CRC for
Water Sensitive Cities

Eric Singleton Bird Sanctuary Constructed Wetland: Monitoring and assessment for optimal stormwater treatment performance

Carlos Ocampo (UWA) & Carolyn Oldham (UWA)

Australian Government
Department of Industry, Science, Energy and Resources

Business
Cooperative Research
Centres Program

Eric Singleton Bird Sanctuary Constructed Wetland: Monitoring and assessment for optimal stormwater treatment performance
Regional Project 6-4

Authors
Carlos Ocampo and Carolyn Oldham (The University of Western Australia)

ISBN 978-1-921912-44-3

Publisher

Cooperative Research Centre for Water Sensitive Cities
Level 1, 8 Scenic Blvd, Clayton Campus
Monash University
Clayton, VIC 3800

p. +61 3 9902 4985
e. admin@crcwsc.org.au
w. www.watersensitivecities.org.au

Date of publication: May 2020

An appropriate citation for this document is:
Ocampo, C.J. and Oldham C.E. (2020) Eric Singleton Bird Sanctuary Constructed Wetland: Monitoring and assessment for optimal stormwater treatment performance. Melbourne, Australia: Cooperative Research Centre for Water Sensitive Cities.

Acknowledgments

This project was funded by the Department of Biodiversity, Conservation and Attractions, City of Bayswater and the Chemistry Centre WA, under CRC Water Sensitive Cities Regional Project 6-4. Support in the field and the laboratory from Yemaya Smythe-McGuiness, Crista Santiago and Andrew van de Ven is gratefully acknowledged.
.

Table of Contents

Table of Contents **3**
List of figures **4**
List of tables **6**
Executive summary **7**
Context 7
Hydraulic performance 7
Nutrient, sediment and metal attenuation 8
Recommendations 8
Future monitoring to improve assessment of ESBS nutrient attenuation performance 9
1 Background **11**
1.1 Aim and scope of work 11
2 Study site **13**
2.1 The wetland components 15
3 Methodology **18**
3.1 Monitoring approach 18
3.2 Hydrological monitoring: Baseflow and stormflows 20
3.3 Water balance approach 22
3.4 Water quality sampling 23
3.4.1 Baseflow sampling 23
3.4.2 Stormflow sampling 24
3.5 Water quality concentration and load targets 25
3.6 Calculation of nutrient attenuation during stormflows 26
4 Water balance analysis **28**
4.1 Rainfall 28
4.2 ESBS inflow and outflows: Preliminary rating curves 28
4.2.1 Inflow rating at PIT1_IN 29
4.2.2 Outflow rating at OVF_W site 29
4.3 ESBS water storage: Baseflow conditions 30
4.3.1 Water levels: October 2017 – December 2018 31
4.3.2 Water levels: January 2019 – August 2019 32
4.4 ESBS water balance: Stormflow conditions 34
4.4.1 Event 1: June 2019 34
4.4.2 Event 2: June 2019 35
4.4.3 Event 3: July 2019 36
4.4.4 Event 4: August 2019 36
4.4.5 Event 5: August 2019 37
4.5 Slade Street drain inflows 38
5 Hydraulic performance **39**
5.1 Changes in inlet and outlet locations and flow rates 39
5.2 Hydraulic function under baseflow conditions 40
5.3 Hydraulic function under stormflow conditions 40
5.4 Impacts of tide 41
5.5 Impact of operations 41
5.6 Water balance findings summary 41
6 Nutrient attenuation **43**
6.1 Baseflow conditions: Concentration attenuation across ESBS treatment train 43
6.1.1 The gross pollutant trap 50
6.1.2 The sedimentation pond 51
6.1.3 The wetland 51
6.1.4 The Slade Street drain 55
6.1.5 Is the wetland achieving concentration targets? 56
6.1.6 Concentrations and attenuation during intended functioning periods 57
6.2 Impact of GPT operations 58
6.3 Baseflow conditions: Load attenuation across ESBS 58
6.4 Stormflow conditions: Concentration attenuation across ESBS 62
6.4.1 Variability of concentration nutrient species 68
6.5 Stormflow conditions: Load attenuation across ESBS 70

6.6 Baseflow conditions: Load attenuation across ESBS 70
7 Metal attenuation across ESBS components 73
8 Conclusions and recommendations 80
8.1 Conclusions 80
8.2 Recommendations 81
8.2.1 Operational functioning and performance of ESBS components 81
8.2.2 Future monitoring to improve assessment of ESBS nutrient attenuation performance 82
References 84
Appendices 86
Appendix 1A – Event hydrographs and water balance summary 86
Appendix 1B – Nutrient and sediment loads: Baseflow conditions 87
Appendix 2 – Surveying work for surface water site reference levels. 89
Appendix 3 – Recommended monitoring activities 91

List of figures

Figure 1 The Eric Singleton Bird Sanctuary constructed wetland location in Perth, Western Australia. Image source: Nearmap. 14
Figure 2 The ESBS wetland treatment components. 15
Figure 3 Design components, showing gross pollutant trap, sedimentation pond, open water and different marsh zones, and outflow and overflow to Bayswater Brook (modified from GHD 2013). 16
Figure 4 Outflow structures controlling discharge from ESBS; a) main Outflow point and associated surface overflow weir, b) Overflow structure and wooden boards weir used in management actions to control water levels in the wetland. 17
Figure 5 Location of monitoring and sampling sites at the Eric Singleton Bird Sanctuary constructed wetland. Image source: Nearmap.. 19
Figure 6 Photographs of monitoring equipment setup; a) water level sensor well at PIT1_IN, b) water level sensor (capacitance probe) at SED_P. 21
Figure 7 Flow meter deployment at a) the outlet pipe of PIT1_IN, and b) the centre pipe at OVF_W. 21
Figure 8 Camera stills used in flow monitoring at ESBS Outlet and Overflow: a) water exchange between Bayswater Brook and the wetland via weir structure, and b) discharge from wetland to Bayswater Brook at OVF_W under high water levels (note floodgates upward allowing discharge). 22
Figure 9 Automatic water sample collection at ESBS for event sampling: a) PIT1_IN, b) OVF_W. 24
Figure 10 Water sample collection strategy implemented at ESBS wetland during the period September 2017 – August 2019: a) monthly sampling for baseflow conditions, b) stormflow sampling during events. Shaded grey area indicates sampling time. 25
Figure 11 Rating curves for PIT1_IN. Orange dots correspond to the period when the GPT was blocked. 29
Figure 12 Rating curves for the OVF_W. Curves correspond to levels in the single central pipe of the culvert structure. Blue dots indicate low water conditions, and orange dots indicate high water conditions, in Bayswater Brook. 30
Figure 13 Water levels from October 2017 to December 2018. Water levels are shown for PIT1_IN, PIT2_B, the sedimentation pond (SED_P) and the wetland inlet (WET_IN). Data gaps represent sensors malfunctioning. 31
Figure 14 Water level time series from October 2017 to December 2018. Data is shown from Wetland inlet (WET_IN), Overflow (OVF_W) and Outflow (OUT_W). Data gaps represent sensor malfunctioning and loss of OVF_W sensor (from late August to early November 2018). 32
Figure 15 Water levels for selected monitoring stations from January to August 2019. Data is shown from the PIT1_IN, PIT2_B, SED_P, and WET_IN. 33
Figure 16 Water levels at selected monitoring stations from February to June 2019. Variation in levels was observed as flows responded to wetland management operations. Wetland levels increased to 0.80 m AHD due to the addition of two boards at the Overflow structure (OVF_W site), and later their removal in preparation for winter season inflows. 34
Figure 17 Event 1 flow hydrographs. The Inflow-BB_OUT hydrograph indicates inflow from Bayswater Brook to the wetland via the large weir at the Outlet structure. 35
Figure 18 Event 2 flow hydrographs. The OVF_W hydrograph shows good discharge conditions over the course of the event. 35
Figure 19 Event 3 flow hydrographs. The wetland responded to multiple rainfall events over the period. 36

Figure 20 Event 4 flow hydrographs. Refer to right y-axis for PIT1_IN hydrograph discharge values. 37
Figure 21 Event 5 flow hydrographs. Refer to the right y-axis for PIT1_IN hydrograph discharge values. 37
Figure 22 Water level for SLADE_S. Data were recorded between 20 June 2018 and 21 August 2019. 38
Figure 23 Total nitrogen concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 44
Figure 24 NO-x concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 45
Figure 25 Total phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 46
Figure 26 Filterable reactive phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 47
Figure 27 Particulate phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site. 48
Figure 28 Total organic carbon concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site. 49
Figure 29 Total suspended solids concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 50
Figure 30 Comparison of nutrient concentrations at OUT_W and OVF_W: a) total nitrogen; b) total phosphorus. Dashed line represents trigger value (ANZECC & ARMCANZ 2000). 53
Figure 31 Comparison of nutrient concentrations at OUT_W and OVF_W : a) NH3-N; b) NOx-N; c) FRP. Dashed line represents trigger value (ANZECC & ARMCANZ 2000). 54
Figure 32 Comparison of TSS concentrations at OUT_W and OVF_W. Trigger value corresponds to interim guideline concentration of 6 mg/L (Department of Water and Environmental Regulation). 55
Figure 33 Comparison of nitrogen species concentrations at the SLADE_S sampling site. Dashed line represents trigger value for TN (ANZECC & ARMCANZ 2000). 56
Figure 34 Phosphorus concentrations at SLADE_S. Dashed line represents trigger value for TP (ANZECC & ARMCANZ 2000). 56
Figure 35 Total nutrient loads for baseflow conditions: a) TN load; b) TP load. 59
Figure 36 Dissolved inorganic nutrient loads for baseflow: a) NH3-N; b) NOx-N; c) FRP. 60
Figure 37 Dissolved organic nitrogen loads for inflow and outflow sites. 61
Figure 38 Total suspended sediments load for inflow and outflow sites. 61
Figure 39 Event 1 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OUT_W. 63
Figure 40 Event 2 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W. 64
Figure 41 Event 3 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W. 65
Figure 42 Event 4 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W. 66
Figure 43 Event 5 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W. 68
Figure 44 Total aluminium concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 75
Figure 45 Total cobalt concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 76
Figure 46 Total copper concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 77

Figure 47 Total nickel concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 78
Figure 48 Total zinc concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline. 79
Figure 49 Areal coverage of the two surveying campaigns. Image sourced by Nearmap (22/11/2015). 89
Figure 50 Survey of the outflow weir structure cross-section and corresponding heights (AHD) values. Image sourced by Nearmap (date 7/09/2015). 90
Figure 51 Stage 1 monitoring and sampling locations for the upper treatment train components. Arrows indicate flow direction. Image source: Nearmap. 91
Figure 52 Stage one monitoring and sampling strategy for the wetland component. Arrows indicate flow direction. Image source: Nearmap. 92
Figure 53 Stage two monitoring and sampling strategy for groundwater interaction and in-within wetland processes. Arrows indicate flow direction. Image source: Nearmap. 93

List of tables

Table 1 Specifications of the different wetland components and associated structures. 15
Table 2 Wetland functional levels. 17
Table 3 Monitoring site descriptions and their codes. 18
Table 4 Summary of monitoring program. 20
Table 5 Nutrient concentration and load targets set over the last 20 years for the Bayswater Main Drain. 26
Table 6 Rainfall characteristics and tidal conditions during monitored events. 28
Table 7 Comparison of nutrient and metal concentrations at different sampling stations allows the assessment of the performance of individual components of the ESBS treatment train. 43
Table 8 Dates of GPT cleaning in 2017, 2018 and 2019. The cleaning on 6 September 2019 was undertaken by a new contractor, using a different method that did not discharge GPT effluent to the sedimentation pond. 51
Table 9 Measured concentrations during periods that corresponded to intended wetland design functioning. ANZECC guidelines are given for nutrients. 57
Table 10 Arithmetic mean concentrations for monitored events. ANZECC guidelines (for estuarine protection) are given for nutrients. Numbers in red represent concentration values exceeding the guidelines. 69
Table 11 Nutrient and sediments loads and EMC values for each event. EMC values in brackets expressed in mg/L. 70
Table 12: Load attenuation (%) across the treatment train. Hydraulic residence time (HR) in days is also provided. Operation refers to inflow rate at PIT1_IN after addition/removal of logs at DIV_W. N for the OVF site indicates weir boards in place (no outflow). N for the OUT site indicates blockage of submerged discharge pipe impeding discharge and (–) indicates lack of flow rate data to compute loads. 71
Table 13: Comparison of historical metal concentrations in Bayswater Main Drain with concentrations measured in the current study. Groundwater concentrations measured in 2008-09 are also shown. ANZECC guideline values are shown in the first column, and concentrations above ANZECC guidelines are highlighted in orange. 74
Table 14 Summary of event characteristics and water balance. Tp refers to time to peak level in wetland, Net Vol. Smax refers to net volume at the time of peak level in the wetland. 86
Table 15 Nutrients and sediments loads corresponding to PIT1_IN and OVF_W sites under baseflow conditions. 87
Table 16 Height values (AHD in meters) for water level control and referencing points. 90

Executive summary

Context

This report presents findings from monitoring and assessment of the performance of the Eric Singleton Bird Sanctuary (ESBS) in Perth, Western Australia.

The ESBS is situated on the banks of the Swan River, and tidal flows and high estuarine water have been observed travelling up Bayswater Brook from the river (GHD 2013). The impact of estuarine flows on the wetland water balance before its reconfiguration was quantified, highlighting the role of the tide in explaining water level variability in the wetland (GHD 2013). However, the wetland hydraulic and nutrient attenuation performance after its reconfiguration has not previously been assessed.

The reconfiguration of the ESBS was completed in October in 2015 and involved partial diversion of water from Bayswater Brook (< 200 L/s, GHD 2013) to a gross pollutant trap (GPT), installed at the inflow of the wetland. Brook water flowed out of the GPT into a sedimentation pond, then a marsh wetland, and finally out to the Bayswater Brook again, before being discharged to the Swan River. This treatment train system was predicted to remove up to 40 tonnes of sediment and rubbish per year, approximately 200 kg of phosphorus per year and 1.3 tonnes of nitrogen per year (GHD 2013), however the actual nutrient attenuation performance of the ESBS has not previously been assessed.

For this project, baseflow performance was determined during 2017 to 2019.

Rainfall event runoff was also intensively monitored and sampled from June to August 2019; a total of 382 mm was recorded over the 3-month period. Over this period, five storm events were monitored: one small (less than 10 mm), two minors (less than 40 mm) and two majors. Event 5 on 16 August comprised the 1-year ARI event, and was particularly important for analysing the wetland’s hydraulic response and nutrient attenuation capacity.

Hydraulic performance

The wetland water pathway for water treatment during rainfall events, is not currently working as intended in the original design during rainfall events. Results suggest that only one-third of the length of the treatment pathway is currently used for all events including the 1-year ARI event, because short-circuiting occurs between the two outflow points.

The major inflow to the wetland during rainfall events is water from the Bayswater Brook entering at the Outflow weir location. This inflow explains between 60% and 80% of the total inflow for the event. It is independent of the rainfall magnitude and is believed to have occurred since late 2017.

The original design considered the possibility of additional inflow during major events and high tide conditions (10-year ARI event). However, we have documented additional inflow for small events (less than 10 mm) and the 1-year ARI event. Under these conditions, it takes up to 48 hours for the wetland to return to normal.

These issues are not related to problems in the design, but to a substantial reduction of the conveyance capacity of a 400 m reach in the Bayswater Brook, around the Outflow weir location. Overgrown vegetation in the channel increased the water level in the brook by more than 0.3 m; water level reached 0.842 m AHD under wet season baseflow levels.

Inflow to the GPT via the diversion structure depends strongly on the GPT conditions. Outflow from the GPT into the sedimentation pond ranged from 5 L/s to 156 L/s under baseflow conditions (when blocked and cleaned respectively). Outflow from the GPT to the sedimentation pond reached 250 L/s during major events.

Water levels in Bayswater Brook, the status of the GPT, stormflows and tidal influence all impacted the wetland’s hydraulic performance. Therefore, we recommend implementing a maintenance strategy for Bayswater Brook, to restore its conveyance and decrease water levels across the wetland. Subsequently, additional flow measurements should be conducted to establish ratings for the site when it is operating as expected.

Nutrient, sediment and metal attenuation

We could not assess nutrient and sediment load attenuation across treatment train components, due to challenges in ratings curves at the inlets and outlets. However, we did analyse concentration reduction across wetland components.

The GPT appears to have been a source of total suspended solids (TSS) and nutrients for some of the monitoring period; at other times the flows through the GPT into the sedimentation pond were minimal. When the flows out of the GPT and into the sedimentation pond were minimal, and the main inflows into the wetland were via the Outflow, the sedimentation pond was not able to perform its expected function.

For part of the monitoring period, the area between the Overflow and Outflow sites (where inflows were occurring) and the Outflow appeared to be a source of nutrients and TSS. This area appears to be a stagnation area, with seasonal fluxes of nutrients and TSS being discharged.

The wetland achieved good attenuation of nutrients and TSS, during baseflow and also during a range of storm events. This result is despite the poor functioning of the GPT, the Outflow forming the main inflow to the wetland for part of the monitoring period, and the Outflow area contributing nutrients and TSS to the discharge. Except for the first storm event monitored (a first flush event), the TN and TP concentrations at the outflow of the wetland were below ANZECC guideline values (TN 1.2 mg/L; TP 0.065 mg/L).

Across the wetland system, the TN loads were on average attenuated during storms events around 20% and the TP loads were on average attenuated around 50%. High TSS load attenuation was achieved, on average, around 61%. This is better performance overall, than predicted by early MUSIC modelling of the system.

Due to the previous practice of discharging effluent from GPT cleaning into the sedimentation pond, it is not possible to determine the ability of the pond to attenuate sediments and nutrients. This need to be reassessed, now this practice has been discontinued.

Dissolved organic nitrogen made up 30–50% of the TN, and the wetland was a source of this nutrient to Bayswater Brook. This shows that attenuation of inorganic nitrogen was much higher than implied by the TN attenuation.

The vegetated marsh area of the wetland is responsible for significant removal of TSS and nutrients, and likely more than the sedimentation pond under the recent operational challenges.

The Slade Street drain discharges high concentrations of nitrate into the ESBS wetland. Concentrations of all other parameters (nutrients and metals) are very low in the drainage water. The high nitrate concentrations are considered a signature of inflows of contaminated groundwater into the brook. The small Slade Street sub-catchment appears not to receive sufficient surface flows to dilute the groundwater nitrate concentrations. While the same groundwater is expected to flow into the Bayswater Brook, the larger catchment area provides much larger volumes of surface runoff, to dilute nitrate concentrations.

The inflows to the ESBS from Bayswater Brook contain an order of magnitude lower metal concentrations than were measured in the brook 30 years ago. However, the inflow concentrations still exceed ANZECC guidelines. While the GPT and sedimentation pond have little impact on metal concentrations, the wetland reduced metal concentrations by 40–60%. In general, higher metal concentrations were observed at the Outflow site compared with the Overflow site.

Recommendations

1. Water levels in Bayswater Brook, the status of the GPT, stormflows and tidal influence all impacted the hydraulic performance of the ESBS wetland component. In the first instance, we recommend implementing a maintenance strategy for Bayswater Brook to restore its conveyance and decrease water levels during baseflow and minor runoff event conditions.

2. The expected attenuation of nutrients and TSS was achieved, even when the wetland was not operating as designed. However, we recommend some relatively inexpensive modifications to operation and maintenance to support additional attenuation; these modifications are outlined below.

3. An inexpensive flow meter (e.g. Starflow) with telemetry capabilities should be installed at the PIT1_IN, to provide real time monitoring of inflows, to inform operational decisions, including clearing Bayswater Brook and cleaning the GPT.

4. Given the key role of the vegetated marsh area in water quality improvement, care should be taken to ensure this area is maintained and supported, i.e. the vegetation is not harvested.

5. Given the good performance of the ESBS despite the poor functioning of the GPT, consideration should be given to the GPT effectiveness, relative its maintenance cost. We recommend exploring the option of installing a simple bypass mechanism inside the GPT pit, to convey water directly from the diversion weir in Bayswater Brook through to the sedimentation pond. If the GPT is kept as a component of the ESBS treatment train, regular monitoring of its condition is critical, along with an associated maintenance schedule. The ability of the GPT to remove gross pollutants should also be assessed.

6. The location and water level control of the outflows should be reviewed. Consideration should be given to simplifying the wetland flow paths and water level control operations. The floodgates are not operating as intended, and the weir boards sometimes exacerbate hydraulic challenges within the wetland. It is possible that simplifying the wetland flow paths will not impact nutrient attenuation performance, as long as the vegetated marsh areas are not impacted. The semi-permanent installation of weir boards at the Overflow structure should be considered, along with removing the floodgates. The ESBS has shown it manages well as an intermittent storm and tidal influenced system; high water induced flooding of the wetland (whether due to tides or storms) subsided within 48 hours, and did not affect the performance of the system. Removing the overflow mechanism and floodgates establishes a simpler and longer flow path for inflowing water, prevents the creation of dead zones within the wetland, and removes the potential for short circuiting.

7. Overall, the system performance will likely benefit from simplified flow paths (bypassing the GPT, leaving the overflow weir boards in place, and removing the tidal flood gates). This approach would also greatly reduce the maintenance costs for ESBS. Any change in the system configuration should be accompanied by ongoing monitoring to assess performance under the new configuration.

Future monitoring to improve assessment of ESBS nutrient attenuation performance

The following actions are recommended to better quantify ungauged inflows and outflows (with or without a change in wetland configuration), improve representativeness of water sampling sites, and improve within-wetland processes understanding:

1. Collecting large composite water samples (i.e. 4–20 L) should be considered during baseflow conditions, now that wetland dynamics are better understood. Three autosamplers should be used to overcome practical limitations of water sample collection at all sampling points.

2. The reliability of the Overflow rating is low, because it is limited to a few flow scenarios and the central pipe of the culvert structure. The rating curve cannot be applied to the site when weir boards are added/removed to the overflow structure. More investigation is recommended to quantify discharge under different wetland operational scenarios.

3. Additional flow measurements should be conducted to establish ratings for ungauged inflow and outflow sites when the treatment train is operating as intended. These sites are primarily the inflow from the Slade Street drainage pipe and the outflows from the sedimentation pond and the Outflow site.

4. Proper quantification of groundwater discharge into the wetland component should be undertaken. This work should include measuring point-source discharge via the Slade Street drainage pipe and diffuse-source discharge at the eastern boundary of the wetland. Recommendations for further investigation of within-wetland processes are provided in the Appendix 3.

5. As the practice of discharging effluent from GPT cleaning into the sedimentation pond has now ceased, it is now possible to assess the performance of the sedimentation pond to attenuate sediments and nutrients. This will require operational maintenance (e.g. dredging) to reinstate the pond to as-intended sediment and water storage capacities.

6. A change in the outlet sampling location of the sedimentation pond is needed to facilitate a more robust assessment of the pond's ability to attenuate concentrations. One possible alternative location is the overflow grate at SED_P, and preliminary sampling should be conducted to confirm the representativeness of this alternative location (see Appendix 3).

7. The installation of a dissolved oxygen logger between the overflow and outlet structures should be considered, to confirm the presence of anoxic waters in the wetland component.

8. Additional monitoring should be considered during the spring season, to determine differences in NH3-N attenuation across the wetland component. Recommended locations for wetland water sampling are provided in Appendix 3.

1 Background

The Eric Singleton Bird Sanctuary (ESBS) wetland is situated adjacent to the Bayswater catchment main drain, known as Bayswater Brook, and the banks of the Swan River, in Western Australia. The location of the ESBS made it suitable to be reconfigured to take flow from Bayswater Brook, and thus become part of an end-of-catchment treatment train.

The ESBS wetland was reconfigured under a $3 million partnership project between the City of Bayswater and the Swan River Trust (now Department of Biodiversity, Conservation and Attractions). The project stemmed from the Bayswater Brook Action Plan and the Bayswater Brook Local Water Quality Improvement Plan (BBWQIP) which identified the ESBS as a priority project to reduce nutrients entering the Swan River. It aimed to remediate a severely degraded wetland into a nutrient stripping wetland, to remove pollutants from the Bayswater Brook and to improve its water quality before discharging into the Swan River. Inflow from the Bayswater Brook also secured a sustainable water source for the wetland and maintained water levels, while aiming to restore, manage and remediate contamination at the site, and improve the ecological value of the wetland.

The ESBS reconfiguration was completed in October of 2015 and involved partial diversion of water from Bayswater Brook (< 200 L/s, Fortier & GHD 2013) to a gross pollutant trap (GPT), installed at the inflow of the wetland. The GPT was predicted to reduce total suspended solid (TSS) loads by 30%, removing up to 28 tonnes of sediment and rubbish per year, and reduce TP loads by 15%, removing approximately 85 kg of phosphorus per year, from the discharge to the Swan River (GHD 2013). However, the GPT's actual performance has not previously been assessed.

After passing through the GPT, 25% of flow was designed to be directed to a sedimentation basin and then the wetland, and 75% of flow was designed to be diverted back to Bayswater Brook (GHD 2013). In the reconfigured wetland, water flows in a circuit from the north western inlet, through alternating sections of open water and vegetated marsh, to the outlet to Bayswater Brook, along the south western edge. The GPT and the rehabilitated overall wetland system was predicted to remove 45% of TSS loads (equating to 32 tonnes of sediment), 35% of total phosphorus loads (equating to 200 kg of phosphorus) and 20% of total nitrogen (equating to 1.3 tonnes of nitrogen), from the diverted brook waters per year (GHD 2013). However the actual performance of the wetland system has not previously been assessed.

Since wetland restoration, groundwater seepage has been observed and monitored along the north eastern edge. The seeps have higher electrical conductivity (EC), lower levels of dissolved oxygen (DO) and higher concentrations of nitrogen and ammonia, than measured in Bayswater Brook. The impact of this seepage on nutrient attenuation performance has not previously been assessed and has not been included in this study.

The ESBS is situated on the banks of the Swan River, and tidal flows and high estuarine water have been observed travelling up Bayswater Brook from the river (Fortier & GHD 2013). The impact of estuarine flows on wetland hydraulics and nutrient attenuation performance has not previously been assessed.

The Department of Biodiversity, Conservation and Attractions (DBCA) identified a number of knowledge gaps relating to the hydrological and nutrient attenuation performance of the wetland. Given the significant investment in the wetland, and its role as a key nutrient intervention point for the catchment, addressing these knowledge gaps was a priority. This project involved collaboration between DBCA, the City of Bayswater (CoB), Chemistry Centre WA (ChemCentre) and the University of Western Australia (UWA), through the Cooperative Research Centre for Water Sensitive Cities and constitutes a significant contribution to the Swan Canning River Protection Strategy.

1.1 Aim and scope of work

This study aimed to answer the following questions:

- What is the hydrological and nutrient attenuation effectiveness of the wetland?
- What is the nutrient attenuation performance of each wetland component?
- What effect do wetland management operations have on the overall wetland performance under baseflow conditions?
- What are the changes in nutrient concentrations and speciation from the inflow to the outflows?
- How does the wetland perform during stormflow events?

- Is the wetland achieving its nutrient attenuation targets?

This project monitored the ESBS from October 2017 to August 2019 to assess the system over a full water year cycle, across rainfall events of different magnitude and under a range of management operation actions, to determine their impact on wetland hydrological and nutrient attenuation performance.

Hydrological data was used to develop a water balance and assess wetland hydrological performance. Water quality data was then used to determine nutrient removal performance and assess overall water quality status for other parameters of interest such as heavy metals. These assessments aimed to inform future management and operational activities by the City of Bayswater, improve maintenance schedules for wetland components to ensure optimal performance, and provide guidance on wetland performance assessment in lowland areas of the Swan Canning River Estuary including those impacted by tide.

2 Study site

The study site occupies approximately 5 ha and is located 9 km north east of the City of Perth (Figure 1). It is bounded by residential property to the west, Guildford Road to the north, Riverside Gardens to the east and the Swan River to the south. The area presents a Mediterranean climate characterised by dry, hot summers and wet winters and receives an average annual rainfall (1980 – present) of 725 mm and pan evaporation of 1800 mm (BOM 2019).

The Bayswater Main Drain (Bayswater Brook) drains a residential catchment of 2600 ha (GHD 2013); the wetland also receives water from the Slade Street sub-catchment (1 ha). The site was previously part of a seasonally wet depression in Riverside Gardens which was drained in the 1950s and then altered to become a permanent wetland in the 1970s when the surrounding land was used for landfill (GHD 2013). It is understood that a clay bund was constructed around the landfills to prevent leachate from entering the wetland site. From 1979 until its reconfiguration, the wetland was artificially recharged with groundwater during summer periods to support a bird population and prevent acidification from in-situ Potential Acid Sulfate Soils (PASS). After the reconfiguration in 2015, the diversion of Bayswater Brook water maintained wet conditions in the wetland (i.e. groundwater replenishment was no longer required). As with other constructed wetlands, it has also improved amenity, habitat and recreational opportunities for the community.

The reconfigured wetland has an elevation range RL -0.85 to +1.2 m Australian Height Datum (AHD), compared with RL -0.5 to +0.5 m AHD before reconfiguration. The site has long been a marsh area and is situated within Alluvium, as light, yellow brown sandy silt, soft when moist and with variable clay content (GHD 2013). The shallow sandy soil can create seasonally perched water tables; the regional water table typically rises over winter to incorporate the perched layers into a continuous saturated soil profile.

The design of the reconfigured wetland was constrained by (GHD 2013):

- a seasonally high water table
- intrusion of river tides and salinity
- a small hydraulic gradient across the lake (± 200 mm) and high water levels during winter baseflow conditions in Bayswater Brook
- an inability to implement control structures within Bayswater Brook
- maintaining a shallow wetland with an average depth of 0.5 m
- lake sediments that are potentially acid sulfate soils
- the area/volume available was smaller than required to treat dry season and annual average flow in Bayswater Brook and its first flush
- the requirement to optimise the use of in-situ material, local provenance material and maintain the pre-existing impermeable clay layer.

The reconfigured design criteria are listed in GHD (2013), and included:

- shallow gradients (mostly 1:6) and stable embankments
- maintenance of a permanent water body
- creation of bird habitats
- active management of potential acid sulfate soils
- maintenance of low water velocities (<0.05 m/s) to minimise sediment resuspension
- wetland length to width ratio > 5:1
- hydraulic efficiency > 0.5
- hydraulic retention time <48 hours; ARI 1 year HRT < 21 hours.

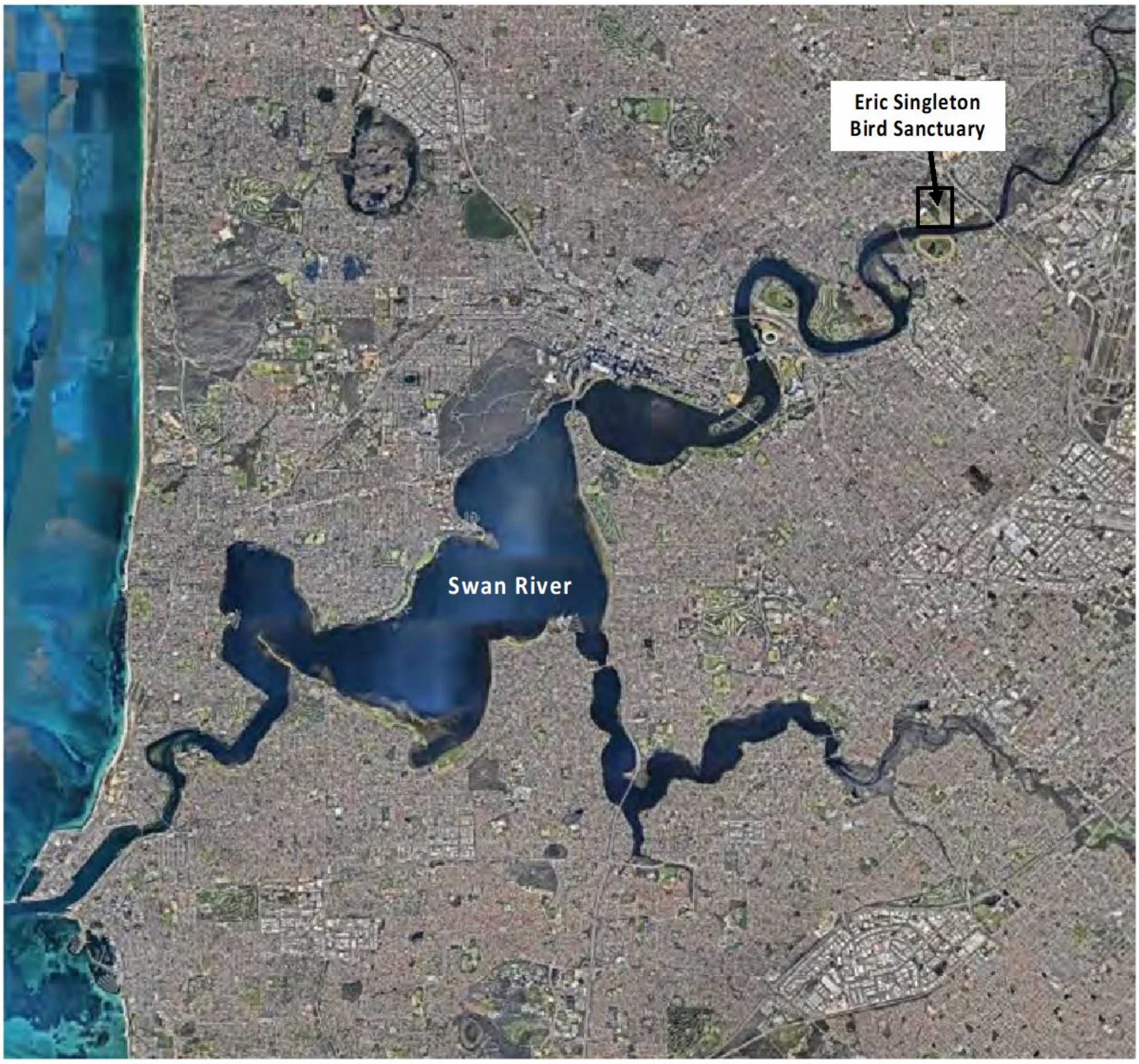

Figure 1 The Eric Singleton Bird Sanctuary constructed wetland location in Perth, Western Australia. Image source: Nearmap.

2.1 The wetland components

The wetland components and associated inflow and outflow structures are presented in Figure 2. Water is diverted directly from the Bayswater Brook via a weir structure and a 0.5 m diameter concrete pipe.

