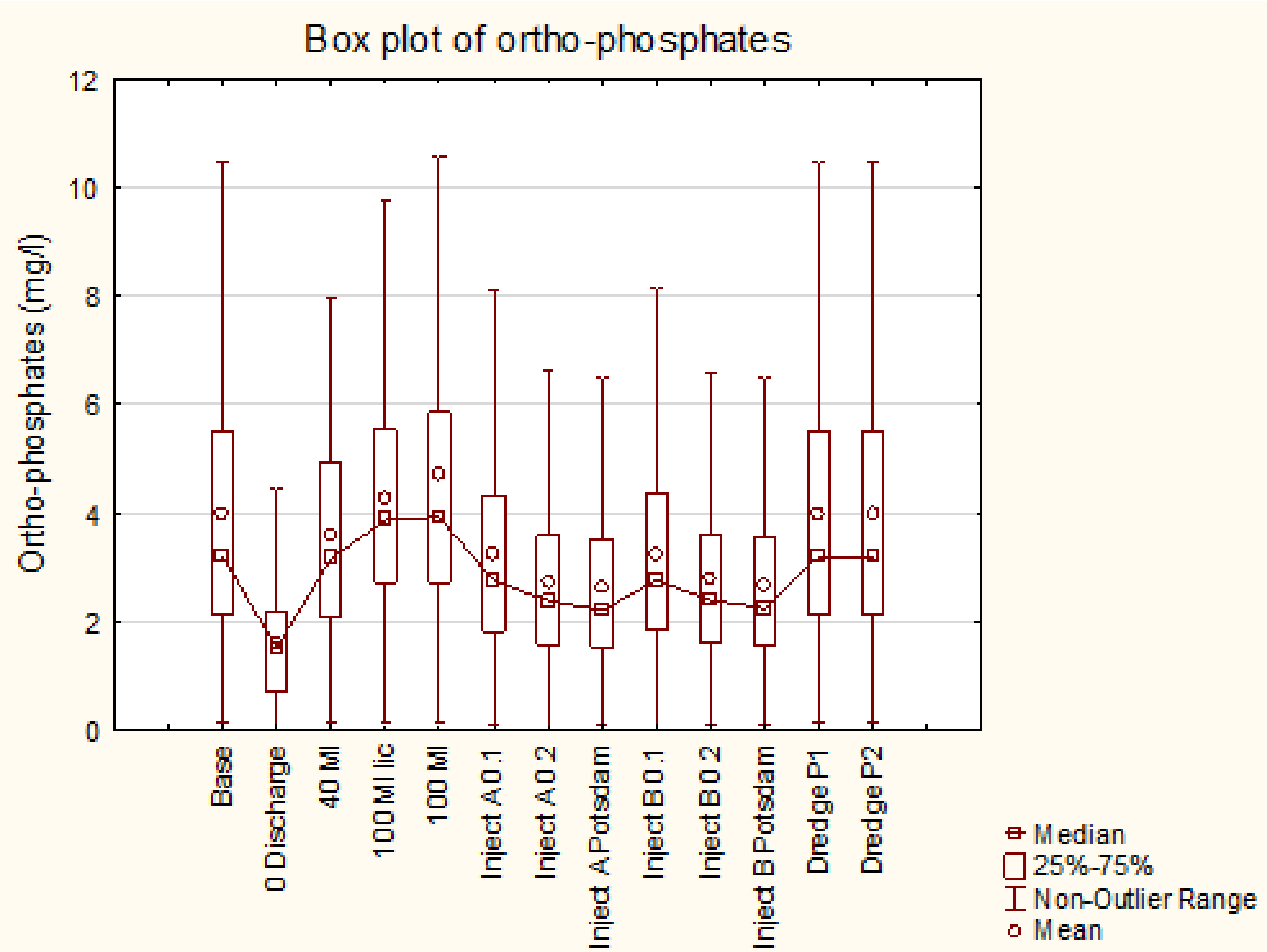


Figure 13-46. Box and whiskers plot for ortho-phosphates

Table 13-8. Descriptive statistics for ortho-phosphates

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	4.00	3.20	0.13	12.76	2.15	5.48	2.55
Option 4: Zero discharge	1.62	1.52	0.00	5.47	0.70	2.20	1.18

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
40 Ml discharge at licence quality	3.58	3.17	0.13	7.94	2.11	4.92	1.82
100 Ml discharge at licence quality	4.29	3.88	0.14	9.75	2.73	5.54	2.07
100 Ml discharge at current poor quality	4.74	3.92	0.14	15.37	2.73	5.87	2.94
Option 3: Seawater addition 0.1 m³/s at Point A	3.24	2.76	0.12	10.21	1.82	4.33	1.95
Option 3: Seawater addition 0.2 m³/s at Point A	2.73	2.39	0.00	7.95	1.58	3.60	1.59
Option 3: Seawater addition matching Potsdam discharge at Point A	2.63	2.21	0.12	7.94	1.53	3.52	1.48
Option 3: Seawater addition 0.1 m³/s at Point B	3.26	2.76	0.00	10.78	1.84	4.36	1.95
Option 3: Seawater addition 0.2 m³/s at Point B	2.79	2.40	0.12	8.63	1.62	3.62	1.66
Option 3: Seawater addition matching Potsdam discharge at Point B	2.66	2.25	0.12	8.21	1.58	3.56	1.50
Option 1: Dredging (below Woodbridge only)	4.00	3.20	0.13	12.76	2.15	5.48	2.55
Option 1: Dredging (whole lagoon)	4.00	3.20	0.13	12.76	2.15	5.48	2.55

The cycle of ortho-phosphates in the system was modelled as organic phosphates settling to the sediments, as well as decaying to inorganic phosphates which is used as a nutrient for algal production. The eutrophic state would be reduced the most by diverting all Potsdam discharge to a marine outfall, followed by seawater injection at Potsdam discharge rates at Location B. All the scenarios of injecting seawater served to decrease the ortho-phosphate concentrations. Dredging had no significant impact on orthophosphate concentrations.

13.7.5 Nitrogen

Figure 13-47 and Table 13-9 are representative of the changes in nitrogen concentrations from the base scenario.

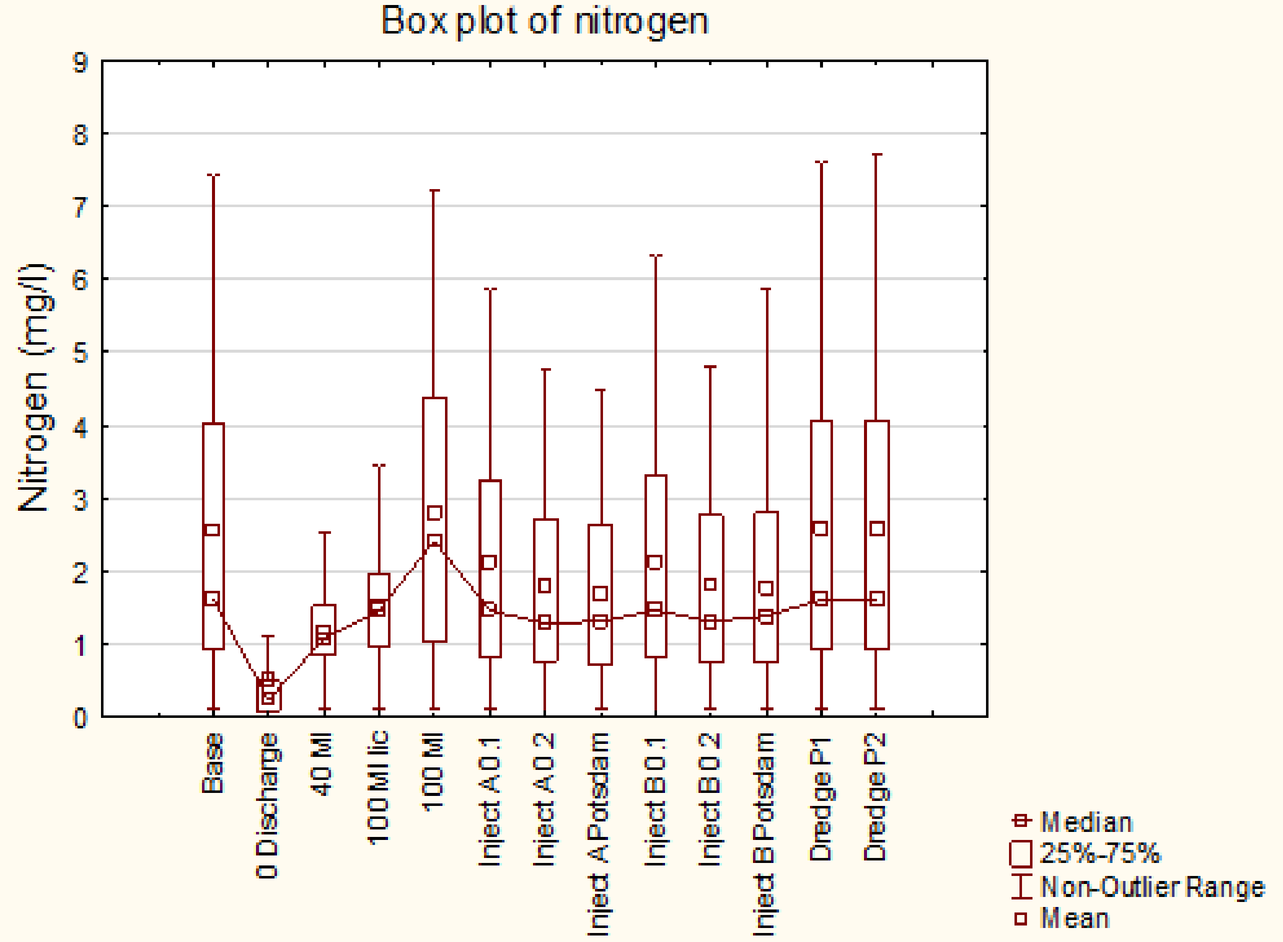


Figure 13-47. The box and whiskers plot for nitrogen

Table 13-9. Descriptive statistics for nitrogen

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	2.55	1.61	0.11	7.43	0.93	4.03	1.97
Option 4: Zero discharge	0.51	0.24	0.00	3.21	0.09	0.49	0.69
40 Ml discharge at licence quality	1.17	1.07	0.11	2.83	0.84	1.52	0.53
100 Ml discharge at licence quality	1.52	1.47	0.12	4.08	0.96	1.96	0.74
100 Ml discharge at current poor quality	2.80	2.41	0.12	7.21	1.03	4.38	1.94
Option 3: Seawater addition 0.1 m³/s at Point A	2.11	1.46	0.11	5.87	0.83	3.25	1.56
Option 3: Seawater addition 0.2 m³/s at Point A	1.78	1.29	0.00	5.77	0.75	2.71	1.26
Option 3: Seawater addition matching Potsdam discharge at Point A	1.69	1.31	0.11	4.47	0.72	2.63	1.17
Option 3: Seawater addition 0.1 m³/s at Point B	2.13	1.46	0.00	7.21	0.84	3.32	1.57
Option 3: Seawater addition 0.2 m³/s at Point B	1.81	1.31	0.11	6.49	0.75	2.77	1.30

