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Urban stormwater inputs to an adapted coastal wetland: role in water treatment and impacts on wetland biota.

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Abbreviations: BOD, biological oxygen demand; CBD, central business district; CVAAS, cold vapour atomic absorption spectrometry; GC-FID, gas chromatography- flame ionisation detection; GC-MS, gas chromatography-mass spectroscopy; ICP-OES, inductively coupled plasma-optical emission spectroscopy; NATA, National Association of Testing Authorities; NO_x, oxidised nitrogen; PAH, polycyclic aromatic hydrocarbon; PCB, poly chlorinated biphenyl; SRP, soluble reactive phosphorus ;TN, total nitrogen; TP, total phosphorus; TPH, total petroleum hydrocarbons.

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Abstract

The Lake Pertobe wetland system is a semi-natural wetland that has been modified primarily for recreational use. However, this lake system receives stormwater from much of the central business district of Warrnambool city (Victoria, Australia), and serves as a buffer zone between the stormwater system and the Merri River and Merri Marine Sanctuary. This work considers the impact of stormwater inputs on Lake Pertobe and the effectiveness of the lake in protecting the associated marine sanctuary. Sediment contaminants (including heavy metals and polycyclic aromatic hydrocarbons (PAHs)) and water quality parameters within the lake, groundwater and stormwater system were measured. Water quality parameters were highly variable between stormwater drains and rain events. Suspended solids rapidly settled along open drains and shortly after entering the lake. Groundwater inputs increased both salinity and dissolved nitrogen in some stormwater drains. Some evidence of bioaccumulation of metals in the food chain was identified and sediment concentrations of several PAHs were very high. The lake acted as a sink for PAHs and some metals and reductions in *Escherichia coli*, biological oxygen demand and total phosphorus were observed, affording some protection to the associated marine sanctuary. Nutrient retention was inadequate overall and it was identified that managing the lake primarily as a recreational facility impacted on the effectiveness of stormwater treatment in the system.

Additional keywords

Heavy metals, sediments, PAH, bioaccumulation, nutrients, constructed wetland.

1. Introduction

Extensive evidence exists to support the use of constructed wetlands for the treatment of contaminated waters (Ghermandi et al., 2007; Vymazal, 2007), however, the use of natural and semi natural wetlands for stormwater processing is widespread but these systems can be poorly characterised. Stormwater treatment is a critical issue, due to the potential for high contaminant loads in urban run-off. A suite of priority pollutants in stormwater have been identified (Eriksson et al., 2007) and these include standard water quality parameters plus a number of metals, polycyclic aromatic hydrocarbons (PAHs), herbicides and other industrially-derived contaminants. The mix of pollutants found in stormwater varies with catchment, land use and the exposure of the water to various surfaces (Barbosa et al., 2012). For example, run-off from roadways can be expected to contain heavy metals from vehicle emissions (Gnecco et al., 2005; Scholz, 2006), and polycyclic aromatic hydrocarbons from tyre wear, exhaust fumes and bitumen surfaces (Aatmeeyata and Sharma, 2010; Ngabe et al., 2000; Takada et al., 1991). Run-off from domestic environments will include pesticides and herbicides, fertilizers and faecal material from pets (Edwards and Withers, 2008) but may include key metals such as zinc due to corrosion of roof surfaces (Gnecco et al., 2005).

The presence of contaminants in stormwater can have negative impacts on the receiving aquatic environments. These impacts can be difficult to assess due to the highly pulsed nature of the input which can result in organisms being exposed to both brief periods of high contaminant concentrations and much lower chronic exposures to accumulated toxins. Brix et al (2010) demonstrated that for some species the lower chronic exposure to Zn from stormwaters had a greater impact than the brief pulses of high concentrations at the beginning of rain events and that in their study environment the risks were rapidly attenuated with distance from the discharge point. Over time, metals that are below the limit of detection in flowing waters can accumulate to measurable

concentrations within wetlands (especially in the sediments) but there may not always be a clear trend between concentration and distance from the inlet (Calijuri et al., 2011). Movement of water through a sequence of wetlands has been shown to result in a gradual reduction in biomarker response associated with the presence of endocrine-disrupting compounds (Norris and Burgin, 2011).

The decrease in concentration of contaminants in stormwaters as they move through wetlands has resulted in their widespread use for stormwater treatment. Wetlands constructed for stormwater treatment make use of large aquatic plants such as *Phragmites* and *Typha* species to assist with the breakdown of wastes and removal of microorganisms and pollutants (Shutes, 2001) and recent work has shown differential uptake of metals into the plant tissues (Klink et al., 2013; Lyubenova et al., 2013). A number of designs exist for the construction of these wetlands and they make use of surface flow, subsurface flow (water is directed through a substrate and the root zone) and floating rafts and systems for urban stormwater often require more complex designs including pre-treatments, due to the variability in both flow rates and water quality (Shutes, 2001). The diversion of water to a treatment facility from a 'first flush' event, where rainfall after a dry period is expected to have considerably higher concentration of contaminants, has also been shown to be a valuable design feature (Kim et al., 2007). The designs for these constructed wetlands, however, reflect that systems receiving urban and highway runoff will accumulate metal and hydrocarbon pollutants and that the sediments will require disposal to landfill after a period of time (Shutes, 2001).