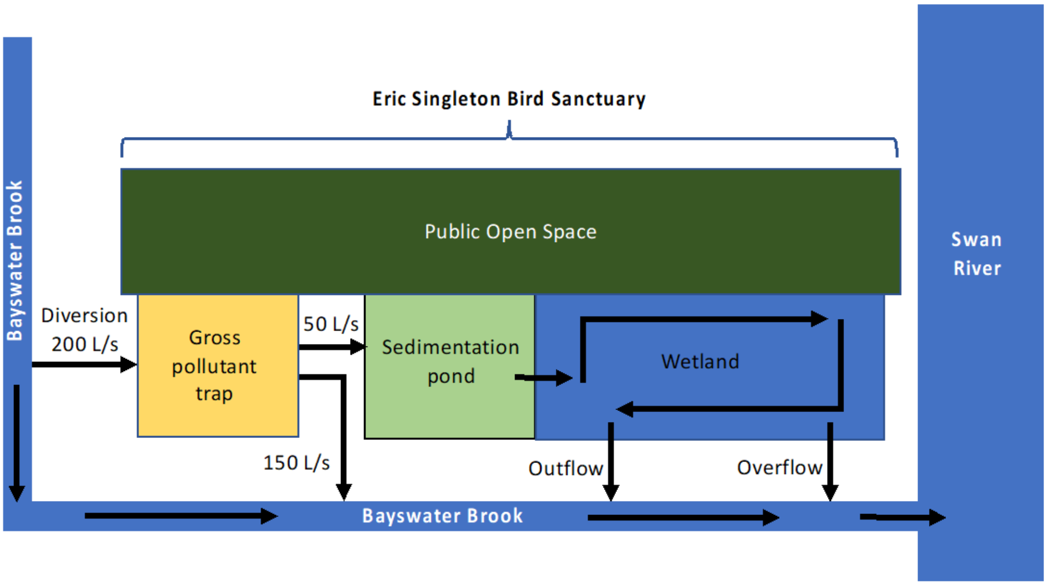

Figure 2 The ESBS wetland treatment components.

Table 1 summarises the main characteristics and associated wetland structures, while Figure 3 and Figure 4 show a schematic of the wetland components and photographs of the outflow structures respectively. Detailed information regarding physical dimension, design specifications, construction materials and landscaping, and vegetation species for the wetland area can be found in original design documents by GHD (2013).

Table 1 Specifications of the different wetland components and associated structures.

ESBS components	
Date of construction	2015
Diversion weir	Concrete weir structure in Bayswater Brook, with removable wooden boards (jarrah stop logs). Diversion of 200 L/s to GPT during baseflow conditions. Water conveyed to the GPT structure via a single concrete pipe (58.5 m length, 0.45 m diameter).
Gross pollutant trap (GPT)	Modified HumeGard GTP with dual outlets. Dimensions are 3 m x 2.5 m x 2.8 m (W,L,H). Concrete inlet and outlet pipes (0.45 m diameter). Flow partitioning returns 150 L/s to Bayswater Brook and 50 L/s enters the sedimentation basin.
Sedimentation pond	Surface area of 410 m^2 and volume of 248 m^3. Maximum depth of 1.3 m. Inflow, from GPT, via a concrete pipe (42 m length, 0.45 m diameter). Outflow to wetland via an overflow pit (e.g. normal level) and additional outflow via surface weir (e.g. high flow during storm events).
Main wetland area	Surface area of 27,000 m^2 and volume of 8,816 m^3 for normal level.

	Surface area for operational levels of 29,700 m^2, and extended detention volume of 14,822 m^3. Vegetated area (shallow pools); Open water areas (depth 1 m) followed by submerged marsh, deep marsh zones (depth 0.45–1 m) and shallow marsh (depth 0.15 m) in the wetland bend areas.
Outlet mechanism	An Outflow structure comprises submerged concrete pipe (0.4 m diameter) and a Penstock watergate. Movable floodgate designed to stop discharge and avoid tidal intrusion or stormflow from the Bayswater Brook. A large overflow weir (10 m wide, 6 m length, flow depth up 0.5 m) allows discharge to Bayswater Brook for high water levels (> 0.9 m AHD). See Figure 4a for details. An Overflow structure comprises a concrete culvert (3 m wide x 5 m length, 3 x 0.45 m pipes with movable floodgates) and removable weir boards (board dimensions 1 m length x 0.1 m high). See Figure 4b for details.

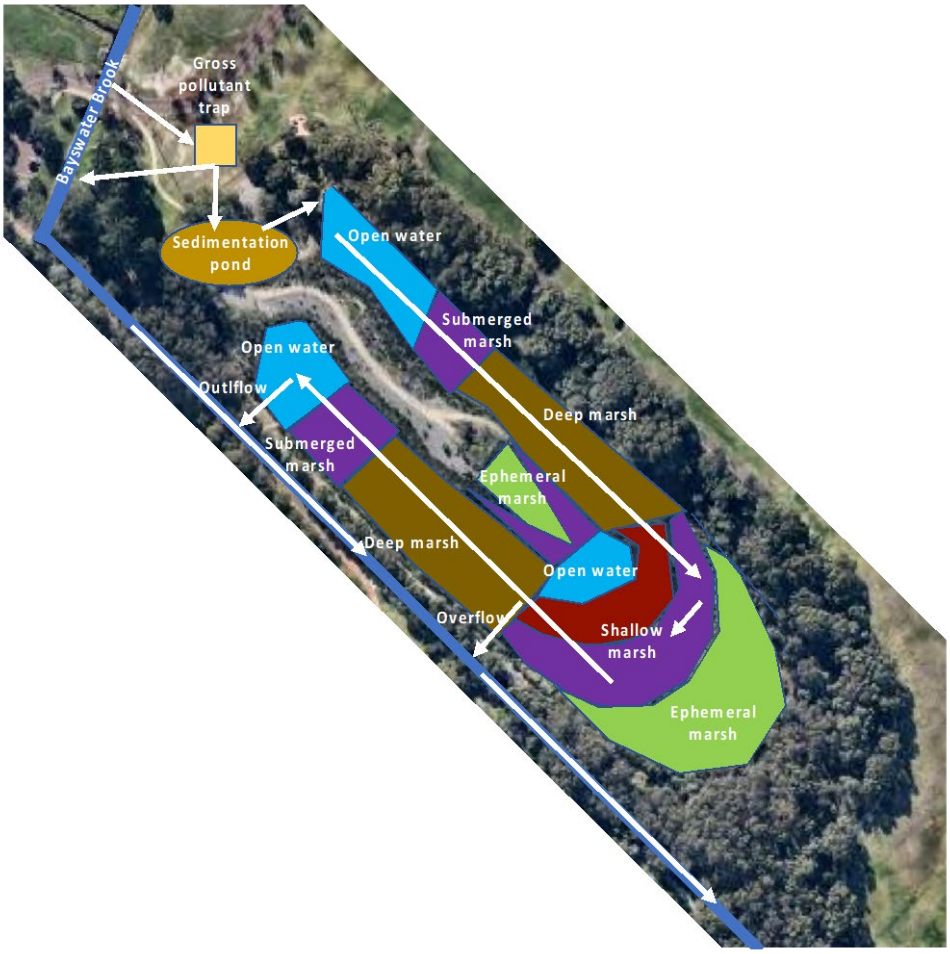

Figure 3 Design components, showing gross pollutant trap, sedimentation pond, open water and different marsh zones, and outflow and overflow to Bayswater Brook (modified from GHD 2013).

Figure 4 Outflow structures controlling discharge from ESBS; a) main Outflow point and associated surface overflow weir, b) Overflow structure and wooden boards weir used in management actions to control water levels in the wetland.

During the wetland design, a hydraulic gradient of 50 mm was considered the minimum acceptable for adequate flow across the sedimentation pond and wetland. This minimum gradient constrained the wetland outlet invert level to 0.4 m AHD (Table 1); this outlet invert level created challenges for the flow dynamics across the wetland as outlined in Section 5.3.

Table 2 Wetland functional levels.

Level ID	Level (m AHD)
Normal water level (summer)	0.45
Operating water level (autumn, winter, and spring)	0.75
Embankment level (Bayswater Brook-ESBS western boundary)	1.1
Diversion weir invert level	0.67
Sedimentation pond inlet invert level	0.45
Wetland outlet invert level (pipes at Outflow and Overflow structures)	0.40

3 Methodology

3.1 Monitoring approach

The monitoring program conducted for this project, was undertaken from October 2017 to August 2019 and involved both baseflow (monthly) and storm events sampling strategies for hydrological and water quality data. Figure 5 shows monitoring site locations and Table 3 lists monitoring site codes used throughout this report. Table 4 presents an overview of the complete monitoring program and a description of the sampling methodology is provided in the following sections.

Table 3 Monitoring site descriptions and their codes.

Monitoring site code	Monitoring site description
DIV_W	Weir in Bayswater Brook that diverts brook flows into the gross pollutant trap (GPT). Location for water sample collection and water quality monitoring (also referred as DIVERSION in Lab reports).
PIT1_IN	Sampling pit downstream of the GPT, in the flow directed to the sedimentation pond. This represents inflow to the sedimentation pond and the wetland system. Water level was monitored.
PIT2_B	Sampling pit downstream of the GPT, in the flow returned to Bayswater Brook. It documents water level variability of the brook.
SED_P	Sedimentation pond. Water samples were collected at its inlet and water level was monitored at its outlet (also referred as SEDPOND).
SLADE_S	Sampling pit at Slade Street, accessing water drained from nearby residential area. Water level was monitored and water samples were collected inside the pit. Water level was monitored.
WET_IN	Main inflow to the wetland, after the sedimentation pond. Water samples were collected at a submerged pipe entering the wetland.
OVF_W	Overflow structure with weir boards, acting as the main outflow from the wetland and monitored during the winter season (also known as OVERFLOW in Lab reports). Water level was monitored ~ 20 m into the wetland and water samples collected at the weir board location.
OUT_W	Outflow structure with large weir, designed as the main outflow from the wetland (also known as OUTFLOW in Laboratory reports). Water level and samples were monitored and collected at the watergate.
BB_OUT	Bayswater Brook water level station opposite the OUT_W. This site was used to monitor water exchange between Bayswater Brook and the wetland via OUT_W.
BB_OVF	Bayswater Brook water level station opposite the OVF_W. This site was used to assess discharge conditions at OVF_W.

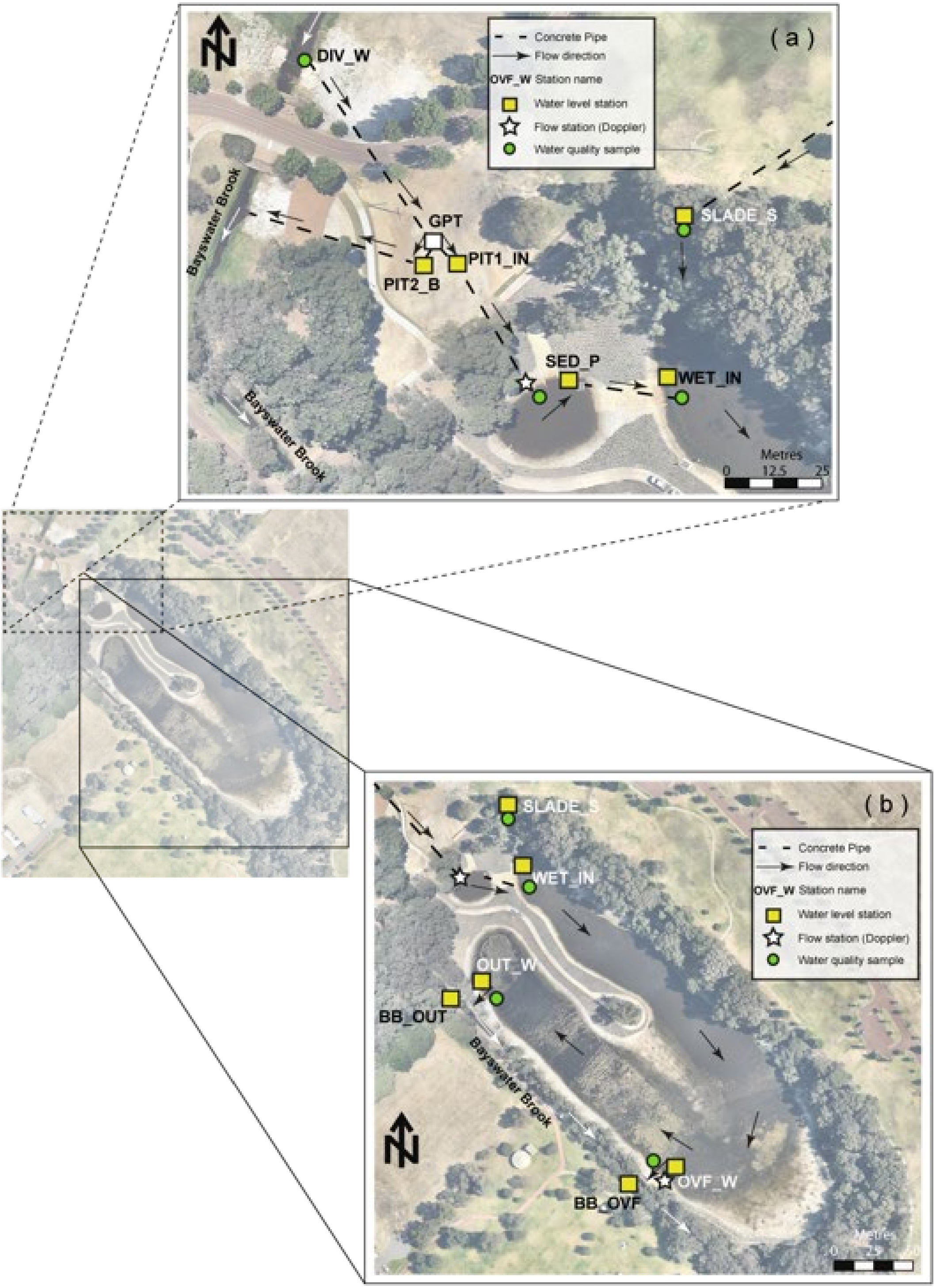

Figure 5 Location of monitoring and sampling sites at the Eric Singleton Bird Sanctuary constructed wetland. Image source: Nearmap.

Table 4 Summary of monitoring program.

Component	Parameters	Frequency	Site codes
Surface water			
Water quality	Total nitrogen (TN) Total phosphorus (TP) Nitrate/nitrite (NOx-N) Total Kjeldahl nitrogen (TKN) Ammonia (NH3-N) Filterable reactive phosphorus (FRP) Total suspended solids (TSS)	Event based sampling	PIT1_IN and OVF_W
	All the above plus: Metals (dissolved and total) Temperature, pH, electrical conductivity (EC), dissolved oxygen, turbidity, and redox potential (ORP)	Monthly	DIV_W, SED_P, WET_IN, OVF_W, OUT_W and SLADE_S
	Temperature Electrical conductivity (EC)	Continuous (10-minute intervals)	PIT1_IN and OVF_W
Hydrology	Water level	Continuous (5- to 10-minute intervals)	PIT1_IN, PIT2_B, SED_P, SLADE_S, WET_IN, OVF_W, OUT_W, BB_OUT and BB_OVF
	Water velocity (Doppler)	Event based sampling (2- to 5-minute intervals)	Outlet pipe of PIT1_IN and OVF_W

3.2 Hydrological monitoring: Baseflow and stormflows

Continuous hydrological monitoring stations were installed initially at nine surface water sites within ESBS wetland (Figure 5) and later reduced to seven, due to data redundancy found at two wetland sites (wetland bend areas). The monitoring undertaken and period at each station is outlined in Table 4 and Appendix 1-A; photographs of the monitoring setup can be seen in Figure 6 and Figure 7.

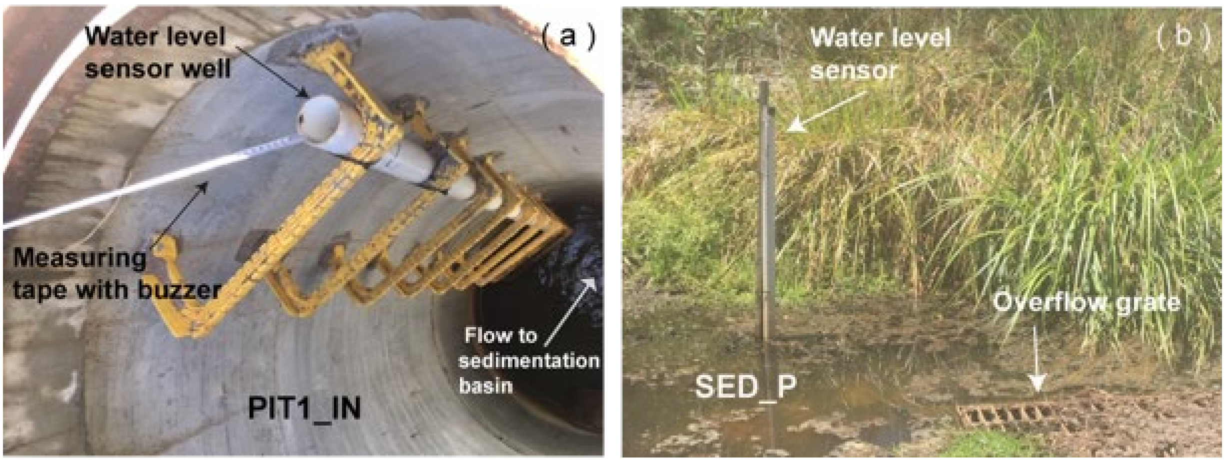

Figure 6 Photographs of monitoring equipment setup; a) water level sensor well at PIT1_IN, b) water level sensor (capacitance probe) at SED_P.

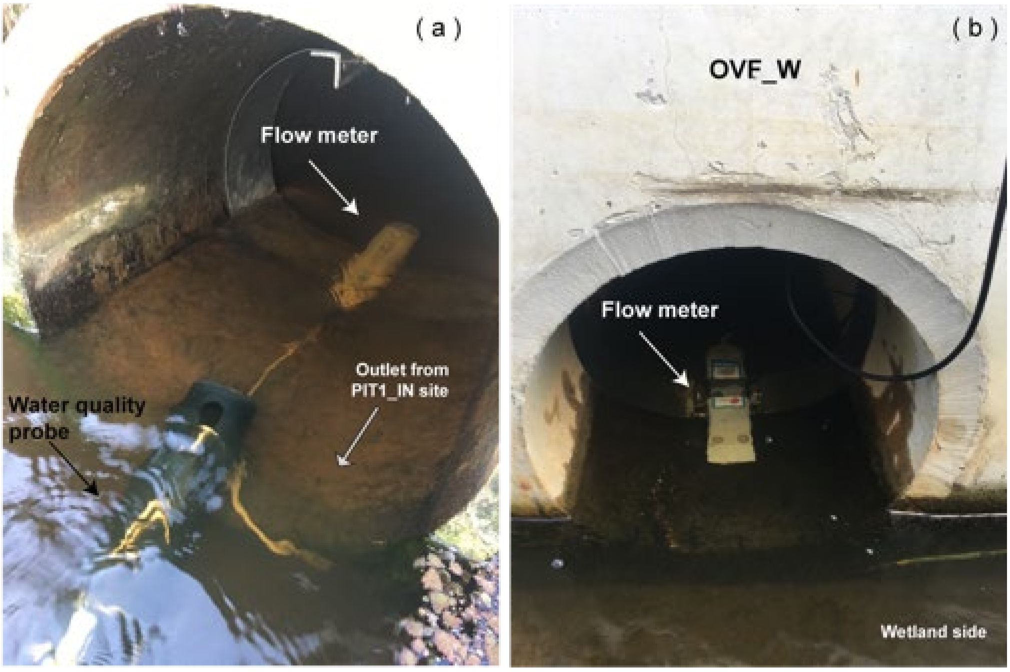

Figure 7 Flow meter deployment at a) the outlet pipe of PIT1_IN, and b) the centre pipe at OVF_W.

Opportunistic discharge measurements (DM) were conducted to develop rating equations for inflow and outflow sites over the course of rainfall events. Acoustic Doppler velocimeters with a pressure transducer water level sensor (Unidata Starflow and Sontek IQ) were deployed inside pipes at PIT1_IN and OVF_W, to record water mean velocity and depth at 2- to 5-minute intervals (Figure 7); flow discharge rate was computed using the Area-Velocity method. Rating of the sites required additional water level sensors at upstream and downstream locations of the Doppler deployment points to assess downstream control on flow discharge. These additional water level measurements were done at SED_P (for PIT1_IN) and at BB_OVF (for OVF_W) to characterise high flow conditions and tidal incursion into Bayswater Brook.

Theoretical discharge values were also estimated for each station based on flow hydraulic conditions and the geometry of pits and pipes following the Bodhaine (1968) method for indirect peak flow estimates. These estimates verified flow type conditions and flow magnitude provided by Doppler measurements.

At OUT_W, the weir structure was also monitored, because water exchange between the wetland and the Bayswater Brook could occur under high water level. Surveying work provided the necessary weir dimensions and elevation thresholds for water overtopping (see Appendix 2), and a water level sensor (BB_OUT, see Figure 5 for location) and field camera recorded water level dynamics for discharge computations and flow direction (e.g. from Bayswater Brook to wetland or vice versa). Discharge was computed using the Area-Velocity method by means of Manning's equations. Figure 8 illustrates flow conditions captured by the cameras during high flow in the Bayswater Brook at the OUT_W and OVF_W.

Figure 8 Camera stills used in flow monitoring at ESBS Outlet and Overflow: a) water exchange between Bayswater Brook and the wetland via weir structure, and b) discharge from wetland to Bayswater Brook at OVF_W under high water levels (note floodgates upward allowing discharge).

Water levels in nearby groundwater bores were not monitored in the present work due to difficulties in finding existing bores from previous groundwater investigations (GHD 2013). The pipes were either decommissioned or the casing caped and buried with soil around the public open space.

Groundwater level dynamics were indirectly monitored by water level inside the Slade Street drainage pit (SLADE_S in Figure 5), recording both stormflow and baseflow variability before discharging into the wetland. Its representativeness of groundwater discharge is discussed in Section 4.5.

3.3 Water balance approach

Two different approaches were used to compute inflow and outflow volumes to the wetland: one for baseflow conditions (during dry weather) and another for stormflow periods (during monitored events).

The water balance for stormflow conditions (rainfall events) aimed to identify the major inflows and outflows that explained the observed change in wetland water storage, through to its peak during an event. This approach followed a simple dynamic mass balance equation, representing changes in water storage as the result of all inflows and outflows to the wetland. The event water balance was defined as:

$$\frac{dS}{dt} = R + Q_{IN} + Q_{UNG} + Q_{OVER} + Q_{OUT} \pm Q_{WEIR} \pm Q_{BANKEX} \quad (1)$$

where *dS/dt* represents the rate of change in wetland water storage (m^3/time unit), *R* is the volume contributed by direct rainfall to the wetland surface (m^3), Q_{IN} is the volume entering the wetland at the sedimentation pond, Q_{UNG} represents ungauged surface runoff generated by surrounding areas (park and residential) over the duration of the rainfall event (m^3), Q_{OVER} and Q_{OUT} correspond to volumes discharged at the wetland Overflow and Outlet structures respectively, and Q_{WEIR} and Q_{BANKEX} represent volumes discharge in or out the wetland (m^3) at the OUT_W weir and the wetland's western boundary embankment respectively.

Some of the water balance terms can be neglected depending on the magnitude of the rainfall event and the height of the water in Bayswater Brook. Groundwater contribution was assumed to be minor and consequently neglected in the event water balance computation.

Major inflows and outflows to the wetland were identified as those that explained the observed variation in the wetland water levels, for example, from pre-event to peak levels, determined as the net volume gained by the wetland, at a time at which inflow and outflow rates are also equal (Chow et al. 1988).

Inflow and outflow hydrographs were computed using the corresponding water level records and rating curves for each site at 10-minute time intervals; the volumetric contribution was then obtained by numerical integration of the event hydrographs.

3.4 Water quality sampling

The ESBS monitoring Sampling and Analysis Plan (SAP) developed by McGuinness (2017) was mostly followed for water quality sampling but later altered to manage logistics and site specifics constraints. The SAP specified collecting a large composite sample (20 L) over a few hours, from which a sub-sample would be obtained. Although conceptually correct to account for a 12-hour residence time in the wetland, the task required approximately 15 hours to complete all six intended locations (see green dots in Figure 5 for locations) by a two-operator team. Knowledge about the potential effect of tidal intrusion into the wetland was not available before the sampling, and this issue needed to be assessed before attempting such a complex strategy for water sampling.

The baseflow (monthly) sampling used the standard grab sampling collection approach, while the stormflow (events) sampling used automatic discrete water sampling collection (sequential) by means of autosamplers.

3.4.1 Baseflow sampling

The baseflow sampling strategy used standard manual grab sample collection, but modified to accommodate two 1 L bottles, one for nutrient analysis and the other for metal analysis. This approach provided sufficient volume for rinsing and collecting filtered and unfiltered samples and yet ensuring that water samples were all collected at the same sampling point.

Research teams from UWA, and occasionally joined by DBCA personnel, conducted the water sample collection and water quality monitoring activities at six locations, as shown in Figure 5.

In-situ measurements of temperature, specific conductance (i.e. electrical conductivity at 25^{o}C), pH, redox potential, dissolved oxygen and turbidity were taken using a multi-parameter probe (YSI EXO2). The multi-parameter probe was attached to a floating frame and pushed into the wetland interior at the WET_IN (see Figure 5) to avoid stirring sediments. As part of the SAP, a Field Observation Form (FOF) for each sampling day was completed on site and emailed to DBCA personnel the following day.

Water samples for dissolved nutrient and metal analyses were filtered in the field (0.45 µm), and all samples were stored on ice and transported to the ChemCentre Laboratory, Bentley, complying with ChemCentre Chain of Custody (CoC) processes.

3.4.2 Stormflow sampling

Water sampling was done using automatic sequential samplers at PIT1_IN and OVF_W (Figure 5) (Hach-American Sigma Inc.). The samplers had 24 1 L bottles for sample collection and were programmed to collect a sample at varying intervals (1 or 2 hours), depending on the rainfall totals and duration forecasted by the Australian Bureau of Meteorology (BOM). Figure 9 shows sampler deployment sites.

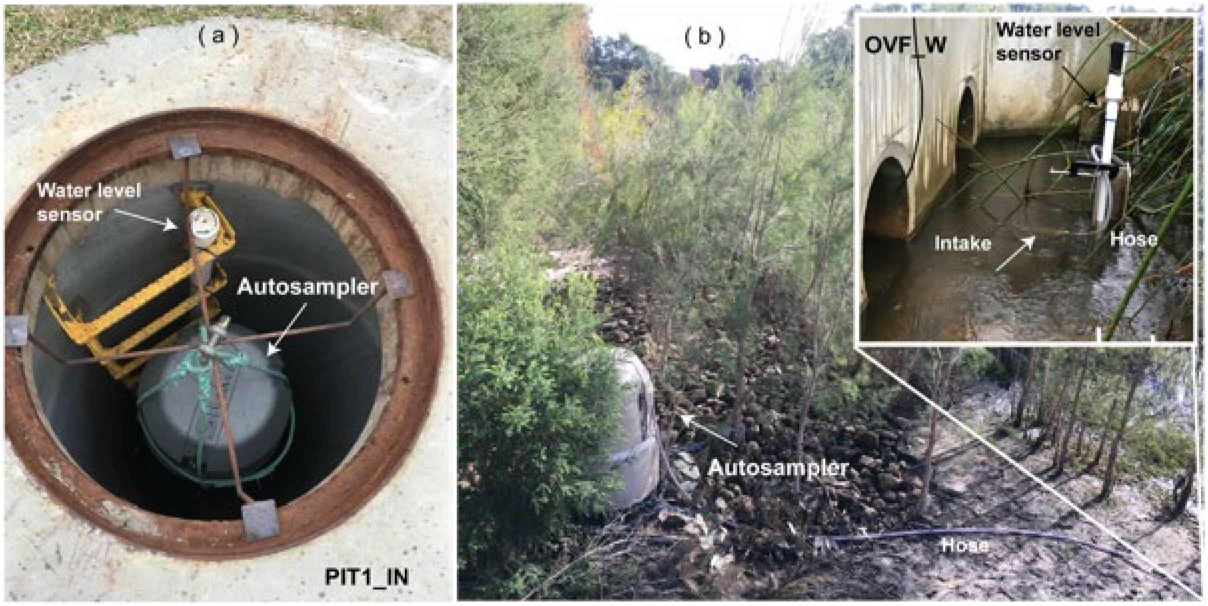

Figure 9 Automatic water sample collection at ESBS for event sampling: a) PIT1_IN, b) OVF_W.

At PIT1_IN, a compact autosampler (Hach-Sigma, Model 900D) was deployed from the top of the pit and supported by a flat cross-metal frame allowing the proper closure of the pit (Figure 9a). This deployment was safe and practical for operating the sampler and protecting against vandalism due its location in a public open space.

A larger sampler (American Sigma, Model 9000) was deployed at OVF_W (Figure 9b). It was positioned on the ground next to the Overflow structure, secured to a large tree (chain and lock) and hidden from the public sight by small tree branches. The sampler's water intake and a water level capacitance sensor were attached to existing metal poles upstream of the overflow structure's centre pipe (picture inset in Figure 9b). Capacitance probe records provided easy and rapid access to the event hydrograph information, used to select water sample bottles for laboratory analysis.

A minimum of six water samples were used to capture nutrient concentration variability across the event, and each sample composited two consecutive bottles, providing sufficient volume for nutrients and TSS analysis. The adopted sampling strategy prioritised event coverage over sample volume, due to the timing of events (90% of events occurring at midnight and weekends).

Five rainfall events (one small, two minors and two majors) were sampled and included one close to a 1-year ARI event; samples were analysed for nutrient concentrations and TSS (as per SAP).

Figure 10 shows the temporal coverage for the baseflow and stormflow sample collections in conjunction with water levels at the PIT1_IN, WET_IN and OVF_W.

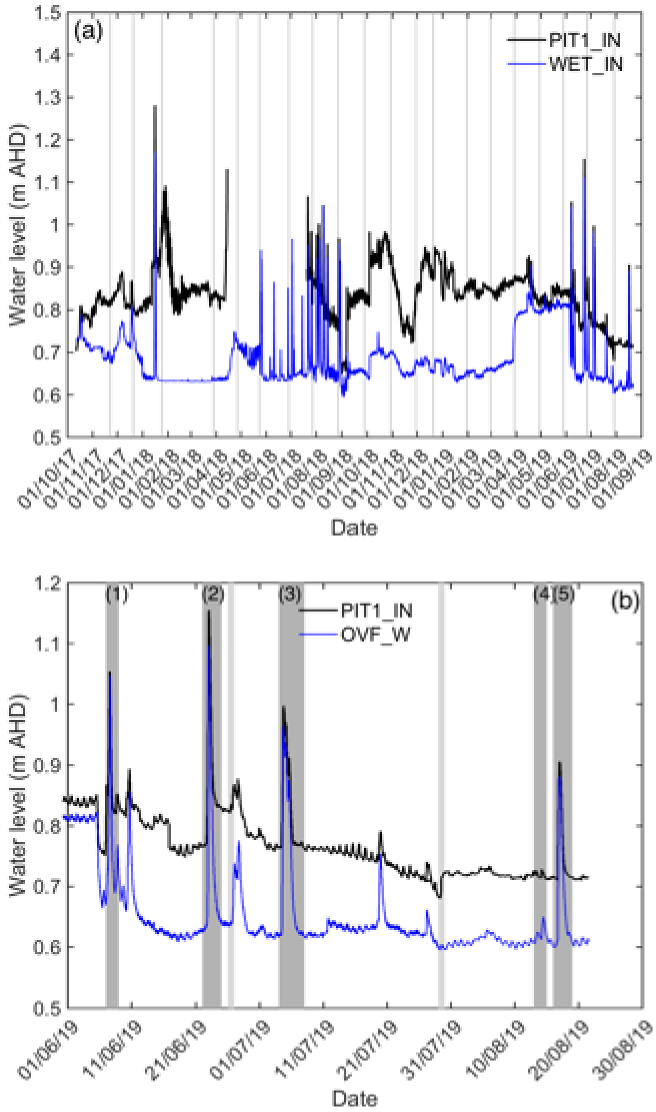

Figure 10 Water sample collection strategy implemented at ESBS wetland during the period September 2017 – August 2019: a) monthly sampling for baseflow conditions, b) stormflow sampling during events. Shaded grey area indicates sampling time.

3.5 Water quality concentration and load targets

Nutrient and metal concentrations measured at the sampling sites (during both baseflow and event conditions), were compared with nutrient concentration established targets for the Swan Canning Estuary, and ANZECC metal guidelines respectively. Water quality targets for Swan and Canning River catchments were established in 1999 (SRT 1999) (Table 5). For the Bayswater Main Drain (Bayswater Brook), a 5-year (2005) concentration target of TN 2.0 mg/L and TP 0.2 mg/L was recommended, and a 20-year (2020) concentration target of TN 1.0 mg/L and TP 0.1 mg/L was recommended. Ten years later, the Bayswater Main Drain was still identified as a priority catchment (SRT 2009), and the previous short-term targets (TN 2.0 mg/L; TP 0.2 mg/L) and long-term targets (TN 1.0 mg/L; TP 0.1 mg/L) were maintained (SRT 2008), based on median concentration values. These concentrations targets were selected to meet the management needs of the estuary as identified in 1999; at the time the ANZECC water quality guidelines were under revision. The range of nutrient concentrations measured in the Bayswater Main Drain for the previous decade (1987–1998) were TN 1.0 – 2.0 mg/L; TP 0.1 – 0.5 mg/L (SRT 1999). The target median concentrations for the Bayswater Main Drain were revised again (SRT 2009) to TN 0.5 mg/L and TP 0.05 mg/L, and the maximum acceptable annual loads were set at TN 4.0 tonnes and TP 0.44 tonnes. These load targets represented a reduction from 1997–2006 average annual loads of 59% and 27% for nitrogen and phosphorus respectively.

A nutrient load reduction target of 30% by 2015 was recommended for the Ellen Brook, Mills Street Main Drain and Southern River catchments (SRT 2008), with the Bayswater Main Drain classified as a second priority catchment category. A Local Water Quality Improvement Plan was established (City of Bayswater 2008) for Bayswater Brook, and annual load targets for 2015 were set at TN 6.7 tonnes and TP 0.65 tonnes (these were based on a targeted 30% reduction of historical (20-year) loads during median flows (TN 9.56 tonnes and TP 0.93 tonnes)). At that time, the median annual concentration targets for 2015 were set at TN 0.7 mg/L and TP 0.07 mg/L.