Option 3: Seawater addition matching Potsdam discharge at Point B	1.75	1.38	0.11	6.81	0.74	2.80	1.22
Option 1: Dredging (below Woodbridge only)	2.57	1.61	0.11	7.61	0.94	4.05	1.99
Option 1: Dredging (whole lagoon)	2.58	1.61	0.11	7.70	0.94	4.05	2.00

In the model, organic nitrogen is hydrolysed to ammonium and inorganic nitrogen which is then further oxidised to nitrites and nitrates. The modelled scenario nitrogen concentration followed a similar trend to the ortho-phosphate concentrations in that with respect to eutrophication, the Potsdam diversion to a marine outfall improved the ortho-phosphate concentration, followed in terms of effect by Potsdam meeting its licence conditions for effluent discharge quality. Seawater injection at location B at the higher rates lowered the nitrogen concentrations followed by location A. Dredging both phases had negligible effect on the nitrogen concentrations.

13.7.6 Ammonium

Figure 13-48 and Table 13-10 are representative of the changes in ammonium concentrations from the base scenario.

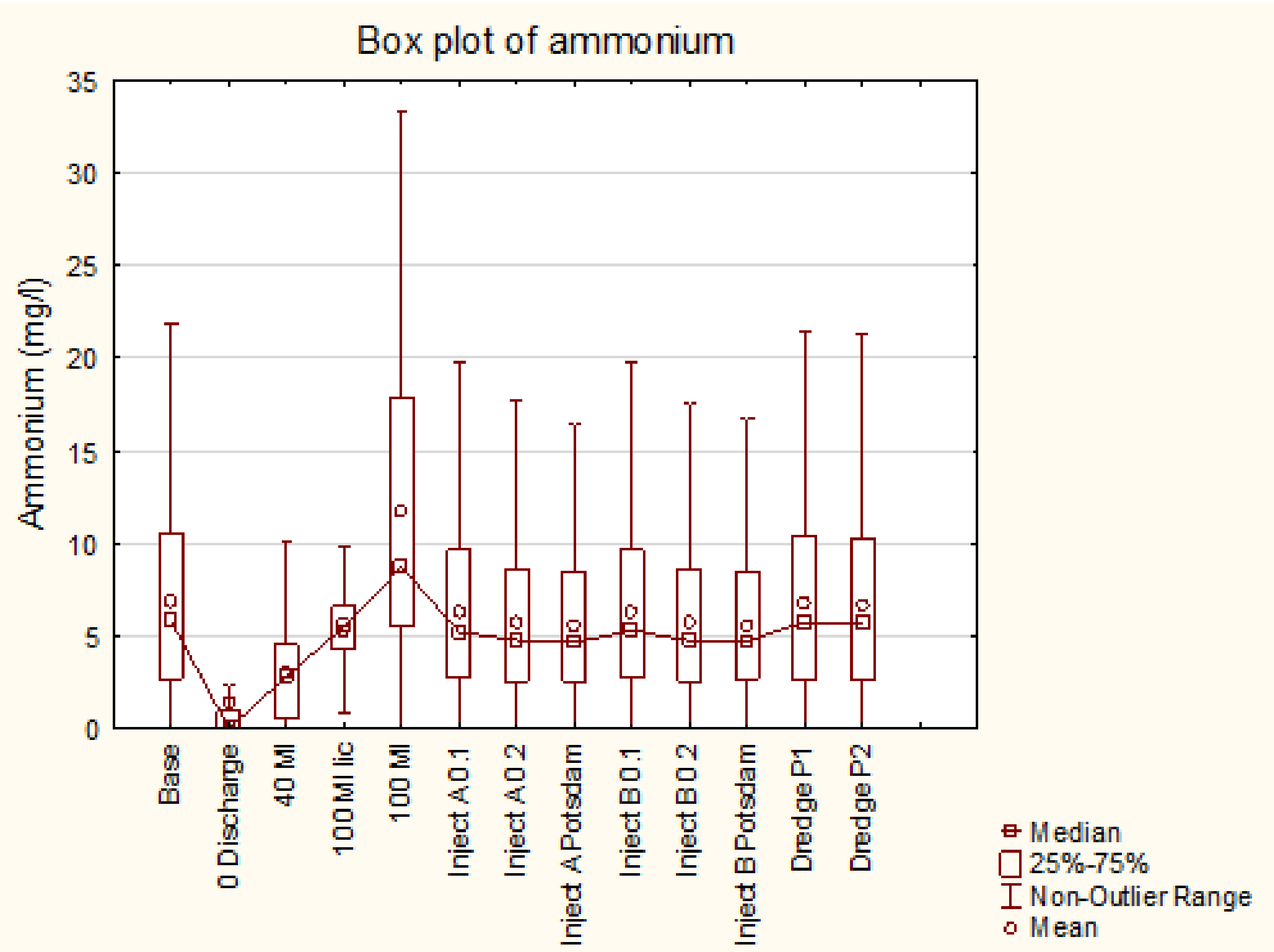


Figure 13-48. The box and whisker plot for ammonium concentrations

Table 13-10. Descriptive statistics for ammonium

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	6.89	5.81	0.00	21.87	2.72	10.53	5.23
Option 4: Zero discharge	1.37	0.02	0.00	10.39	0.00	0.97	2.66
40 Ml discharge at licence quality	2.99	2.84	0.00	10.06	0.63	4.51	2.60
100 Ml discharge at licence quality	5.25	5.50	0.00	9.90	4.30	6.64	2.20
100 Ml discharge at current poor quality	11.67	8.73	0.00	33.25	5.50	17.91	8.34
Option 3: Seawater addition 0.1 m³/s at Point A	6.27	5.20	0.00	23.30	2.75	9.67	4.58
Option 3: Seawater addition 0.2 m³/s at Point A	5.70	4.79	0.00	17.73	2.58	8.63	4.02
Option 3: Seawater addition matching Potsdam discharge at Point A	5.49	4.66	0.00	16.43	2.57	8.46	3.80
Option 3: Seawater addition 0.1 m³/s at Point B	6.24	5.30	0.00	21.08	2.73	9.63	4.57
Option 3: Seawater addition 0.2 m³/s at Point B	5.70	4.75	0.00	20.86	2.57	8.58	4.08
Option 3: Seawater addition matching Potsdam discharge at Point B	5.52	4.72	0.00	16.74	2.69	8.42	3.80
Option 1: Dredging (below Woodbridge only)	6.74	5.76	0.00	21.48	2.65	10.39	5.06
Option 1: Dredging (whole lagoon)	6.65	5.72	0.00	21.26	2.62	10.26	4.96

Ammonium in the water column uses dissolved oxygen for oxidation to nitrites and nitrates, so it was expected that lower ammonium concentrations correlated to improved dissolved oxygen concentrations. The greatest improvement in ammonium concentrations was for the diversion of Potsdam effluent to a marine outfall, followed by Potsdam discharging effluent in accordance with its licence limits. Seawater injection tended to dilute the ammonium concentrations nitrification in the water column and location B was preferred over A. The ammonia concentrations peaked for the simulation of Potsdam discharging 100 Ml of poor-quality effluent.

13.7.7 Algal concentrations

Figure 13-49 and Table 13-11 are representative of the changes in algal concentrations from the base scenario.