This highlights a conflict between the design of constructed wetlands and the adaptation of natural or semi-natural wetlands- in these latter systems there is often no planning for sediment disposal or wish to disturb the habitats these wetlands provide. Rather than high levels of plant cover and subsurface flow that are often present in constructed wetlands, natural and semi-natural systems will often have larger areas of open water, with varying amounts of fringing vegetation. Budd et al (2011) identified that retention of

suspended sediment was critical for the retention of some pesticides within constructed wetlands, but that degradation rates within the wetland were low enough for accumulation over time (particularly under dry or aerobic conditions) and for the system to potentially act as a reservoir of bioavailable pesticides under conditions of re-suspension or desorption. This would suggest that to maximise protection of downstream assets from pesticides, a wetland would need to have conditions of low flow, little open water (to minimise re-suspension) and sustained periods of sediment anoxia. These are not the conditions commonly found in wetlands that are also maintained for recreational purposes (and anoxia may limit the effectiveness of metal retention in systems receiving diverse inputs, such as stormwaters). There is also the implication that while protecting the downstream assets, there is some degradation of the ecological values within the wetland itself- a point requiring careful consideration where a natural wetland is adapted to receiving stormwater, rather than using a purpose-built treatment wetland.

The wetland system considered in this work is composed of a coastal saltmarsh which has been modified to form a system of lakes within parkland. While managed primarily for recreational and aesthetic purposes, the coastal lake system receives stormwater from an urban centre and acts as a buffer between the drainage system and a nearby estuary and marine sanctuary.

This study aimed to assess:

1. The effectiveness of a recreational wetland system in modifying the contaminant loadings in stormwater runoff from an urban centre before it reaches the nearby marine sanctuary.
2. If the stormwater generates a detrimental effect on the wetlands and associated biota.
3. How groundwater inputs moderate the effectiveness of the lake in treating stormwater.

2. Materials and Methods

2.1. Site Description

Lake Pertobe (38°23'20" S, 142°28'40" E) is located in the coastal city of Warrnambool (population approximately 30,000), Victoria, Australia. The Lake Pertobe recreational precinct includes a three basin, permanent, shallow, coastal wetland system made up of Mill Lake (14.5 ha, 0.5 m depth), Main Lake (12.2 ha, 1.5 m depth) and Children's Lake (3.2 ha, 1 m depth) (Figure 1 and supplementary material). Lake Pertobe is situated between the Warrnambool central business district (CBD) and the Merri River estuary and associated Merri Marine Sanctuary.

Main Lake and Children's Lake are predominantly used for recreation (e.g. boating and canoeing) and are managed for this purpose. Each lake has multiple inlets and outlets and all three are intermittently interconnected. Water from a nearby bore is regularly added to both Main and Children's Lakes to maintain water levels. Children's Lake is infrequently drained into Main Lake, for macrophyte control purposes. During winter, when precipitation is regular, water may also be released from Main Lake into Mill Lake. Mill Lake may be back-filled from the Merri River estuary when the freshwater section of the stratified water column is high enough to flow into Mill Lake (2 – 3 times per year, depending on conditions) and also receives stormwater from the city CBD (estimated catchment area of 180 ha). Mill Lake also receives runoff from nearby sports grounds via drains from Harris Street reserve. The outlet channel from Mill Lake receives several drain inputs from sporting and open recreational areas prior to discharging into the Merri River. During winter and high flow periods, water levels in Mill and Main Lakes may be so high that water overflows across the dividing levees between the lakes. It may also be possible for water to percolate through the levees between the two lakes.

Mill Lake receives stormwater inputs from three major drains: Japan Street drain delivers runoff from residential and commercial districts east of the Warrnambool CBD; Gilles Street drain delivers runoff from the Warrnambool CBD; Fairy Street drain delivers runoff from part of the Warrnambool CBD and residential districts to the west of the CBD, as well as highway runoff from the Princes Highway. The Japan Street drain empties into a channel that also receives water from the Gilles Street drain before emptying into the north-east corner of Mill Lake through a macrophyte bed. The Fairy Street drain empties directly into Mill Lake from the north-west corner across a grassed area. Due to continual inputs of spring water, the receiving channel for the Japan and Gilles Street drains is constantly discharging water into Mill Lake.

Several bores are located in the vicinity of the lake system and water is extracted from a shallow aquifer (2 m) to top up wading pools and both Main Lake and Childrens's Lake. Natural groundwater inputs from the northern and western sides of the lake system are possible due to drainage from elevated groundwater storage. Groundwater on the western side of Mill Lake potentially drains

from the old woollen mill site (contaminated soils) and the Harris Street Reserve (a former landfill site).

Figure 1. Map of the Lake Pertobe recreational wetland system with open water areas marked in grey. Routine water quality monitoring sites marked with pentagons. Sampling sites for sediment and water quality in April 2008 marked with squares, September water sites marked with stars.

2.2. *Sampling*

Sites for water sampling and sediment sampling are marked in Figure 1 and sampling dates are described in Table 1. Figure 2 illustrates the timing of sampling events relative to rainfall over the sampling period. No attempt was made to capture stormwater during night events due to safety considerations and brief duration rain events could not be sampled due to the very short transit time for stormwater to move from the catchment to the wetlands.

2.2.1 *Water Samples*

Water samples were collected by hand or using a sample pole: general water samples in 1 L Nalgene bottles, nutrient samples in 250 mL Nalgene bottles and *E.coli* samples in 60 mL sterile polypropylene screw-cap jars. Samples were stored on ice and transported to the laboratory immediately. Bottles for trace metal and phosphorus samples were washed in hot water and Merck Extran MA03 cleaner, rinsed, and then soaked in approx 4M hydrochloric acid solution for 24 hours and triple rinsed with Millipore water. Samples for metal analysis were frozen immediately upon return to the laboratory.

2.2.2 Sediment Samples

Nine sediment samples were taken: 3 from Main Lake and 6 from Mill Lake (Figure 1). Samples were collected from a boat using a sampling pole with attached corer. The PVC corer was approximately 20 cm long and 7 cm in diameter and was thoroughly rinsed with Millipore water between each sample. Samples were stored in acid-soaked Nalgene bottles and put into an ice cooler for transport and frozen immediately upon return to the laboratory.