More recently, it was suggested that nutrient concentrations discharged from the Bayswater Main Drain should be compared with ANZECC trigger values for slightly disturbed lowland rivers of the southwest of Australia (ANZECC and ARMCANZ 2000). These concentrations were considered to be most applicable to the Bayswater Main Drain (City Bayswater/DPaW 2016). The ANZECC trigger values for lowland rivers are TN 1.2 mg/L and TP 0.065 mg/L. For this report, these will be taken as the current targets for TN and TP.

Table 5 Nutrient concentration and load targets set over the last 20 years for the Bayswater Main Drain (Bayswater Brook).

			Targets			
	SRT (1999) 2005 target	SRT (1999) 2015 target	CoB (2008) 2015 target	SRT (2008) 2015 target	SRT (2009)	CoB/DPaW (2016)
Median TN concentration (mg/L)	2.0	1.0	0.7	1.0	0.5	1.2
Annual TN load under median flow conditions (tonnes)	–	–	6.7	–	4.0	–
Median TP concentration (mg/L)	0.2	0.1	0.07	0.1	0.04	0.065
Annual TP load under median flow conditions (tonnes)	–	–	0.65		0.44	–

3.6 Calculation of nutrient attenuation during stormflows

The event mean concentration (EMC), defined as the total nutrient load (mass) divided by the total runoff volume of an event, has been recommended by guidelines to assess nutrient removal by structural elements such as constructed wetlands (Winer 2000, DoW 2004). EMC values for the inflow and outflow are used to evaluate the percentage of nutrient removal as:

$$\text{EMC attenuation (\%)} = \left(\frac{EMC_{IN} - EMC_{OUT}}{EMC_{IN}} \right) \times 100 \qquad (2)$$

where EMC_{IN} is the nutrient (total or species) mean event concentration at the inflow and EMC_{OUT} is the nutrient (total or species) mean event concentration at the outflow.

Estimating EMC requires temporal resolution of nutrient concentration over the course of an event, which was obtained from sequential sampling techniques. For systems with multiple inflows and outflows, EMC also requires accounting for all inflow and outflow volumes and concentrations for a proper attenuation assessment. This aspect becomes limiting when using equation (2) for a treatment train in a large wetland system, such as the ESBS wetland, particularly during wet weather (events).

The present work used the change in wetland water storage from pre-event conditions to the peak storage value to identify inflows and outflows that explained close to 90% of the event water balance. Additional inflows and outflows may contribute to the water balance, depending on rainfall totals and duration, but most importantly, the water level conditions in Bayswater Brook during storm and high tide events. Water volumes and concentrations for each of the identified inflows and outflows were assigned, and used in equation (2) to calculate EMC.

The usefulness of EMC attenuation for comparison with guideline concentration trigger values is acknowledged, but nutrient load attenuation is also a critical performance measure and of interest for operating and managing wetlands. The change in nutrient mass is also relevant during dry weather conditions, when less variability in the wetland hydrological conditions is expected and wetland operational actions could potentially have a large impact on its nutrient dynamics. The change in nutrient (nitrogen and phosphorus) mass from inflow to outflow was therefore also used to assess nutrient removal efficiency. For this study, the percentage removal or load attenuation was defined as:

$$\textbf{Load attenuation (\%)} = \left(\frac{L_{IN} - L_{OUT}}{L_{IN}} \right) \times 100 \qquad (3)$$

where L_{IN} is the nutrient load at the inflow, calculated as the product of the inflow time-variant nutrient concentrations over the course of the event and the inflow volume, and L_{OUT} is the nutrient load at the outflow, calculated as the product of the outflow time-variant nutrient concentrations over the course of the event and the outflow volume.

Each rainfall event was defined, and then loads were calculated at 10-minute intervals by multiplying the total volume of the event water at the inflow (or outflow) by the corresponding nutrient concentrations (linearly interpolated from time-variant concentration series). Load attenuation (%) value was then calculated using equation (3).

Equation 3 was also used to assess load attenuation during the dry weather conditions (baseflow). In this case, L_{IN} is the nutrient load at the inflow calculated as the product of the time-variant inflow rate over the course of a 24-hour period and its reported concentration for the sampling date, and L_{OUT} is the nutrient load at the outflow, calculated as per L_{IN} using time-variant outflow rate and the reported outflow concentration.

4 Water balance analysis

4.1 Rainfall

The medium-term (1975 – present) mean annual rainfall for the area is approximately 725 mm, based on observations at the nearby BOM station (Perth Airport station 9021, BOM 2019). The rainfall during the study period (September 2017 – August 2019) totalled 729.4 mm and 743.8 mm for 2017 and 2018 respectively.

Above average monthly rainfall occurred between December 2017 and January 2018, and between July and August 2018. The highest daily rainfall was recorded on 15 January 2018, resulting from the remains of ex-cyclone Joyce; the 92 mm event was close to the 50-year ARI event and trigged wetland operational actions to mitigate flooding (removal of weir boards at the Overflow structure). Other significant events for 24-hour and 48-hour durations were 35.6 mm and 59 mm, recorded on 21 July 2018 and 25–26 May 2018, respectively.

In summary, ESBS wetland was subject to average annual rainfall, however with mostly minor rainfall events (e.g. less than 40 mm daily totals) occurring over the monitoring period. Rainfall antecedent conditions and rainfall event characteristics are required for context, before the analysis of water quality results and wetland operational actions that changed water levels. These are presented below.

Rainfall event runoff was intensively monitored and sampled from June to August 2019; a total of 382 mm was recorded over the 3-month period. Table 6 presents rainfall characteristics and tidal conditions for the five monitored events: one small (less than 10 mm), two minors (less than 40 mm) and two major events. Event 5 on 16 August 2019 comprised the 1-year ARI event, and was particularly important for analysing the hydrological response and nutrient attenuation capacity of the wetland during the design event.

Table 6 highlights tide conditions found at the time of the relevant rainfall events for the water balance assessment; i.e. each event's potential effect on outflows discharge and temporary ungauged inflows and outflows to the wetland.

Table 6 Rainfall characteristics and tidal conditions during monitored events.

Event	Duration (hr:min)	Rainfall (mm)	Rain1h (mm)	Imax_15 (mm/hr)	Tide	Tide observations^
7–8 June 19	10:00	26.2 +9.3	5.6	8.1	Very high	Rising tide (0.32 m) affecting wetland functioning – Peak at 1.01 m
22–23 June 19	7:45	46 +7.6	13.2	23.4	Average	Receding tide (0.31 m) affecting wetland discharge
4–5 July 19	13:50	33.5 +18.5	19.3	10.1	High	Peak tide (0.8m) affecting wetland discharge on following day
13-14 Aug 19	3:10	4.6 +8.9	4.3	7.1	Average	Receding tide (0.2 m) with minor effect on wetland discharge
16 Aug 19*	6:00	24.4 +1.5	16.0*	47.5	Low	Receding tide (0.2 m) with no effect on wetland discharge

** Event close to the 1-year Average Recurrence Interval (ARI).*

^ Tidal level corresponds to Australian Height Datum (AHD). Source: Australian Bureau of Meteorology (BOM).

4.2 ESBS inflow and outflows: Preliminary rating curves

Gauging activities provided preliminary discharge data for rating curves, explored suitability of the sites for flow measurements, and documented the tidal effect on outflow discharge.

These rating curves are considered preliminary for several reasons. First, no independent discharge measurements in the field were conducted to corroborate flow meters' performance for velocity or mean velocity factors (e.g. velocity factors required for Starflow meter). Second, there are issues with the stability control for discharge provided by concrete pipes in inflow and outflow structures. The rating showed the effect of downstream control (backwater) by rising water levels and a decrease in velocity for the same discharge. Finally, these ratings represent winter flow conditions when the Overflow structure (e.g. OVF_W) acted as the main wetland outflow. Weir boards are commonly used to control water levels between October–April each year, however such scenarios were not documented.

Flow meters were tested before field deployment using a flume at the UWA School of Engineering Hydraulic Laboratory, and later in the field using a state-of-the-art, in-pipe Doppler velocimeter (Sontek IQ-Pipe). Starflow meter flow measurements agreed closely with those measured by the Sontek IQ-pipe, under similar flow conditions.

The reliability of the developed rating is low due to the limited amount of data collected; however, it is considered sufficient to assess the event water balance, EMC computations, and obtain nutrient loads to assess wetland performance. A brief description of the rating follows.

4.2.1 Inflow rating at PIT1_IN

Figure 11 shows two rating curves developed for PIT1_IN. The water level (m AHD) corresponded to the upstream location and measured at PIT1_IN. The blue and grey dots show flow measurements for low- and mid-range discharges up to 60 L/s (blue dots) and theoretical discharge estimates for high water levels (grey dots) using observed water levels at PIT1_IN (upstream) and the SED_P (downstream control). The orange dots show a shift due to partial blockage of the GPT detected by the end of July 2019, that resulted in a low inflow value of 5 L/s (1/10 of design operational flow) and significant reductions in high flows. This rating was applied to event flows and baseflow recorded during August 2019.

The GPT cleaning operation took place in early September 2019 and afterwards high flow and water levels were recorded in both PIT1_IN and SED_P (0.95 m AHD) for a measured discharge of 156 L/s into the sedimentation pond. This result confirmed that such flow magnitudes could be expected during storm events, as predicted by the theoretical rating.

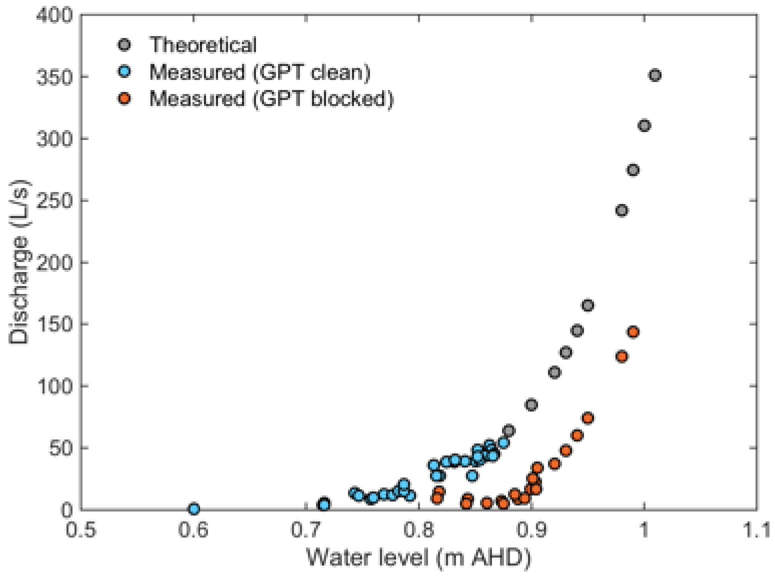

Figure 11 Rating curves for PIT1_IN. Orange dots correspond to the period when the GPT was blocked.

4.2.2 Outflow rating at OVF_W site

Three concrete pipes at the Overflow structure (OVF_W) acted as control structures when low water levels were observed in Bayswater Brook (including low tides); but high tides or high water levels in Bayswater Brook partially or fully drown these three outflow pipes. Such tailwater complexity was recorded during Event 2 (22–26 June 2019) and used, in conjunction with other events with low tide conditions, for rating the site.

Figure 12 shows two rating curves related to the water levels measured in the central pipe of the Overflow structure (m AHD) or at the staff gauge plate adjacent to the culvert. The blue dots represent outflow discharge from the pipe, when there is neither tidal influence nor high water levels in Bayswater Brook. The rating extends to 0.95 m AHD, before the water reaches the top of the pipe at the upstream end. Theoretical flow discharge estimates were used to extrapolate the curve for water levels greater than 1.1 m AHD. Estimates for theoretical discharge values used observed water levels at upstream, inside the pipe and downstream points. The orange dots provide discharge values influenced by high water levels in Bayswater Brook (either tidal, stormflow or a combination of both). These results demonstrate a large difference in wetland discharge when water levels are less than 0.9 m AHD, depending on flow conditions in Bayswater Brook.

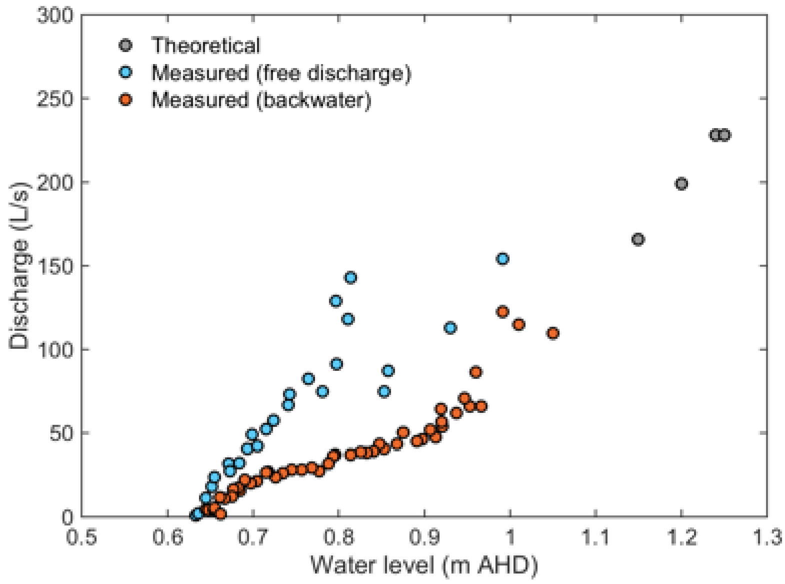

Figure 12 Rating curves for the OVF_W. Curves correspond to levels in the single central pipe of the culvert structure. Blue dots indicate low water conditions, and orange dots indicate high water conditions, in Bayswater Brook.

A range of discharge scenarios can occur for different combinations of water levels in the wetland and Bayswater Brook. The ratings display hysteresis (looping) and, even for a single event, discharge can shift from one curve to the other, as illustrated by points situated between the two curves.

The reliability of these ratings is low, because they are limited to a few flow scenarios and to the centre pipe of the culvert structure. These curves do not apply to the site when weir boards are added/removed to the overflow structure. More work and investigation will be required to monitor and quantify discharge under different wetland operational scenarios.

4.3 ESBS water storage: Baseflow conditions

Water levels in the wetland rapidly responded to changes in inflow and outflow rates and operational activities during baseflow conditions. These responses are presented below for two periods, during which different wetland operational conditions were experienced.

4.3.1 Water levels: October 2017 – December 2018

Figure 13 shows water level recorded at PIT1_IN, PIT2_B, SED_P and the WET_IN for the period October 2017 – December 2018. Data gaps resulted from sensor removal due to malfunctioning (e.g. initial capacitance probes); two new level sensors (Solinst LT) were deployed on 20 July 2018 at PIT1_IN and WET_IN.

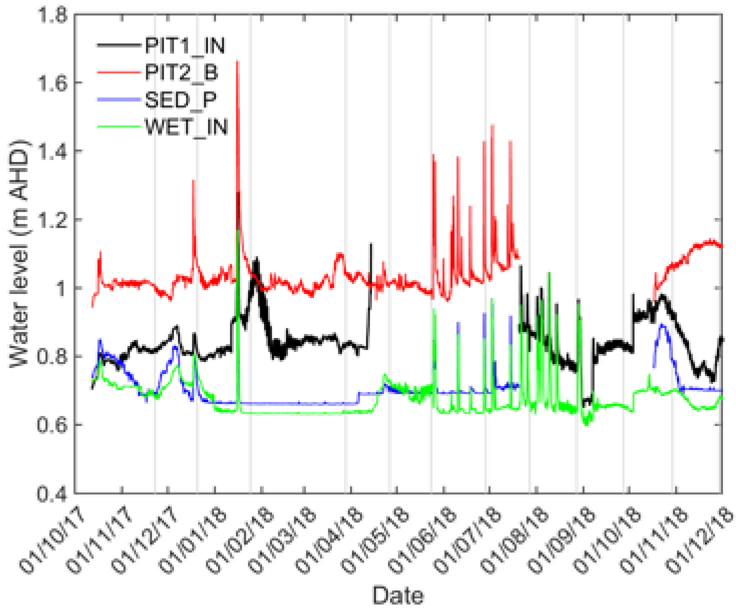

Figure 13 Water levels from October 2017 to December 2018. Water levels are shown for PIT1_IN, PIT2_B, the sedimentation pond (SED_P) and the wetland inlet (WET_IN). Data gaps represent sensors malfunctioning.

The wetland maintained a water level of 0.71 m AHD (on average) from 10 October 2017 to 2 December 2018, increased by 0.1 m after 2 December 2018 and dropped rapidly before a minor event on 18 December 2018 (22.4 mm). This water level dynamics in the wetland resulted from changes at the PIT1_IN and SED_P water levels (Figure 13) and is believed to reflect adding and then removing one log at the DIV_W in Bayswater Brook (documented on a field visit).

The wetland water level remained below the operational level at 0.635 m AHD, from early January to mid-April 2018. The significant change in water level on 15 January 2018 (peak of 1.170 m AHD) was in response to a 50-year ARI rainfall event. Weir boards at the Overflow structure (OVF_W) were in place at the time of this event, but were removed on 16 January 2018 to mitigate wetland flooding. Weir boards were not reinstated after that date and OVF_W became the main wetland outlet point for the rest of this period.

Baseflow levels in the wetland slightly increased in the winter season to 0.65 m AHD (on average) and 12 events were recorded over the winter season with the largest peak on 9 August 2018 at 1.04 m AHD. No discharge was observed from OUT_W due to blockage of the submerged outlet pipe by sediments and high water in Bayswater Brook.

The wetland experienced a drop in water level of 0.05 m following the last event of the winter in late August 2018 and it operated at 0.6 m AHD for the first week in September 2018. Water levels experienced a two-step increase: first to 0.650 m AHD (until 10 September) and then to 0.70 m AHD on 4 October 2018 (Figure 13). Water levels at PIT1_IN confirmed the wetland level dynamics did not result from management operations at DIV_W in Bayswater Brook, but resulted from small rainfall events in September and October 2018.

The water level receded to 0.65 m AHD level by the middle of November, and increased by 0.03 m in response to small events towards the end of November. The wetland baseflow level was maintained at 0.65 m AHD by adding one log at the DIV_W in the Bayswater Brook (documented by photographs) for the summer season.

Figure 14 shows water levels observed at monitoring stations WET_IN, OVF_W and OUT_W for the period under analysis. All stations displayed similar dynamics and responses to rainfall events and operation of the diversion weir in Bayswater Brook (DIV_W).

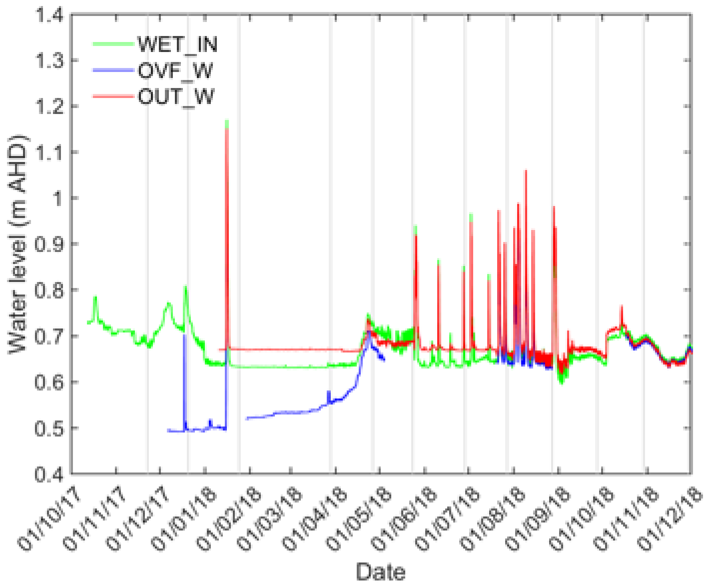

Figure 14 Water level time series from October 2017 to December 2018. Data is shown from Wetland inlet (WET_IN), Overflow (OVF_W) and Outflow (OUT_W). Data gaps represent sensor malfunctioning and loss of OVF_W sensor (from late August to early November 2018).

Wetland water level records confirmed that higher water levels at its inlet point drove flow towards OVF_W and OUT_W. The wetland discharge point at the Overflow structure was also indicated by its water level that was lower than at OUT_W. Weir boards were not in place after the 16 January event.

4.3.2 Water levels: January 2019 – August 2019

The wetland water level (WET_IN) slightly dropped in summer, to a minimum value of 0.633 m AHD on 20 January 2019, but it then increased until 18 March before stabilising at 0.68 m AHD for 10 days (Figure 15). This unexpected increase resulted from small rainfall events 7–10 March 2019. The diversion weir with one log was the only inflow control at that time, while the Overflow structure (OVF_W) was the wetland's only discharge point (the OUT_W outlet pipe was submerged and being blocked).

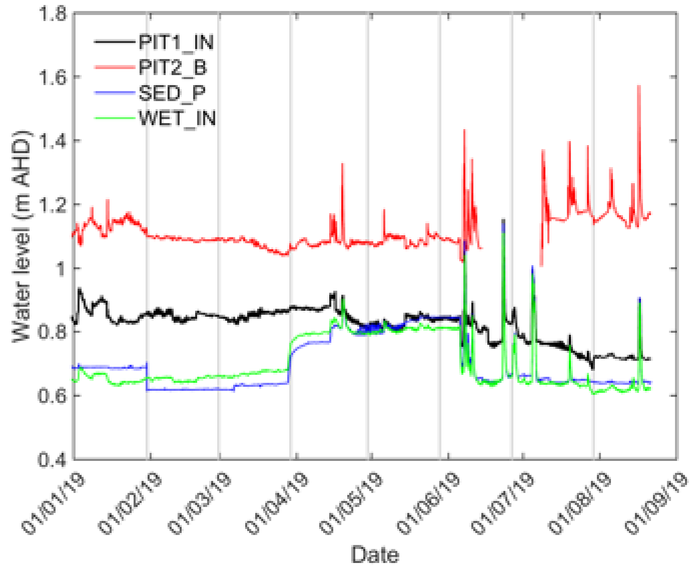

Figure 15 Water levels for selected monitoring stations from January to August 2019. Data is shown from the PIT1_IN, PIT2_B, SED_P, and WET_IN.

Wetland maintenance activities and management actions changed the wetland dynamics and reinstated its expected hydraulic functioning on 28 March 2019. These actions included adding a second log at the diversion weir in Bayswater Brook, closing the Overflow structure (two boards were added to the weir) and removing the blockage of the Outflow submerged pipe to re-activate its function as a discharge point (OUT_W). These operations resulted in a rapid (in 28 hours) increase in water level (by 0.08 m) followed by a steady increase (over a 5-day period) until the water level reached a plateau of 0.8 m AHD (see Figure 15 and Figure 16 for details). A few small rainfall events between 15 April – 20 April set a new wetland high water level of 0.820 m AHD (not observed in the previous year except for the January storm event). A GPT clean up operation was conducted on 16 April 2019, but it remains unclear whether this contributed to the increase in wetland water level (Figure 15).

High water levels continued across all wetland components until the end of May; OUT_W continued its discharge via the submerged pipe but outflow progressively slowed down and ceased altogether by 4 June 2019, when the floodgate plate was closed by high water levels in the Bayswater Brook. Removing two boards at the Overflow structure on 5 June 2019 (see OVF_W in Figure 16) decreased the wetland water level from 0.82 m to 0.67 m AHD in 20 hours. This action established the winter wetland water level at 0.64 m AHD, to manage inflows from expected winter events. However, the water level unexpectedly dropped on 29 July to 0.610 m AHD (on average). This decrease was caused by reduced inflow from the GPT to the sedimentation pond and resulted in low water levels until the end of August 2019. Inflow hydrographs for PIT1_IN and PIT2_B (Figure 15) clearly reflected the effect of the GPT blockage by the opposite direction on their dynamics; i.e. PIT2_B increased as PIT1_IN decreased. Water levels alone cannot identify the location of the blockage within the GPT, because the increase in PIT2_B water levels can result from either a blockage of the Diversion pipe (e.g. forcing water from brook to enter the GPT via PIT2_B pipe) or a partial blockage of the PIT1_IN pipe to the sedimentation pond (e.g. forcing the diverted inflow to return back to the brook resulting in increase in water level at PIT2_B).

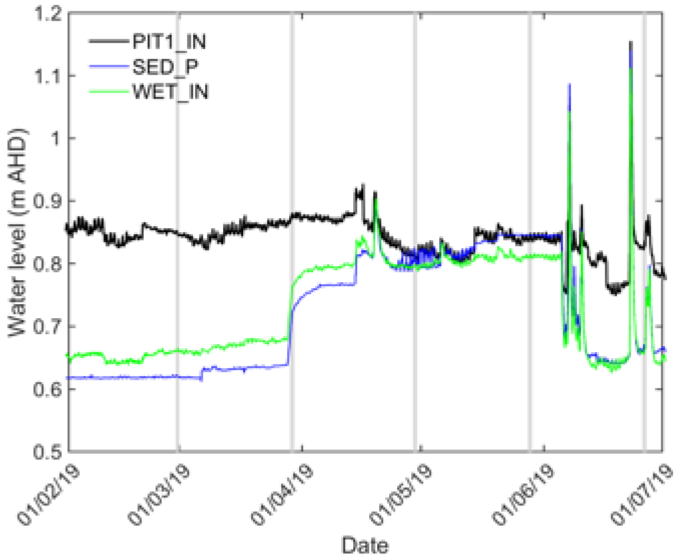

Figure 16 Water levels at selected monitoring stations from February to June 2019. Variation in levels was observed as flows responded to wetland management operations. Wetland levels increased to 0.80 m AHD due to the addition of two boards at the Overflow structure (OVF_W site), and later their removal in preparation for winter season inflows.

4.4 ESBS water balance: Stormflow conditions

This section presents the water balance for the five monitored events. We focus on identifying and quantifying the main inflows and outflows contributing to the observed changes in wetland water storage, over the course of an event. A brief description of each event provides context for water quality assessment in the following sections. We need to quantify inflows and outflows to estimate nutrient load attenuation across ESBS (see Section 3.6).

4.4.1 Event 1: June 2019

Event 1 started on the evening of 6 June 2019, increased its intensity at midnight and continued until the early hours of the next day, totalling 26.2 mm. It coincided with a very high tide on the morning of 7 June 2019, reaching a “Minor flooding condition” for the Swan River. A field visit corroborated high water levels in the wetland; pathways around ESBS were partially flooded, there were water inflows from Bayswater Brook, and the Overflow structure outlet pipes were fully submerged (little outflow at OVF_W). Wetland water levels peaked at 1.02 m AHD (by 2:20 pm on 7 June) with a net increase of 0.374 m in 17 hours. Figure 17 shows hydrographs for the event at different sites.

The water balance results indicated 60% of the wetland volume came from Bayswater Brook inflows through the Outlet structure weir, followed by 21% from PIT1_IN, and contributions of 14% and 5% from surface runoff from adjacent areas and direct rainfall into wetland respectively.

The Bayswater Brook peak level was 1.148 m AHD (measured at BB_OUT) and remained higher than 1.10 m AHD for about 2.5 hours. Water exchange across the embankment (at several points along the western boundary of Bayswater Brook) likely occurred over the 2.5-hour period.

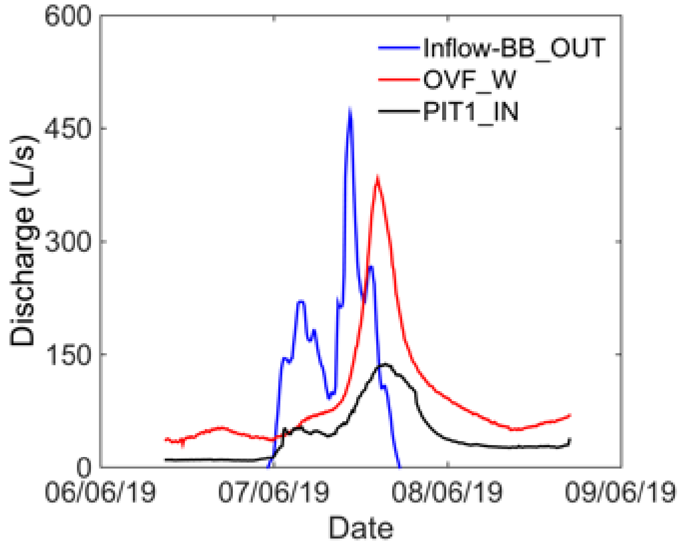

Figure 17 Event 1 flow hydrographs. The Inflow-BB_OUT hydrograph indicates inflow from Bayswater Brook to the wetland via the large weir at the Outlet structure.

4.4.2 Event 2: June 2019

Event 2 started in the afternoon of 22 June 2019 and finished at midnight, totalling 46 mm over approximately 7.5 hours (a major event). It coincided with a receding tide, but the high rainfall magnitude resulted in high levels in the Bayswater Main Drain and flooding of wetland interior areas.

The wetland recorded peak water level at 1.116 m AHD (at 12:20 am on 23 June) as the water level increased by 0.465 m over the 7.5-hour period. A field visit corroborated that water level conditions were similar to Event 1, but good discharge was observed at OVF_W. The wetland returned to pre-event water levels within 42 hours. Figure 18 shows hydrographs for the event at different wetland sites.

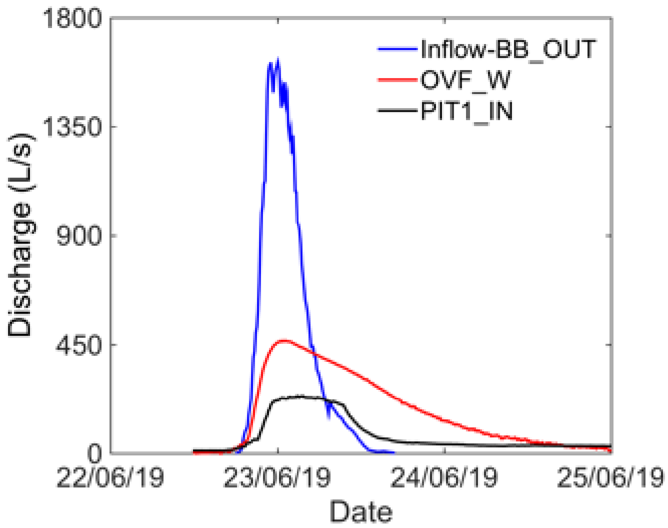

Figure 18 Event 2 flow hydrographs. The OVF_W hydrograph shows good discharge conditions over the course of the event.

The water balance results indicated the inflow was dominated by Inflow-BB_OUT at the weir (74%), with equal contributions (10%) from PIT1_IN and surface runoff from nearby areas, and a 6% contribution from direct rainfall to the wetland.

Bayswater Brook water levels peaked at 1.224 m AHD (at BB_OUT) and high water levels continued for about 22 hours. The above water balance over-predicted the observed net gain in wetland water level by approximately 20% (i.e. outflows were underestimated). The results suggested the volume excess could be discharged from the wetland at approximately 150 L/s over the subsequent 22 hours, across the lowland downstream areas situated between the wetland and the Bayswater Brook.

4.4.3 Event 3: July 2019

Event 3 was comprised of multiple periods of rain over two days; it started on 3 July 2019, with 33.5 mm over the next 14 hours, and another 18.5 mm on the following day. The event coincided with high tide conditions, reflected by high water levels in Bayswater Brook.

Wetland water levels peaked at 0.981 m AHD in just 9.5 hours, resulting in a water level increase of 0.347 m. High water continued for the duration of event, with the wetland returning to pre-event water levels after 48 hours. Figure 19 shows hydrographs for the event, measured at different sites.

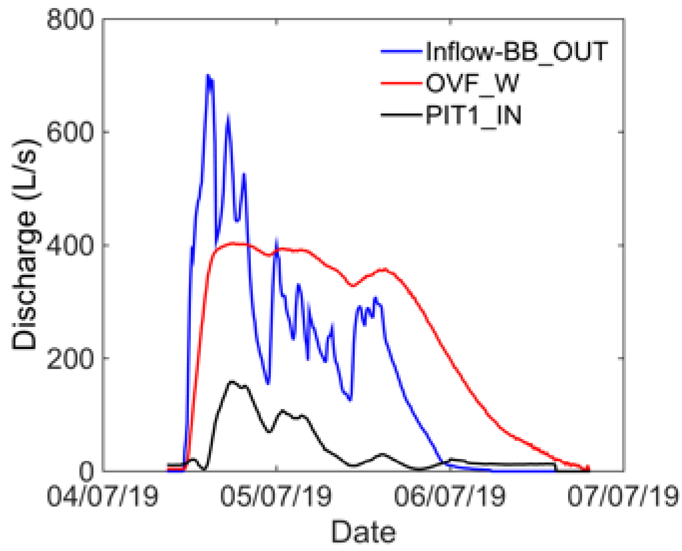

Figure 19 Event 3 flow hydrographs. The wetland responded to multiple rainfall events over the period.

The water balance results showed the inflow was dominated by inflow-BB_OUT flows through the weir (66%), and surface runoff from nearby areas (19%), with only minor contributions from PIT1_W (10%) and 5% from direct rainfall (27 mm) to the time of the peak level in the wetland.

The Bayswater Brook water level peaked at 1.147 m AHD (at BB_OUT) and these high water levels caused flow into the wetland for 42 hours. The water balance slightly over-predicted the wetland maximum level by 2% and it reflected good discharge condition at the Overflow structure (OVF_W); this result shows that all inflows were successfully managed by the wetland storage capacity (e.g. level at 0.981m AHD) for the 2-day event.

4.4.4 Event 4: August 2019

Event 4 started on 14 August 2019 at 4:00 am, with 4.6 mm of rainfall over a 3-hour period; it continued with more showers over the day adding another 8.9 mm. Low tide conditions were observed at the Swan River and Bayswater Brook (receding tide).

Wetland water levels increased by 0.04 m in response to the event, peaking at 0.657 m AHD after 7.5 hours. Figure 20 shows the resulting hydrographs.

The water balance results showed the wetland inflows were dominated by inflow-BB_OUT flows over the weir (80%), even for this small event, then by rainfall (11%) and flows coming from PIT1_IN (9%). Local runoff at a rate of 100 L/s, as suggested by GHD (2013), over-predicted an increase in wetland water levels and was not supported by observed data. Therefore local runoff for this low intensity (4.3 mm/h) and short duration event was not included in the water balance presented here.