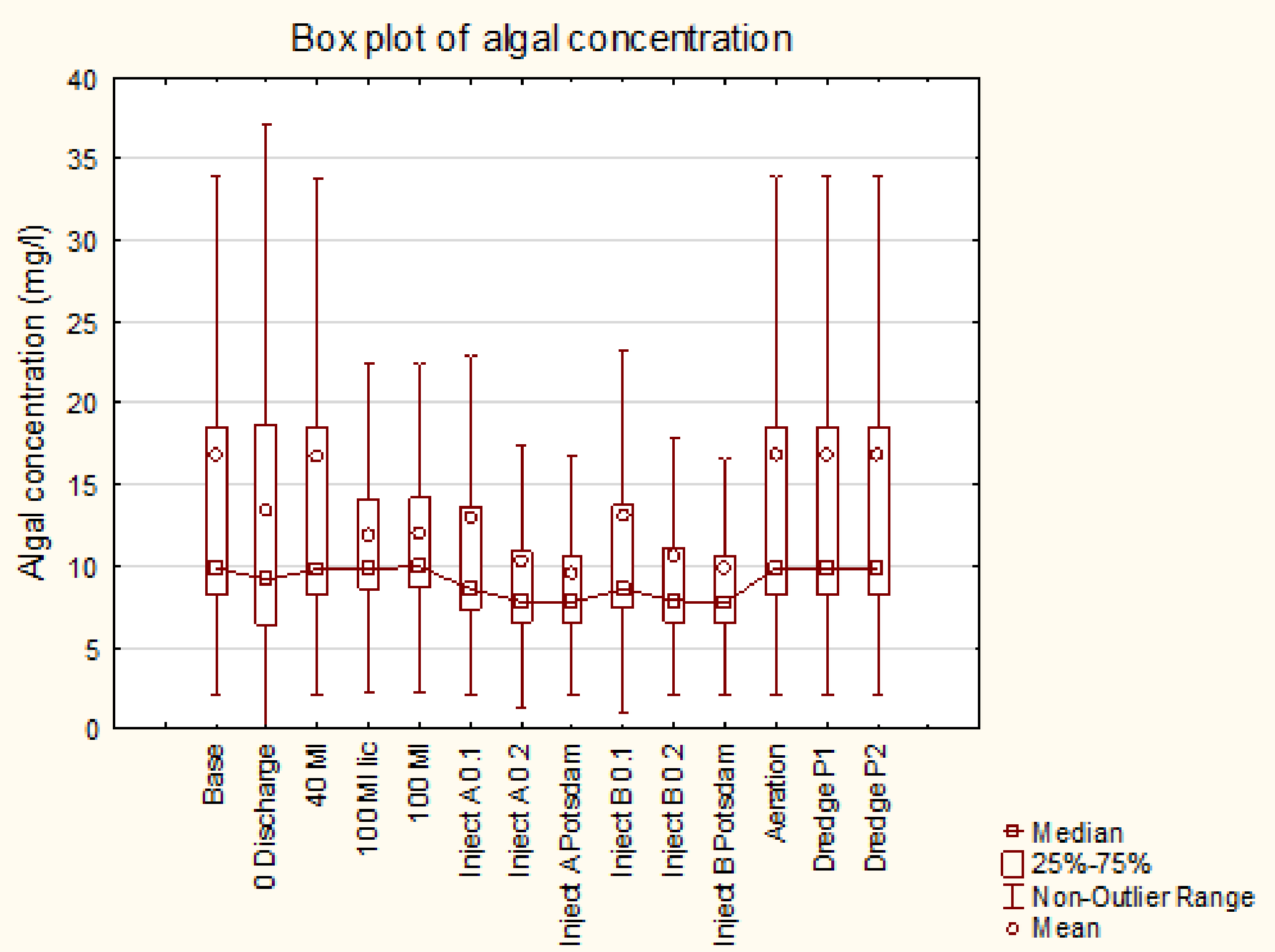


Figure 13-49. The box and whisker plot for algal concentrations

Table 13-11. The descriptive statistics for algal concentrations

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	16.8	9.9	2.1	62.4	8.2	18.6	13.5
Option 4: Zero discharge	13.4	9.2	0.0	50.1	6.4	18.7	9.8
40 Ml discharge at licence quality	16.7	9.8	2.1	62.3	8.2	18.5	13.4
100 Ml discharge at licence quality	11.9	9.8	2.2	28.7	8.6	14.1	4.8
100 Ml discharge at current poor quality	12.0	10.0	2.2	28.3	8.7	14.2	4.9
Option 3: Seawater addition 0.1 m³/s at Point A	12.9	8.6	2.1	44.2	7.4	13.6	9.2
Option 3: Seawater addition 0.2 m³/s at Point A	10.3	7.8	1.3	38.2	6.6	10.9	6.1
Option 3: Seawater addition matching Potsdam discharge at Point A	9.5	7.8	2.1	26.7	6.5	10.6	4.7
Option 3: Seawater addition 0.1 m³/s at Point B	13.1	8.5	1.0	44.2	7.4	13.7	9.5

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Option 3: Seawater addition 0.2 m^3/s at Point B	10.6	7.9	2.1	32.4	6.6	11.1	6.5
Option 3: Seawater addition matching Potsdam discharge at Point B	9.8	7.7	2.1	41.3	6.5	10.6	5.2
Option 1: Dredging (below Woodbridge only)	16.8	9.9	2.1	62.4	8.2	18.6	13.5
Option 1: Dredging (whole lagoon)	16.8	9.9	2.1	62.4	8.2	18.6	13.5

The data show that the greatest improvement in eutrophication was from Potsdam discharging to a marine outfall, followed by seawater injection which effectively lowered algal concentrations. Algal concentrations were at a minimum for increased salinity in the river (all injection scenarios) as well as for Potsdam discharge diversion which would also allow greater ingress of seawater. Increased Potsdam effluent discharge reduced algal concentration, but only as a result of dilution not decreased growth. Dredging the bed resulted in similar algal concentrations to the baseline scenario as the nutrients have not been removed from the inflowing water. The data suggest that the dissolved oxygen improvement is not a result of increased algal production over the baseline scenario.

13.7.8 Salinity

Figure 13-50 and Table 13-12 are representative of the changes in salinity from the base scenario.

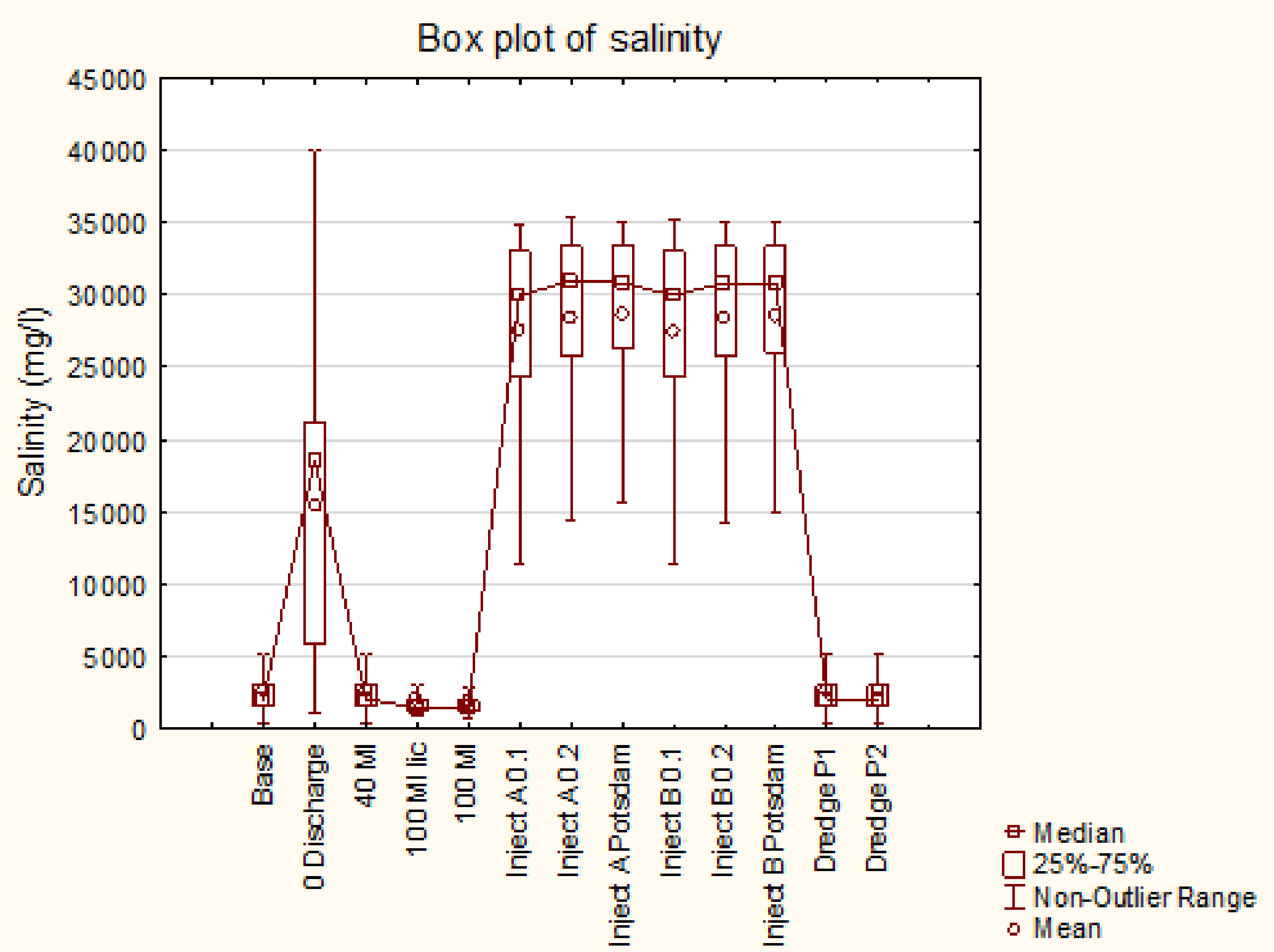


Figure 13-50. The box and whiskers plot for salinity

Table 13-12. Descriptive statistics for salinity

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Base	2702	2702	2052	360	11842	1615	3030
Option 4: Zero discharge	15449	15449	18571	1050	39971	5978	21141
40 Ml discharge at licence quality	7882	7882	5984	1049	34538	4710	8837
100 Ml discharge at licence quality	2020	1490	122	11739	1308	2021	1441
100 Ml discharge at current poor quality	5694	5694	4234	1988	34246	3758	5626
Option 3: Seawater addition 0.1 m³/s at Point A	27513	27513	29964	4862	34930	24354	32991
Option 3: Seawater addition 0.2 m³/s at Point A	28427	28427	30969	6161	35449	25796	33367
Option 3: Seawater addition matching Potsdam discharge at Point A	28694	28694	30816	3599	34936	26315	33464
Option 3: Seawater addition 0.1 m³/s at Point B	27490	27490	29948	4929	35216	24366	33013
Option 3: Seawater addition 0.2 m³/s at Point B	28353	28353	30837	6105	34938	25707	33342

Scenario	Descriptive Statistics						
	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Standard Deviation
Option 3: Seawater addition matching Potsdam discharge at Point B	28575	28575	30770	3412	34938	25983	33400
Option 1: Dredging (below Woodbridge only)	7882	2702	2052	360	11842	1615	3030
Option 1: Dredging (whole lagoon)	7882	2702	2052	360	11842	1615	3030