2.2.3 In-lake Water Quality

Spot measurements were taken of water quality parameters from in-lake sites on two occasions (Table 1, Figure 1). A Yeo-Kal Model 611 Intelligent Water Quality Analyser was used to record temperature, dissolved oxygen, salinity, conductivity, pH and turbidity. Due to the shallow well-mixed nature of the lakes, only surface readings were recorded.

2.2.4 Biota

Plant and animal samples were taken during May 2008 at the fringe of Mill Lake and returned to the laboratory for sample preparation. At the laboratory, the plants were separated into their constituent parts (leaves, roots, and rhizomes). Each section was then thoroughly washed to remove any trace sediment. Parts were then separately freeze dried prior to analysis. Similar parts were then combined to form a pooled sample. Freshwater shrimp and water boatmen were also collected from the fringe of Mill Lake by sweeping a dip net under the overhanging bank of the lake. These samples were placed in a cooler of water and taken back to the laboratory for sample preparation. Shrimp and water boatmen were removed from the water, frozen, and then freeze dried.

Table 1. Sampling dates for water and sediment collection.

Sample Date	Days (size mm) since previous event *	Rainfall immediately prior (mm)	Size of sampled rainfall event (mm)	Timing of sampling within rainfall event	Samples Taken
20/2/2008	13 (16)	7.2	8.8	mid to late - summer storm event	Stormwater-routine
21/2/2008	1 (8.8)	0	0	24 hours after event	Stormwater-routine plus metals Bore water
10/4/2008	na	0	0	Non-event sampling	Sediments, In-lake water
30/4/2008	7 (5.4)	1.6	8.4	Front end of winter rain event	Stormwater-routine
3/9/2008	na	0	0	Non-event sampling	In-lake water
22/9/2008	2 (4.2)	2.6	8.0	Front end of spring rain	Stormwater-routine

				event	
12/12/2008	7 (11.6)	8.0	64.0	Front- middle of summer rain event	Stormwater- routine
14/3/2009	9 (4.4)	16.6	16.6	Back end of autumn rain event	Stormwater- routine

Previous rain events of at least 4mm. na- not applicable – stormwater not sampled.

2.3. Analysis

Routine water analysis was undertaken by the (NATA accredited) Deakin University Water Quality laboratory. Parameters measured in all stormwater samples included: pH, conductivity, suspended solids, biological oxygen demand (BOD), total phosphorus (TP), soluble phosphorus (SRP), total nitrogen (TN), oxidised nitrogen (NO_x), chloride ions (Cl⁻), potassium ions (K⁺), soluble calcium (Ca²⁺), soluble magnesium (Mg²⁺), total iron (Fe), fluoride (F⁻), sulfate (SO₄²⁻), turbidity and *E. coli*.

Sediment samples were sent to the NATA accredited, Advanced Analytical Australia Laboratories, for the following analyses: Trace metals (aluminium (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), tin (Sn) and zinc (Zn)) by ICP-OES and mercury (Hg) by CVAAS; total petroleum hydrocarbons by GC-FID, PAHs (19), PCBs (11) and organochlorine (21)

and organophosphate (21) pesticides by GC-MS. Stormwater samples were also analysed for the same suite of metals at this laboratory.

Trace metals in biota samples were analysed by ICP-OES at the National Measurement Institute, Pymble, NSW.

Project constraints prevented collection of replicate samples at each sampling site on every sampling date. Variability associated with sample collection and analysis was assessed with duplicate samples on two different sampling dates (in each case from a stormwater outlet and the outlet of Mill Lake). Minor variation was observed between system input and output replicate samples taken sequentially (in some cases 20-30% where values were quite small, but frequently 5-10%). Single spot samples can be considered to be representative of instantaneous values at a given site. See Supplementary materials for raw data.

3. Results and Discussion

3.1. In-lake Water Quality

Main Lake water quality measurements indicated that salinity was relatively consistent across the wetland which was generally well mixed. Oxygen levels at all sites were supersaturated. pH was high (above 8.5) at all sites and was positively correlated with oxygen levels; this suggested that high pH values resulted from extensive photosynthesis by the dense growth of the macrophyte *Ruppia megacarpa* (Paradise, 1998).

Mill Lake water quality measurements indicated that the mean salinity in Mill Lake was approximately 50% of that in Main Lake. Salinity values recorded at different sites indicated that the lake was generally well mixed with some variation between sites.

Oxygen levels at all sites were supersaturated on both sampling dates and the highest oxygen values (above 180% saturation) occurred around the edges of the lake where filamentous green algae growth on the bottom was clearly visible and most dense. Salinity was highest (4.8 ppt) in the middle of the lake. Mean turbidity was higher than for Main Lake and turbidity was highest (90 NTU) at the lake outlet. Wind mixing and re-suspension of fine sediments appeared to be much less pronounced than in Main Lake due to the stabilising effect of the dense growth of *R. megacarpa*. pH was high (above 8.0) at most sites on both sampling occasions and was positively correlated with oxygen level, with highest pH (8.5 – 8.89) values occurring where oxygen supersaturation was highest; this suggested that high pH values resulted from photosynthesis by filamentous algae. Mean pH was slightly lower than in Main Lake. Despite the high potential for wind mixing of the lake, site to site variation in oxygen level and pH

Figure 2. Rainfall totals for the study period. Markers indicate sampling dates. Data Accessed from <http://www.bom.gov.au/climate/data> for station 90186 (Warrnambool airport).