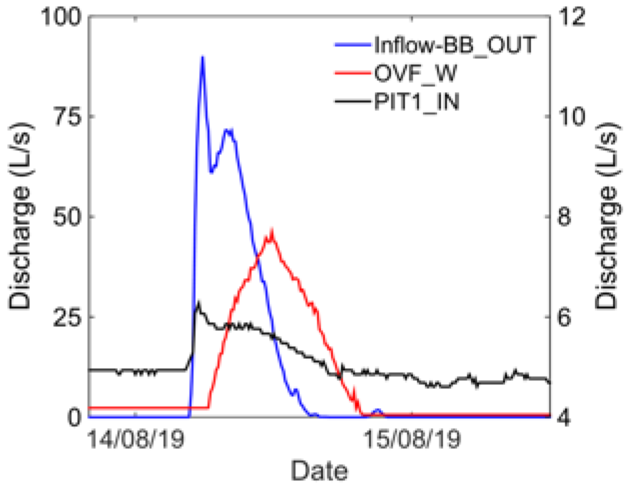

Figure 20 Event 4 flow hydrographs. Refer to right y-axis for PIT1_IN hydrograph discharge values.

An important reduction of inflow via PIT1_IN (discharge peaked at only 6.5 L/s) impacted water levels across all wetland treatment zones. Closer inspection of water levels showed decreasing water levels since 29 July, due to blockage of the GPT (see Section 4.2.1).

Despite low tide conditions and low runoff from the event, water level in the Bayswater Brook (at BB_OUT) peaked at 1.014 m AHD and this contributed inflow to the wetland for approximately 12 hours.

4.4.5 Event 5: August 2019

This 24.4 mm event was recorded over 6 hours on 16 August 2019, and comprised a single rainfall burst of 16 mm in one hour, followed by a few showers on the following day. A closer inspection of the event indicated its similarity to the 1-year ARI 1-hour event; this was of specific interest for designing treatment wetlands and WSUD at source control structural elements. Low tide and water levels in the Bayswater Brook (at BB_OVF) ensured free discharge from the Overflow structure outlet pipes (OVF_W).

The wetland water level increased by 0.27 m from its pre-event condition and peaked at 0.892 m AHD in approximately 9.5 hours. Figure 21 shows the recorded hydrographs for the event.

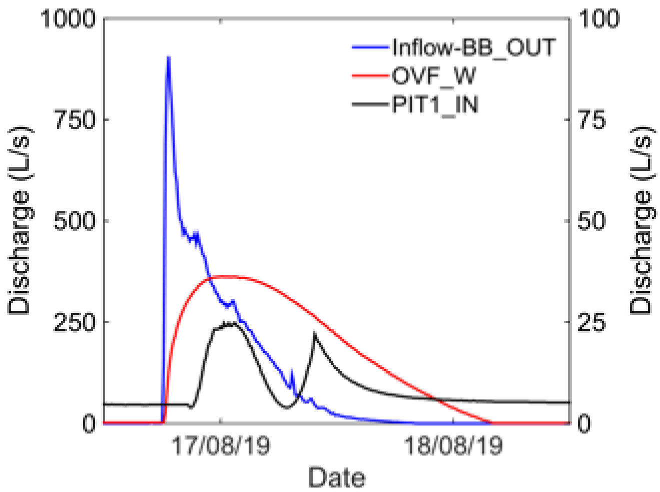

Figure 21 Event 5 flow hydrographs. Refer to the right y-axis for PIT1_IN hydrograph discharge values.

The water balance results showed the wetland inflow during the event was dominated by inflow-BB_OUT flows coming in via the large weir (76%), followed by surface runoff from nearby areas (16%) and minor contributions from direct rainfall (5.5%) and PIT1_IN (2.5 %). The latter contribution was confirmed by Doppler measurements during the event. The low contribution from PIT1_IN highlighted that the GPT blockage significantly reduced the design inflow rate (50 L/s), and compromised wetland operational water levels.

Stormflow runoff from the event resulted in high water levels in the Bayswater Brook, peaking at 1.166 m AHD (at BB_OUT); these high water levels caused inflow to the wetland for approximately 26 hours.

4.5 Slade Street drain inflows

The Slade Street drainage pipe conveys runoff from a 6.9 ha urbanised area (GHD 2013), via a 0.3 m diameter pipe entering the ESBS wetland at its northern end (Figure 5). A concrete pit (1.2 m diameter) collects runoff and then discharges it via seepage through a landscaped rocky-embankment. The SLADE_S created difficulties for rating, as runoff takes the form of shallow sheetflow over a vegetated bed; using the traditional volumetric method (i.e. variation in volume inside the pit, per unit of time) resulted in overestimated and unrealistic flow discharge of about 2 L/s for a large event. Water level in the pit is not affected by wetland water levels due to its higher position in the landscape.

Water level recorded at SLADE_S is shown in Figure 22. Its variability reflected both the rapid response of the Slade Street catchment to rainfall events and the seasonal change in baseflow. Water level variability under baseflow was consistent with annual rainfall totals and monthly distributions for 2018 and 2019; water levels in August and September 2018 were observed to be higher than for the same months in 2019. This baseflow seasonal variability reflected antecedent wetness conditions in the catchment and corresponded with shallow groundwater dynamics (i.e. the timing of peak level) reported by previous studies in the area (GHD 2009; URS 2010).

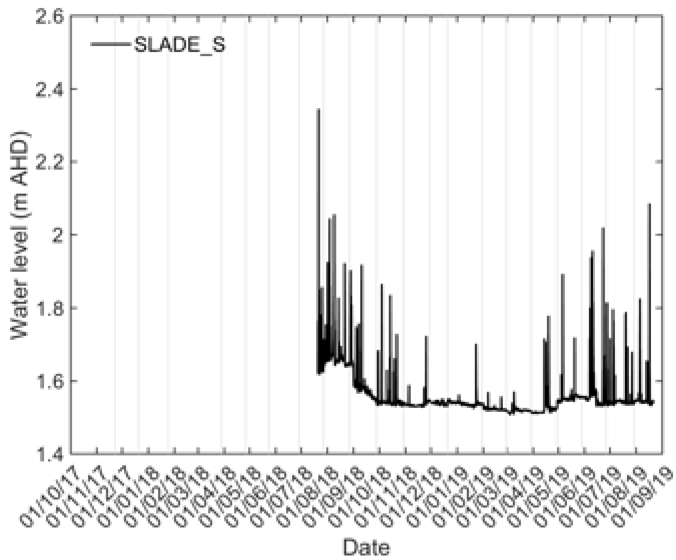

Figure 22 Water level for SLADE_S. Data were recorded between 20 June 2018 and 21 August 2019.

The water level data suggested groundwater intersected the drainage pipe and discharged into the wetland. The lower baseflow water level observed in the pit during August–September 2019 (Figure 22) likely reflected the drier 2019 winter, with decreased groundwater recharge and discharge to the drain. Water quality and nutrient concentration data can be used to confirm this finding and will be presented in Section 7.1.

5 Hydraulic performance

The design of the wetland treatment train and associated structures (GHD 2013) considered the constraints of the site, for example, the proximity to the Swan River estuary, wetland site characteristics (e.g. low hydraulic gradients, shallow water depth, high water tables) and those posed by Bayswater Brook at its western boundary (e.g. high baseflow, tidal incursion).

The treatment train comprises structures controlling inflows and outflows, and an embankment next to Bayswater Brook at a 1.1 m AHD; all of these were designed to achieve normal and operational water levels within the wetland of 0.45 m AHD and 0.75 m AHD respectively (GHD 2013; Table 2). The design also intended to reduce the possibility of additional inflows from Bayswater Brook under different combinations of tides, Swan River flood levels, and Bayswater Brook stormflow levels, for events up to the 1:5-year ARI. Previous hydraulic modelling of Bayswater Brook showed its conveyance capacity exceeded 3 m^3/s for water levels above 1.10 m AHD. Further, the design anticipated partial or total flooding of the wetland area for short periods of time, without affecting its functionality.

The diversion weir in Bayswater Brook (DIV_W) was designed to facilitate inflow of approximately 200 L/s to the GPT, with 50 L/s entering the sedimentation pond via an inbuilt weir and a plate with a 300 mm orifice inside the GPT. The remaining 150 L/s was designed to return to Bayswater Brook (PIT2_B in the present study). The amount of inflow diverted to the GPT is controlled by a series of jarrah logs at DIV_W, manually added or removed by City of Bayswater personnel, as part of the wetland's operational plan.

The above information will be used to discuss the wetland hydraulic performance and functioning under baseflow and stormflow conditions.

5.1 Changes in inlet and outlet locations and flow rates

Wetland hydraulic functioning was negatively affected by changes in inflow and outflow magnitudes and locations, which resulted in wetland water levels that were outside the expected range. The perturbation of the wetland hydraulic performance was variously caused by:

- blockage of the GPT structure that reduced inflows to the sedimentation pond
- disruption of flow conditions in Bayswater Brook that resulted in the diversion of large volumes of water into the wetland (e.g. inflow-BB_OUT flows), and
- closure of wetland outflow structures (e.g. high water levels in the brook pushing against floodgates).

Doppler measurements in December 2018 confirmed good hydraulic functioning of the GPT, with it delivering flows of 41 L/s to the wetland under dry weather baseflow conditions. However, it decreased at the onset of the wet season in 2019, from 35 L/s in June, to just 5 L/s by the end of July. Minimum wetland water levels were observed at 0.60 m AHD from 30 July – 6 September 2019. The reduced inflows could be attributed to partial blockage of the GPT and resulted in the lower water levels in the wetland. The associated water balance agreed with these observations, showing the inflow volumetric contribution to wetland storage decreased from 21% in June, to 10% in July, and to 2.5% in August, for a 1-year ARI event.

The GPT was thoroughly cleaned on 6 September 2019 by a new contractor, and this maintenance reinstated flow through the GPT. Doppler measurements conducted after the maintenance recorded a flow rate of 156 L/s entering the wetland (significantly higher than the 50 L/s expected), and additional data (not shown) revealed the wetland reached a maximum water level of 1.08 m AHD at WET_IN. This high level resulted from the combination of the high diverted inflow rates into the wetland and the closure of the Overflow structure (two boards were added at OVF_W).

The wetland water level was reinstated to 0.8 m AHD on 20 October 2019, by removing one board at the Overflow structure. The Water Corporation undertook management actions in December 2019, to clear vegetation in Bayswater Brook. This was expected to reduce water levels in the brook which, combined with log removal at DIV W weir, could reduce the inflow rate to the wetland to its designed 50 L/s. However, the hydraulic functioning after this maintenance is not reported here.

These results highlight the importance of inflow rates to the wetland; ongoing monitoring of flow rates should inform the operation of the diversion weir (DIV_W). For this purpose, we recommend installing an inexpensive flow meter (e.g. Starflow) with telemetry capabilities at PIT1_IN, to provide real time monitoring of inflows. Ongoing monitoring of the condition of the GPT is also critical, along with an associated appropriate maintenance schedule.

5.2 Hydraulic function under baseflow conditions

Wetland hydraulic functioning under baseflow conditions can only be assessed from 28 March 2019 to 4 June 2019, when all wetland components were operating as intended. Water level data only was used for this assessment due to difficulty of obtaining discharge measurements for outflow sites during baseflows conditions, when there is simultaneous discharge from the submerged pipe at the Outflow structure (OUT_W) and leaks through weir board gaps at the Overflow structure (OVF_W).

The average inflow over the period was approximately 45 L/s and it was close to the designed 50 L/s (GHD 2013). Inflow rates ranged between 30 L/s and 58 L/s with the higher value observed in May 2019; this wetland inflow was verified via water velocity measurements.

The wetland water level for the period was at 0.80 m AHD; this level is slightly higher than the designed operational level of 0.75 m AHD (GHD 2013). Water level records confirmed water levels at the wetland inlet (e.g. SED_P) drove the flow to the Outflow and Overflow structures. All wetland monitoring stations displayed daily variation in water level of a few centimetres; this is believed to be the result of tidal influence in Bayswater Brook, and the (only) partial closing of the outlet pipe floodgates at OVF_W.

5.3 Hydraulic function under stormflow conditions

As discussed in the previous section, high water levels in the Bayswater Brook resulted from reduction in the conveyance capacity of the brook due to overgrown macrophytes (for baseflow and mid-flow ranges) and a few fallen trees across the brook banks that produced backwater effects (for high flows). Pre-storm event water levels in Bayswater Brook were as high as 0.82 m AHD at BB_OUT, but 600 m downstream in a sandy bed section at BB_OVF, water levels were lower at 0.31 m AHD. Under these conditions, a small rise of 0.15 m in water level at BB_OUT can trigger additional inflows into the wetland; this condition was achieved for small and minor events. Further, these conditions placed the BB_OVF site water levels at 0.52 m AHD, well above the Overflow structure concrete base level of 0.45 m AHD.

Due to the high water at BB_OUT, inflow volumes were recorded for all monitored events through the Outlet structure weir; this inflow contributed 71% (on average) of the wetland maximum water storage over the course of an event. This finding was unexpected because all monitored events had ARI of less than 1:5-year event. This indicates the wetland is not operating as intended during rainfall events. Short-circuiting occurred between the Outlet structure (acting as the main inflow point under these conditions) and the Overflow structure (acting as main outflow under these conditions); this reduced the pathway length for stormflow treatment to one-third of the total wetland length, even for small and minor events.

Site visits and camera still images revealed the complexity of measuring and predicting outflows at OVF_W. Flow rates depended on both wetland and Bayswater Brook water levels and the time of the tide arrival to the Overflow structure location (monitored at BB_OVF).

5.4 Impacts of tide

Tides affected water level of the Bayswater Brook near the wetland Overflow structure (OVF_W), increasing it from 0.02 m to 0.28 m, under low baseflow conditions (i.e. when water levels were at 0.30 m AHD at BB_OVF). A 0.28 m increase in water level was a common occurrence and sufficient to reach the Overflow outlet pipes at OVF_W. Doppler data confirmed the transitional conditions with the arrival of the tide. Initially, the wetland discharged to the Bayswater Brook; with the incoming tide, the water level inside the pipe slightly increased and slowed the discharge. This tidal perturbation of water levels, propagated upstream and within the wetland, was captured by water level sensors across the wetland.

The discharge at OVF_W is very sensitive to the arrival of the tide, as well as the height of water in the wetland. Little outflow from the wetland was observed on the morning of 7 June 2019 (Event 1) coinciding with a very high tide. The wetland water level then increased as the result of inflows and reached a threshold point, after which the outflow discharge increased. This is reflected in the shape and timing of the outflow hydrograph for Event 1 at OVF_W. This situation was not observed for other events because a sufficient outflow rate was established earlier in the event and the outflow continued, even with high water levels in Bayswater Brook recorded at BB_OVF.

5.5 Impact of operations

The rationale behind the current management strategy of removing weir boards at the Overflow structure during winter flow is not clear, although it may be a response to additional flows entering the wetland via the weir at the Outflow structure. Water level data for monitored events in 2019 showed water levels exceeded 1 m AHD at WET_IN despite good outflow rates at OVF_W. The results suggested removing the boards does not mitigate flooding of the wetland.

Restoring the conveyance capacity of Bayswater Brook will reduce the occurrence and magnitude of the additional inflows observed during storm events. It will also reduce blockage of the submerged pipe acting as a discharge mechanism for the wetland at the Outflow structure. This single pipe was operational for a few months a year with a minor discharge; it was blocked most of the time by sand accumulation and the closure of the floodgate by high water levels in Bayswater Brook (as recorded at BB_OUT). The Outflow structure was not functioning as an outflow point for most of the time over the monitoring period, and it created a storage zone for nutrient and sediment accumulation.

Given all the factors that have affected flow discharge from outflow points, and therefore the hydraulic performance of the wetland, we recommend, in the first instance, implementing a maintenance strategy for Bayswater Brook, to restore its conveyance and decrease water levels across the wetland. In addition, given the wetland has demonstrated its ability to manage transient high water levels, we recommend reconfiguring ESBS slightly, to establish a simpler and more reliable flow path through the system. Subsequently, additional flow measurements should be conducted to establish ratings for the site when it is operating as expected.

5.6 Water balance findings summary

1. During rainfall events, the wetland water pathways were not working as intended in the original design. Results suggest that one-third of the length of the nutrient treatment pathway is currently used for small and minor events, including the 1-year ARI event. Short-circuiting occurs between the two outflow points.

2. The major inflow to the wetland during rainfall events is water from Bayswater Brook entering at the Outflow structure weir. This inflow explains 60–80% of the total inflow for the event. It is independent of the rainfall magnitude and is believed to have occurred since monitoring began in late 2017.

3. The design considered the possibility of additional inflows during major events and high tide conditions (10-year ARI event). However, additional inflow also occurred for small events (less than 10 mm) and the 1-year ARI event. Further, the wetland takes up to 48 hours to return to normal level.

4. These issues do not relate to problems in the design but to a substantial reduction of the conveyance capacity of a 400 m reach in Bayswater Brook around the Outflow structure. Overgrown vegetation in the channel increased the water level in the brook by more than 0.3 m; water levels reached 0.842 m AHD at BB_OUT under wet season baseflow levels.

5. Inflow to the wetland via the diversion structure highly depends on the GTP conditions. Flow discharge ranges from 5 L/s to 156 L/s for baseflow conditions under blockage and cleaning conditions respectively. Discharge to the sedimentation pond can reach 250 L/s under major events.

6 Nutrient attenuation

6.1 Baseflow conditions: Concentration attenuation across ESBS treatment train

Nutrient concentrations under baseflow conditions were measured across the ESBS treatment train, allowing assessment of each component's effectiveness at attenuating concentrations (Table 7).

Table 7 Comparison of nutrient and metal concentrations at different sampling stations allows the assessment of the performance of individual components of the ESBS treatment train.

Wetland component	Indicator of concentration attenuation
GPT	Difference between DIV_W and SED_P
Sedimentation pond	Difference between SED_P and WET_IN
Wetland	Difference between WET_IN and OUT_W (or OVF_W)
Dead zone around OUT_W	Difference between OUT_W and OVF_W
Overall ESBS	Difference between DIV_W and OVF_W (or OUT_W)

The ESBS component performance under baseflow conditions, when data is averaged across all sampling dates, highlights specific dynamics across nutrient species (see e.g. Figure 23a). The changes in the function of the ESBS treatment train with season is indicated by the water quality measured across the components on each baseflow sampling date (see e.g. Figure 23b).

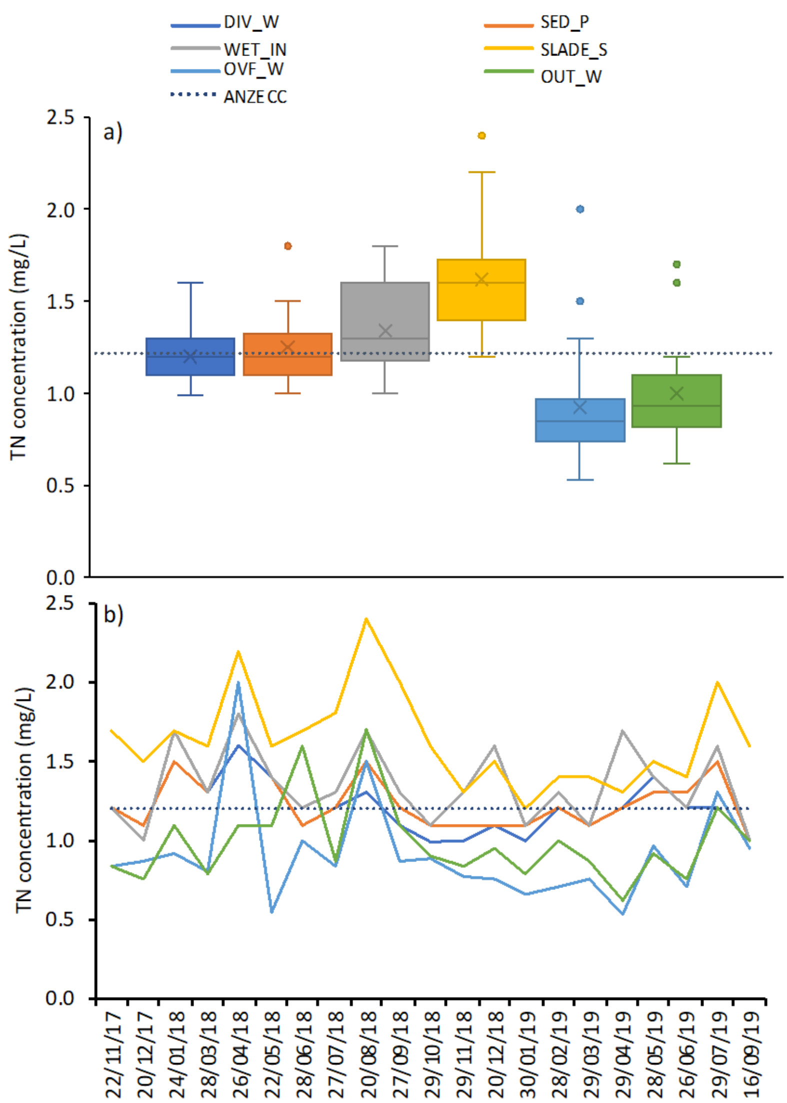

Figure 23 Total nitrogen concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

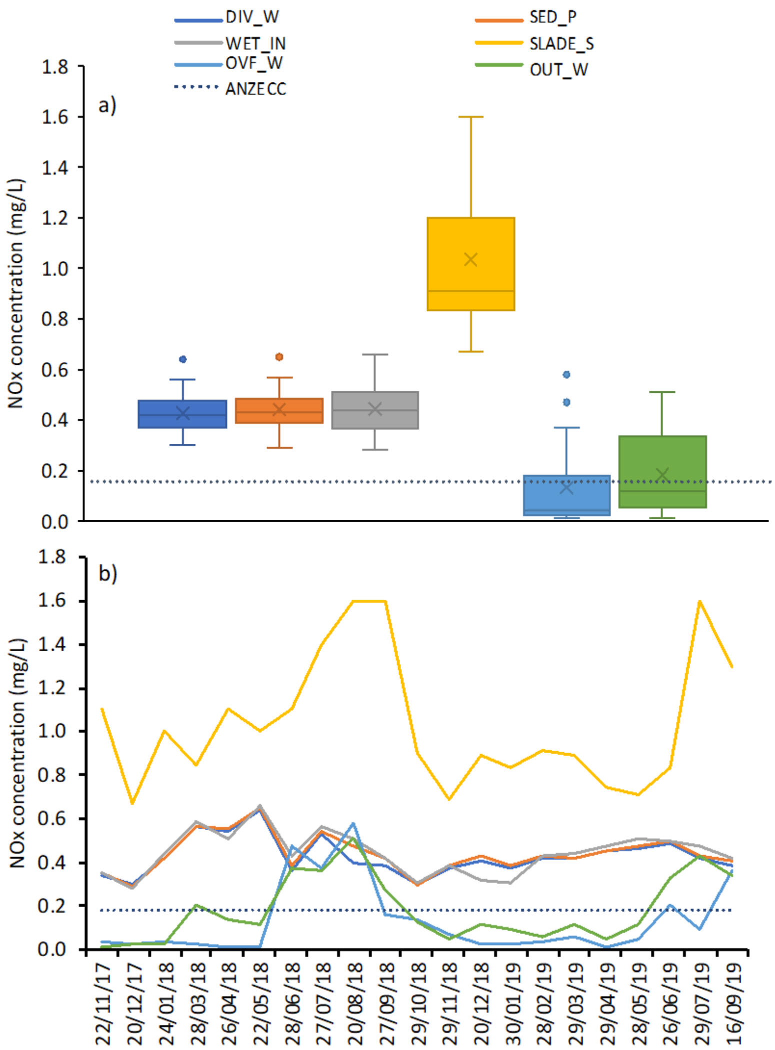

Figure 24 NO-x concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

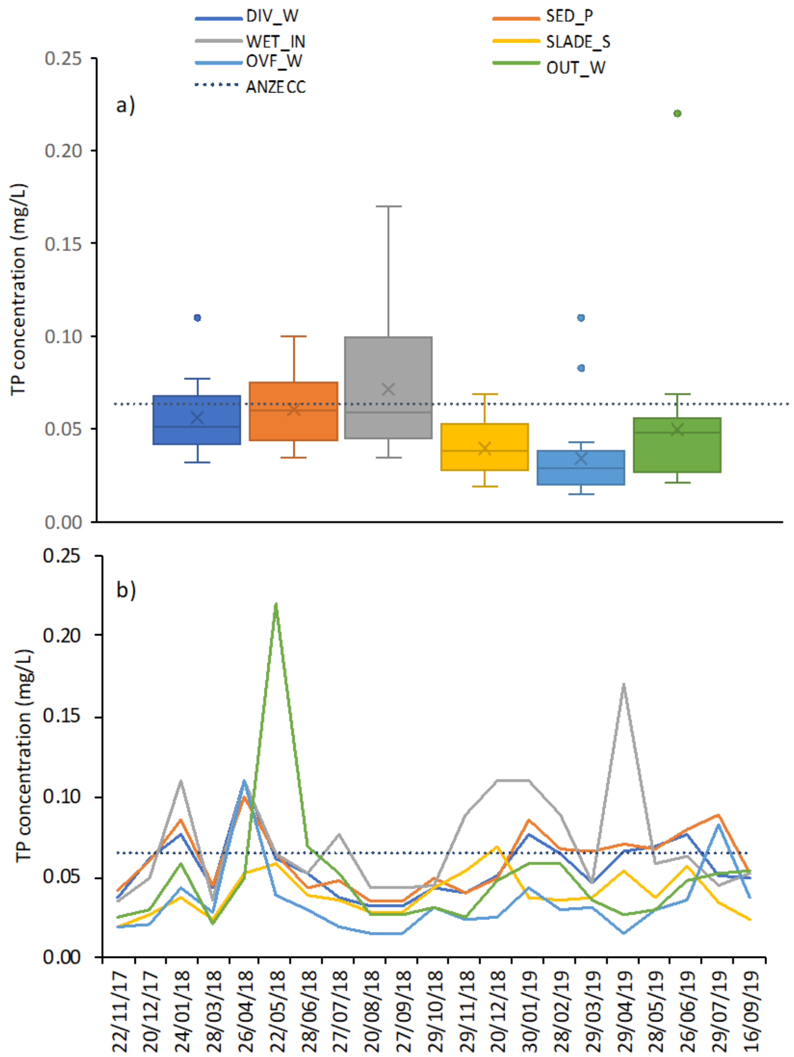

Figure 25 Total phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

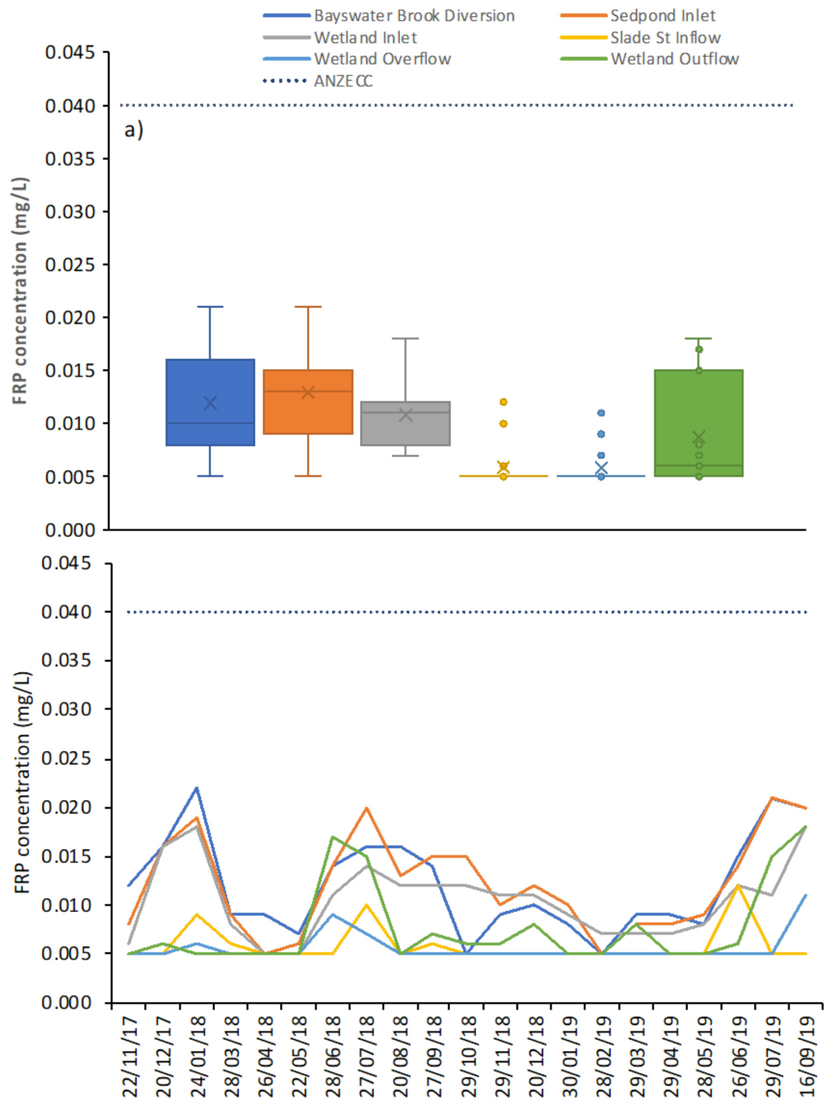

Figure 26 Filterable reactive phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

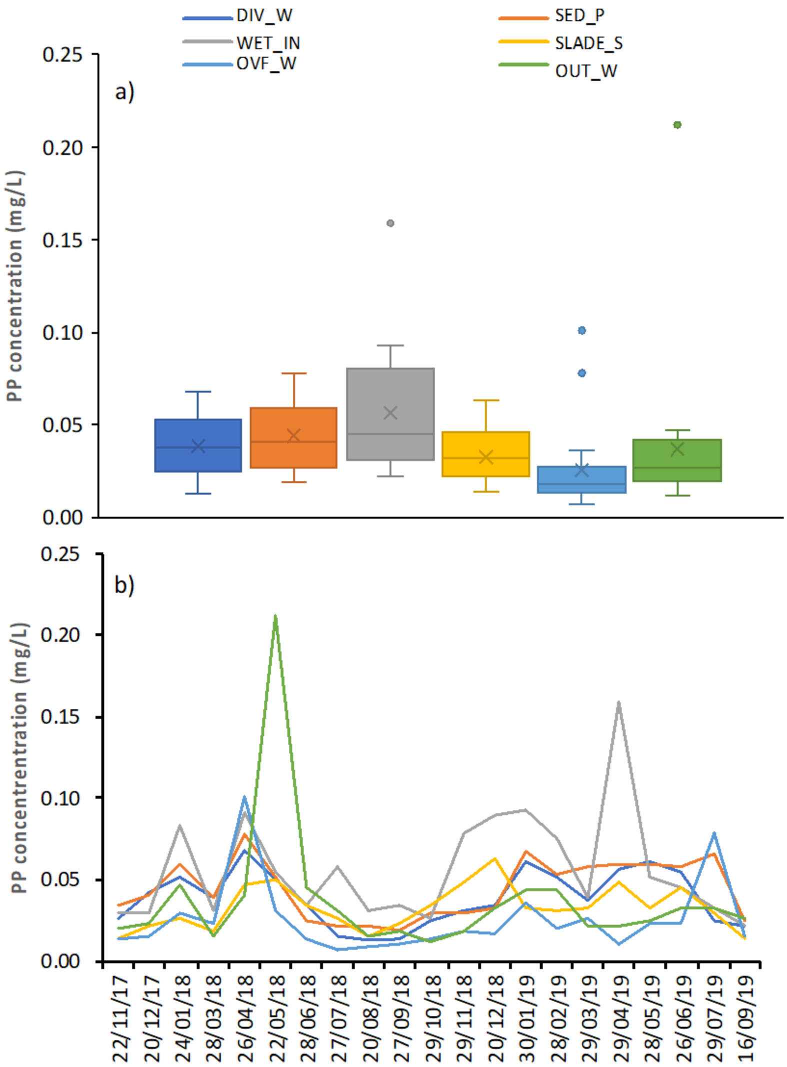

Figure 27 Particulate phosphorus concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site.

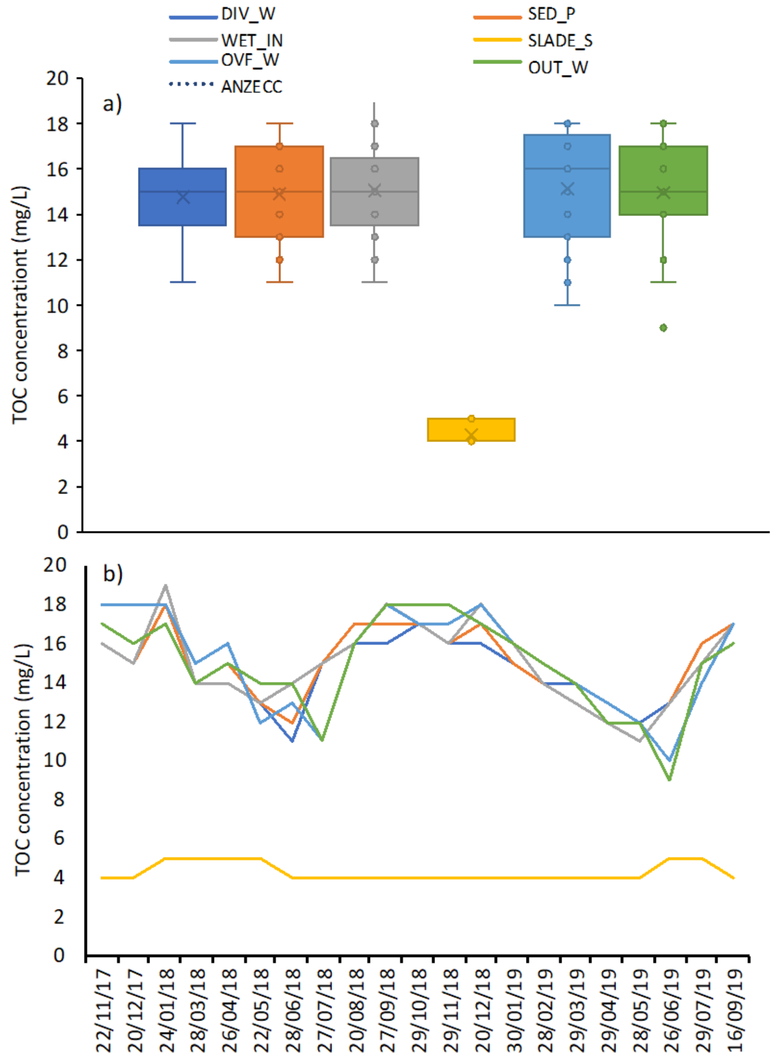

Figure 28 Total organic carbon concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site.