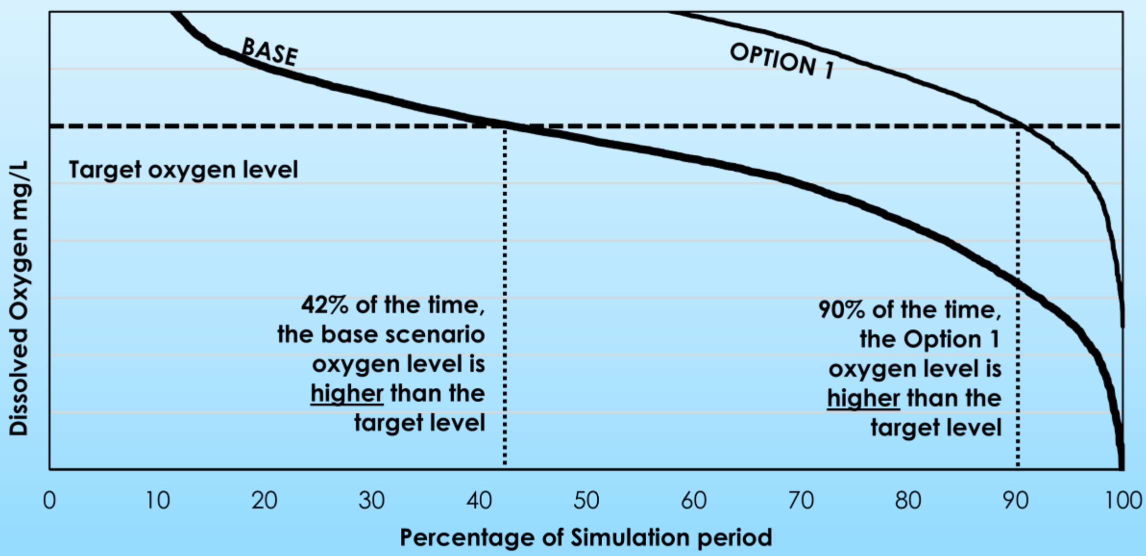
Even though RTV10 is close to the mouth and ocean, the background salinity of the system was very low as the summer flows are dominated by the Potsdam effluent discharge and the mouth very seldom closes, which in a functioning estuary would increase salinity. The average sampled conductivity from December 2018 to August 2019 at RTV10 was 752 mS/cm (approximately 4.7 PSU). For any injection of seawater, the salinity increased in the system. Similar effects are seen for the diversion of Potsdam effluent to a marine outfall. Increasing the Potsdam discharge to 100 Ml/day resulted in a salinity reduction over the baseline but this was due to increased conductivity in the effluent rather than to marine exchange. Dredging had no effect on modelled salinity as the mouth would not be dredged (see Figures Figure 13-40 and Figure 13-41) thereby not affecting seawater exchange.

13.7.9 Performance plot

To summarise the previous figures as well as to assist managers with the choice of interventions to improve the key water quality constituent, namely dissolved oxygen, the performance of the various interventions is presented in Figure 13-51 in the form of an **exceedance plot**, which shows the percentage of the modelled timeframe (a period of 9 months) during which the simulated dissolved oxygen level at sample point RTV10 was **higher** (i.e. 'better') than a given value. Refer to Box 13-1 for an explanation of interpretation.

Box 13-1. Interpretation of exceedance plots

The exceedance plot used to reflect water quality displays data as follows:



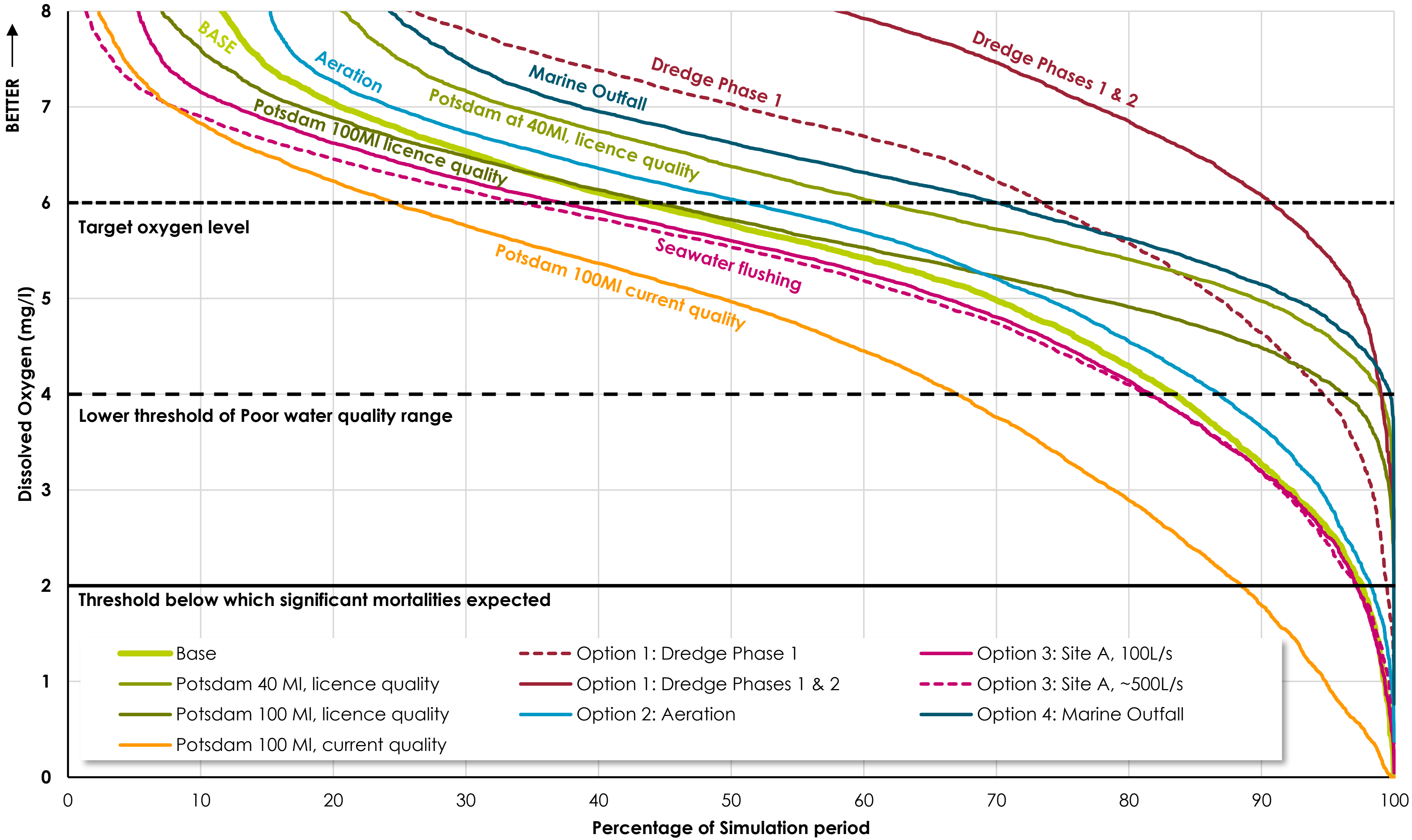


Figure 13-51. Exceedance plot for dissolved oxygen in selected model scenarios. Refer to Box 13-1 for interpretation guidance and note that oxygen thresholds are lower thresholds – higher values are better water quality.

The inverse of Figure 13-51 is also a useful way to assess scenario effectiveness, as set out below. The base scenario showed a dissolved oxygen concentration above 4mg/l for 86% of the simulation time, or **less than 4mg 14% of the simulation time**.

The proposed interventions are ranked in order of decreasing beneficiation:

- A fully dredged lagoon spent only 0.8% of the simulation time below 4 mg/l.
- Licence-quality discharge at 40 Ml/day had dissolved oxygen below 4mg/l for only 0.9% of the simulation and for 100Ml/day it was increased to 3.3% of the time simulated..
- The diversion of Potsdam discharge resulted in dissolved oxygen levels less than 4mg/l for only 2.6% of the simulation.
- Phase 1 dredging only had dissolved oxygen levels below 4 mg/l only 4.6% of the time.
- Seawater injection at A resulted in the dissolved oxygen being less than 4mg/l for 15.8% of the simulation and at location B 14.6% of the simulation time.
- 100 Ml of Potsdam discharge at the current poor quality resulted in dissolved oxygen levels below 4 mg/l for 30.4% of the simulation time.

Table 13-13 shows the dissolved oxygen performance for 4mg/l and below.

Table 13-13. Dissolved oxygen performance

Scenario	Percentage of time dissolved oxygen was <u>less</u> than			
	4 mg/l	3 mg/l	2 mg/l	1 mg/l
Base	14.0	6.6	**2.0**	0.6
Option 4: Zero discharge	2.6	0.8	**0.1**	0.0
40 Ml discharge at licence quality	0.9	0.1	**0.0**	0.0
100 Ml discharge at licence quality	3.3	0.3	**0.0**	0.0
100 Ml discharge at current poor quality	30.4	19.3	**10.2**	4.4
Option 3: Seawater addition matching Potsdam discharge at Point A	15.8	7.2	**2.4**	0.3
Option 3: Seawater addition matching Potsdam discharge at Point B	14.6	6.2	**2.0**	0.3
Option 2: Aeration	13.4	5.5	**1.9**	0.4
Option 1: Dredging (below Woodbridge only)	4.6	1.4	**0.5**	0.0
Option 1: Dredging (whole lagoon)	0.8	0.1	**0.0**	0.0

The table shows that the greatest improvement in dissolved oxygen concentrations was when Phase 2 dredging was simulated as the dissolved oxygen concentration did not drop below 2 mg/l. If the Potsdam WWTW were to discharge licence quality effluent, the dissolved oxygen concentration would improve substantially over the baseline scenario as well as being better than discharging to marine outfall. Aeration also showed improvement in raising minimum concentrations reached over the baseline, justifying its use as short-term intervention to increase dissolved oxygen concentrations in the lagoon area.

13.8 Summary

The modelling showed that additional inflows to the system over the baseline resulted in increased water elevations whilst diversions decreased the surface water elevations. The dredging of the system did not influence the water elevations.

The proposed scenarios did not alter the water temperatures significantly, but injecting seawater into the system had a minor cooling effect on water temperatures. The scenario where Potsdam discharged directly to the ocean allowed the area to reach its minimum temperature as there was maximum penetration of cooler oceanic waters.

The base scenario showed a **dissolved oxygen** concentration less than 4mg/l for 14% of the simulation time at RTV10 and the proposed interventions were ranked in order of decreasing beneficiation as:

- Phase 2 dredging was only 0.8% of the simulation time below 4 mg/l.
- The 40 Ml good quality discharge was less than 4mg/l for 0.9% of the simulation
- The 100 Ml good quality discharge was less than 4mg/l for 3.3% of the simulation
- The 0 Potsdam discharge was less than this for 2.6% of the simulation
- Phase 1 dredging was only 4.6% of the time below 4 mg/l
- Seawater injection at A resulted in the dissolved oxygen being less than 4mg/l for 15.8% of the simulation and at location B 14.6% of the simulation time.
- 100 Ml poor quality was 30.4% of the simulation below 4 mg/l

It was concluded that dredging the whole of the lagoon would improve the dissolved oxygen concentrations more than the other proposed interventions.