3.2. *E. coli*

The *E. coli* levels in stormwater drains (Table 2) were close to the lower end of the range expected for urban stormwaters under wet conditions (ARMCANZ and ANZECC, 2000) but considerably higher than *E. coli* levels found in the lakes on most occasions. The concentrations found at in-lake sites are generally within a range acceptable for primary contact (swimming) and well within the range acceptable for boating (ANZECC, 2000), although there are a limited number of samples upon which to base this assessment. *E. coli* levels in the stormwater drains exceeded the threshold concentrations (150 faecal coliform organisms/100 mL for primary contact and 1000 faecal coliform organisms/100 mL for secondary contact) on most occasions (noting that *E. coli* is expected to make up 97% of faecal coliform organisms (ANZECC, 2000)) and there is a clear reduction in *E. coli* concentration as stormwater enters the lake system. Up to 25% of indicator bacteria introduced to estuarine systems from stormwater have been found to attach to suspended sediments and settle out of the water column with the level of adsorption influenced by both species and particle size (Jeng et al., 2005). Sedimentation also occurs as stormwater passes over grass and through vegetation beds which may be accelerating the removal of these bacteria as they enter the lake system. However, it has been suggested that the bottom sediments may provide a protective environment for these bacteria and that re-suspension may result in degradation of water quality (Jeng et al., 2005). Further, it can be expected that salinity levels will influence the adsorption of bacteria to particle surfaces through alteration of the surface properties of the sediments. Die-off of bacteria may also be influenced by salinity,

sunlight and temperature (Jeng et al., 2005), so the change in salinity as stormwater enters the lake, and moves across large open water areas, may also contribute to this aspect of stormwater processing.

Table 2. *E. coli* (organisms/100 mL) on various sampling dates.

Site	21/02/2008	30/08/2008	22/09/2008	12/12/2008
Japan A	2900	50*	1100	1500
Japan B	2600	20	290	18
Gillies		160	1300	3600
Gillies Japan		20	1200	2000
Fairy		2500	5800	2000
Mill Lake	8			
Mill Lake Outlet	12	160	<10	120
Main Lake	1		<10	120
Mill Bird		<20	<10	10

* duplicate samples- range 40-60 organisms/100 mL

3.3. Basic Water Quality Parameters and Nutrients

The pH of stormwater sampled across the study period fell in the range 6.6-7.9 with a median value of 7.3 (see supplementary data for more detail). There was a slight increase in pH between Japan A and Japan B sampling sites. The pH of the lake water increased on each sampling date with a median across the study period of 8.3 (range 7.3 to 9.5), reflecting the influence of both the underlying limestone geology and very high rates of photosynthesis associated with extensive submerged macrophytes and shallow water depth.

Changes in Biological Oxygen Demand are shown in Table 3. BOD is variable between drains on any given sampling date, and varies across dates for each drain. ANZECC (2000) does not provide trigger levels for BOD and notes that a number of factors determine dissolved oxygen levels and ecosystem specific studies are warranted. Trigger values are concentrations of key parameters above which further investigation of ecosystem impact or management actions should be triggered. BOD is generally lower in the lake and at the outlet than in the stormwater drains, suggesting that bacterial metabolism of introduced organic matter is occurring within the drain and at introduction to the lakes and is not resulting in problematic elevated oxygen demand within the lake system or in water exported to the estuary.

Change in water quality along the Japan Street drain is more evident in the conductivity data (Table 4). There is a clear increase in conductivity at the Japan B site on each sampling occasion, suggesting the springs which input water to this channel have a higher salinity than the stormwater. This increase is most pronounced during the lower flow periods during summer and autumn. The bore water used to top up Main and Children's lakes on occasion is also much higher in salinity than the stormwater, but it is not known if the bore is connected to the same aquifer as the springs. Groundwater conductivity is at the lower end of the brackish range typical for the coastal limestone/marly limestone aquifer that extends from Peterborough to Port Fairy (Department of Minerals and Energy, 1982). Ions in the bore water are dominated by sodium and chloride. Examination of the analysis of common ions (see

supplementary tables) indicates the raise in conductivity with the springwater inputs is due to contributions from Ca^{2+} , Na^+ , Cl^- , some K^+ , and on occasion Mg^{2+} , Fe (measured as total), and SO_4^{2-} . Mill Lake, which receives stormwater inputs, has consistently lower conductivity than Main Lake, despite these lakes being occasionally interconnected. Possible explanations for this trend include dilution in Mill Lake due to stormwater inputs contrasted with concentration effects in Main Lake due to evaporation and input of bore water to Main Lake increasing salinity.

The concentrations of suspended solids (Table 5) at sites closest to the input drains suggest sedimentation occurs quite rapidly after the stormwater enters the lake system, likely assisted by both a reduction in water velocity and an increase in salinity. There is consistently a substantial reduction in suspended solids in the channel between Japan A and Japan B and generally also between the Gilles St drain and the point where this mixes with the channel carrying water from the Japan St drain. However, where sampling occurred at the beginning of a rain event, it is possible that stormwater inflows had not fully impacted the Japan B and Gilles mix sites. The concentration of suspended solids is highly variable between storm events at all sites and the Lake sites had elevated suspended sediments on some occasions, particularly at the outlet to the Merri River and these results are also generally reflected in the turbidity data (see supplementary material). This variability is expected, given the shallow water depth, fine nature of the sediments, large open water areas and exposed coastal position. However, it does suggest that materials adsorbed on the sediments may not be permanently retained in the wetlands but may be periodically re-mobilised, especially during periods of rainfall accompanied by high winds.