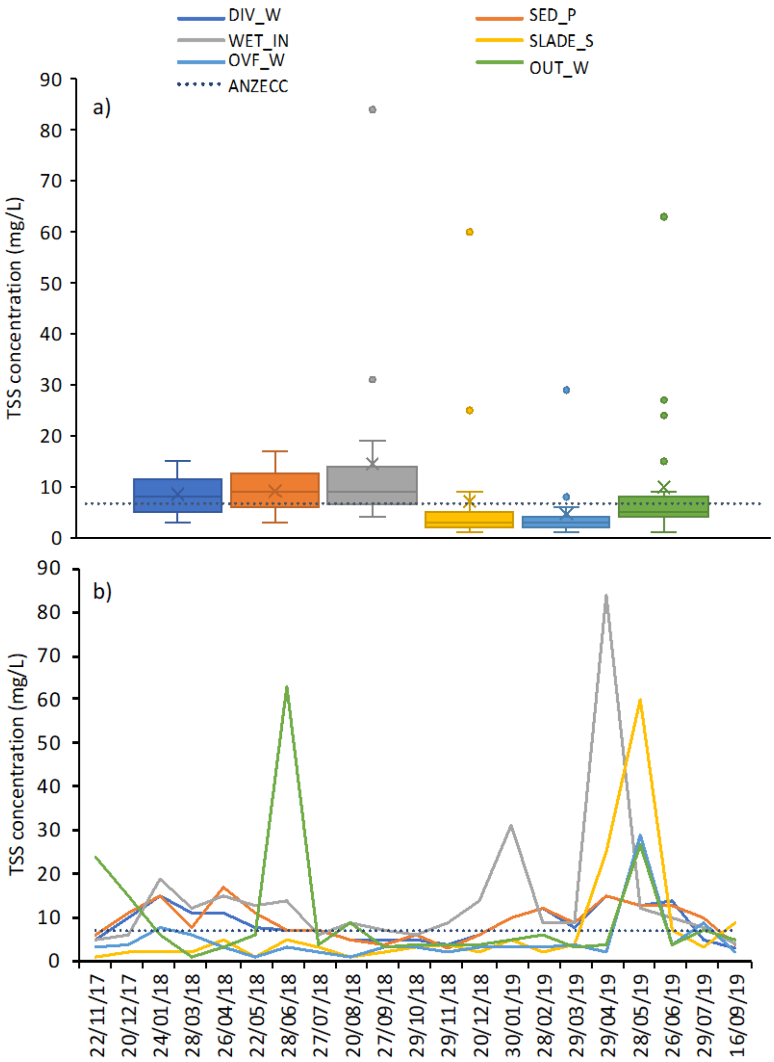

Figure 29 Total suspended solids concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

6.1.1 The gross pollutant trap

The GTP was implemented in ESBS with the prediction that it would reduce TSS loads into the wetland by 30%, and therefore reduce TP loads by 15%. However, TSS mean concentrations were 8.8 mg/L (± 4.1 mg/L) and 9.3 mg/L (± 4 mg/L) for DIV_W and SED_P respectively (Figure 29a). TP mean concentrations were 0.056 mg/L (± 0.02 mg/L) and 0.061 mg/L (± 0.02 mg/L) for the DIV_W and SED_P respectively (Figure 25a). These data indicate the GPT at times acted as a TSS and TP source to the ESBS. This situation was observed on multiple occasions throughout the sampling period (Figure 29b and Figure 26b).

TSS concentrations at DIV_W displayed no seasonal pattern and they seemed to depend more on the magnitude of baseflow or the occurrence of rainfall events ahead of the sampling dates (Figure 29b). TP concentrations at DIV_W displayed high values in April and May and lower concentrations over the winter months to early spring (e.g. high flow season) (Figure 26b).

TSS concentration attenuation was found on four sampling dates; PIT1_IN water levels suggested a decrease in inflow rate to below 35 L/s could be responsible for this performance, but this behaviour was not further investigated.

TSS concentration values at SED_P were below the interim DWER guidelines of 6 mg/L (URS 2010) in 25% of the sampled dates in the spring and early summer seasons in 2018.

As expected, the GPT had minimal impact on other nutrient concentrations (Figures 23a, 24a, 25a, 26a, 27a, 28a and 29a).

6.1.2 The sedimentation pond

This component has a single inlet and one outlet point for water levels of less than 0.94 m AHD, by means of an overflowing weir connected to a submerged pipe with final discharge to the wetland (WET_IN). Using this outlet as representative of outflows from the sedimentation pond is questionable, because a water sample was likely a mixture of wetland water as well as sedimentation pond water.

Also, the attenuation of nutrient and TSS concentrations, as indicated by the difference between SED_P and WET_IN, was impacted by the practice of discharging GPT effluent into the sedimentation pond, after cleaning the GPT. Over the sampling period, this practice occurred regularly (Table 8), therefore would likely have strongly affected the concentrations of TSS and nutrients in the pond over the sampling period. Evidence of the impact of this practice was observed on 29 April 2019, when very high TSS concentrations were measured in the pond. This was two weeks after the GPT cleaning followed by a 22 mm minor event; these may have contributed to sediment build up and reduced GPT capacity, and exacerbated the reduction of inflow to the sedimentation pond.

Table 8 Dates of GPT cleaning in 2017, 2018 and 2019. The cleaning on 6 September 2019 was undertaken by a new contractor, using a different method that did not discharge GPT effluent to the sedimentation pond.

Year	Dates of GPT cleaning
2017	21 February, 22 March, 18 April, 16 June, 3 August, 6 September, 4 October, 1 November, 14 December
2018	7 March, 4 April, 6 June, 15 October
2019	16 April, 6 September

The data showed concentration attenuation for NH3-N of 32% (on average) between October 2018 and June 2019, with higher values achieved in summer months. FRP concentration was consistently attenuated by 18% on average for the entire monitoring period (Figure 26b). There was no attenuation in concentrations for TN (Figure 23b), NOx-N (Figure 24b), and TP (Figure 25b).

We recommend changing the outlet sampling location, to facilitate a more robust assessment of concentration attenuation. One possible alternative location is the overflow grate at SED_P (Figure 6b) and preliminary sampling should be conducted to confirm if this alternative location is representative.

6.1.3 The wetland

Inlet to Overflow

The wetland component receives surface water inflows from the sedimentation pond and the Slade Street drain (SLADE_S), and a minor contribution from groundwater (GHD 2013).

TN concentrations did not show a clear seasonal pattern at the WET_IN (Figure 23b); its mean concentration value was 1.36 mg/L with a standard deviation of ± 0.26 mg/L. This contrasted with concentration at OVF_W that showed higher values for the wet winter season than those for the summer season (Figure 23b); TN mean concentration was 0.9 mg/L (± 0.34 mg/L), which was 33% lower than measured at the inlet (Figure 23a).

TP consistently showed low concentrations in winter and early spring, coinciding with the high flows at the WET_IN site (Figure 25b); no clear seasonal pattern in concentration was observed at the OVF_W sampling site but substantial attenuation occurred (Figure 25b). TP mean concentrations were 0.071 mg/L (± 0.03 mg/L) and 0.034 mg/L (± 0.02 mg/L) for the WET_IN and OVF_W sites respectively; TP concentration showed 52% attenuation along the wetland component (Figure 25a).

TN was made up of 14% of NH3-N, 33% of NOx-N and 34 % of dissolved organic nitrogen at WET_IN and this partitioning was consistent with that of the diverted Bayswater Brook waters. Inorganic nitrogen concentrations displayed slightly higher values in winter, coinciding with early flows of the wet season, but concentration values decreased over the summer months (Figure 24b). Mean concentration values of 0.2 mg/L (± 0.1 mg/L) and 0.44 mg/L (± 0.1 mg/L) were obtained for NH3-N and NOx-N (Figure 24a) respectively.

TN composition and inorganic nitrogen concentrations were different at OVF_W; NH-3-N and NOx-N represented 4% and 14% of TN respectively, with dissolved organic nitrogen making up 63% of TN. During the winter months, inorganic nitrogen concentrations were 72% (on average) of the inlet concentrations (Figure 24a); the NOx mean concentration of 0.47 mg/L likely resulted from winter inflows to the wetland. Dissolved organic N mean concentration at OVF_W was 0.57 mg/L (± 0.13 mg/L), which was 21% higher than measured at WET_IN.

FRP constituted a small proportion of TP at 12% (on average) for the wetland waters and it did not display a clear seasonal pattern in concentrations (Figure 26b). However, 70% attenuation was observed along the wetland, and resulted in a mean concentration of 0.003 mg/L (± 0.002 mg/L) at OVF_W (Figure 26a).

TSS mean concentrations were 11.4 mg/L (± 6 mg/L) and 4 (± 3 mg/L) at WET_IN and OVF_W respectively (Figure 29a). Concentrations attenuation across sampling dates indicated the wetland achieved 72% attenuation on average, suggesting the wetland acts as sink for sediments.

The wetland experienced changes in functioning due to management operations and they affected the Outflow structure capacity to discharge into Bayswater Brook. At the Outflow structure location, the wetland presents a deep-open water area and discharge occurred from late March to early June 2019 (see Section 5.5). This provides context for the analysis of results for nutrient concentrations.

Comparison of water quality at Overflow and Outlet structure sites

For ease of analysis, TN and TP concentrations at the Outflow and the Overflow sites are presented in Figure 30.

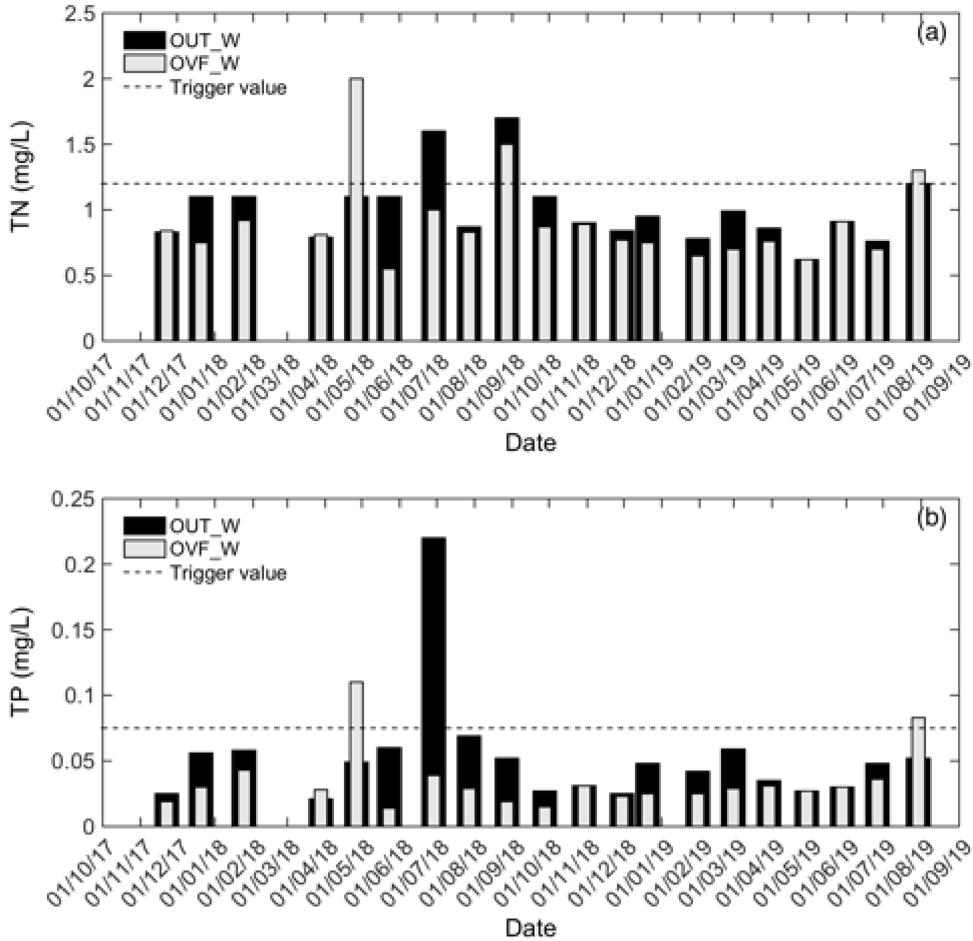

Figure 30 Comparison of nutrient concentrations at OUT_W and OVF_W: a) total nitrogen; b) total phosphorus. Dashed line represents trigger value (ANZECC & ARMCANZ 2000).

TN mean concentration at OUT_W was 1 mg/L (± 0.27 mg/L) (Figure 30a) and it was made up of 8% as NH3-N, 19% of NOx-N and 54% of dissolved organic nitrogen. An increase in the proportion of inorganic nitrogen (NH3-N and NOx-N) corresponded with a drop in dissolved organic nitrogen; these partitions in TN composition were higher than those at OVF_W.

The TP mean concentration was 0.052 mg/L (± 0.04 mg/L) at OUT_W, an increase of 52% over what was measured at OVF_W. It also showed more seasonal variability compared with OVF_W and WET_IN (Figure 30b).

Concentrations for inorganic nitrogen and FRP are shown in Figure 31. Concentrations at OUT_W were consistently higher than those at OVF_W; the exception being mostly winter samples in 2018 for NOx-N (Figure 31b).

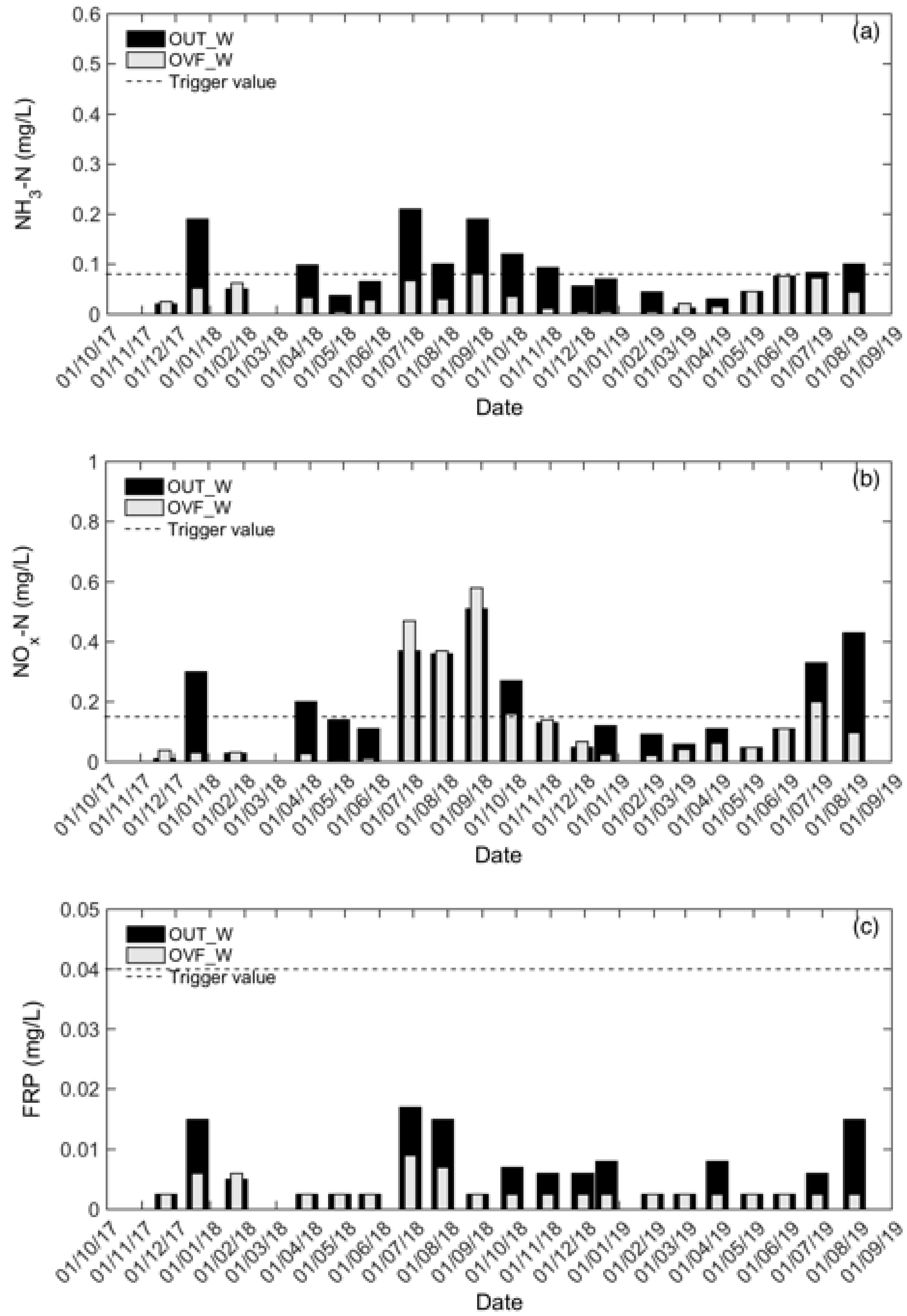

Figure 31 Comparison of nutrient concentrations at OUT_W and OVF_W : a) NH3-N; b) NOx-N; c) FRP. Dashed line represents trigger value (ANZECC & ARMCANZ 2000).

FRP mean concentration also increased by 130 % at the OUT_W (Figure 31c), with a mean value of 0.007 mg/L (± 0.005 mg/L). This increase was also reflected in TP composition with FRP accounting for 13% of TP.

TSS mean concentration at OUT_W was 6 mg/L (± 5 mg/L), slightly higher and more variable than that measured for the OVF_W (Figure 32).

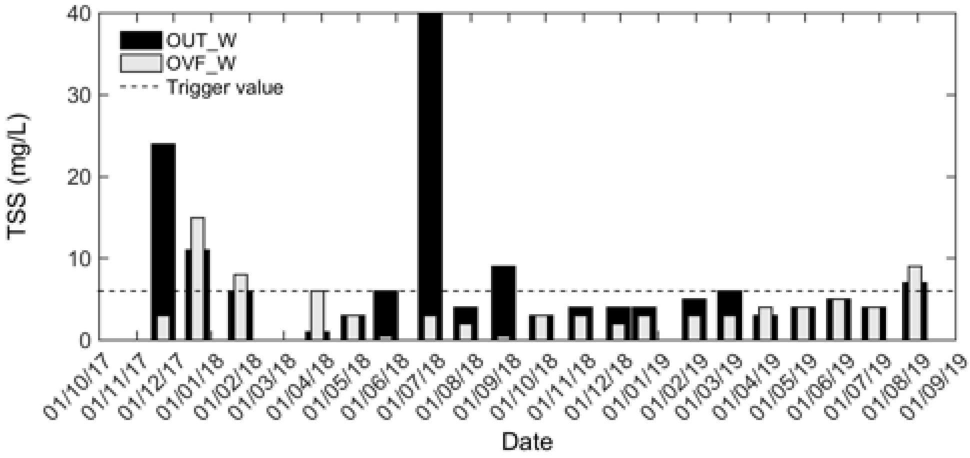

Figure 32 Comparison of TSS concentrations at OUT_W and OVF_W. Trigger value corresponds to interim guideline concentration of 6 mg/L (Department of Water and Environmental Regulation).

The comparison of concentrations at OUT_W and OVF_W and their temporal variability suggested an accumulation of nutrients around the Outflow structure area; this area acted as a flow stagnation point with no discharge for most of the monitoring period. The temporal variability of TN, TP and NOx also suggested OUT_W may be more impacted by direct inflow and nutrient loading from Bayswater Brook during events. This hypothesis requires further investigation.

6.1.4 The Slade Street drain

Unlike the other sampling sites, the Slade Street drain inflow to the wetland (SLADE_S) displayed a seasonal pattern in nutrient concentrations that reflected a different composition of water entering the wetland component. Nitrogen and phosphorus totals and species concentrations are presented in Figure 33 and Figure 34.

TN concentration displayed lower values in the summer, increased over the autumn and winter seasons and peaked by August or September each year (Figure 33). TN mean concentration was 1.64 mg/L (± 0.3 mg/L) and higher than those recorded for the other sampling sites. The TN composition changed remarkably, with an increase in the proportion of inorganic forms: 63% as NOx-N, 19% as NH_3-N, and only 16% of dissolved organic nitrogen. Mean concentration for NOx-N was 1.04 mg/L (± 0.3 mg/L) and all nitrogen concentrations were above trigger values.

The Slade Street drain exhibited nitrate contamination, with maxima of 1.6 mg/L observed in winter in 2018 and 2019. Interestingly, GHD (2009) showed local groundwater was contaminated with very high concentrations of TN (up to 148 mg/L), dominated by organic nitrogen and ammonia-N (up to 44 mg/L). Groundwater nitrate concentrations were all less than 0.03 mg/L. The high nitrate concentrations and low ammonia-N concentrations observed in the present study suggested oxygenated conditions in the drain waters, and has promoted nitrification that has transformed groundwater ammonia-N into nitrate-N.

TP concentration showed no clear seasonal pattern as high concentrations were observed in winter and summer seasons (Figure 34); TP mean concentration was 0.04 mg/L (± 0.01 mg/L) and most of the P was in particulate form (58%). FRP mean concentration was 0.004 mg/l (± 0.003 mg/L) and it made up 11% of TP. Unlike nitrogen, P concentrations were all below trigger values.

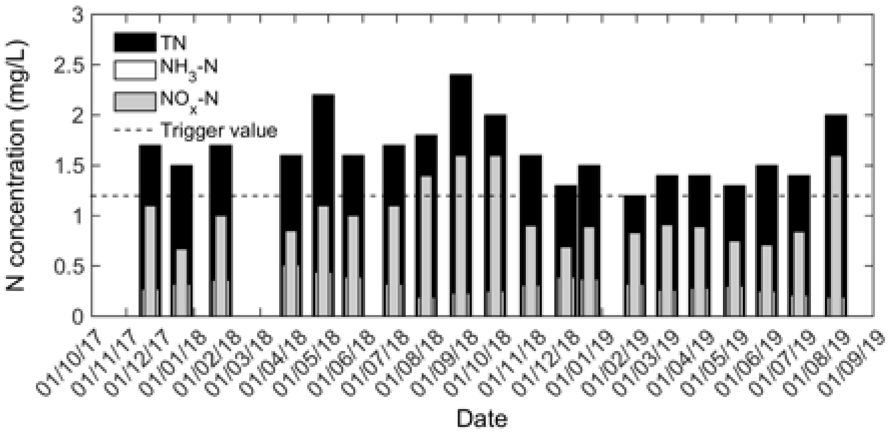

Figure 33 Comparison of nitrogen species concentrations at the SLADE_S sampling site. Dashed line represents trigger value for TN (ANZECC & ARMCANZ 2000).

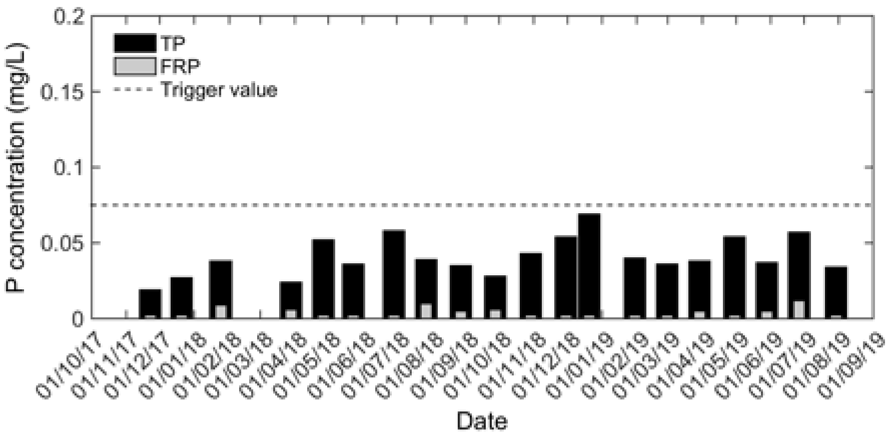

Figure 34 Phosphorus concentrations at SLADE_S. Dashed line represents trigger value for TP (ANZECC & ARMCANZ 2000).

Nutrient concentration speciation, seasonal pattern and magnitude seem to agree with the water level dynamics observed at SLADE_S (see Section 4.5), in that groundwater was intersected by the drainage pipe and discharged into the wetland component area. Previous studies showed the local shallow groundwater system was characterised by high TN concentrations and electrical conductivity (EC) in the range of 1395-2360 µS /cm (GHD 2009; URS 2010). Measured EC values at SLADE_S were consistently higher than those recorded at the other sampling sites and ranged from 887–1021 µS/cm, representing a mixture of surface water runoff (from events) and local shallow groundwater. Further work is required to properly quantify the groundwater component discharge to the wetland via the Slade Street drainage pipe.

6.1.5 Is the wetland achieving concentration targets?

Overall the wetland component reduced nutrient and TSS concentrations to below target concentrations (ANZEEC & ARMCANZ 2000) at OVF_W. The high efficiency in TSS removal seems to also help the wetland achieve P concentration targets.

The mean concentration at OVF_W for TN of 0.9 mg/L was below the target value of 1.2 mg/L, and this included winter samples taken during periods of high TN loading from the catchment. The wetland achieved guideline concentration for long-term targets in 85% and 80% of the water samples for TN and NOx-N concentrations respectively. Mean concentration values for both TN and NOx-N can achieve below guideline values if winter samples (particularly those in 2018) were not included in the calculation.

Importantly, the concentrations were lower than those reported for the wetland area before its rehabilitation work. Wetland nutrient concentrations historically exceeded 3 mg/L for TN (mostly as NH3-N) and 0.2 mg/L for TP (NMCG 2007; URS 2010); concentrations under dry summer conditions exceeded the guidelines by up to 20 and 7 times for TP and TN respectively (URS 2010).

6.1.6 Concentrations and attenuation during intended functioning periods

All wetland section components were operating (or close to) as intended functioning from 29 March to 5 June 2019, before weir boards were removed at the Overflow structure to manage the winter flow event season. Monthly average inflow rates to the sedimentation pond were 47 L/s for April 2019 and 32 L/s for May 2019 for baseflow periods. The drop in inflow rate in May represented 36% of the intended functioning inflow rate (e.g. 50 L/s), but average inflow increased to 38 L/s (± 5 L/s) in the week before sampling on 29 May 2019. During these conditions, concentration attenuation between the GPT and wetland outlet can be used to assess the intended performance of the treatment train (Table 9).

Table 9 Measured concentrations during periods that corresponded to intended wetland design functioning. ANZECC guidelines are given for nutrients.

Date		TP [mg/L]	TN [mg/L]	FRP [mg/L]	NOx-N [mg/L]	NH_3-N [mg/L]	DON [mg/L]	TSS [mg/L]
Targets		0.065	1.2	0.04	0.15	0.08		6
29/3/19	SED_P	0.066	1.100	0.008	0.420	0.190	0.490	9
	OVF_W	0.031	0.760	0.002	0.062	0.015	0.553	4
	OUT_W	0.035	0.860	0.008	0.110	0.030	0.620	3
29/4/19	SED_P	0.070	1.200	0.008	0.450	0.270	0.280	15
	OVF_W	0.015	0.530	0.002	0.005	0.014	0.431	2
	OUT_W	0.027	0.620	0.002	0.047	0.045	0.358	4
28/5/19	SED_P	0.068	1.300	0.009	0.470	0.220	0.610	12
	OVF_W	0.029	0.960	0.002	0.042	0.014	0.624	3
	OUT_W	0.030	0.910	0.002	0.110	0.076	0.494	5
Average	SED_P	0.068	1.200	0.008	0.450	0.230	0.460	12
	OVF_W	0.025	0.750	0.002	0.036	0.014	0.536	3
	OUT_W	0.030	0.800	0.004	0.090	0.050	0.490	4

* Corresponds to un-ionised ammonia-N at 20°C.

Results indicated both OVF_W and OUT_W concentrations were below guidelines for all nutrients.

The ESBS treatment components from GPT to OUT_W sites achieved average concentration attenuations of 78% for NH-3, 80% for NOx-N and 33% for TN; the lower TN attenuation reflected the large proportion of dissolved organic nitrogen entering and available in the system. High concentration attenuation was also observed for FRP (50%), TP (56%) and TSS (67%). A slightly higher attenuation (e.g. 12% increase on average) was observed for the OVF_W site with the exception of dissolved organic nitrogen (DON); the wetland component alone sourced 17% more DON than the inflow (Table 9).

6.2 Impact of GPT operations

The GPT was cleaned out twice during the monitoring period in 2019: 16 April 2019 and 6 September 2019 (CoB *personal communication*; Santiago 2019). Different contractors were used each time, with different cleaning methods. On 16 April 2019, the operator vacuumed a single quadrant of the GPT, due to restricted access caused by the GPT lid. The extracted material from the GPT was temporarily stored in the operator's truck, before the dirty water was pumped back into the sedimentation pond (Santiago 2019).

On 6 September 2019, a different contractor used a different cleaning method. The operator plugged the inlet pipe at the diversion weir, and two outlets of the GPT, restricting flows in and out. One person entered the GPT, and manually shovelled the material trapped in the inaccessible quadrants into the single quadrant that was accessible for vacuuming. The operator vacuumed that quadrant, completely emptying the GPT, and then removed the extracted material from the site. No extracted liquid was pumped back into the sedimentation pond.

The GPT maintenance schedule and cleaning method impacted its hydraulics and effectiveness in removing pollutants, including sediments. When the GPT was full, it appeared to prevent the full 200 L/s being diverted from Bayswater Brook, and also blocked off the return flow back to Bayswater Brook. The GPT also became a source of sediment to the ESBS. The practice of waste release to the sedimentation pond, as implemented by the first contractor, triggered a major spike in TSS concentration that was still evident in the sedimentation pond two weeks later. Interestingly, the wetland appears to have removed these sediments from the water, before discharge back to Bayswater Brook.

6.3 Baseflow conditions: Load attenuation across ESBS

Nutrient and sediment loads for the inflow PIT1_IN and outflow OVF_W were computed, using equation 3 (see Section 3.6) and used to assess attenuation across ESBS under baseflow conditions. Values correspond to daily total mass (e.g. kg/day) for each sampling date and are presented in Table 15 (Appendix 1-B). Outflow load computations were not possible on six sampling dates due to lack of a rating curve for OUT_W, which was at that time acting as the main discharge point; the sampling dates shown below corresponded to *as-designed* conditions, and so PIT1_IN loads were computed for these dates. Outflow load computations could be performed when a OUT_W rating becomes available.

Nutrient and sediment loads displayed larger variability than observed for concentrations, reflecting changes in inflow and outflow rates to the wetland; these flow rates responded to seasonal changes in hydrology and also management operations.

TN loads into PIT1_IN were low, at 1.48 kg/day and 0.72 kg/day in August 2018 and July 2019 respectively (Figure 35a); low inflow rates were attributed to GPT blockage and therefore low flow rates. The wetland was functioning *as-designed* in the spring 2017 and autumn 2019 seasons; TN load at PIT1_IN during these periods averaged 4.6 kg/day and 4.1 kg/day respectively and displayed more variability in autumn (due to small rainfall events).

The largest TN load of 10.5 kg/day was observed in October 2018 and resulted from a high inflow rate of 109 L/s reflecting high baseflow conditions in the Bayswater Brook (see section 4.3.1). TN loads at OVF_W were lower than measured at PIT1_IN, except for four sampling dates; TN loads ranged from 0.6 kg/day to 9.3 kg/day with the largest value observed in October 2018 (Figure 35a).

TP loads at PIT1_IN and OVF_W are presented in Figure 35b. TP load variability across years closely followed that previously described for TN; it was driven by inflow and outflow rates changes, in response to the hydrology and wetland management operations.

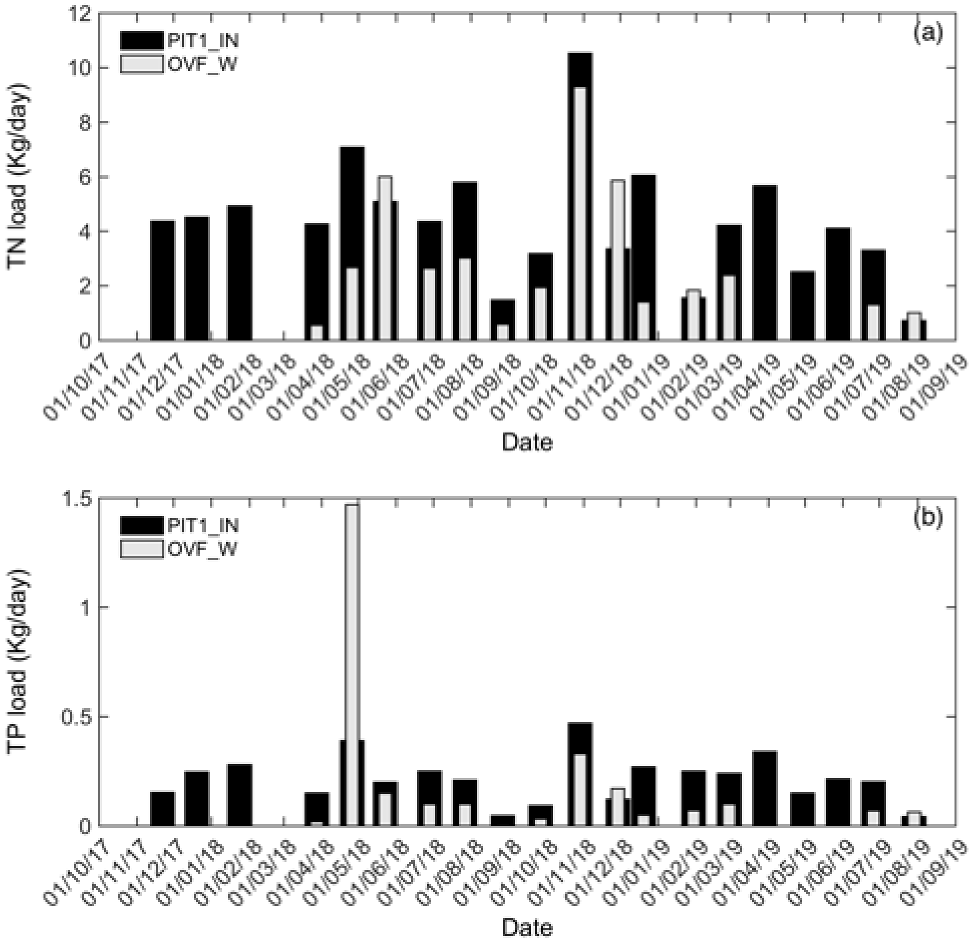

Figure 35 Total nutrient loads for baseflow conditions: a) TN load; b) TP load.

TP load ranged from 0.04 kg/day to 0.45 kg/day at PIT1_IN; these values corresponded to low and high inflow rates experienced by the ESBS. Unlike TN, there was no difference in mean TP load over the period of *as-designed* wetland functioning, with an average value of 0.23 kg/day (Figure 35b). In general, TP loads were lower at the OVF_W site than those entering the treatment train, except for a value of 1.47 kg/day sampled in April 2018. This load resulted from an increase in concentrations and also outflow rates at OVF_W. A close examination of water levels in the wetland suggested the possibility of a management operation after the GPT clean up in early April, which involved adding and later removing weir boards at OVF_W. TP loads were also slightly higher than those for the inflow site on two occasions (Figure 35b), when inflow rates were reduced at 35 L/s and as low as 5.5 L/s due to GPT blockage.