Ranking the improvements in terms of minimum values and lower quartile over the baseline scenario for the RTV10 sample point near the mouth resulted in the following:

1. Phase 2 dredging
2. 40 Ml good quality effluent
3. 100 Ml good quality effluent
4. Phase 1 dredging
5. Potsdam diversion to marine outfall
6. Inject seawater at B at Potsdam discharge rates
7. Inject seawater at A at Potsdam discharge rates
8. Inject seawater at A at 0.2 m^3/s
9. Aeration but only improved the minimum dissolved oxygen concentration
10. Inject seawater at B at 0.2 m^3/s
11. Injecting seawater at A or B at 0.1 m^3/s
12. 100 Ml of poor effluent discharge from Potsdam resulted in the lowest overall dissolved oxygen concentrations.

The eutrophication state would benefit the most by diverting Potsdam discharge to marine outfall followed by seawater injection at Potsdam discharge rates at Location B.

All the scenarios of injecting seawater served to decrease all the nutrients concentrations by virtue of dilution.

Even though RTV10 is close to the mouth and ocean, the background salinity of the system was very low as the summer flows are dominated by the Potsdam effluent discharge.

For any injection of seawater, the salinity improved in the system as well as for the diversion of Potsdam. Increasing the Potsdam discharge to 100 Ml/day resulted in an increase over the baseline but this was due to increased releases of conductivity in the effluent.

The environmental performance of dissolved oxygen shows that biggest improvements were from phase 2 dredging the system.

13.9 Notes on model performance

The study goal was to assess the performance of proposed interventions that would improve the water quality of the Milnerton Lagoon. This was accomplished by using the HEC-RAS model and calibrating it to the baseline scenario from where all further simulations would be compared. The HECRAS model is a deterministic model which calculates events exactly, without the involvement of randomness. The HEC-RAS model reproduced the expected hydraulics as well as the sampled water quality data (as calibration and validation) with the flow data generated by this study and the City's sampled water quality data as boundary conditions. **Any modelled scenario compared to the original must be considered a relative change and not an absolute.**

The validated model reproduced the sampled data for long-term and diurnal water temperatures, salinity, dissolved oxygen concentrations, algal concentrations and nutrients. The following data improvements would increase the future accuracy of model predictions:

» Meteorological data is a driver for long term as well as diurnal water temperatures and the data sourced for the study was obtained from climate reanalyses data. The data does not incorporate shading data for riparian vegetation thus some water temperatures simulated are greater than the sampled data.

» The boundary conditions are the main drivers for the hydraulics of model and in order to improve their accuracy, flow gauges should be installed on the Diep River as well as quantifying the groundwater interaction and an estimation of stormwater flows.

» Salinity and dissolved oxygen concentrations of the ocean at the mouth were not sampled and, in their absence, constant values of 35 PSU (35 000 mg/l) and 7mg/l DO were used as inputs for the simulations. Improving this boundary condition with sampled data would improve accuracy of the salinity and dissolved oxygen simulations.

» Salinity in the estuary is dependent on dispersion which generally depends on the river and tidal inflows, thus in-situ longitudinal dispersion coefficients need to be measured for the varying river flowrates and tides in order to more accurately reproduce salinity results. The use of handheld conductivity meters that can convert the measured conductivity to PSU and TDS compensating for temperature is recommended.

» The bathymetry of Diep River estuary is constantly changing due to sediment erosion and deposition, especially the mouth area and thus any long-term water quality modelling should account for a changing bathymetry including scour and deposition.

14 ASSESSMENT OF REMEDIATION OPTIONS

14.1 Assessing options for implementation

The factors considered in selecting and prioritising interventions to address the water quality in the Milnerton Lagoon are outlined below.

14.1.1 Modelled efficacy

Water quality modelling was undertaken to assess the performance of proposed interventions in relation to the water quality of the Milnerton Lagoon. The HEC-RAS model was used, calibrated to a baseline scenario against which options were tested. The HEC-RAS model reproduced the expected hydraulics as well as the sampled water quality data (as calibration and validation) with the flow data generated by this study and the City's sampled water quality data as boundary conditions. Options 1, 2, 3 and 4 were modelled (with a number of variations) and compared with the baseline situation for parameters including dissolved oxygen, salinity, nutrients, and algae. A relative improvement or deterioration in these parameters is the basis for the assessment of expected effects of the intervention.

14.1.2 Expected benefits

The expected positive outcomes of the intervention, including but not limited to the modelled effects on specific water quality parameters, are qualitatively considered. This incorporates the expected benefits of smaller-scale interventions not expected to significantly affect the water quality of the lagoon but which may address some of the cumulative sources of pollution in the catchment. It also includes the largely-unknown effects of proposed bioremediation by inoculation of the lagoon's sediments with microbial and enzymatic products.

14.1.3 Estimated costs

Financial costs of interventions have been estimated based on high-level concepts and engineering assumptions.

14.1.4 Time to start

The time required to conclude design, approvals and procurement processes is a key driver of whether an intervention is a viable short-term solution to the Milnerton Lagoon's water quality issues. Refer to Table 14-3 for a summary of expected timeframes.

14.1.5 Time to achieve effect

Implementation of some of the options would be a lengthy process, with construction timeframes affecting the point at which effects on the water quality may be expected. As the primary focus of this plan is on short-term interventions, time to achieve an effective result is a key consideration.

14.1.6 Ecological risks

If an intervention poses a significant ecological risk in addition to (potentially) improving water quality, its positive effects must be weighed up against the harm it may cause.

14.1.7 Risk of failure

For interventions with a higher financial or ecological cost, the decision whether or not to implement them should consider the probability of success – or, put another way, the risk of the intervention not achieving the desired or expected outcomes.

14.1.8 Other negative impacts

Impacts other than ecological and financial costs may determine the desirability of a particular option – for instance, the level of disturbance, noise, or access difficulties caused by its implementation.

14.1.9 Approvals required

Most of the proposed interventions will be subject to approval in terms of environmental legislation. Refer to section 4.4 for an overview of the types of authorisation which may be applicable to the proposed options, and to sections 5 through 11 for further details of their applicability to each option. Table 14-1 summarises the authorisation requirements for each option.

Table 14-1. Authorisation requirements for all options

Option / Approval	1: Dredging	2: Aeration	3: Seawater flushing	4: Marine outfall	5: Biofilters	6: Treatment wetlands	7: Inoculation
Environmental authorisation (NEMA)[†††] *Listed activities*	**Yes - BA** *19,19A of LN1*	**Possible - BA** *14 of LN3** ***depends on extent of electrical infrastructure**	**Yes – full EIA** *12, 15, 19, 19A of LN1 14 of LN2 14 of LN3*	**Yes – full EIA** *10, 12, 15, 19, 19A of LN1 6, 14 of LN2 14 of LN3*	**Yes - BA** *19,19A of LN1*	**No*** ***if outside watercourses**	No
Water Use authorisation (NWA)[‡‡‡] *Water uses*	**Yes: Probable W.U.L.** *21(c), (i), (g)?*	**Yes: Probable G.A.** *21(c), (i)*	**Yes: Probable W.U.L.** *21(c), (i)*	**Yes: Probable W.U.L.** *21(c), (i)*	**Yes: Probable G.A.** *21(c), (i)*	**Yes: Probable G.A.** *21(c), (i)*	No
Coastal waters discharge permit (ICMA)	Possible	No	No	**Yes**	No	No	No
Waste Management Licence (NEMWA)	Possible	No	No	No	No	No	No
Minimum statutory timeframes	**8-9 months**	**8-9 months**	**9-12 months**	**9-12 months**	**8-9 months**	**1-3 months**	None

The following section summarises the model outputs and expected costs and benefits. **Table 14-2 on page 276 summarises the assessment of all options.**

[†††] 'BA' = Basic Assessment; 'full EIA' = Scoping and Environmental Impact Assessment

[‡‡‡] 'W.U.L.' = Water Use Licence; 'G.A.' = General Authorisation

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14.2 Summary of options and recommendations

14.2.1 Option 1: Dredging

Dredging of the lagoon in up to three phases, depending on budgetary constraints, was investigated. Dredging will remove contaminated, nutrient-rich sediment with a high oxygen demand from the system and the models suggest **significant benefits for oxygen levels would result**. If the lower lagoon from Woodbridge to the mouth is dredged (cost estimated at R 77 million), it is expected to take no more than ten years for organic sediments to accumulate to similar levels in this area. If the entire lagoon between Otto du Plessis and the mouth is dredged (cost estimated at R 133 million), it is expected to take up to 20 years for sediments to accumulate to current levels again. Repeated or maintenance dredging will be needed in future – increasing the costs significantly - unless the inflows of organic sediment are stopped at the same time that dredging is implemented. This requires significantly improving the treatment of effluent at the Potsdam WWTW.

Risks associated with the proposed dredging include the availability of sites for disposal of dredged material. Dredging will also impact on amenity during implementation, both due to the noise and disturbance of the process itself, and the need for a large area to be dedicated to dewatering and handling of the dredged material – it is suggested that the Loxton Road park is the only suitable location for this. Other impacts include disturbance of habitats and short-term resuspension of sediments during dredging.

Given the anticipated benefits for water quality and the pressing need to remove built-up sediments from the lagoon, **it is recommended that dredging be implemented in the Milnerton Lagoon to the extent possible.** Authorisations and detailed design will be required, and is expected that at least 24 months will be required for these processes and the procurement of contractors. Dredging of the lagoon will, however, provide only short-lived improvements if the discharge at Potsdam WWTW of under-treated effluent containing significantly elevated nutrients and solids is not ceased.

14.2.2 Option 2: Aeration

Low or non-existent dissolved oxygen levels in the lagoon are a significant contributing factor to the change in bacterial activity that results in sulphurous odours in the lagoon (section 2.6.4), as well as to the high ammonia concentrations that cause toxicity for aquatic life. Artificial aeration of the lagoon is recommended. Depending on the technology option selected, capital cost estimates are between R 1.5 and R 4.14 million), either of which could be implemented near the Erica Road outfall or at the Otto du Plessis road bridge where the channel is relatively deeper and narrower. Due to the low contact time between the flowing water and the aerators, the best possible outcome is an increase in oxygen levels to approximately 4 mg/L – still low in absolute terms.