Total phosphorus does not appear to consistently sediment out near the stormwater outlets although on occasion is lower at Japan B than Japan A (see supplementary data). The range of TP concentrations measured in stormwater (0.09-0.99 mg/L) is consistent with that expected for stormwaters from urban areas (ARMCANZ and ANZECC, 2000). TP concentrations in the lake water were

also variable across an order of magnitude (0.04-0.34 mg/L), but while the range is narrower than in the stormwater, it was not always the case that all lake sites had lower TP concentrations than stormwater outlets on a given sample date. However, the outlet to Mill Lake generally presented higher TP than the outlet to Main Lake, while still indicating some retention of phosphorus in the lake. The significance of the TP concentrations in these wetlands is difficult to interpret as there is currently no water quality Trigger Value for this parameter in wetlands in South Eastern Australia. The trigger value for lowland rivers is 0.050 mg/L and for estuaries 0.03 mg/L and our wetland data falls below the estuarine trigger level on all but one sampling occasion.

The effect of the interaction between the stormwater and the lake system on nitrogen concentrations was more interesting and unexpected (Tables 6 and 7). Stormwater nitrogen concentrations increased substantially within the open channel, prior to entering the lake. It was found that the highest concentrations of TN and NO_x were measured in the bore water and in drain sites impacted by input from the groundwater springs. Groundwater was a significant source of nitrogen to the open channel collecting water from both Japan and Gilles St drains. Nitrogen concentrations in the bore water are very high compared to slightly disturbed surface waters (ANZECC, 2000), but lower than reported in other coastal aquifers in Australia (Sewell, 1982; Thorburn et al., 2003), and considerably lower than has been reported in groundwaters from pastoral areas in south east South Australia (Dillon, 1988). The stormwater TN concentrations were towards the lower end of the range expected from urban catchments (ARMCANZ and ANZECC, 2000). Nitrate concentrations in stormwater were generally higher when sampled at the front of a rain event, consistent with the highly soluble nature of nitrates (these would be expected to be mobilised with the first flush). The lowland river trigger level for TN is 0.5 mg/L and the estuarine is 0.3 mg/L and the nitrogen values in all lake sampling sites exceeded these trigger values, however it is noted that this is not dependent on stormwater inputs.

3.4. *Organic Contaminants*

Concentrations of PAHs and total petroleum hydrocarbons in sediments of the Lake Pertobe system are presented in Table 8. Twenty different PAH compounds were present in sediment samples including all 16 priority PAHs. Many PAHs were found at concentrations exceeding the low trigger levels for sediment set by ANZECC (2000) and at some sites the high trigger level was exceeded (darker shaded cells). The Japan Street drain channel and Mill Lake had considerably higher concentrations of PAHs than Main Lake (which does not receive direct stormwater inputs but is at times connected to Mill Lake). The highest concentrations of all PAHs in the Lake Pertobe system (with the exception of fluorene and coronene), were recorded at site 6 in Mill Lake. This site is closest to the inputs from the Japan and Gilles St drains and concentrations exceeded the low ANZECC trigger level for all PAHs, and the high trigger level for 5 PAHs. All PAH concentrations are higher at this site than in the Japan Street drain, and the average concentrations of 8 PAHs were higher in Mill Lake than in the Japan Street drain. Possible explanations for the lower PAH concentrations in the Japan Street drain are removal of the top layer of sediment by high flow in the channel, or the PAHs may be largely associated with fine particles or colloidal oils that don't settle before they are flushed from the channel and are subsequently deposited in Mill Lake.

Table 3 Biological Oxygen Demand (BOD) (mg/L) in stormwater and lake water

Site	20/02/08	21/02/08	30/08/08	22/09/08	12/12/08	14/03/09
Japan A	20	7	4	10	25	7
Japan B	25	5	<2	4	6	15
Gillies	8		10	8 to 9	8	5
Gillies Japan			<2	10	10	15

Fairy	15	80	10	7	7
Mill Lake	9				
Mill Lake					
Outlet	15	9	3 to 4	5	7
Main Lake	4		5	<4	7
Mill Bird		5	<2	<4	5

Table 4. Water conductivity results ($\mu\text{S/cm}$) across the study period

Site	20/02/08	21/02/08	30/08/08	22/09/08	12/12/08	14/03/09
Japan A	140	170	630	560	260	100
Japan B	530	360	860	710	870	390
Gillies	150		150	150	100	95
Gillies Japan	610		820	210	380	380
Fairy	140		160	150	75	130
Mill Lake		1020				
Mill Lake Outlet		1020	660	695	880	1280
Main Lake		3650		2060	2400	4350
Bore 1		2400				
Mill Bird			650	800	890	1300

Lake sediments in the vicinity of the Japan/Gilles Street drain have higher concentrations of PAHs than sediments in the vicinity of the Fairy Street drain. This may reflect a difference in the quality of stormwater from Japan/Gilles Street subcatchments relative to Fairy Street subcatchment and/or the influence of the large area of terrestrial grasses and aquatic emergent vegetation between the Fairy Street drain outlet and the open water of Mill lake which may act as a filtration system prior to stormwater entering the open water of the lake. PAHs were generally higher at site 6 than at site 3; suggesting that the former Woollen Mill site might not be a dominant source of PAHs. Overall, PAH concentrations in the sediments of the Lake Pertobe system were typical of, or at the upper end of, those recorded from a range of estuarine sites including polluted sites reported by Kennish (1997).

There were decreases in all PAH concentrations across the lake from site 6 to site 2 to site 1. This trend corresponds with previous research on sediment concentrations of PAHs in stormwater wetlands (Zheng et al., 2002) which has found a gradual decrease in PAHs away from the source of wastewater in a tidal wetland due to sedimentation effects. A number of the PAHs found in high concentrations at this site have been associated with roadway runoff (Tromp et al., 2012) and there was a high retention efficiency of these compounds in a constructed treatment wetland. In contrast to the work of Neary and Boving (2011), higher concentrations of higher molecular weight PAHs, relative to low molecular weight PAHs, were found in this study. While the previous study found the higher molecular weight PAHs were sorbed to a finer sediment fraction and were more readily flushed from a detention pond system, these compounds appear to have been retained in the fine sediments of Lake Pertobe. The higher retention may be related to the high organic matter content of the sediments in Lake Pertobe and the relative insolubility of larger PAHs (Neary and Boving, 2011). A complicating factor in assessing PAHs in the sediments of the Lake Pertobe system is the potential impact of construction

activities (creation of artificial islands etc.) across the system, disturbing existing sediments and moving them around the various components of the system, and bringing material onto the site from elsewhere.