Loads for NH3-N, NOx-N and FRP at the PIT1_IN and OVF_W sites are shown in Figure 36. Inorganic nitrogen loads displayed a similar seasonal pattern as those for concentrations but they increased in winter when concentrations typically were low (see Figure 24b and Figure 26b).

At PIT1_IN, NH3-N load ranged from 0.16 kg/day to 1.43 kg/day and these two values resulted from inflow rates outside *as-designed* wetland functioning (Figure 36a). Average NH3-N loads of 0.71 kg/day and 0.75 kg/day were found for the spring season in 2017 and the autumn season in 2019 respectively. These loads were close to the overall average load for the inflow, of 0.7 kg/day. A substantial reduction in NH3-N load was observed at the OVF_W site (Figure 36a) with values ranging from 0.009 kg/day to 0.30 kg/day; the largest load value occurred in May 2018 coinciding with high water levels in the wetland (0.707 m AHD).

Figure 36b shows NOx-N loads for PIT1_IN and OVF_W. At the inflow, NOx-N loads ranged from 0.21 kg/day to 2.9 kg/day, with a mean value of 1.58 kg/day. NOx-N loads during periods of *as-designed* functioning were slightly lower than the average load for the site, at 1.4 kg/day. Low loads observed on five sampling dates corresponded to inflow rates less than 30 L/s. Attenuation of NOx-N load was observed at OVF_W, with loads ranging from 0.019 kg/day to 1.47 kg/day. High NOx-N loads observed in winter 2018 were attributed to increases in concentrations at OVF_W rather than outflow rates. This result contrasted with the high load observed in November 2018 when a large inflow and low NOx-N concentrations occurred, with the flow driving the increase in NOx-N load.

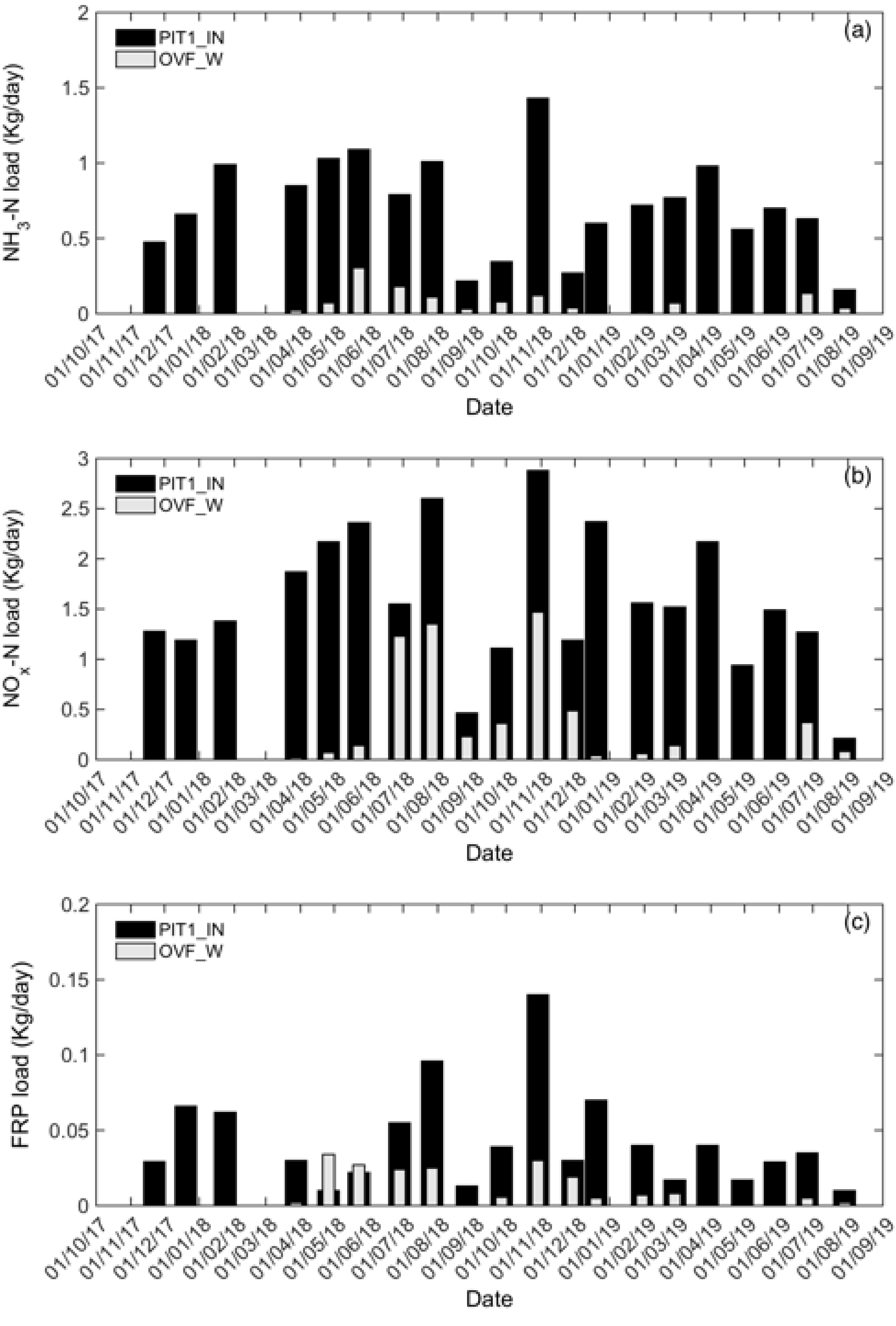

Figure 36 Dissolved inorganic nutrient loads for baseflow: a) NH3-N; b) NOx-N; c) FRP.

FRP loads for PIT1_IN and OVF_W are shown in Figure 36c. FRP loads at PIT1_IN ranged from 0.01 kg/day to 0.14 kg/day, with a mean value of 0.04 Kg/day. Unlike the TP loads that showed no change in mean load during periods of *as-designed* functioning, FRP loads showed higher values in the spring season 2017 than in the autumn season 2019; FRP loads were 0.05 kg/day and 0.03 kg/day for these periods respectively (Figure 36c). High FRP loads corresponded with high inflow rates observed at PIT1_IN rather than increases in concentration.

A large FRP load attenuation was observed at OVF_W (Figure 36c). Loads ranged from 0.0001 kg/day to 0.034 kg/day. FRP loads were higher than those measured at the inflow on two sampling dates; these corresponded to an increase in observed TP loads, as above presented.

Dissolved organic nitrogen loads at PIT1_IN and OVF_W are presented in Figure 37. As discussed in Section 6.1.3, DON made up a large proportion of TN and likely affected TN load attenuation. DON loads at PIT1_IN were driven mainly by inflow rate variability rather than concentrations; the largest load of 5.2 kg/day was observed on 29 October 2018 at times of a high inflow rate (109 L/s). Inflow DON loads during *as-designed* functioning periods were on average 2.1 kg/day and 1.7 kg/day for the spring 2017 and autumn 2019 seasons respectively. DON loads at OVF_W were higher than for those at PIT1_IN on six sampling occasions and were driven by both high outflow rates and concentration values.

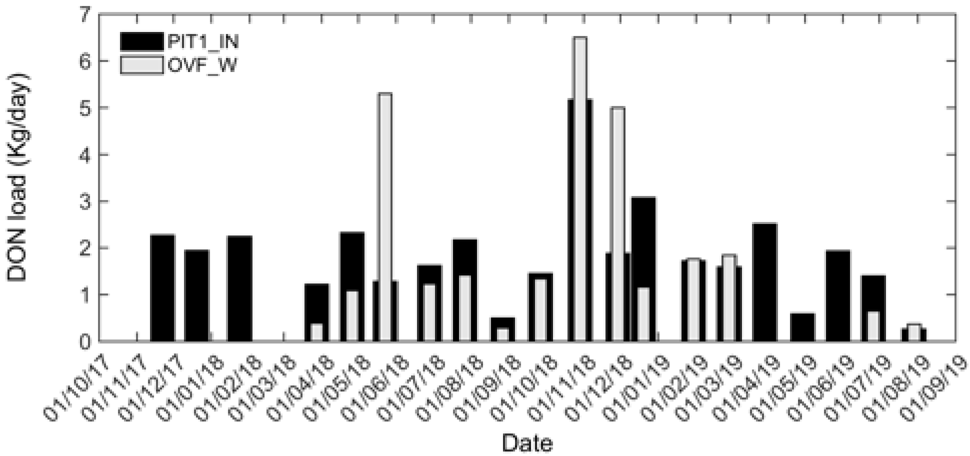

Figure 37 Dissolved organic nitrogen loads for inflow and outflow sites.

Figure 38 shows TSS loads at PIT1_IN and OVF_W. PIT1_IN loads ranged from 5 kg/day to 67 kg/day, with a mean value of 33 kg/day. The largest load was observed on 26 April 2018, and corresponded to an increase in TSS concentration, under close to expected inflow rate condition (50 L/s). There was no difference in average TSS load between the two periods in spring 2017 and autumn 2019 seasons under *as-designed* functioning condition; however, the spring season showed a trend of increasing load driven by increases in concentration. TSS load for these periods was 39 kg/day on average.

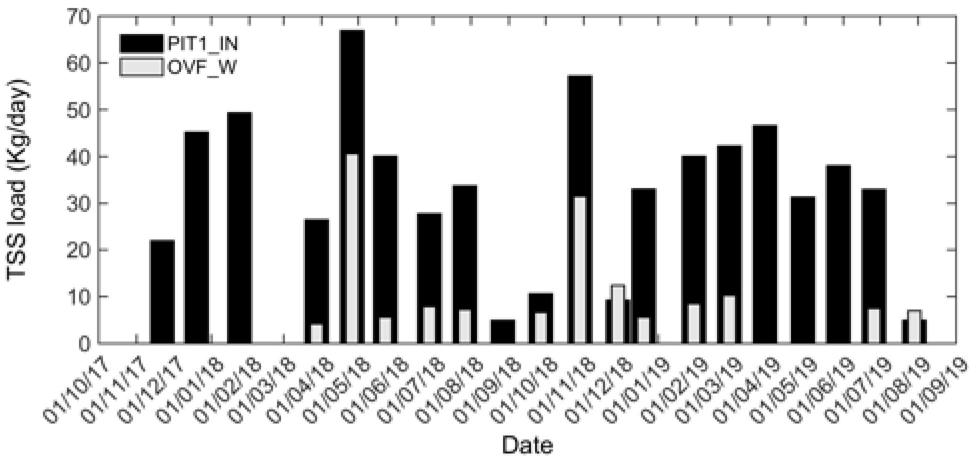

Figure 38 Total suspended sediments load for inflow and outflow sites.

TSS loads were attenuated by the time OVF_W was reached and loads at that outflow ranged from 0.2 kg/day to 41 kg/day; the mean TSS load at the outflow was 11 kg/day. Two large TSS loads resulted mainly from high flow rates at OVF_W but also from an increase in TSS concentrations on 26 April 2018 (Figure 38), likely to be the result of wetland management operations as discussed for TP load.

Finally, reported loads in this study were compared with previous load estimates used to assess ESBS's nutrient balance before its rehabilitation work; these loads were estimated at 5.75 kg/day and 0.3 kg/day for TN and TP respectively (GHD 2009). The present study found PIT1_IN TN loads of 4.6 kg/day and tp load of 0.23 kg/day under *as-designed* functioning periods were slightly lower than those reported by GHD (2009). However, observed average loads at OVF_W were very different to those reported. The observed loads at OVF_W of 2.8 kg/day for tn and 0.1 kg/day for TP were substantially larger than the loads of 0.57 kg/day and 0.07 kg/day estimated by GHD (2009) for TN and TP respectively. Current observations confirmed a lower attenuation capacity of the system, compared with what was predicted after ESBS wetland rehabilitation (GHD 2009).

6.4 Stormflow conditions: Concentration attenuation across ESBS

For completeness, this section presents a brief overview of concentration patterns and concentration attenuation (e.g. initial vs final) for TN, TP and TSS over the course of each event.

Please note we assessed nutrient attenuation performance during a period of high water levels in Bayswater Brook, so the system did not perform hydraulically as intended. Nutrient attenuation is expected to improve when maintenance activities reduce the water height in the brook.

Event 1 was the first significant rainfall event after the long dry spell of the summer and autumn seasons. Inflow concentrations displayed a typical "flushing" pattern with concentrations peaking before the flow discharge peak (Figure 39a-c). TN concentration dropped from 3.1 mg/L to 1 mg/L from the start of the event to its peak discharge but slightly increased to 1.1 mg/L towards the end of the event hydrograph (Figure 39a). TP concentration initially displayed the same pattern, with a drop from 0.94 mg/L to 0.15 mg/L at the time of the peak but concentration decreased to 0.062 mg/L by the end of the event (Figure 39b).

TSS concentration also showed a flushing pattern with a 10-fold reduction from its initial value of 260 mg/L to that at the peak flow of 25 mg/L; concentration further decreased by 76% by the end of the event. The similarities in TSS and TP patterns suggested most TP attenuation can be related to particulate form of P or dissolved reactive P attached to sediments.

Outflow concentrations for TN, TP and TSS are shown in Figure 39d-f; concentrations were slightly higher than those observed for the inflow. This is particularly true over the first part of the event with values of up to 3.4 mg/L, 1.1 mg/L and 340 mg/L for TN, TP and TSS respectively. Higher concentrations could result from resuspension and/or mobilisation by high runoff rates of particulate material available on the ground surface and at the wetland edges. Overall, concentrations temporal patterns remained the same as those observed for the inflow.

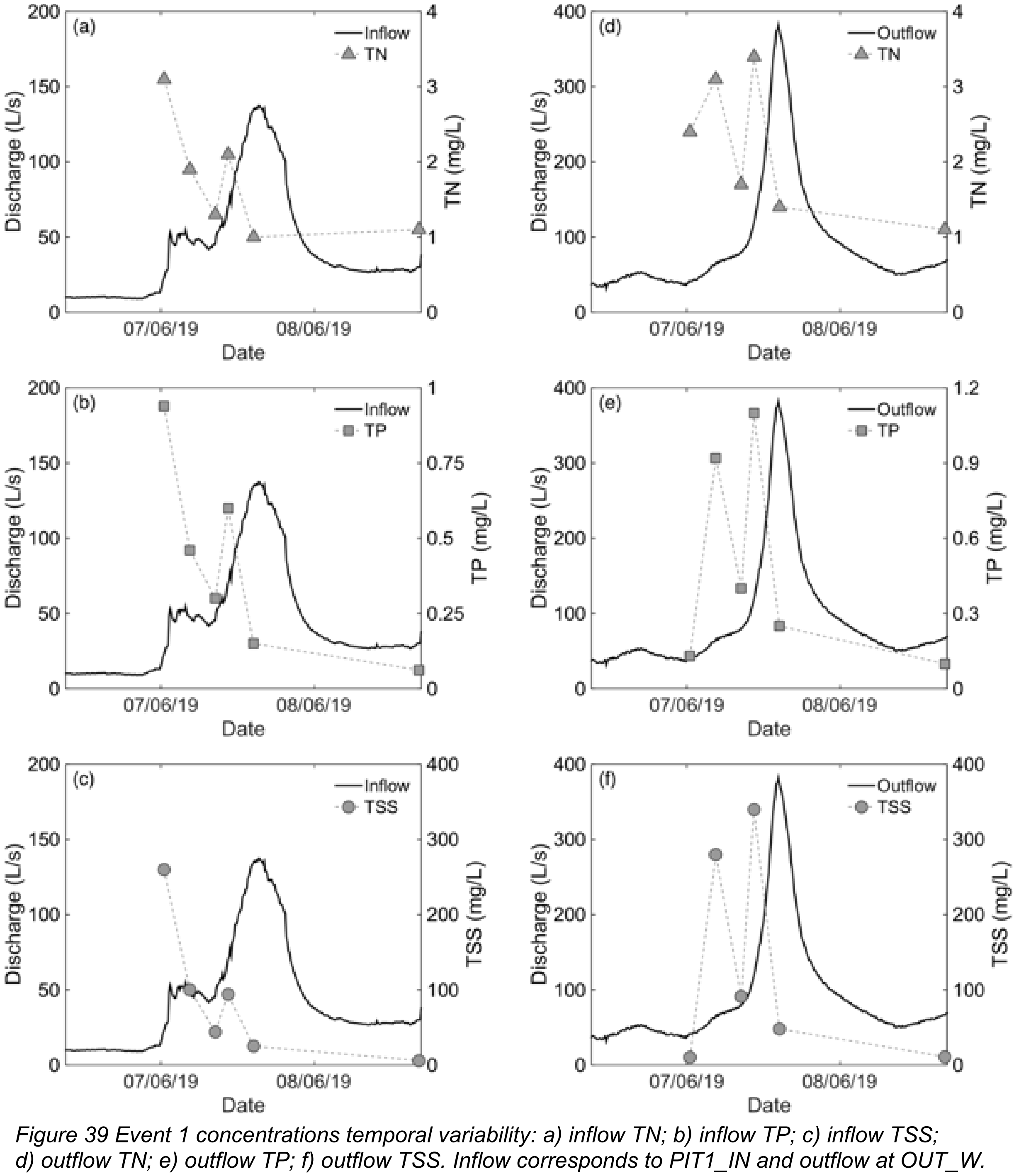

Figure 39 Event 1 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OUT_W.

Event 2 occurred under wetter antecedent conditions than those for Event 1 and concentration patterns began to show clear differences for TN compared with TP and TSS (Figure 40). Initial TN concentrations were lower than those for Event 1 by 60%, and decreased towards the peak flow but recovered to pre-event values by the end of the event (i.e. a typical dilution pattern, Figure 40a). TP and TSS concentrations displayed decreasing concentrations towards the end of the event (Figure 40b,c) and had similar attenuations (84% and 89% respectively).

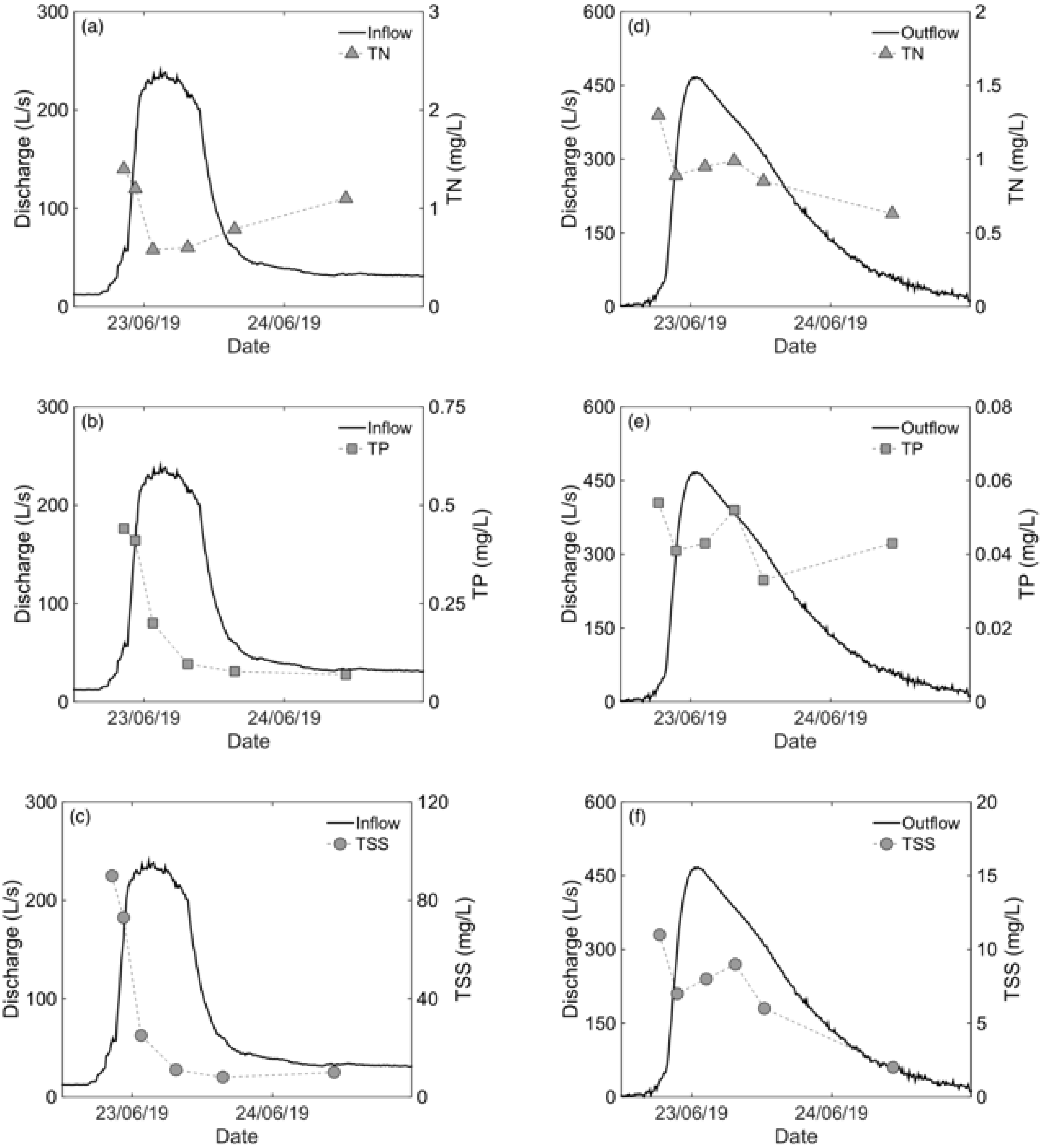

Figure 40 Event 2 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W.

The TN outflow concentration pattern differed from that of the inflow as concentration values continued their decline towards the end of the event (Figure 40d). An overall attenuation in concentration of 57% (e.g. within the event for this location) was recorded at the outflow station. TP concentration did not display a clear pattern (Figure 40e) as that for the inflow station but concentrations remained in the range 0.035–0.055 mg/L over the course of the event. Despite the large influx of TP at the inflow site, the outflow concentrations showed an attenuation of approximately 50% (e.g. initial vs final average concentrations). TSS did follow the same declining pattern as that for Event 1 with a reduction of 83% between initial and final concentrations values (from 12 mg/L to 2 mg/L, Figure 40f).

Event 3 inflow concentrations are shown in Figure 41a-c. We observed no change in overall temporal patterns for inflow TN, TP, and TSS concentrations when compared with those for Event 2; however initial concentrations for the event dropped by 48% for TP and 43% for TSS. This result suggested less TP and TSS were available for transport from the catchment and contrasted with TN, which experienced a minor concentration change across winter flow events. The attenuation in concentration values (e.g. initial vs final) was found at 79% and 91% for TP and TSS respectively.

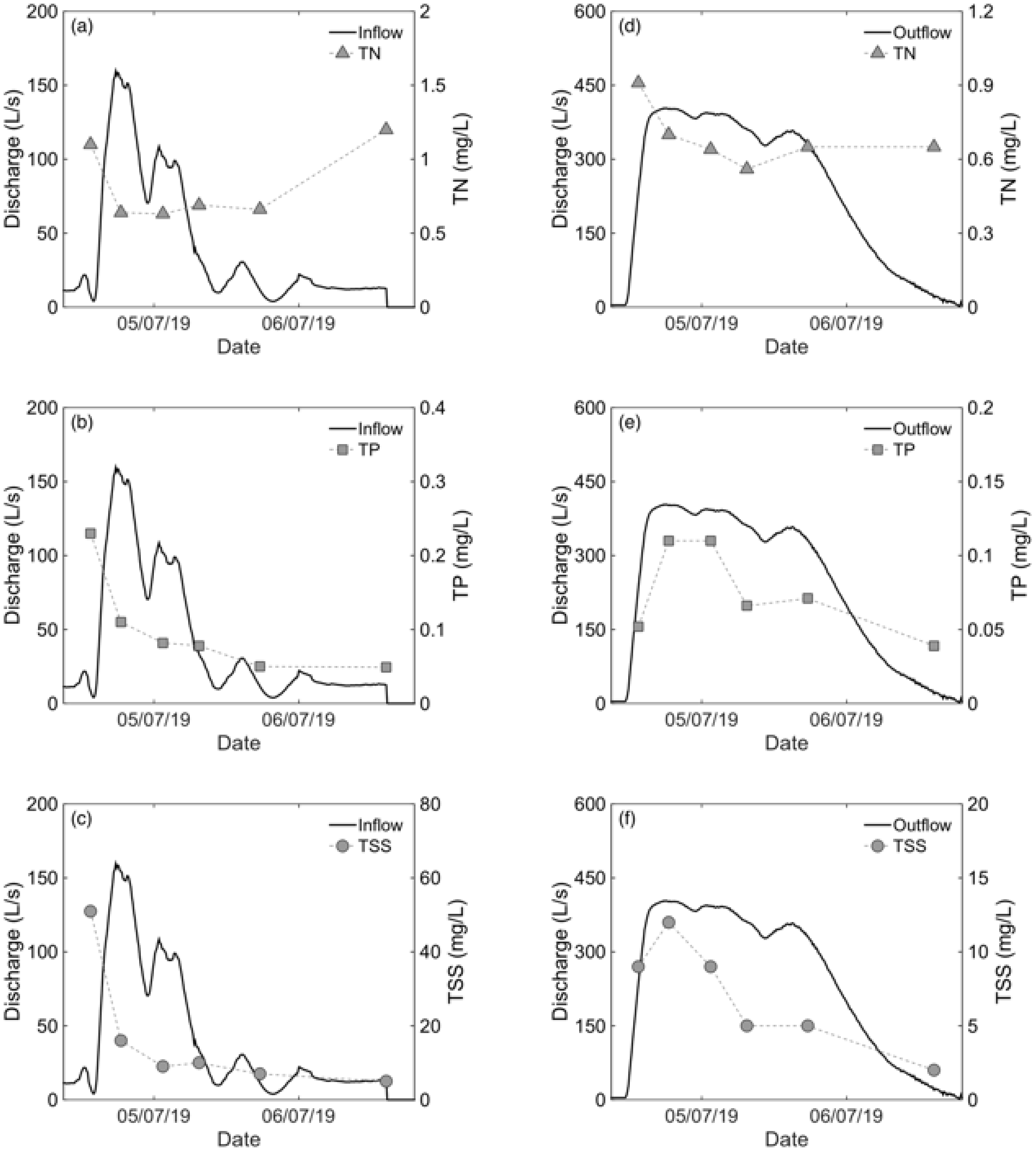

Figure 41 Event 3 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W.

Outflow TN concentration changed neither temporal pattern nor magnitude of concentration values and it contrasted with observations for TP and TSS (Figure 41d-f). TP and TSS concentrations increased over early stages of the event hydrograph (Figure 41e,f), responding to a large input of TP and TSS observed for the inflow hydrograph. However, they reversed this trend towards the end of the event. The attenuation in concentrations (e.g. initial vs final) was found at 64% and 83% for TP and TSS respectively.

Event 4 inflow concentrations are shown in Figure 42a-c.

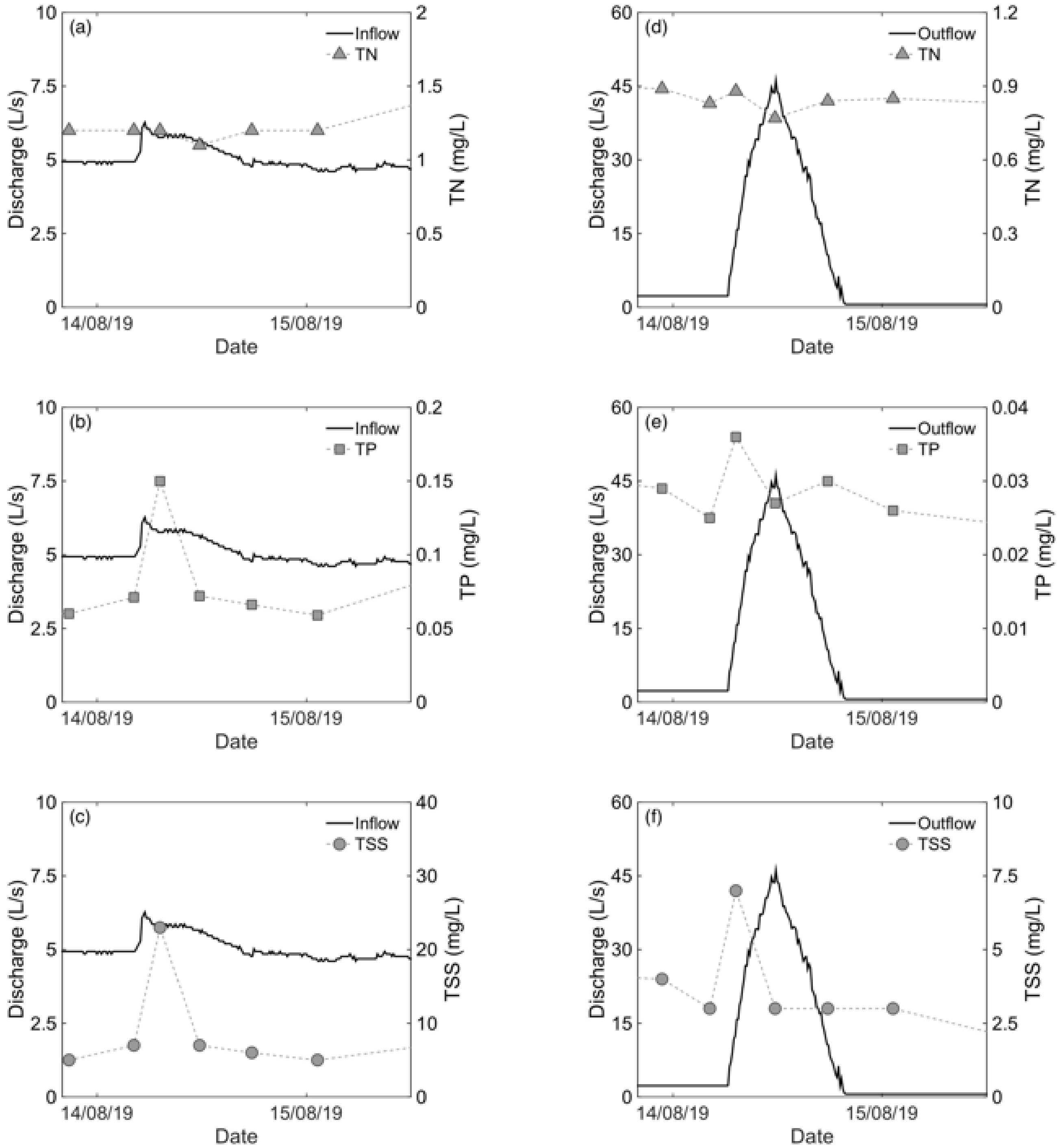

Figure 42 Event 4 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W.

As for previous events, TN concentration continued its pattern of no reduction in concentrations, with an average value of 1.1 mg/L. However, TP and TSS displayed a minor change in pattern with a peak in concentration lagging that of the flow hydrograph. This result is likely to be mobilisation of P and sediments from a localised source. Initial concentration values for TP and TSS at the early stages of the event dropped by a further 35% and 51% respectively from those observed for Event 3.

TN outflow concentration temporal pattern did not change and the event presented an average concentration value of 0.83 mg/L (Figure 42d). Peaks in TP and TSS outlet concentrations were observed over the rising limb of the flow hydrograph but rapidly dropped and remained at lower concentrations towards the end of the event (Figure 42e,f). Concentrations experienced attenuation of 27% and 57% for TP and TSS respectively over the course of the event.

Event 5 inflow concentrations are presented in Figure 43a-c. Although temporal patters remained the same as those for previous events, concentration values at early stages of the event increased for TN, TP and TSS. Higher initial concentrations resulted in higher values towards the end of the event and overall mean concentration values. TN mean concentration was 1.1 mg/L while initial concentrations for TP and TSS increased to 0.51 mg/L and 110 mg/L respectively (Figure 43b,c). Attenuation in concentrations (e.g. initial vs final) increased slightly to 89% for TP and 94% for TSS, relative to those for Event 2. It is likely the characteristics of the event (short duration-high intensity) contributed to the mobilisation of nutrients and sediments resulting in high concentration values.

Figure 43d-f shows outflow concentrations for TN, TP and TSS. TN mean concentration for the event dropped slightly to 0.82 mg/L (Figure 43d). Concentrations values for TP and TSS increased and reflected those observed in the inflow (Figure 43e,f). Nevertheless, attenuation in concentration (e.g. initial vs final) over the course of the event remained high at 79% and 93% for TP and TSS respectively.

Dissolved organic nitrogen (DON) contributed on average 32% of the inflow TN concentration but 55% of the outflow TN concentrations.

These observations led us to the following conclusions:

- The lack of change in event temporal pattern and concentrations for TN (e.g. between events 2 to 5) at both PIT1_IN and OVF_W suggests the Bayswater Brook catchment area contains a sustained source of nitrogen that the wetland has less capacity to treat. A large proportion of DON (e.g. 32% and 55% for inflow and outflow respectively) and the short residence time available for treatment of the wetland component during events likely explain the lack of TN event concentration variability.
- Consistency in TP and TSS concentration patterns across the winter events (events 2 to 5) highlight a limited source of TP and TSS from the Bayswater Brook catchment area. The reduction in concentrations from inflow to outflow suggests that the wetland component is highly efficient at removing sediment and that particulate P and possibly FRP attached to sediment particles.

The following sections will present results of event nutrient and sediments mass balances, event mean concentration (EMC) values, and load computations to assess wetland treatment performance.