Risks associated with the proposed aeration include the risk of failure or only minimal improvement to the situation, especially in respect of odour. The mechanical equipment may be subject to theft or vandalism, and its operation will be affected by power availability during load-shedding. Other more minor impacts include noise, obstruction to boating, disturbance of habitats and possible resuspension of sediments.

Although oxygen levels in the lagoon will only be significantly and lastingly improved by reducing the oxygen demand in the Potsdam effluent, it is recommended that artificial aeration be trialled as a short-term measure to increase the oxygen concentrations in the water and contribute to reducing odours and other impacts. Authorisations will however be required from the authorities, increasing the timeframes to as much as 8-9 months for implementation. If pursued, aeration of the lagoon may alleviate some of the symptoms of the pollution, but will not have a lasting effect. It is recommended that aeration be pursued if authorisation processes and the installation of electrical supply and procurement of aerators are feasible within 6 to 8 months.

14.2.3 Option 3: Seawater flushing

Amongst the impacts of urban development around the Milnerton Lagoon is reduced marine interchange, which results in lower salinities and less flushing of the lagoon by seawater. To counter this change, it is proposed that seawater be pumped into the lagoon upstream of the Woodbridge. The intention is to increase the flushing of sediments, and more importantly to increase the oxygen concentration of water in the lagoon by mixing with oxygen-rich and saline seawater. Modelled benefits include a reduction in nutrient concentrations (by dilution) and algal growth (due to dilution and salinity changes). A reduction in the levels of ammonia toxicity is also expected.

Estimated costs for this intervention depend on the rate of seawater pumping required; modelled scenarios considered a rate of 100 to ~500 litres per second and found the greatest effect with the higher rate of pumping. The costs of implementation are estimated in the region of R42 million for a 100 L/s flow rate, or R 70 million for a 500 L/s flow rate, for installation of an offshore intake, beach pumpstation and pipeline, with flows pumped to a point approximately in line with Erica Road.

For the lower flow rate of 100 L/s, beach wells installed along the back of the beach are a more viable and cost-effective option with an installation cost of approximately R 10 million.

Risks include the loss of freshwater habitat (largely reedbeds) at and downstream of the outfall in the lagoon, and impacts would also include some initial disturbance along the pipeline route and on the beach. Environmental authorisation would certainly be required, from the national authority, and timeframes for implementation are estimated at between three and four years. Due to the **high capital cost and long implementation timeframes relative to the intended improvements at Potsdam, seawater flushing is not strictly a short-term remediation measure for the Milnerton Lagoon.** The combination of low ecological risks and potential significant ecological benefits means that seawater pumping is recommended for a trial if financially feasible.

14.2.4 Option 4: Marine Outfall

The primary purpose of the marine outfall option would be to redirect the flow of treated, but poor-quality effluent from the Milnerton Lagoon directly to the ocean via a new pipeline. Removing the largest source of pollution loading in the lagoon would have a number of effects, the most significant of which would be an improvement in the water quality in the lagoon – modelled simulations without any flow from Potsdam suggest significant reductions in nutrient pollution and algal concentrations. Conversely, the reduction in flow into the lagoon is expected to reduce the flushing of the system and it is likely that the mouth would close during the summer if not actively managed through dredging. In the absence of summer flows from the Diep River catchment, this may result in a situation where contaminated stormwater runoff continues to enter the lagoon and is not flushed out to sea.

The construction of a new marine outfall would have significant risks in respect of regulatory approvals – recent public concern in respect of existing outfalls that discharge primary treated effluent (i.e. treated to a much lower degree than would be the case at Potsdam) has reference. Environmental authorisation would certainly be required from the national authority, in addition to a coastal waters discharge permit. Timeframes for implementation are estimated at between three and four years at minimum, and costs are estimated at R 190 million, higher than any other option.

Removing the largest source of pollution entirely may improve water quality in the lagoon, but its costs are significant and it is unlikely to be implemented sooner than the major capital upgrades proposed for Potsdam WWTW. It would increase maintenance requirements for the lagoon by altering the hydrology significantly. Based on the **high capital cost and long implementation timeframes relative to the intended improvements at Potsdam, together with the regulatory risks around authorisation for new offshore disposal of treated effluent, a marine outfall is not recommended as a short-term remediation measure for the Milnerton Lagoon.**

14.2.5 Option 5: Biofiltration

Retention and filtration of polluted runoff using large-scale filters packed with biochar or similar media shows some promise in small-scale trials implemented downstream of informal settlements elsewhere in the Western Cape. They mimic processes that are found in young wetland systems, but are dependent on in-situ conditions and variability in water quality and retention time to achieve the best performance.

It is suggested that a gravity-fed diversion channel be implemented on an experimental basis to divert a small proportion of the Milky Way stormwater flows through a biofiltration system before its discharge to the lagoon. Costs are estimated at approximately R 400 000. The implementation of a such an experimental system may serve to contribute to the incremental reduction of cumulative impacts arising from urban runoff, while increasing the state of knowledge of such systems through a comprehensive monitoring programme. It is not, however, anticipated to have a significant effect on water quality in the Milnerton Lagoon.

14.2.6 Option 6: Treatment Wetlands

The potential for constructed treatment wetlands to improve the quality of runoff before it enters Milnerton Lagoon was investigated. Treatment wetlands do not perform well with very high pollutant loadings, and would not be appropriate for catchments where the bulk of stormwater runoff is grey or black water. In the present situation, treatment wetlands might be used only for polishing of stormwater runoff from catchments with lower pollution levels, such as Killarney and parts of Montague Gardens. Given the high level of existing impacts on natural wetlands within the estuarine area, it is not proposed that existing natural wetlands should be used for treatment.

Areas available for this purpose include reedbeds no. 2 and 3, located between Potsdam WWTW and the Theo Marais Canal. These would first require extensive remediation to remove contaminated sediments and re-establish reedbed vegetation. This is a strongly recommended intervention, as these historic ponds have been contaminated by a history of receipt of raw sewage overflows during surcharges at the WWTW, and are assumed to play a role in actively contributing to pollution loads into Milnerton Lagoon. It is recommended that the ponds should be redesigned to include capacity for the isolation of WWTW overflows within contained areas of the wetland, rather than allowing such surcharges to spread over into and contaminate the entire wetland. Stormwater flows from the refinery area might also reasonably be treated within this wetland, provided that suitable ongoing management can be implemented. Additional treatment wetlands might also be created in the area between reedbed no. 3 and Koeberg Road, although this area is currently in use for various stockpiling purposes and may be required as a dewatering site for dredging of the lagoon. **Remediating the Potsdam reedbeds is recommended as an essential contamination remediation measure, but it is not short-term, is likely to require environmental and water use authorisation, and would not in itself be likely to bring about any measurable change in the current condition of the Milnerton Lagoon.**

14.2.7 Option 7: Inoculation

Microbial bioremediation is the use of microorganisms to degrade and remove pollutants. Because of the proprietary nature of, and scarcity of detailed technical information regarding commercial preparations of microbes and enzymes used for bioremediation, this study could not rely on published literature to recommend products for implementation. Prospective suppliers were asked via a formal, public process to submit information to assist in determining which commercially available products may assist with the breakdown of organic matter in sediments into non-toxic substances that can easily be flushed out of the water body with natural flow. Submissions were assessed to determine whether evidence was provided of their efficacy in saline or brackish environments, whether they demonstrated effectiveness in flowing systems with short retention times, and whether the information provided was scientifically and technically robust. Three of the 12 submissions received by the City show some promise on these bases and were subjected to ecotoxicity testing to determine any potential ecological risk. At least two of the three are recommended for **in-situ testing, which will be the basis for a final recommendation in respect of microbial bioremediation implementation in the lagoon.** An experimental design for in-situ testing of the selected products is set out in this report, for testing of their efficacy under in-situ conditions.

14.3 Summary of assessment

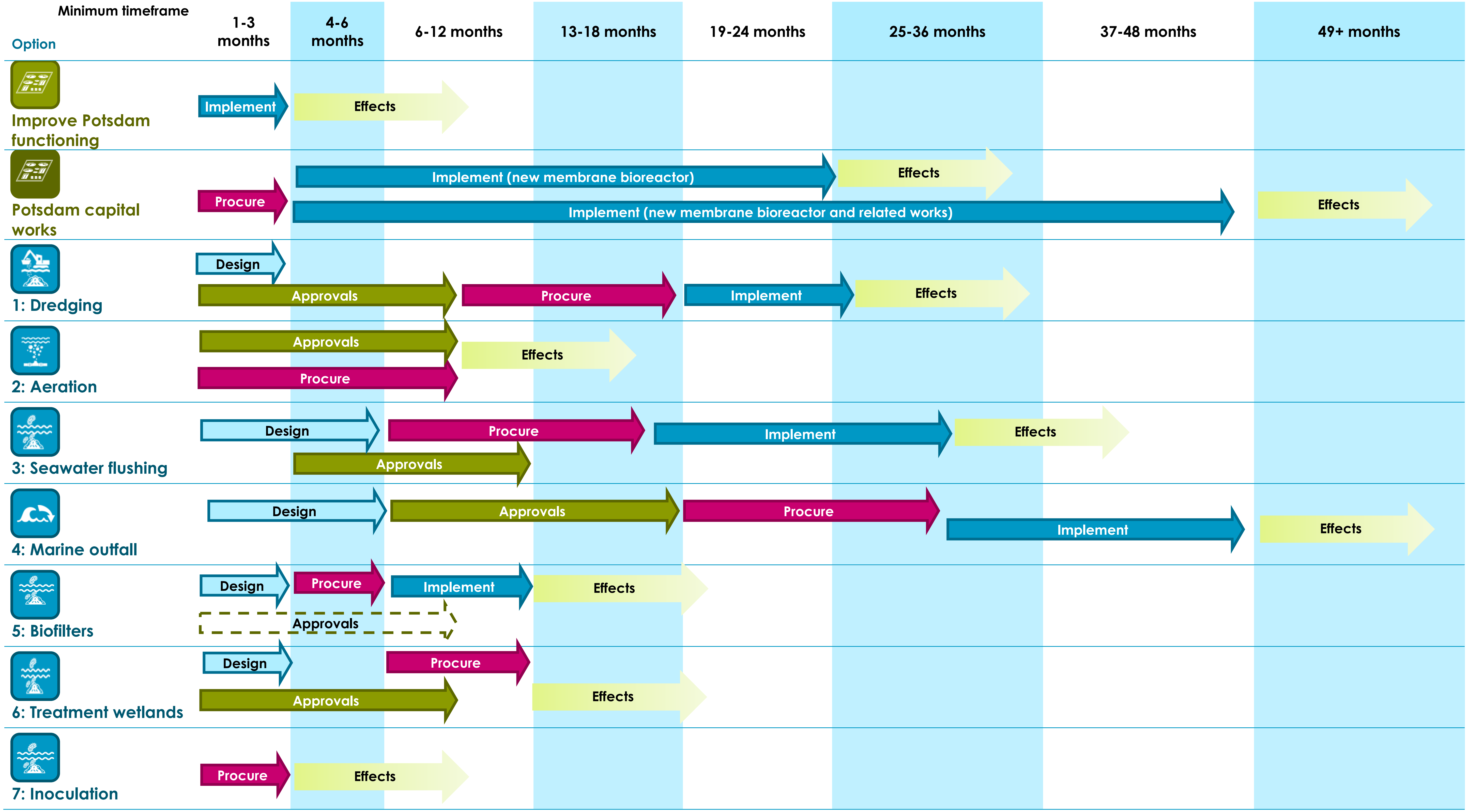
Table 14-2. Summary of options, high-level risks, costs and benefits for assessment