Table 5. Suspended solids in water from stormwater outlets and lake sites (mg/L)

Site	20/02/08	21/02/08	30/08/08	22/09/08	12/12/08	14/03/09
Japan A	40	7.8	9.1-8.3	11	11	60
Japan B	10	4.5	<2.5	5.2	13	17
Gillies	22		160	25-24	50	50
Gillies Japan	6		<2.5	23	8.9	16
Fairy	35		85	30	8.3	55
Mill Lake		7.1				
Mill Lake		12	170	4.6-5.0	40	200
Outlet						
Main Lake		6.3		13	65	<8.7
Mill Bird			45	9.8	26	25

Table 6. Total Nitrogen concentrations in water from stormwater and lake sites (mg/L)

Site	20/02/08	21/02/08	30/08/08	22/09/08	12/12/08	14/03/09
Japan A	<1.0	<0.8	1.2	0.85	1	0.86

Japan B	2.6	<1.2	8.6	8.3	5	1
Gillies	<1.0		1.5	0.91	1.2	0.48
Gillies Japan	3.4		7.8	1.3	2.6	1.5
Fairy	<1.0		2.2	1.2	0.62	0.65
Mill Lake		1				
Mill Lake		0.8	1.6	0.56	0.95	2.4
Outlet						
Main Lake		1.2		1.4	0.97	1.6
Bore 1		2.4				
Mill Bird			0.95	0.81	0.94	1.6

Total Petroleum Hydrocarbons were generally higher in Mill Lake and the Japan St drain than in Main Lake. The highest concentrations were found in the vicinity of the old Woolen Mill and the Japan St drain. TPH were not well correlated with PAHs across the Lake Pertobe system.

A suite of 21 organochlorine pesticides were analysed in all sediment samples but were below the detection limit of 10 $\mu\text{g/kg}$.

Twenty one organophosphorus pesticides were all below the detection limit of 200 $\mu\text{g/kg}$. Ten PCB congeners and total PCB congeners were all below the detection limit of 5 $\mu\text{g/kg}$. The ri
of detection for these parameters.

3.5. Trace Metals

Analysis of stormwater inputs (sampled on 21/02/08) for heavy metals gave an instantaneous snapshot of metal concentrations for a summer flow event. Results are presented in Table 9 alongside 80% and 99% trigger values for slightly – moderately disturbed freshwater lake ecosystems (ANZECC, 2000). Eighty percent trigger values afford protection for 80% of the organisms present while 99% trigger values afford protection for 99% of organisms present.

Table 7 NO_x concentrations (mg/L) in water samples

Site	20/02/08	21/02/08	30/08/08	22/09/08	12/12/08	14/03/09
Japan A	0.17	<0.01	0.29	0.39	0.02	0.2
Japan B	0.66	0.44	7.3	5.4	7	<0.01
Gillies	0.23		0.51	0.34	0.47	0.14
Gillies Japan	0.72		6.8	0.45	2.1	0.43
Fairy	0.23		0.44	0.33	0.23	0.2
Mill Lake		<0.01				
Mill Lake Outlet		<0.01	0.04	0.01	0.17	0.03
Main Lake		0.02		0.02	0.12	0.02
Bore 1		3.1				

Mill Bird	0.01	<0.01	0.18	0.02
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With the exception of mercury and tin, concentrations of metals were highest in stormwater from the Fairy Street drain; stormwater in the Gilles Street drain tended to have the lowest metal concentrations. This reinforces the apparent differences in water quality of stormwater between the different subcatchments draining into the Lake Pertobe system. The Fairy Street commercial subcatchment had poorer water quality across a range of water quality parameters.

With the exception of arsenic, mercury and tin, metal concentrations decreased from Japan A to Japan B suggesting losses to the sediments via settling of suspended materials from stormwater. This is supported by the high concentrations of aluminium, arsenic, cadmium, chromium, nickel and lead in Japan Street drain sediments relative to other sites in the systems (see Table 10). In this regard the Japan Street drain serves as a preliminary settlement chamber within the overall Lake Pertobe system. Note however, that the increase in arsenic may be further evidence for some input of groundwater: other sites in the district demonstrate an increase in arsenic concentration associated with groundwater intrusions (Howitt et al., 2007).

The highest metal concentrations in water samples relative to 80% trigger values were observed for aluminium, copper and zinc. High aluminium concentrations may result from the effect of the digestion process on clay minerals (Sutherland, 2002) and may not represent a bioavailable fraction of aluminium. In all drains, copper concentrations were low and lead concentrations were very low compared to expected stormwater concentrations for both residential and commercial catchments reported elsewhere (Burton and Pitt, 2002; Gnecco et al., 2005; Thurston, 1999). Zinc levels were high, especially in Gilles and Fairy Street drains, compared to

stormwater levels for commercial catchments reported elsewhere (Burton and Pitt, 2002). High concentrations of zinc have been associated with runoff from roof surfaces (Gnecco et al., 2005).