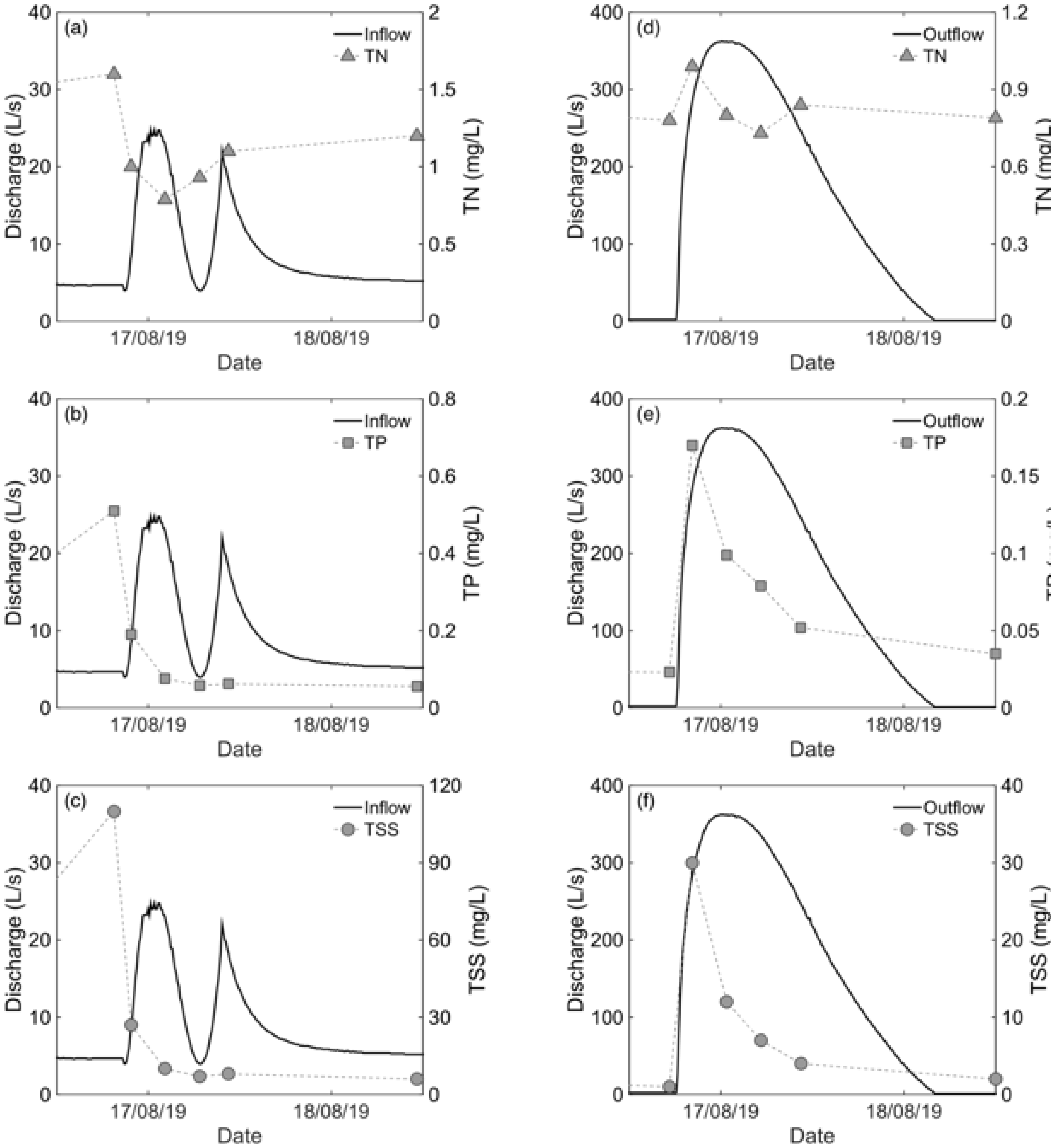

Figure 43 Event 5 concentrations temporal variability: a) inflow TN; b) inflow TP; c) inflow TSS; d) outflow TN; e) outflow TP; f) outflow TSS. Inflow corresponds to PIT1_IN and outflow at OVF_W.

6.4.1 Variability of concentration nutrient species

Table 10 shows the mean concentrations of nutrient species in the inflow and outflow for all events, together with a comparison of our measured mean concentrations for events with the ANZECC 2000 guidelines triggers.

TP mean concentrations were reduced by the wetland in all events but Event 1. TP concentrations at the Outflow were below the TP target level for all events except Event 1. The wetland reduced the mean TP concentration by 54% (on average).

FRP reduction closely followed that for TP. FRP concentration decreased by 42% on average, with FRP making up 24% of TP on average. The largest reduction of FRP was 73% for Event 2, which closely aligned with the 79% reduction in TP.

TN concentrations outflow were below the short-term target (1 mg/L) for all the events except Event 1. The mean TN concentration was reduced by 21% (on average), with the largest reduction (~31%) achieved for the small event (Event 4).

Ammonia was a small fraction of TN (~ 12%) in the inflow, and an average reduction of 39% at the outflow was achieved for all events except Event 1. The largest reduction in NH3 concentration (63%) occurred in the last two events in August 2019, at times when the inflow concentration increased.

Table 10 Arithmetic mean concentrations for monitored events. ANZECC guidelines (for estuarine protection) are given for nutrients. Numbers in red represent concentration values exceeding the guidelines.

Event		TP [mg/L]	TN [mg/L]	FRP [mg/L]	NOx-N [mg/L]	NH_3-N [mg/L]	DON [mg/L]	TSS [mg/L]
ANZECC		0.065	1.2	0.005	0.045	0.04*	n/a	n/a
Event 1	Inflow	0.420	1.750	0.022	0.290	0.096	0.424	88
	Outflow	0.483	2.183	0.017	0.273	0.150	0.545	130
Event 2	Inflow	0.210	0.945	0.037	0.253	0.095	0.228	36
	Outflow	0.044	0.935	0.010	0.167	0.094	0.407	7
Event 3	Inflow	0.090	0.940	0.026	0.305	0.133	0.318	14
	Outflow	0.064	0.687	0.024	0.178	0.096	0.317	6
Event 4	Inflow	0.080	1.214	0.021	0.414	0.157	0.436	9
	Outflow	0.028	0.841	0.005	0.089	0.050	0.618	4
Event 5	Inflow	0.158	1.103	0.035	0.300	0.185	0.367	28
	Outflow	0.076	0.821	0.025	0.202	0.078	0.452	11

* Corresponds to un-ionised ammonia-N at 20ºC.

Nitrate made up 27% (on average) of TN in the inflow and it reduced to 18% of TN mean concentration for the outflow. The wetland consistently decreased NOx-N mean concentration by 39%.

DON contributed 30% and 55% (on average) of the inflow and outflow TN concentrations respectively. On average the wetland discharged 35% more DON than arrived via the inflow. This result may be linked to the events mobilising DON accumulated in the wetland. If the wetland was not acting a source of DON, the attenuation of TN would be significantly higher.

TSS mean concentrations consistently decreased from the inflow to the outflow, for all events but Event 1. On average, the wetland reduced concentration by 64%, with the largest drop during Event 2 (80%) followed by the 1-year ARI event (61%). High flow conditions experienced by the wetland across the events did not affect its capacity to retain sediments, which highlights the effectiveness of vegetated marsh in trapping sediments. The role of the sedimentation pond in reduction may be small, because most of the inflow volume and sediments entered the wetland directly at the Outflow structure location. Sediments load reduction is estimated below, to quantify the wetland's sediment storage.

6.5 Stormflow conditions: Load attenuation across ESBS

Since inflow via the weir at the Outlet structure location accounted for 71% of the total inflow to the wetland for all events, the overall performance of the wetland is better quantified by using nutrient load data and Equation 3. Inflow load computation assumed concentrations values at the Outflow structure weir are equal to those measured at the main inflow (PIT1_IN). This assumption holds because Event 1 outflow concentrations were similar to those at the inflow when the autosampler collected wetland water at the weir of the Outlet structure.

The wetland achieved average load attenuations of 53% for TP and 19% for TN (Table 11). Low attenuation for TN was also estimated by mean concentrations reduction at 21% (Table 10). The wetland attenuated the load of FRP on average by 53% (except for Event 3). Only Event 4 achieved DON load attenuation at 16%, with the remaining events underperforming (e.g. increasing DON load) by 20% on average. The data indicated high variability for NOx-N load attenuation as the wetland shifted from sink to source upon event characteristics. An early event (Event 2) achieved high load attenuation of up to 43%, but it was followed by increases in NOx-N outflow loads by 20% on average. Only the small Event 4 achieved 90% load attenuation for NOx-N.

Load attenuation for NH3-N was also variable but improved as the wet season progressed. The wetland initially showed low attenuation or acted as a source, but it increased to 52% (on average) for the two August events; a similar degree of attenuation was observed for mean concentrations values as previously presented. It is unclear what mechanism operated late in the season, and how the increase in inflow loading contributed to attenuation improvements. More work and monitoring of events in the spring season are needed.

Average load attenuation for TSS was 61%. Poor TSS attenuation (only 6%) was observed for Event 1 (Table 11) but it increased as the wet season progressed, reaching 88% for Event 2. TSS load was attenuated to 68% for Event 5 (e.g. close to 1-year ARI event). Overall, the wetland was efficient for TSS removal and it likely explains the high TP attenuation achieved for the events.

Table 11 Nutrient and sediments loads and EMC values for each event. EMC values in brackets expressed in mg/L.

Event	Station	TP [kg]	TN [kg]	FRP [kg]	NOx-N [kg]	NH_3-N [kg]	DON [kg]	TSS [kg]
Event 1	Inflow	7.5 (0.38)	32.1(1.6)	0.4(0.023)	5.3(0.3)	1.8(0.1)	8.4(0.4)	1479.4(75)
	Outflow	5.8 (0.33)	31.5 (1.8)	0.3(0.017)	4.8(0.3)	2.2(0.1)	9.7(0.5)	1386.8(77)
Event 2	Inflow	13(0.23)	46.2(0.8)	2.6(0.05)	11.2(0.2)	4.2(0.1)	12.7(0.2)	2031.8(36)
	Outflow	1.5(0.04)	32.2(0.9)	0.4(0.01)	6.4(0.2)	3.5(0.1)	15.0(0.4)	241.2(6)
Event 3	Inflow	4.9(0.11)	32.6(0.7)	1.3(0.031)	7.8(0.2)	4.4(0.1)	13.9(0.3)	818.6(19)
	Outflow	3.9(0.08)	32.3(0.7)	1.6(0.033)	9.3(0.2)	4.7(0.1)	15.6(0.3)	346.6(7)
Event 4	Inflow	0.23(0.1)	2.6(1.2)	0.05(0.05)	0.9(0.3)	0.3(0.2)	0.9(0.4)	29.8 (44)
	Outflow	0.04(0.03)	1.1(0.8)	0.01(0.03)	0.08(0.2)	0.07(0.1	0.8(0.6)	5 (10)
Event 5	Inflow	4.2(0.23)	20.7(1.1)	0.8(0.05)	4.9(0.3)	3.0(0.2)	6.7(0.4)	799(44)
	Outflow	2.1(0.08)	20.6(0.8)	0.7(0.03)	5.9(0.2)	2.1(0.1)	11.3(0.5)	254(10)

6.6 Baseflow conditions: Load attenuation across ESBS

Table 12 presents load attenuation (%) achieved by the ESBS treatment train for each sampling date, together with the estimated hydraulic residence time (HR) and wetland management operations undertaken at the time. These operations included addition/removal of weir logs at the Diversion weir structure in the Bayswater Brook, and the outflow structures (OVF and OUT) acting (Y) or not (N) as main outflow points.

Table 12: Load attenuation (%) across the treatment train. Hydraulic residence time (HR) in days is also provided. Operation refers to inflow rate at PIT1_IN after addition/removal of logs at DIV_W. N for the OVF site indicates weir boards in place (no outflow). N for the OUT site indicates blockage of submerged discharge pipe impeding discharge and (–) indicates lack of flow rate data to compute loads.

Date	TP	TN	FRP	NOx-N	NH_3-N	DON	TSS	HR	Operation		
	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[days]	DIV	OVF	OUT
22/11/17	–	–	–	–	–	–	–	–	42 L/s	N	Y
20/12/17	–	–	–	–	–	–	–	–	47 L/s	N	Y
24/01/18	–	–	–	–	–	–	–	–	38 L/s	N	Y
28/03/18	87	87	93	99	97	68	84	28.3	38 L/s	Y	N
26/04/18	**–277**	62	**–240**	97	93	53	39	1.0	45 L/s	Y	N
23/05/18	25	–18	–23	94	72	**–314**	86	1.2	38 L/s	Y	N
28/06/18	60	40	56	21	77	24	72	4.3	45 L/s	Y	N
27/07/18	52	48	74	48	89	35	78	3.3	55 L/s	Y	N
27/08/18	85	60	99	51	86	42	96	28	11 L/s	Y	N
27/09/18	65	39	86	68	77	8	37	5.2	30 L/s	Y	N
29/10/18	30	12	79	49	92	–26	45	1.2	109 L/s	Y	N
29/11/18	–40	–75	37	59	86	**–166**	–34	1.6	35 L/s	Y	N
20/12/18	81	77	93	98	99	62	83	6.2	63 L/s	Y	N
30/01/19	72	–18	83	96	99	-2	79	4.1	46 L/s	Y	N
28/02/19	58	43	53	91	91	-16	76	3.4	40 L/s	Y	N
29/03/19	–	–	–	–	–	–	–	–	59 L/s	N	Y
29/04/19	–	–	–	–	–	–	–	–	24 L/s	N	Y
28/05/19	–	–	–	–	–	–	–	–	36 L/s	N	Y
26/06/19	65	60	86	71	79	54	77	6.4	29 L/s	Y	N
29/07/19	–50	–40	80	61	77	–41	–41	13.5	5.5 L/s	Y	N
Average	**45***	**27**	**69***	**72**	**87**	**45***	**56**				

* Indicates that outliers were not included in the average load attenuation estimate (load attenuation outliers shown in bold).

The ESBS treatment train achieved 27% load attenuation on average for TN (Table 12). It underperformed on four sampling dates when inflow rates were outside intended operation (e.g. either too high or too low) or after high performance periods (e.g. above 50% attenuation), possibly due to TN accumulation within the system. Winter samples in 2018 showed a higher average attenuation of 47% but dropped to 34% for the summer months. Poor TN attenuation in spring coincided with low attenuation of dissolved organic nitrogen (discussed below).

An average attenuation of 45% for TP loading was achieved. On a seasonal scale, TP load attenuation for the winter 2018 was 66% and slightly higher at 70% for summer 2019. Inflow rates over these two seasons were on average 38 L/s and 50 L/s, with average wetland water level at 0.656 m AHD. As previously discussed, large variability in wetland water levels (caused by changes in inflow and outflow rates) reduced TP load attenuation. Low attenuation is likely associated with short hydraulic residence times, except for the 29 July 2019 sample, which was influenced by an extremely low inflow rate (due to the blocked GPT).

High dissolved inorganic nitrogen load attenuation was observed, with seasonal differences. NH3-N load was consistently attenuated and achieved 87% on average; average attenuation varied across seasons from 82% in winter to above 90% for the summer and autumn months. NOx-N load attenuation along the treatment train was

72%, although there was a clear seasonal difference in average attenuation, ranging from 96% in summer and autumn to just below 50% in winter and spring. Interestingly, there was no direct correlation between attenuation performance and hydraulic residence time. Inorganic nitrogen concentrations were high at OVF_W in winter and were likely associated with additional loading occurring in the proximity to OVF_W during storm events; it also could result from nitrogen-rich groundwater discharging into the wetland.

The treatment train attenuated FRP load by 69% on average; this estimate excluded the result from 26 April 2018. There was no seasonal pattern in FRP attenuation as high performance (above the 90%) was achieved for all seasons. This high attenuation was associated with inflow rates outside design functioning (mostly low rates), resulting in longer hydraulic residence time (e.g., more than 5 days).

The treatment train achieved 56% attenuation of TSS load on average. No seasonality in TSS load attenuation was found, however high values were consistently observed in summer 2018-19 (79% on average). Inflow and outflow rates during this time were close to *as-designed* operational values and resulted in longer hydraulic residence time (e.g., more than 3 days). The close relationship between TSS and TP load attenuation aligned closely with concentration reductions (see Section 6.1.3), highlighting that most TP was particulate.

DON load was poorly attenuated, at 22% on average. This low attenuation likely impacted TN attenuation, because DON made up a large proportion of TN. This finding requires further investigation because it could be related to in-within wetland processes or groundwater discharge to the wetland. Increasing efficiency for DON attenuation should help achieve the TN attenuation target.

7 Metal attenuation across ESBS components

Concentrations of a number of metals were monitored across the sampling period under baseflow conditions. Most were below ANZECC guideline values; however, Al, Co, Cu, Ni and Zn were consistently elevated.

Metal concentrations in the Bayswater Main Drain were first reported by Klemm and Deeley (1991), then by Tan and Oldham (2007), and more recently by URS (2010) in preparation for re-engineering ESBS. GHD (2008) reported on adjacent groundwater quality. A summary of historical results for Al, Co, Cu, Ni and Zn is provided in Table 13, along with mean concentrations measured in the current study.

Clearly, metal concentrations in Baywater Main drain/Bayswater Brook have reduced over the last 30 years, as a result of significant catchment cleanup of industrial waste by City of Bayswater and state regulatory agencies. Groundwater concentrations are generally low except for Al.

Despite the typically order of magnitude decreases in these metal concentrations over the past 30 years, some metals in the inflows to the ESBS, as measured from 2017–2019, still exceed ANZECC guidelines.

The trends in metal concentrations across the wetland components, as averaged data across the sampling period, and showing seasonal variability, are shown in Figures 44–48. In general, there was minimal change in metal concentrations as the drain water flowed from Bayswater Brook, into the GPT and then into the wetland. Significantly decreased metal concentrations were consistently observed across the wetland to the Overflow site. Overall, the metal concentration results highlight the ESBS wetland performed very well, attenuating metal concentrations by 40–60% under baseflow conditions. Typically, concentrations increased from the Overflow to the Outflow site.

Clear seasonal patterns were observed for all four metals, with maximum baseflow concentrations measured in April and May 2018. Minimum concentrations were measured in all metals between October 2018 and March 2019. Much lower seasonal increases were observed in 2019. Interestingly the concentrations of Al, Co and Zn measured in Bayswater Brook were higher than those measured in adjacent groundwater in 2008-09. While the number of groundwater samples was low (and thus the results need to be treated with caution), the lower groundwater concentrations, and the peak concentrations occurring during first flush conditions, suggested elevated metals concentrations may be driven by surface flows, rather than groundwater discharge to the brook.

Table 13: Comparison of historical metal concentrations in Bayswater Main Drain with concentrations measured in the current study. Groundwater concentrations measured in 2008-09 are also shown. ANZECC guideline values are shown in the first column, and concentrations above ANZECC guidelines are highlighted in orange.

		Bayswater MD/Bayswater Brook							Groundwater
	ANZECC (2000) slightly disturbed aquatic systems	May 1990 – January 1991	April 2000 – January 2002	April 2000 – January 2002	April 2000 – January 2002	30 March 2010	November 2017 – September 2019	November 2017 – September 2019	August 2008 and February 2009
Mean values		Baseflow	Baseflow maximum	Storm event	All flows	Baseflow	Baseflow maximum	Baseflow	Maximum
		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Aluminium	0.055	n/a	4.1	0.78	4.2	0.05	1.4	0.7	35
Cobalt	0.0014	n/a	n/a	n/a	0.006	0.006	0.006	0.003	0.002
Copper	0.0014	0.022	0.083	0.063	0.056	<0.001	0.006	0.0025	n/a
Nickel	0.0011	n/a	n/a	n/a	0.015	0.007	0.006	0.004	0.01
Zinc	0.008	1.96	0.649	0.545	0.469	0.043	0.07	0.035	0.015
n		12	6	306	408	3	5	21	6
Reference		Klemm and Deeley (1991)	Tan and Oldham (2007)	Tan and Oldham (2007)	Tan and Oldham (2007)	URS (2010)	Current report	Current report	GHD (2008)

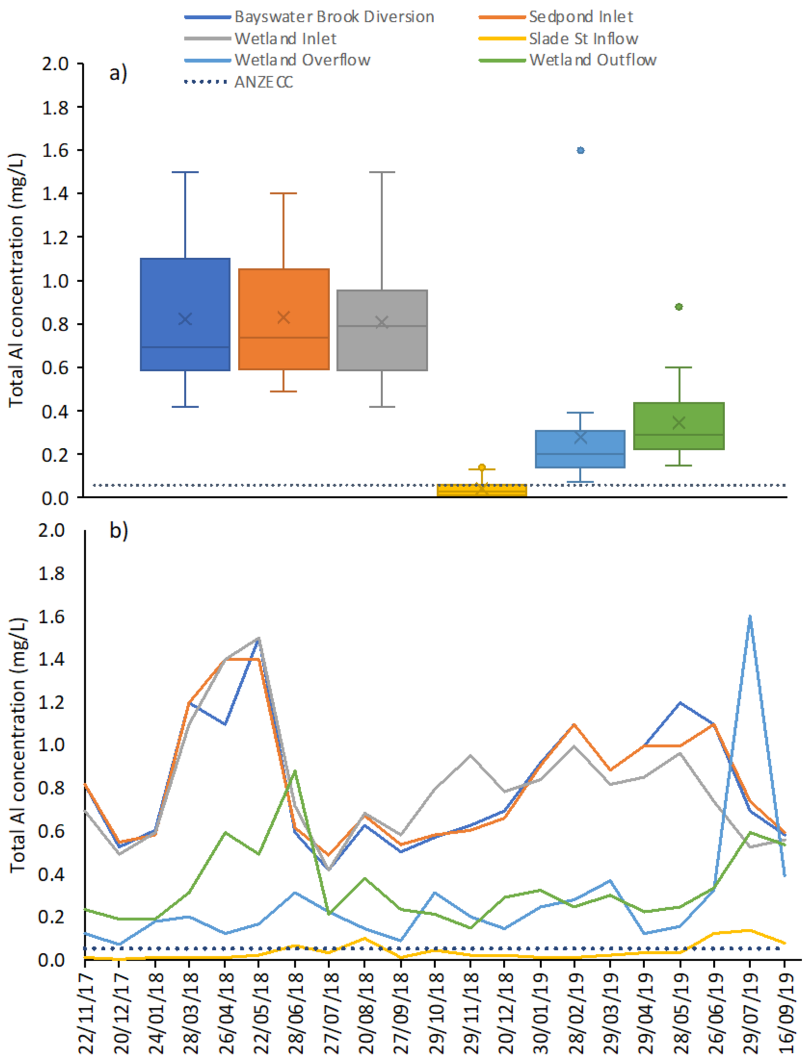

Figure 44 Total aluminium concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

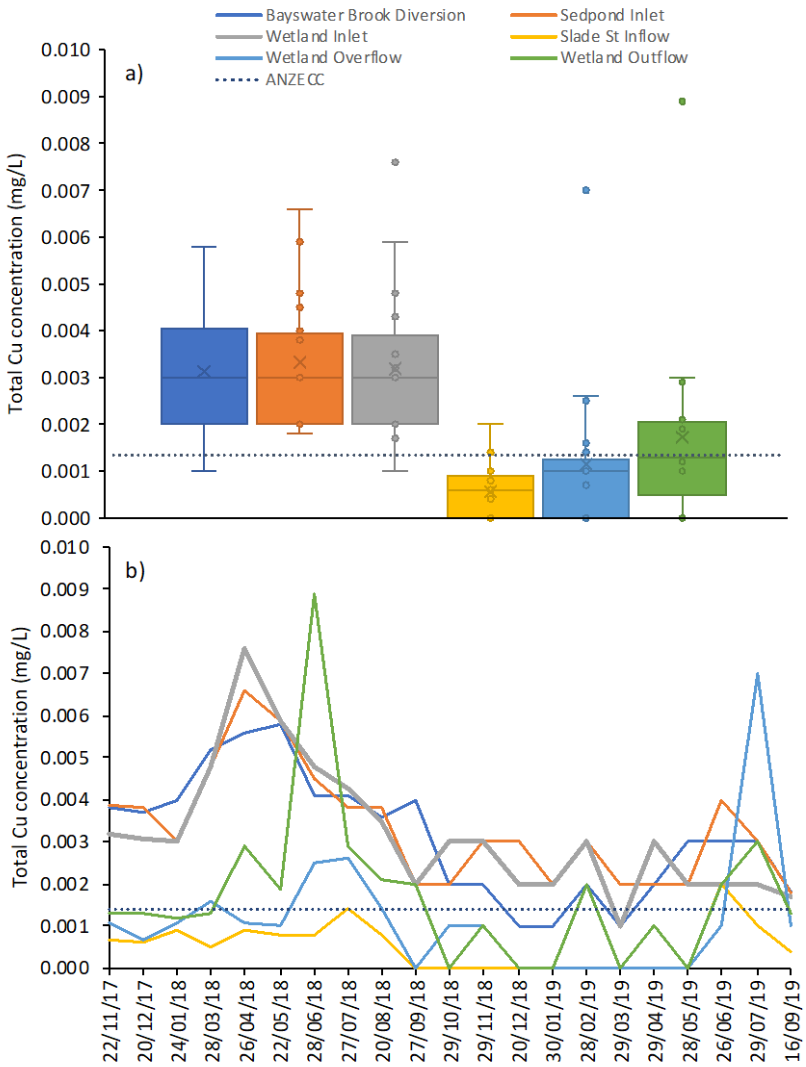

Figure 45 Total cobalt concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

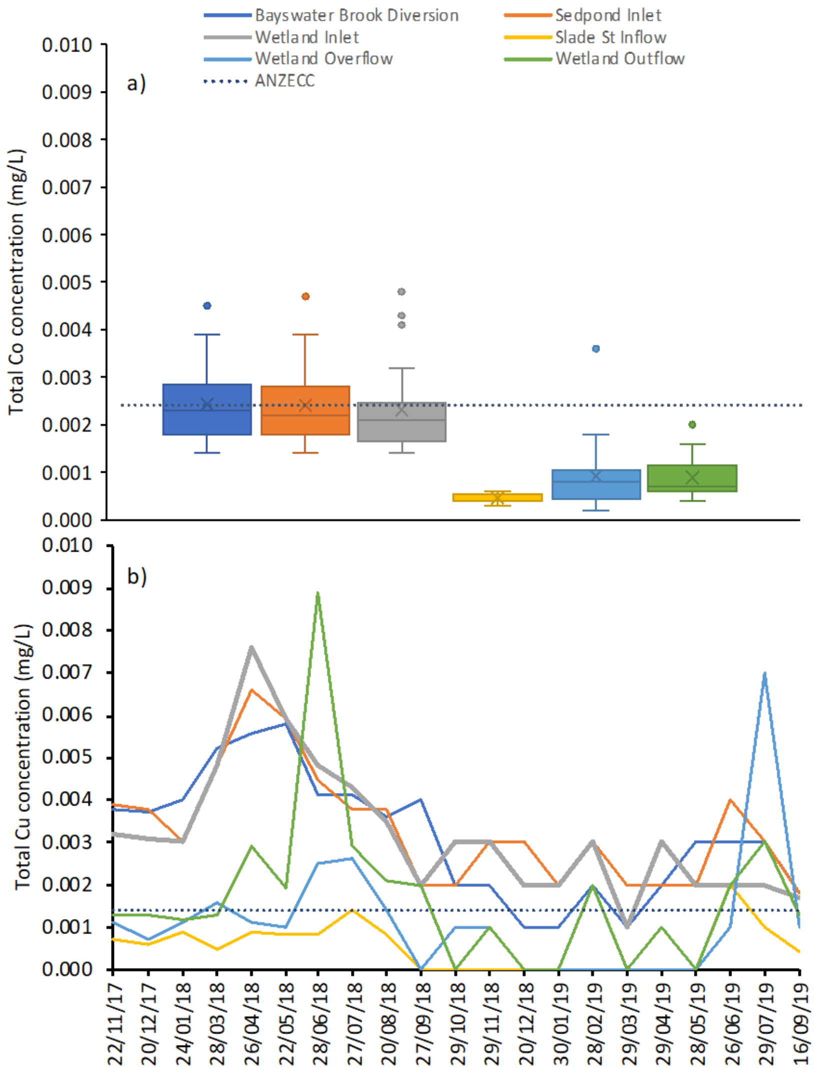

Figure 46 Total copper concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

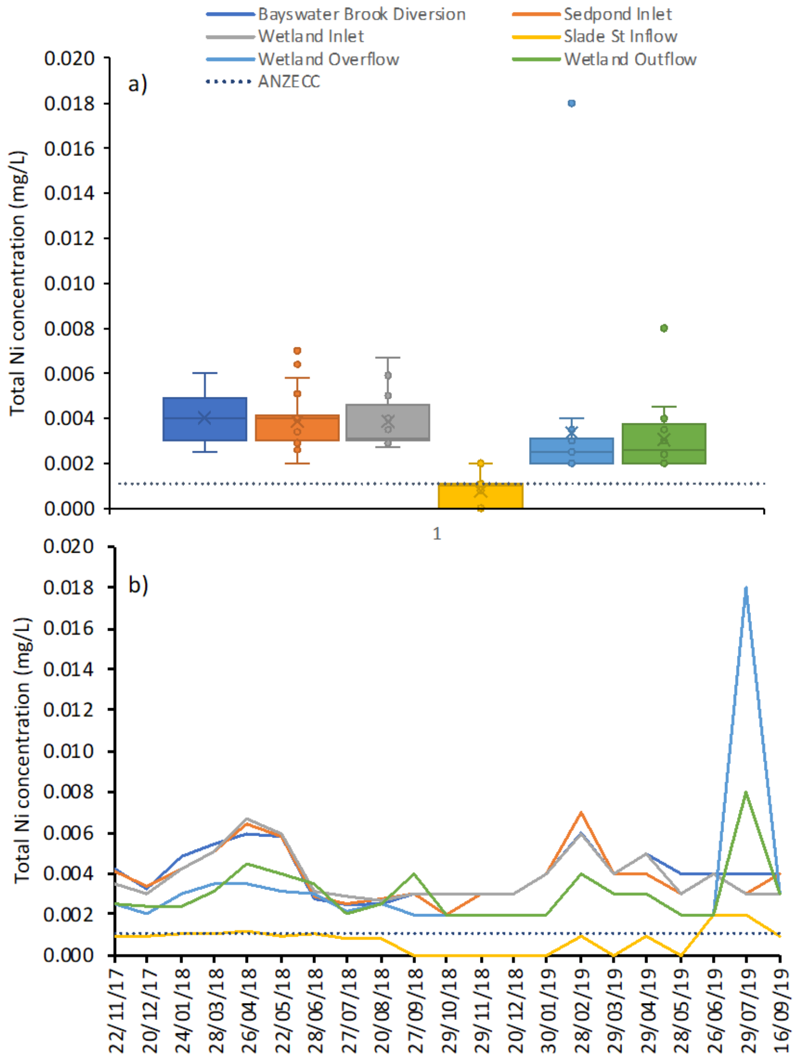

Figure 47 Total nickel concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

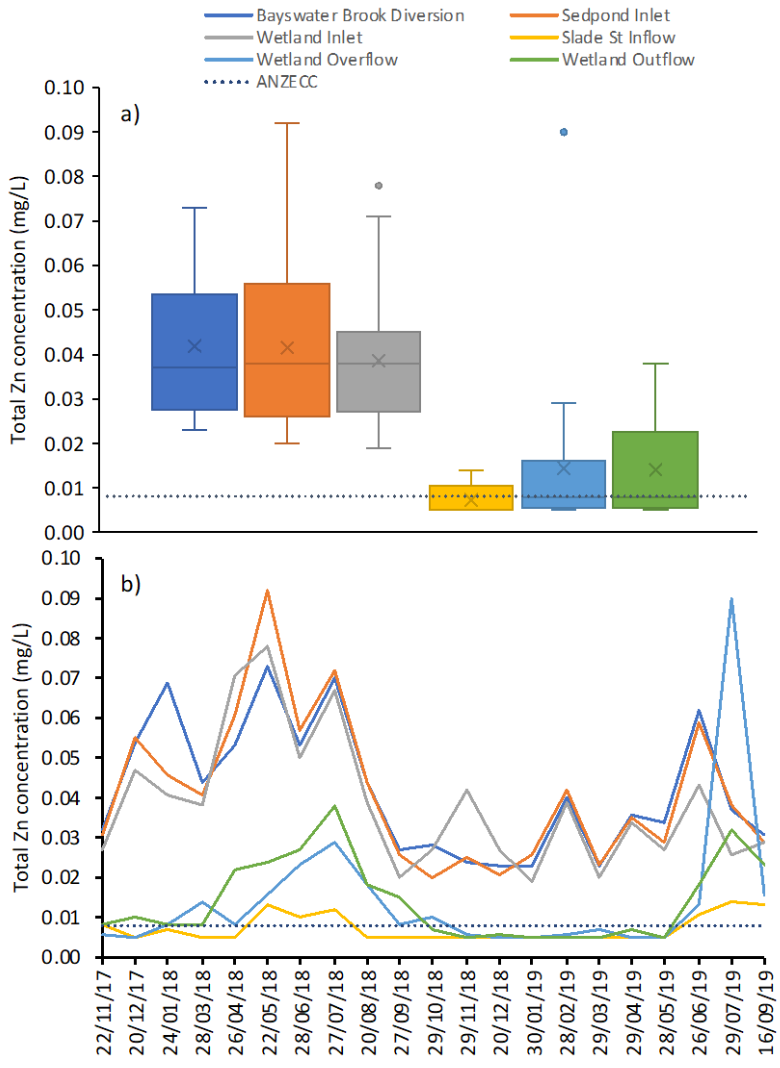

Figure 48 Total zinc concentrations a) at sampling sites along the treatment train, averaged over the sampling period, showing mean, quartiles and outliers; b) on each sampling event. The colours indicate sampling site, and the dashed line indicates the ANZECC guideline.

8 Conclusions and recommendations

8.1 Conclusions

The wetland water pathway for water treatment is not working as intended in the original design during rainfall events. Results suggest that on-third of the length of the treatment pathway is currently used for small and minor events, including the 1-year ARI event. Short-circuiting occurs between the two outflow points.

The major inflow to the wetland during rainfall events is water from the Bayswater Brook entering at the Outflow weir location. This inflow explains between 60% and 80% of the total inflow for the event. It is independent of the rainfall magnitude and is believed to have occurred since late 2017.

The original design considered the possibility of additional inflow during major events and high tide conditions (10-year ARI event). However, we have documented additional inflow for small events (less than 10 mm) and the 1-year ARI event. Under these conditions, it takes up to 48 hours for the wetland to return to normal.

The above issues are not related to problems in the design, but to a substantial reduction of the conveyance capacity of a 400 m reach in the Bayswater Brook, around the Outflow weir location. Overgrown vegetation in the channel increased the water level in the brook by more than 0.3 m; water level reached 0.842 m AHD under wet season baseflow levels.

Inflow to the GPT via the diversion structure highly depends on the GPT conditions. Outflow from the GPT into the sedimentation pond ranged from 5 L/s to 156 L/s under baseflow conditions (when blocked and cleaned respectively). Outflow from the GPT to the sedimentation pond reached 250 L/s during major events.

Water levels in Bayswater Brook, the status of the GPT, stormflows and tidal influence all impacted on the hydraulic performance of the wetland. We recommend, in the first instance, implementing a maintenance strategy for Bayswater Brook, to restore its conveyance and decrease water levels across the wetland. Additional flow measurements should then be conducted to establish ratings for the site when it is operating as expected.