Option / Factor	1: Dredging	2: Aeration	3: Seawater flushing	4: Marine outfall	5: Biofilters	6: Treatment wetlands	7: Inoculation	Improve Potsdam functioning
Intended outcome	**Remove organic sediment to reduce oxygen demand and nutrient availability**	**Increase oxygen concentration in the water column to reduce odour**	**Increase oxygen concentration and salinity in water column, increase flushing of sediments**	**Remove largest source of pollution by bypassing the lagoon to discharge treated effluent at sea.**	**Reduce pollutant loading in selected influent stormwater systems**	**Reduce pollutant loading in selected influent stormwater systems**	**Speed breakdown of organic sediments to reduce oxygen demand**	**Return Potsdam WWTW to functioning at licensed effluent standards**
Expected benefits	Increased oxygen levels; improved scour;	Short-lived and localised effect in dissolved oxygen	Increased salinity; reduction in nutrients, algae; reduction in ammonia toxicity	Significantly reduced nutrient pollution and increased salinity in the lagoon; reduced flow and scour	Unlikely to alter situation significantly	Unlikely to alter situation significantly; remediation of reedbeds will provide for improved WWTW management	*May contribute to reduction in sediment buildup in shallow margins*	Significantly reduced suspended solids and ammonia concentrations
Modelled efficacy	**High** for oxygen levels **Negligible** for nutrients and algae	**Low positive** for oxygen levels	**Low** for oxygen levels **Moderate** for nutrients **High** for algae Reduces temperature	**Low** for oxygen levels **High** for nutrients and algae – greatest effect on eutrophic state	*Not modelled*	*Not modelled*	*Not modelled*	**High** for oxygen levels, nutrients and algae
Estimated costs	R 77 154 000 to R 133 137 000	R 1 524 000 to R 4 145 000 [capital] R 45 000 to R 550 000 per month [running]	R 10 000 000 to R 70 000 000	R 190 000 000	R 400 000 [Pilot]	*Not priced here – subject of separate project by W&S under contract 194C*	*Not priced here*	*Not priced here*
Minimum time to start	24 months	9 to 12 months	20 to 30 months	24 to 30 months	3 months	12 months	3 months	Various interventions already **completed or underway**
Time to achieve effect	36 months	Immediate	36 months	48 months	12 months	24-36 months	*Pending testing*	3 months
Ecological risks	Disturbance of benthic and shoreline habitats	None anticipated	Some loss of freshwater habitat (largely reedbeds) at and downstream of outfall	Impacts on benthic habitat along route of outfall pipeline; marine pollution impacts to be determined	None anticipated	None anticipated	***Unknown***; *to be determined by ecotoxicity testing*	None anticipated
Risk of failure	**Low** (but will require maintenance dredging)	**Moderate**	**Moderate**	**Moderate** (due largely to regulatory risk)	**High**	**High**	**Moderate**	**Low** (but will require ongoing maintenance)
Other negative impacts	Availability of disposal sites not confirmed Noise, disturbance, and increased turbidity during implementation	Noise and obstruction Possible increase in entrainment of sediments	Short-term disturbance of shoreline and benthic habitat along pipeline route and at intake / outfall	Reduced scour and flow into lagoon; summer mouth closures	None anticipated	None anticipated	*Product-dependent: to be determined by testing*	None anticipated
Approvals required	Environmental authorisation Water use authorisation	Environmental authorisation Water use authorisation	Environmental authorisation Coastal permits	Environmental authorisation Coastal Waters Discharge Permit	None (to be confirmed with authorities)	Environmental authorisation Water use authorisation	None (to be confirmed with authorities)	None

14.4 Timeframes for implementation

As set out in the terms of reference for this Remediation Plan (section 1.2), the focus is on short-term actions to address the poor and declining water quality. Although 'short-term' is a relative term, actions that will take longer to implement than the Potsdam WWTW capital upgrades are unlikely to be an effective use of funds and timeframes impact on the feasibility of actions. A generalised set of implementation timeframes is shown in Table 14-3.

Table 14-3. Indicative timeframes for implementation



15 RECOMMENDATIONS

15.1 Options recommended for implementation

Table 15-1 summarises the recommended actions in respect of each option assessed.

Table 15-1. Summary of recommended options

Intervention	Recommendation
Improve Potsdam functioning (remediate maturation ponds)	Implement with urgency (underway as of April 2023)
Potsdam capital works (expansion and upgrade)	Implement with urgency
1: Dredging	Proceed with design and approvals, to implement in parallel with WWTW capital works
2: Aeration	Implement **pilot** as soon as possible
3: Seawater flushing	Not a short-term measure; proceed with design for medium-term implementation
4: Marine outfall	**Not recommended** due to high project costs and long timeframes
5: Biofilters	**Recommended** non-critical intervention – implement a **pilot project**
6: Treatment wetlands	Not feasible at scale – but **remediate existing reedbeds** at Potsdam
7: Inoculation	**Recommended for *in situ* trials** before implementation

15.2 Recommended next steps

Water quality in the Milnerton Lagoon is a complex problem impacted by multiple pollution sources. Current efforts to address these sources of pollution are the subject of a suite of existing plans including the Lower Diep River Transversal Action Plan (dated April 2021), the Diep River Estuarine Management Plan (dated October 2022), and the Potsdam WWTW Improvement Plan (dated February 2023). **It is strongly recommended that the various plans and responses in preparation by the City be aligned, including through the incorporation of the findings of this Remediation Plan into the respective catchment- and estuary-scale planning.**

If the Potsdam WWTW works are concluded on schedule and function as expected, the Milnerton Lagoon would experience improved water quality from approximately 2025 onward, reaching optimal effluent standards by 2028. These capital upgrades are strongly supported, and should be pursued with urgency.

Bypassing and remediation of the maturation ponds at the Potsdam WWTW is a critical intervention that could result in an improvement in Milnerton Lagoon – or at least allow other in-lagoon measures to achieve a long-term improvement. The delays in obtaining the required equipment were significant, and it is recommended that the City procure sufficient standby pumps of the required specification to provide for future incidents of this nature at Potsdam or elsewhere.

Reedbed remediation at Reedbeds 1 to 3, and potentially separation of the reedbeds into isolated zones that contain contaminated inflows rather than allowing them to re-contaminate the whole reedbed are necessary interventions. These measures are not rapid and will not on their own improve lagoon quality, but should be pursued with urgency.

Diversion of low flows from the Milky Way pond and Du Noon into sewer is a necessary intervention but not capable of rapid implementation and would not provide immediate relief to the lagoon ecosystem.

While implementation of the catchment-scale pollution abatement strategy is clearly required, such interventions will not bring about immediate relief to the condition of the Milnerton Lagoon or facilitate the efficacy of short-term in-lagoon interventions considered in this report. **Table 15-2 overleaf sets out a recommended short-term implementation plan specific to the remediation measures considered in this report, together with indicative timeframes.**

Table 15-2. Implementation Plan for the first 18 months for remediation measures recommended in this report

Months 1-6	» Conclude remediation of maturation ponds at Potsdam and reinstate their use, together with UV disinfection. » Conclude remediation planning for the reedbeds at Potsdam with a focus on improving their capacity for management of surcharges and contaminated flows. » Procure and commence statutory processes for environmental and water use authorisations of aeration. » Procure and undertake feasibility design for dredging of Milnerton Lagoon. » Engage internally with responsible departments to confirm dewatering and disposal sites for dredged material – recommended sites are Loxton Road Park and Vissershok, respectively. » Procure and commence statutory processes for environmental and water use authorisations of dredging. » Engage the authorities to confirm statutory requirements for aerator installation » Procure and install aerators in the lagoon upstream of Erica Road. » Procure and undertake feasibility design for marine intakes and seawater flushing. » Test and determine the efficacy of selected bioremediation products for in situ use. If effective and feasible, procure and commence their use in the shallower areas of the lagoon.
Months 7-12	» Conclude detailed design for dredging of Milnerton Lagoon. » Complete environmental impact assessment and water use authorisation applications for dredging and submit for decisions. » Complete environmental impact assessment and water use authorisation applications for aeration and submit for decisions. » Procure and commence statutory processes for environmental and water use authorisations of marine intakes and seawater flushing.
Months 13-18	» Commence procurement for dredging of the lagoon » Conclude statutory processes for environmental and water use authorisations of marine intakes and seawater flushing

15.3 Legal alternatives: Emergency, disaster or incident provisions

Environmental legislation provides for alternative mechanisms in situations where an incident, emergency, or pollution event requires more urgent interventions than the standard authorisations set out in sections 4.4 and 14.1.9.