Aluminium, copper, lead and zinc levels in Mill Lake exceeded the 80 percent trigger level for slightly to moderately disturbed freshwater lake ecosystems. With the exception of cadmium, mercury and tin, all metals were found at lower concentrations at Main Lake outlet compared to Mill lake outlet. With the exception of arsenic and nickel, metal levels were lower at the outlet of Mill Lake than in any of the stormwater drain input sites suggesting retention of metals in Mill Lake. Only aluminium and copper exceeded the 80% trigger level at the outlet of Mill Lake suggesting that generally Mill Lake provides benefit in reducing the loads of metals to the Merri River discharge site. Aluminium exceeded the trigger level at all drain and Mill Lake sites. However, as our analysis involved a complete digestion of unfiltered samples, we do not have data at this time regarding the proportion of the aluminium that is in a biologically available form and the proportion that is tightly bound in the clay structure.

Metal concentrations in water only represent an instantaneous 'snapshot' of actual contaminant inputs. The quality of stormwater runoff is highly variable. The timing of sampling and other factors such as the nature of the precipitation event will influence the constituents and their concentrations in stormwater runoff (Lee et al., 2007). Sediment metal concentrations provide a more integrated measure of contaminant input over time (Table 10). Nickel exceeded the lower trigger value set for sediments at all sites on Mill Lake. Mercury exceeded the trigger level at all sites except Mill 5. Arsenic also exceeded the lower trigger level in the Japan Street drain sample. Other metals such as copper, chromium and tin had relatively high concentrations in relation to the trigger levels but did not exceed them. All metals in both lakes, with the exception of mercury, tin and zinc in Main Lake, and tin in Mill Lake, were at levels exceeding the probable effect level (PEL) as applied in the United States (Burton and Pitt, 2002), but no metals exceeded the severe effects level (SEL) at any site in either lake.

Table 8. Concentrations of EPA Priority 16 PAHs ($\mu\text{g/kg}$) and Total Petroleum Hydrocarbons (mg/kg) in sediment samples taken from Main Lake, Mill Lake and the Japan St drain within the Lake Pertobe system on 10/04/08. For comparison, low and high trigger values for sediments are provided (ANZECC, 2000) and samples exceeding these trigger levels are indicated by light and dark shading, respectively.

PAH	Main 1	Main 2	Main 3	Mill 1	Mill 2	Mill 3	Mill 4	Mill 5	Mill 6	Japan	ANZECC Low	ANZECC High
Naphthalene	55	84	44	210	170	240	130	98	250	100	160	2100
Acenaphthylene	97.5	160	67	290	230	480	310	370	620	250	44	640
Acenaphthene	5.5	11	5	12	10	18	9	16	21	11	16	500
Fluorene	18.5	40	18	71	67	100	43	110	78	74	19	540
Phenanthrene	200	410	160	910	700	1300	710	1500	1800	1600	240	1500
Anthracene	91.5	160	66	300	260	530	310	610	690	570	85	1100
Fluoranthene	750	1460	590	2440	2140	3090	1850	3690	3900	2810	600	5100
Pyrene	740	1450	470	2390	1800	3130	1790	3210	3830	2850	665	2600
Benzo(a)anthracene	245	480	170	940	750	1300	920	1700	1800	1500	261	1600
Chrysene	280	540	200	1300	990	1600	1000	1300	1800	1600	384	2800
Benzo(b)&(k)fluoranthene	630	1100	420	2400	1800	3100	2300	2200	3400	2400	N/A	N/A
Benzo(a)pyrene	330	600	250	1000	710	1400	1200	1400	1800	1700	430	1600
Indeno(1,2,3-cd)pyrene	280	420	230	790	490	930	850	650	1400	730	N/A	N/A

Dibenzo(ah)anthracene	72.5	110	63	220	120	280	200	160	340	210	63	260
Benzo(ghi)perylene	335	530	270	770	490	970	840	780	1400	840	N/A	N/A
TPH C10-14	<10	<10	12	12	<10	<10	<10	<10	10	11	N/A	N/A
TPH C15-28	130	140	170	330	190	290	210	66	290	320	N/A	N/A
TPH C29-36	180	200	250	430	260	380	280	51	350	330	N/A	N/A

The Japan Street drain channel contained higher concentrations of some heavy metals. Arsenic was almost 5 times higher concentration than at most other sites. As discussed, it is thought As is entering via the groundwater and then adsorbing onto the sediments. However, with only a snapshot of storm water quality, it can't be ruled out that the stormwater has been the source of As over time and has accumulated in the sediments to such an extent that they can now act as a local source of contamination when relatively clean water arrives in the system. The result is high in comparison to other sites in the region (Howitt et al 2007). Aluminium, chromium and nickel were all much higher in Japan Street drain than other sites. As the stormwater flows directly into the channel from the drain it is the first stage of removal for material in the runoff. The rate of flow decreases in the channel allowing settling and other removal mechanisms to begin. Copper, mercury, tin and zinc were lower in the Japan Street drain channel compared to the lake samples, possibly indicating association with finer sediments or organic matter which settles later or is more readily re-mobilised during periods of high flow. During sampling it was observed that the bottom of the channel was very solid and had no soft top layer of sediment while the lake samples contained a higher proportion of fine material at the surface. Alternatively, it possible that the Japan St drain is not the dominant source of copper, mercury tin and zinc into the lake.

Table 9. Concentrations of heavy metals detected in stormwater and lake water samples taken 21/02/08. Also included are trigger values set for slightly to moderately disturbed freshwater habits by ANZECC (2000).