Nutrient and sediment load attenuation across individual treatment train components could not be assessed due to challenges in ratings curves at the inlets and outlets. However, we did analyse concentration reduction across wetland components.

The GPT appears to have been a source of TSS and nutrients for some of the monitoring period; at other times the flows through the GPT into the sedimentation pond were minimal. When the flows out of the GPT and into the sedimentation pond were minimal, and the main inflows into the wetland were via the Outflow, the sedimentation pond could not perform its expected function.

For part of the monitoring period, the area between the Overflow (where inflows were occurring) and the Outflow appeared to be a source of nutrients and TSS. This area appears to be a stagnation area, discharging seasonal fluxes of nutrients and TSS.

Despite the poor performance of the GPT, the outflow weir location forming the main inflow to the wetland for part of the monitoring period, and the Outflow area contributing nutrients and TSS to the discharge, the wetland achieved good attenuation of nutrients and TSS, during baseflow and also during a range of storm events. Except for the first storm event monitored (a first flush event), the TN and TP concentrations at the outflow of the wetland were below ANZECC guideline values (TN 1.2 mg/L; TP 0.065 mg/L).

The wetland system appears to attenuate nutrients and TSS better under the as-intended operational setting (pulsed stormflows), than during (relatively constant) winter baseflow conditions. The estimated improvement during pulsed flows needs further investigation to confirm the assumption made of equal inflows and outflows for as-intended functioning computations. Across the wetland system, the TN loads were on average attenuated during storms events (pulsed flows) around 20%, the TP loads were on average attenuated around 50%, and TSS loads were on average attenuated around 60%. This is better overall performance than predicted by early MUSIC modelling of the system.

Due to the previous practice of discharging effluent from GPT cleaning into the sedimentation pond, it is not possible to determine the ability of the pond to attenuate sediments and nutrients. This need to be re-assessed, now this practice has been discontinued.

Dissolved organic nitrogen made up 30–50% of the TN, and the wetland was a source of this nutrient to Bayswater Brook. This shows that attenuation of inorganic nitrogen was much higher than implied by the TN attenuation.

The vegetated marsh area of the wetland is responsible for significant removal of TSS and nutrients, and likely more than the sedimentation pond under the recent operational challenges.

The Slade Street drain discharges high concentrations of nitrate into the ESBS wetland. Concentrations of all other parameters (nutrients and metals) are very low in the drainage water. The high nitrate concentrations are considered a signature of inflows of contaminated groundwater into the drain. The small Slade Street sub-catchment appears not to receive sufficient surface flows to dilute the groundwater nitrate concentrations. While the same groundwater is expected to flow into the Bayswater Brook, the larger catchment area provides much larger volumes of surface runoff, to dilute nitrate concentrations.

The inflows to the ESBS from Bayswater Brook contain an order of magnitude lower metal concentrations than were measured in the Bayswater Main Drain 30 years ago. However, the inflow concentrations still exceed ANZECC guidelines for some metals. While the GPT and sedimentation pond have little impact on metal concentrations, the wetland reduced metal concentrations by 40–60%. In general, higher metal concentrations were observed at the outflow site compared with the overflow site.

8.2 Recommendations

The findings from this two-year period of monitoring and assessment of the ESBS constructed wetland lead to the following recommendations:

8.2.1 Operational functioning and performance of ESBS components

1. Water levels in Bayswater Brook, the status of the GPT, stormflows and tidal influence all impacted the hydraulic performance of the ESBS wetland component. In the first instance, we recommend implementing a maintenance strategy for Bayswater Brook to restore its conveyance and decrease water levels during baseflow and minor runoff event conditions.

2. The expected attenuation of nutrients and TSS was achieved, even when the wetland was not operating as designed. However, we recommend some relatively inexpensive modifications to operation and maintenance to support additional attenuation; these modifications are outlined below.

3. An inexpensive flow meter (e.g. Starflow) with telemetry capabilities should be installed at the PIT1_IN, to provide real time monitoring of inflows, to inform operational decisions, including clearing Bayswater Brook and cleaning the GPT.

4. Given the key role of the vegetated marsh area in water quality improvement, care should be taken to ensure this area is maintained and supported, i.e. the vegetation is not harvested.

5. Given the good performance of the ESBS despite the poor functioning of the GPT, consideration should be given to the GPT effectiveness, relative its maintenance cost. We recommend exploring the option of installing a simple bypass mechanism inside the GPT pit, to convey water directly from the diversion weir in Bayswater Brook through to the sedimentation pond. If the GPT is kept as a component of the ESBS treatment train, regular monitoring of its condition is critical, along with an associated maintenance schedule. The ability of the GPT to remove gross pollutants should also be assessed.

6. The location and water level control of the outflows should be reviewed. Consideration should be given to simplifying the wetland flow paths and water level control operations. The floodgates are not operating as intended, and the weir boards sometimes exacerbate hydraulic challenges within the wetland. It is possible that simplifying the wetland flow paths will not impact nutrient attenuation performance, as long as the vegetated marsh areas are not impacted. The semi-permanent installation of weir boards at the Overflow structure should be considered, along with removing the floodgates. The ESBS has shown it manages well as an intermittent storm and tidal influenced system; high water induced flooding of the wetland (whether due to tides or storms) subsided within 48 hours, and did not affect the performance of the system. Removing the overflow mechanism and floodgates should establish a simpler and longer flow path for inflowing water, prevent the creation of dead zones within the wetland, and remove the potential for short circuiting.

7. Overall, the system performance will likely benefit from simplified flow paths (bypassing the GPT, leaving the overflow weir boards in place, and removing the tidal flood gates). This approach would also greatly reduce the maintenance costs for ESBS. Any change in the system configuration should be accompanied by ongoing monitoring to assess performance under the new configuration.

8.2.2 Future monitoring to improve assessment of ESBS nutrient attenuation performance

The following actions are recommended to address hydraulic and nutrient attenuation assessment performances for specific components of the treatment train; these relate to quantification of ungauged inflows and outflows, improve representativeness of water sampling sites, and additional monitoring to improve within-wetland process understanding:

1. Collecting large composite water samples (i.e. 4–20 L) should be considered during baseflow conditions, now that wetland dynamics are better understood. Three autosamplers should be used to overcome practical limitations of water sample collection at all sampling points.

2. The reliability of the Overflow rating is low, because it is limited to a few flow scenarios and the central pipe of the culvert structure. The rating curve cannot be applied to the site when weir boards are added/removed to the overflow structure. More investigation is recommended to quantify discharge under different wetland operational scenarios.

3. Additional flow measurements should be conducted to establish ratings for ungauged inflow and outflow sites when the treatment train is operating as intended. These sites are primarily the inflow from the Slade Street drainage pipe and the outflows from the sedimentation pond and the Outflow site.

4. Proper quantification of groundwater discharge into the wetland component should be undertaken. This work should include measuring point-source discharge via the Slade Street drainage pipe and diffuse-source discharge at the eastern boundary of the wetland. Recommendations for further investigation of within-wetland processes are provided in the Appendix 3.

5. Given that the practice of discharging effluent from GPT cleaning into the sedimentation pond has ceased, it is now possible to assess the performance of the sedimentation pond to attenuate sediments

and nutrients. This will require operational maintenance (e.g. dredging) to reinstate the pond to as-intended sediment and water storage capacities.

6. A change in the outlet sampling location of the sedimentation pond is needed to facilitate a more robust assessment of the pond's ability to attenuate concentrations. One possible alternative location is the overflow grate at SED_P, and preliminary sampling should be conducted to confirm the representativeness of this alternative location (see Appendix 3).

7. The installation of a dissolved oxygen logger between the overflow and outlet structures should be considered, to confirm the presence of anoxic waters in the wetland component.

8. Additional monitoring should be considered during the spring season, to determine differences in NH3-N attenuation across the wetland component. Recommended locations for wetland water sampling are provided in Appendix 3.

References

ANZECC & ARMCANZ (Australian New Zealand Environment Conservation Council & Agricultural and Resource Management Council of Australian and New Zealand), (2000). National water quality management strategy: Australian and New Zealand water quality guidelines for fresh and marine water quality. Vol 1, the Guidelines (Chapters 1–7).

BOM (Australian Bureau of Meteorology) (2019). Climate data Online for Perth Airport Station (Station number: 009021): Daily rainfall data. Accessed at http://www.bom.gov.au/climate/data/index.shtml.

Bodhaine G.L. (1968). Measurement of peak discharge at culverts by indirect methods. Techniques of Water Resources Investigations- 03-A3. USGS. DOI10.3133/TWRI03A3.

City of Bayswater (2008) Local Water Quality Improvement Plan – Bayswater Brook.

City of Bayswater & Department of Parks and Wildlife (2016) Sampling and analysis plan: Eric Singleton Bird Sanctuary Water Quality Monitoring.

Chow, VT., Maidment D.R. & Mays L.W. (1988). Applied Hydrology. McGraw-Hill, pp. 572.

DoW (2004). Stormwater Management Manual for Western Australia: Performance, Monitoring and Evaluation. Department of Water, Government of Western Australia, Perth.

GHD (2009). Eric Singleton Bird Sanctuary Groundwater Monitoring. Report prepared for City of Bayswater (61/22351/00/90741). November 2009, pp. 83.

GHD (2013). Eric Singleton Bird Sanctuary Redesign: Technical Specifications. Report prepared for City of Bayswater (61/29071). October 2013, pp. 278.

Klemm V.V. & Deeley D.M. (1991). Water quality in the Bayswater Main Drain: a working paper for the Bayswater Integrated Catchment Management Steering Committee. Swan River Trust, Government of Western Australia, Perth.

McGuinness Y. S. (2017). Pollutant reduction performance assessment of the Eric Singleton constructed wetland and stormwater treatment train. Appendix 1: Sampling and Analysis Plan. Master Thesis Research Proposal, The University of Western Australia.

NMCG (North Metro Conservation Group) (2007). Eric Singleton Bird Sanctuary Wetland Water and Sediment Quality Investigation: August 2007 Snapshot. Pp 42.

Santiago C.A. (2019). Quantification of the performance of the gross pollutant trap at the Eric Singleton Bird Sanctuary. MPE thesis, The University of Western Australia.

SRT (1999). Swan Canning Cleanup Program: An action plan to clean up the Swan Canning Rivers and Estuary. Swan River Trust, Government of Western Australia.

SRT (2008). Healthy Rivers Action Plan: an Action Plan to improve water quality in the Swan Canning river system. Swan River Trust, Government of Western Australia.

SRT (2009) Swan Canning Water Quality Improvement Plan. Swan River Trust, Government of Western Australia.

Tan C.K. & Oldham C.E. (2007). Water quality of the main drains of Bayswater and Chapman St, May 2000 – December 2001. Report to City of Bayswater.

URS (2010). Design and Implementation of an Environmental Water Monitoring Program. Eric Singleton Bird Sanctuary, City of Bayswater. Stage 1: Baseline Sampling. September 2010, pp. 297.

Winer R. (2000). National Pollutant Removal Database for Stormwater Treatment Practices. Second edition. Center for Watershed Protection. Ellicott City, MD.

Appendices

Appendix 1A – Event hydrographs and water balance summary

Table 14 Summary of event characteristics and water balance. Tp refers to time to peak level in wetland, Net Vol. Smax refers to net volume at the time of peak level in the wetland.

	Tp [hr]	**WET_IN Max level** [m AHD]	**Δh** [m]	**BB_OUT Level** [mAHD]	**Net Vol. Smax** [m^3]	**Inflow BB_OUT** [m^3]	**Inflow PIT1_IN** [m^3]	**Rain** [m^3]	**Runoff** [m^3]	**Outflow OVF_W** [m^3]
Event 1	17	1.026	0.374	1.148	1,1505.4	10,340	3533	778.1	2,520	5,666.0
Event 2	7	1.116	0.465	1.224	1,6755.8	18,287	2512	1,366.2	2,520	7,929.7
Event 3	9.5	0.981	0.347	1.147	1,0526.0	11,470	1692.8	813.78	3,348	6,798.5
Event 4	8.5	0.657	0.040	1.014	1,186.7	1,410	167.5	190.0	–	580.8
Event 5	9.7	0.892	0.272	1.166	7,569.0	10,539	330.1	724.7	2,160	6,187.0

Appendix 1B – Nutrient and sediment loads: Baseflow conditions

Table 15 Nutrients and sediments loads corresponding to PIT1_IN and OVF_W sites under baseflow conditions.

Date	Station	TP [kg]	TN [kg]	FRP [kg]	NOx-N [kg]	NH_3-N [kg]	DON [kg]	TSS [kg]
22/11/17	PIT1_IN	0.15	4.38	0.03	1.28	0.48	2.27	21.9
	OVF_W	–	–	–	–	–	–	–
20/12/17	PIT1_IN	0.25	4.53	0.066	1.19	0.66	1.94	45.2
	OVF_W	–	–	–	–	–	–	–
24/01/18	PIT1_IN	0.28	4.94	0.062	1.38	0.99	2.24	49.3
	OVF_W	–	–	–	–	–	–	–
28/03/18	PIT1_IN	0.15	4.28	0.03	1.87	0.85	1.22	26.5
	OVF_W	0.020	0.5	0.002	0.019	0.022	0.390	4.2
26/04/18	PIT1_IN	0.39	7.09	0.01	2.17	1.03	2.32	66.9
	OVF_W	1.47	2.67	0.034	0.07	0.071	1.09	40.6
23/05/18	PIT1_IN	0.2	5.1	0.022	2.36	1.09	1.38	40.1
	OVF_W	0.15	6.01	0.027	0.14	0.304	5.30	5.6
28/06/18	PIT1_IN	0.25	4.37	0.055	1.55	0.79	1.68	27.8
	OVF_W	0.10	2.64	0.024	1.23	0.180	1.23	7.9
27/07/18	PIT1_IN	0.21	5.80	0.096	2.60	1.01	2.17	33.8
	OVF_W	0.1	3.02	0.025	1.35	0.11	1.42	7.3
27/08/18	PIT1_IN	0.047	1.48	0.013	0.46	0.22	0.50	4.9
	OVF_W	0.007	0.59	0.0001	0.23	0.03	0.29	0.20
27/09/18	PIT1_IN	0.093	3.18	0.039	1.11	0.34	1.45	10.6
	OVF_W	0.033	1.94	0.0056	0.36	0.078	1.34	6.6
29/10/18	PIT1_IN	0.47	10.53	0.14	2.88	1.43	5.17	57.2
	OVF_W	0.33	9.30	0.03	1.37	0.120	6.50	31.5
29/11/18	PIT1_IN	0.122	3.35	0.030	1.19	0.27	1.88	9.2
	OVF_W	0.171	5.87	0.019	0.49	0.038	5.00	12.4
20/12/18	PIT1_IN	0.270	6.06	0.070	2.37	0.60	3.08	33.1
	OVF_W	0.050	1.41	0.005	0.04	0.009	1.16	5.6
30/01/19	PIT1_IN	0.25	1.56	0.040	1.56	0.72	1.72	40.1
	OVF_W	0.0710	1.84	0.007	0.06	0.01	1.76	8.5
28/02/18	PIT1_IN	0.240	4.23	0.017	1.52	0.77	1.59	42.2

	OVF_W	0.100	2.39	0.008	0.14	0.07	1.84	10.3
29/03/19	PIT1_IN	0.034	5.68	0.040	2.17	0.98	2.52	46.6
	OVF_W	–	–	–	–	–	–	–
29/04/19	PIT1_IN	0.150	2.51	0.017	0.97	0.56	0.59	31.3
	OVF_W	–	–	–	–	–	–	–
28/05/19	PIT1_IN	0.215	4.11	0.029	1.49	0.70	1.93	38.0
	OVF_W	–	–	–	–	–	–	–
26/06/19	PIT1_IN	0.202	3.30	0.035	1.27	0.63	1.40	33.0
	OVF_W	0.070	1.31	0.005	0.37	0.135	0.65	7.5
29/07/19	PIT1_IN	0.043	0.72	0.010	0.21	0.16	0.24	4.9
	OVF_W	0.064	1.01	0.002	0.08	0.037	0.37	7.0

Appendix 2 – Surveying work for surface water site reference levels.

Two surveying campaigns were conducted on 5 December and 12 December 2018 to link heights (Australian Height Datum, AHD) to water levels at monitoring stations and other flow control structures (e.g. overflow grates, pipes inverts and overflow weirs).

The survey used an automatic compensating level (Leica 560) and AHD benchmark points (georeferenced to Perth Coastal Grid 94) provided by City of Bayswater. Figure 49 shows areal coverage of both surveying campaigns.

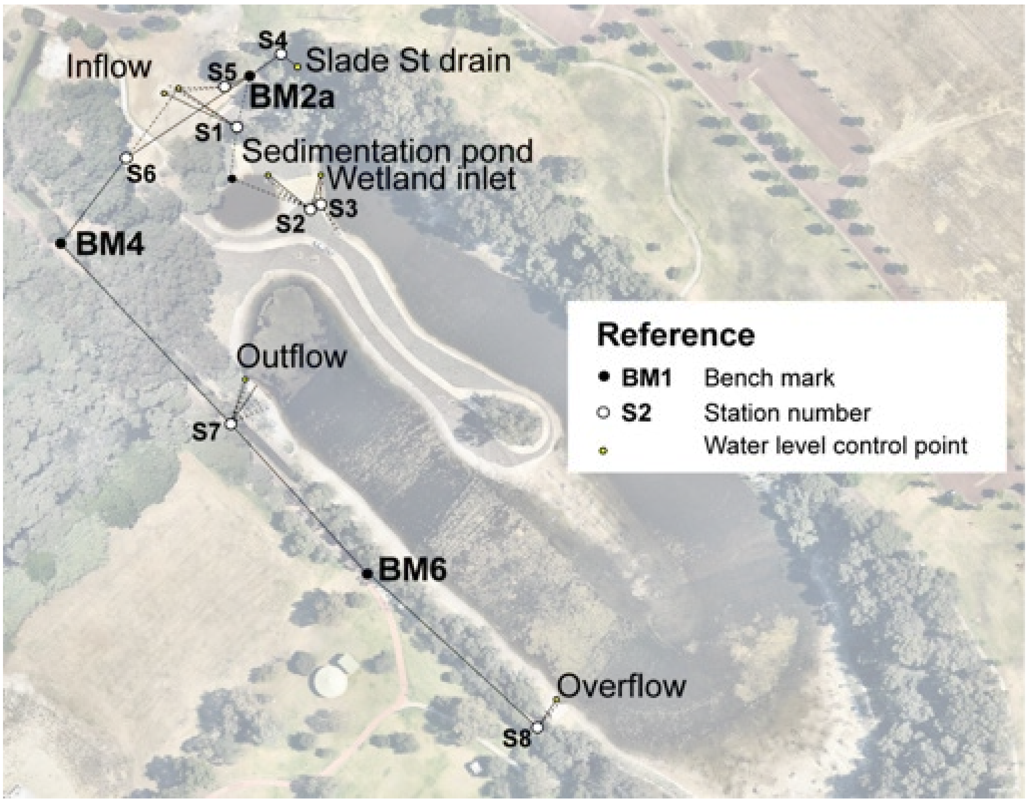

Figure 49 Areal coverage of the two surveying campaigns. Image sourced by Nearmap (22/11/2015).

The results showed height differences between 1 and 8 cm for selected points compared with those reported by GHD (2013) and the constructed technical drawings. The largest difference was found at the Outlet structure (top of concrete wall) which was 8 cm lower than the GHD reported values. Average differences of 4.5 cm were found at the inflow Pits 1 and 2 (denoted D3 and D4 in GHD drawing 61-29071-C010 Rev:3) but minor differences of up to 1 cm were found for all key points at the Overflow weir structure (GHD drawing 61-29071-C013 Rev:1).

A survey of the Outflow weir structure (cross-section) was undertaken to determine the height required by water to overtop the weir (Wetland control endwall as GHD drawing 61-29071-C014 Rev:1). Figure 50 presents the results (inset table) and shows that the left side of the weir (points 5 to 10) maintains a constant value of around 0.971 m but it becomes uneven for the right side with a drop of up to 8 cm in height.

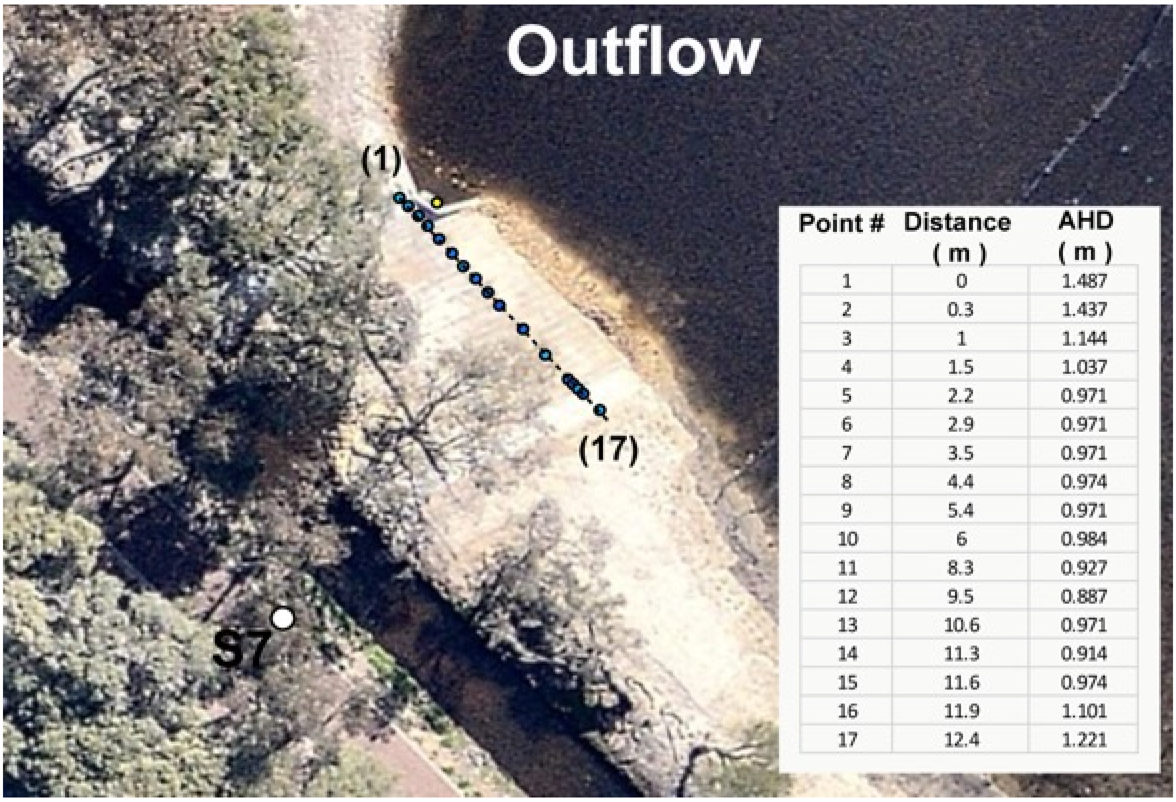

Figure 50 Survey of the outflow weir structure cross-section and corresponding heights (AHD) values. Image sourced by Nearmap (date 7/09/2015).

These results suggest local subsidence is occurring at this location and it could affect the Outflow drainage via the submerged pipe that has been blocked by sand accumulation at the Bayswater Brook side.

Table 16 shows measured height values corresponding to water level monitoring locations within the ESBS wetland. These values were used to establish AHD values for water levels recorded by water depth sensors (relative to probe position) and also for quality control and assurance for manual water level readings (measuring tape with buzzer) obtained during site visits.

Table 16 Height values (AHD in meters) for water level control and referencing points.

Site name	Top lid/grate (AHD m)	Top casing (PVC) (AHD m)
Inflow 1 (Pit 1 to Sedimentation Pond)	3.747	3.452
Inflow 2 (Pit 2 to Bayswater Brook)	3.747	3.412
Sedimentation pond	–	1.562
Wetland inflow	–	1.462
Overflow (left side concrete corner)	1.536	–
Outflow (right corner metal plate)	1.547	–
Slade Street drain	2.677	–
Staff gauge (City of Bayswater) +	0.643	–

+ Staff gauge installed by City of Bayswater in September 2018. Reported value corresponds to AHD height for the gauge zero value.

Appendix 3 – Recommended monitoring activities

Recommended locations for water sample collection, flow measurement (rating), groundwater monitoring and sediment and macrophyte samples are presented in Figure 51, Figure 52 and Figure 53. These activities should be undertaken in two stages over two years to ensure monitoring of a full flow season.

Recommended changes in locations for monitoring and water sampling collection are shown in Figure 51, and include:

- Water level monitoring at DIV_W and full documentation of weir board operations
- Flow meter with telemetry capability at PIT_1
- Flow monitoring (rating) for SED_P site outflows
- Relocation of WET_IN sampling point to overflow grate of the sedimentation pond
- Removal of PIT2_B water level station.

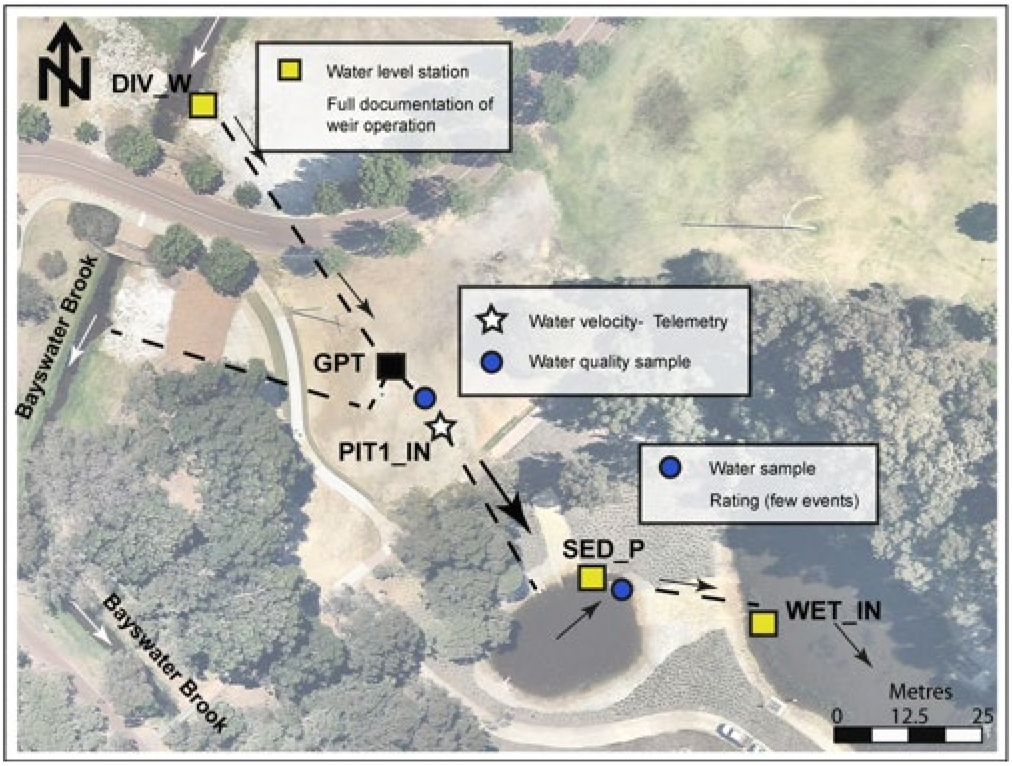

Figure 51 Stage 1 monitoring and sampling locations for the upper treatment train components. Arrows indicate flow direction. Image source: Nearmap.

The monitoring activities recommended for stage one are listed below and should focus on determining the function of the wetland component under *as-intended* conditions. These recommendations assume the wetland flow path has been simplified as recommended (shown in Figure 52).

- Flow monitoring (rating) at the OUT_W site: This includes monitoring water levels at the Bayswater Brook (e.g. BB_OUT site) and water levels at OVF_W with weir boards in place.
- Relocation of wetland water sampling point to 25 m downstream from WET_IN site

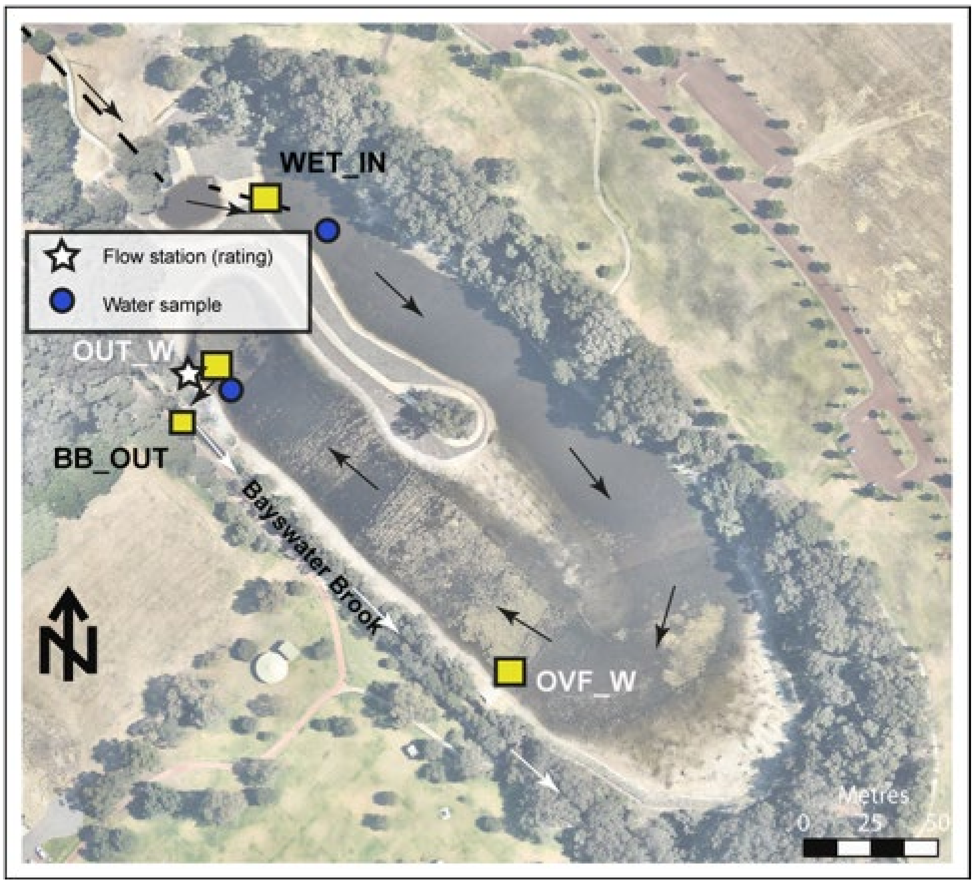

Figure 52 Stage one monitoring and sampling strategy for the wetland component. Arrows indicate flow direction. Image source: Nearmap.

Finally, monitoring activities recommended for stage two are listed below, and should focus on quantifying groundwater inflows from point and non-point sources (e.g. Slade Street drainage pipe) and in-within wetland processes. The locations around the wetland component expected to account for most of the nutrient and sediment attenuations are shown in Figure 53:

- Flow monitoring (rating) of Slade Street drainage pipe inflow
- Monitoring of groundwater (e.g. water levels, water quality and nutrients)
- Sediment and macrophyte sampling be extended to other locations such as the sedimentation pond and the wetland area between OUT_W and OVF_W sites
- Move water sample point from OVF_W site to a new upstream location apart by 100 m (e.g. outlet of the proposed macrophyte sampling area in Figure 53).

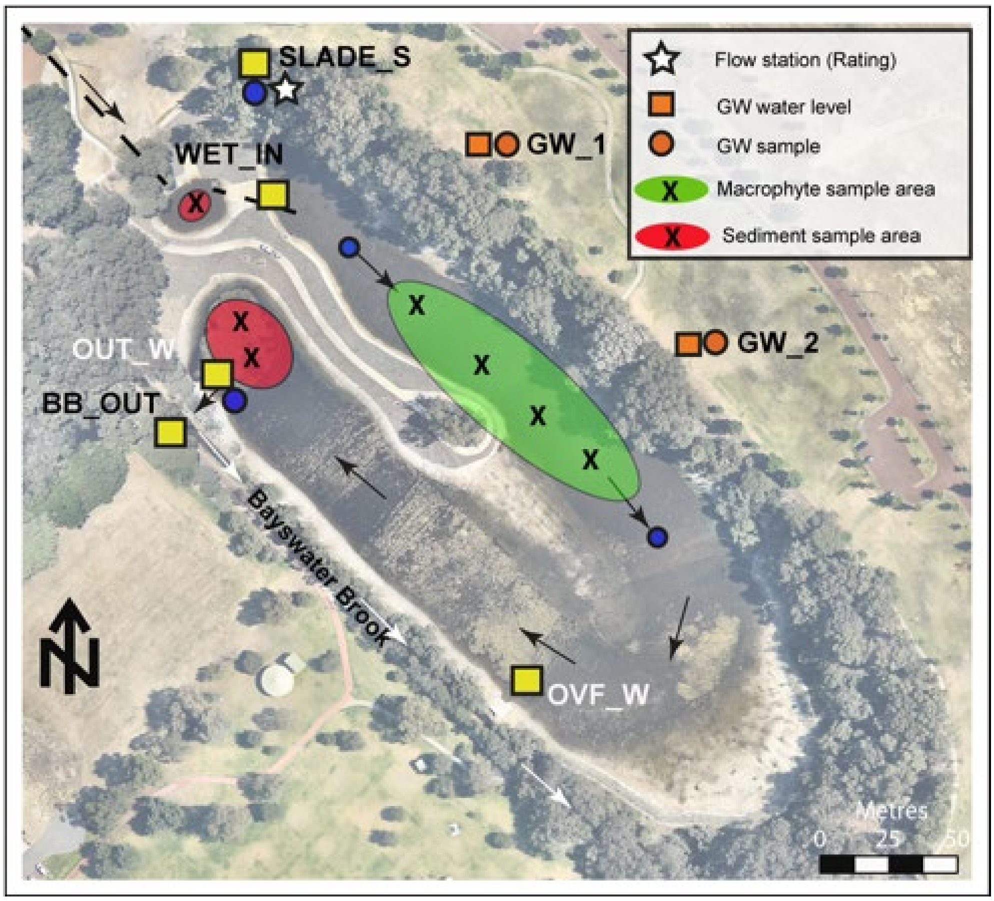

Figure 53 Stage two monitoring and sampling strategy for groundwater interaction and in-within wetland processes. Arrows indicate flow direction. Image source: Nearmap.

CRC for Water Sensitive Cities

Cooperative Research Centre for Water Sensitive Cities

Level 1, 8 Scenic Boulevard
Monash University
Clayton VIC 3800

info@crcwsc.org.au

www.watersensitivecities.org.au