Section 30A of NEMA provides for emergency situations, where

(1) The competent authority may on its own initiative or on written or oral request from a person, direct a person verbally or in writing to carry out a listed or specified activity, without obtaining an environmental authorisation contemplated in section 24(2)(a) or (b), in order to prevent or contain an emergency situation or to prevent, contain or mitigate the effects of the emergency situation.

An "emergency situation" means a "situation that has arisen suddenly that poses an imminent and serious threat to the environment, human life, or property, including a "disaster" as defined in section 1 of the Disaster Management Act 57 of 2002, but does not include an incident referred to in section 30" of NEMA. There is no judicial authority on the interpretation of the phrase "emergency situation" in section 30A.

The pollution in the Milnerton Lagoon can be said to be a "serious threat to the environment", and while it is not a "situation that has arisen suddenly" it does appear to date back to specific incidents at the Potsdam WWTW. The phrase "emergency situation" means a "sudden" situation that includes a "disaster" as per the Disaster Management Act. The definition of "disaster" in the Disaster Management Act is as follows –

A progressive or sudden, widespread or localised, natural or human-caused occurrence which –

(a) Causes or threatens to cause –
  - *(i) Death, injury, or disease;*
  - *(ii) Damage to property, infrastructure, or the environment; or*
  - *(iii) Disruption of the life of a community; and*

(b) Is of a magnitude that exceeds the ability of those affected by the disaster to cope with its effects using only their own resources.

The pollution of the Milnerton Lagoon is a progressive, localised, human-caused occurrence which causes damage to the environment but is arguably not of a magnitude that exceeds the ability of those affected by the disaster (i.e. the City of Cape Town) to cope with its effects using only their own resources. It therefore does not fit into the ambit of the "disaster" definition in the Disaster Management Act.

Section 30A(2) provides for exactly what the request from the person referred to in subsection (1) must include (where known). These are –

(a) The nature, scope, and possible impact of the emergency situation.
(b) The listed or specified activities that will be commenced with in response to the emergency situation.
(c) The cause of the emergency situation.
(d) The proposed measures to prevent or to contain the emergency situation or to prevent, contain, or mitigate the effects of the emergency situation.

Once section 30A(2) is satisfied, the competent authority may direct the person to undertake specific measures within a specific time period in order to prevent or contain the emergency situation or to prevent, contain or mitigate the effects of the emergency situation. In the present scenario, this could include the remediation of the Milnerton Lagoon as remediation would have the effect of preventing, containing, or mitigating the effects of the pollution. However, the word "may" is important, as this gives the competent authority the discretion on whether to issue a directive or not. If the competent authority does not agree that the pollution of the Milnerton Lagoon is an "emergency situation" as described above, the directive will not be issued.

The National Water Act, 1998 also provides for the control of emergency incidents in section 20. Section 20(1) defines "incident" to include "any incident or accident in which a substance pollutes or has the potential to pollute a water resource, or has, or is likely to have, a detrimental effect on a water resource". Section 20(2) defines the "responsible person" to include "any person who is responsible for the incident, owns the substance involved in the incident, or was in control of the substance involved in the incident at the time of the incident". However, the Act is unclear as to what constitutes an "emergency".

Section 20(4) directs a "responsible person" to –

(a) Take all reasonable measures to contain and minimise the effects of the incident;
(b) Undertake clean-up procedures;
(c) Remedy the effects of the incident; and
(d) Take such measures as the catchment management agency may either verbally or in writing direct within the time specified by such institution.

Once the catchment management agency has issued the above directive, the person responsible for the emergency incident must comply with the above section.

Implications
Due to the poor and deteriorating water quality in the lagoon, it may be advantageous for the environmental competent authorities to consider alternative methods for the authorisation and/or enforcement of remedial measures. These could include measures in terms of the NEMA Duty of Care or provisions relating to the remediation of incidents or disasters.

15.4 Other recommendations related to model confidence

Based on the conclusions and modelling presented from this report the following recommendations are proposed for future study:

- The estuary as a system receives inflows from the river, stormwater system and the ocean whilst the authorities tasked with improving the water quality have no direct control over diffuse pollution or accidental spills. Phase 2 dredging is recommended as a practical method to improve the overall water quality of Diep River estuary, based on the water quality simulations presented in this study.
- The river inflow into the system should be gauged to accurately record the inflows.
- Additional water quality constituents such as (silicon and iron) should be measured at the inflows as well as more frequent sampling would be required.
- The ocean in the vicinity of the mouth should be sampled for water quality constituents
- A weather station in the area should be installed to collect all meteorological data for any future modelling. (The current Davis unit at the Nature reserve office needs to be recommissioned).
- In-situ SOD measurements should be performed to allow more accurate representation of the dissolved oxygen dynamics in the system.
- Salinity dispersion coefficients should be measured to improve the salinity simulations
- Additional field data should be collected to improve the current model or to implement a 2 dimensional or even 3-dimensional model such as:
    - Measured roughness coefficients (Manning or Chezy) as these are used to calibrate the hydraulics
    - Measured expansion and contraction coefficients
    - Weir flow coefficients especially for the high flows
    - Flow velocities (axially, lateral and vertical)

- The following in-situ measurements are required to improve the water quality simulations:
    - For dissolved oxygen the production, consumption and atmospheric reaeration
    - CBOD decay and settling rates
    - The Redfield ratio of the biomass in the estuary
    - Algal biomass maximum growth rates, respiration rates, nutrient preference, light growth limitation, nutrient growth limitation, light extinction and settling rate
    - Nutrient dynamics and settling rates

The previous surface water quality modelling is based on the following constraints:

- The Diep River flow is not gauged and all flows were derived from the SHETRAN modelling.
- Potsdam effluent discharge (volumetric) is not gauged or measured but rather calculated as is evident from the negative discharges recorded.
- No groundwater interaction with the system is accounted for in the modelling as there is no data on the volumes or water quality.
- No stormwater flows and quality are accounted for which have a direct influence on the water quality in the system.
- No spills into the system are accounted for as there is no sampled data on diffuse sources and the City's water quality sampling is too infrequent to capture these events
- The meteorological data is from ECMWF (European Centre for Medium-Range Weather Forecasts) which includes reanalysis of climate data based on a 80 km grid size and spliced with observed data.

- The tidal data is from the Hydrographic Institute and data gaps are patched to ensure a continuous data series
- No water quality sampling of the ocean was performed or available for the main water quality constituents viz. temperature, dissolved oxygen concentration, nutrients, salinity
- No sediment oxygen demand (SOD) was measured and in the model this variable was used as the final calibration factor for dissolved oxygen concentrations.
- The model does not account for a changing bathymetry which would affect water elevations, flow velocities and water quality.
- The scenario modelling showed the results of proposed management interventions to relative changes in the baseline scenario.

The rationale of this study was to investigate the effect of various management scenarios on the water quality of the Milnerton Lagoon. Even in the absence of the stormwater flows and quality, the model predicted that water quality improvements were possible especially for dissolved oxygen, the key water quality variable.

The confidence in the model can be increased if better input data is provided as this would increase the confidence in calibration as well as predictions.

Based on the conclusions and modelling presented from this report the following recommendations are proposed for future study:

- All flows into the system should be gauged to record the inflows.
- A study to estimate the volume and quality of the groundwater flows into the system.
- Surface water elevation must be logged to ensure the accuracy of the water balance.
- Tracer/dye studies (time of travel) should be undertaken for various flow conditions to further calibrate the model.
- Additional water quality constituents such as (algal concentration, silicon, suspended solids, organic carbon and iron) should be measured at the inflows as well as more frequent sampling would be required.
- A weather station in the area should be installed to collect all meteorological data for any future modelling.
- In-situ SOD measurements should be performed to allow more accurate representation of the dissolved oxygen dynamics in the system.

The following in-situ measurements are required to improve the water quality simulations:

- the number of algal groups, diatoms and zooplankton in Flamingo Vlei;
- Algal biomass maximum growth rates, respiration rates, nutrient preference, light growth limitation, nutrient growth limitation, light extinction and settling rate; and
- CBOD decay and settling rates.

Ideally, a model should be used as a starting point for limnological investigations of a waterbody, with the data and formulations continuously refined to reflect the increased understanding of the system and processes gained over time. Since water quality compartments are coupled, calibration of one compartment may affect other compartments making calibration difficult.

The general guidelines for future in-lagoon water quality sampling are summarised in the following table.

Table 15-3. Parameter guidelines and frequency required water quality sampling constituents (Cole and Wells, 2018)

Boundary condition		
Minimum parameter	**Additional parameter**	**Sampling frequency**
Inflow and outflow temperature	Conductivity, DO, pH, TDS	Daily or continuous
Total organic carbon	Dissolved and particulate organic carbon	Weekly and storm sampling
Total phosphorus	Total dissolved phosphorus Total inorganic phosphorus Dissolved inorganic phosphorus	Weekly and storm sampling
Nitrate+nitrite Ammonium nitrogen	Total Kjeldahl nitrogen Filtered total Kjedahl nitrogen	Weekly and storm sampling
	TSS Inorganic and volatile SS	Weekly and storm sampling
	Chlorophyll-a Dissolved silica Alkalinity	Weekly and storm sampling
In-Lake		
Temperature DO pH Conductivity	TDS	Monthly
Chlorophyll-a	Phytoplankton biomass and type	Monthly
Total organic carbon	Dissolved and particulate organic carbon BOD	Monthly
Soluble reactive Phosphorus Total phosphorus	Total dissolved phosphorus Total inorganic phosphorus Dissolved inorganic phosphorus	Monthly
Nitrate+nitrite Ammonium nitrogen	Total Kjeldahl nitrogen Filtered total Kjedahl nitrogen	Monthly
	Secchi depth	Monthly
	Total inorganic carbon Alkalinity	Monthly
	Total suspended solids Inorganic and volatile suspended solids	Monthly
	Dissolved/total iron Dissolved/total Manganese Dissolved/total Silica Total dissolved sulphide Sulphate Iron sulphide	Monthly

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ANNEXURES

A Sediment Quality Data

B Bathymetric Survey Report

C Sediment Grading Data

D Water Quality Data

E Request for Information

F Hydrodynamic Model Outputs

G Water Quality Model Outputs

H Decision Framework for Aeration Option