Heavy Metal (mg/L)	Mill Outlet	Mill Bird	Main Outlet	Japan A	Japan B	Gilles	Fairy	ANZECC 99% protection	ANZECC 80% protection
Aluminium	0.92	0.67	<0.03	1.75	0.23	1.05	2.5	0.027	0.15
Arsenic	0.006	0.006	0.003	0.001	0.002	0.001	0.0015	0.001	0.36
Cadmium	<0.0001	<0.0001	<0.0001	0.0001	<0.0001	0.0001	0.00015	0.00006	0.0008
Chromium	0.003	0.001	<0.001	0.005	<0.001	0.0045	0.006	0.00001	0.04
Copper	0.003	0.003	0.001	0.018	0.004	0.0155	0.021	0.001	0.0025
Mercury	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.00006	0.0054
Nickel	0.006	0.006	<0.001	0.0045	0.001	0.002	0.0065	0.008	0.017
Lead	0.002	0.002	<0.001	0.0096	<0.001	0.0085	0.0115	0.001	0.0094
Tin	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02		
Zinc	0.013	0.008	0.007	0.255	0.07	0.385	0.435	0.0024	0.031

Mill and Main lakes exhibited similar levels of aluminium, arsenic, cadmium, and chromium in the sediments, whereas Mill Lake had higher levels of copper, mercury, nickel, lead, tin and zinc in the sediments. There was little evidence of trends in site to site differences within lakes, and little evidence of decreasing sediment metals levels towards the outlet of Mill Lake.

Despite higher levels of almost all metals (except mercury and tin) in Fairy Street drain stormwater (based on 21/02/08 sample), sediments in the vicinity of the Japan/Gilles Street drain input contained higher concentrations of all metals (except nickel and tin) compared to sediments in vicinity of Fairy Street drain input. This might suggest a historical change in the quality of stormwater from Japan/Gilles Street subcatchments relative to the Fairy Street subcatchment. Historically, the Fairy Street subcatchment would have included more motor vehicle garages, a brass foundry and other light industries. The major difference between the Fairy Street and Japan/Gilles Street drain discharge sites in Mill Lake is the large area of terrestrial grasses and emergent aquatic vegetation between the Fairy Street drain outlet and the open water of Mill Lake. Stormwater must travel through this vegetation before reaching the open water of Mill Lake. This effectively constitutes a de facto surface flow reed bed system such as is used for wastewater renovation and may be responsible for some of the differences observed in sediment metal levels in Mill Lake. Higher metal levels at sites 2, 3 and 4 in Mill Lake relative to those at site 5 (closest to the Fairy Street drain), may reflect the influence of Japan Street flows; Fairy Street drain flows could be isolated in the north western corner of the lake given the vegetative cover in this area. If the presence of vegetation is contributing to the reduction of contaminants in stormwater inputs then management of grassed areas/aquatic vegetation is critical to operation of the lake as a stormwater treatment system. It was also

noted that a number of metals were present at higher concentrations at site 3 (closest to the Woollen Mill site) relative to site 6 (closest to the Gille/Japan Street flows), possibly reflecting contamination from the former Woollen Mill site on the western side of the lake.

Table 10. Concentrations of heavy metals in sediment samples taken from Lake Pertobe on 10/04/08. Also included are the Japan Street drain sediment sample concentrations and the low and high trigger values for sediments set by ANZECC.

Heavy Metal (mg/kg)	Main 1	Main 2	Main 3	Mill 1	Mill 2	Mill 3	Mill 4	Mill 5	Mill 6	Japan	ANZECC Low	ANZECC High
Aluminium	11,000	17,000	19,000,	17,000	12,000	17,000	17,000	2,100	15,000	23,000		
Arsenic	6	8.3	9	10	8	10	7.1	6.1	11	54	20	70
Cadmium	0.57	0.8	0.78	1	0.7	1	0.87	0.43	0.95	0.94	1.5	10
Chromium	31	46	50	47	31	46	44	15	38	62	80	370
Copper	18	26	21	50	30	48	34	10	49	15	65	270
Mercury	0.07	0.11	0.09	0.22	0.17	0.26	0.16	0.1	0.23	0.03	0.15	1

Nickel	20	29	29	38	29	40	33	44	40	54	21	52
Lead	39	44	34	110	63	110	64	30	9.7	98	50	220
Tin	1.1	1.6	1.4	2.9	3.1	3.7	2	3	3.5	<1.0	5	70
Zinc	83	110	84	300	160	290	200	84	270	67	200	410

Overall, metals in the sediments in the Lake Pertobe system were either typical of or at the lower end of those recorded from a range of estuarine sites reported by Luoma et al. (2008) and Kennish (1997). No metals in sediments of the Lake Pertobe system were at the high end of the range for polluted estuarine environments.

A range of biota were sampled and analysed for trace metals to assess the potential impact of stormwater inputs on the lake ecology. Concentrations of trace metals in biota from Mill Lake are summarised in Table 11. Primary producers were represented by *Typha domingensis*, *Phragmites australis* and algae (not identified- small sample used for limited analysis only). With the exception of mercury, the highest metal concentrations occurred in roots of macrophytes. Concentrations of most metals were considerably higher in roots than in rhizomes, which in turn were considerably higher than in stem or leaves. This accumulative concentration of metals in underground tissues, especially roots, is consistent with studies on similar species (Klink et al., 2013; Lyubenova et al., 2013). Metals concentrations were similar in both species of macrophyte. The relative levels of metals in macrophytes were: aluminium >> zinc > lead, copper > arsenic, nickel, chromium > cadmium, tin > mercury. Metal levels in the detritivore shrimp (*Paratya australiensis*), indicated relative concentrations of: aluminium, copper > zinc >> arsenic > nickel, lead > chromium > mercury, tin > cadmium. This pattern differed slightly from the carnivorous Water boatmen (Family *Corixidae*) relative metal concentrations of: aluminium > zinc >> copper > lead > arsenic, nickel > chromium, mercury > cadmium, tin. These results clearly indicate that some fraction of the elevated aluminium concentrations found in the lake system is bioavailable and assimilated in the bodies of the organisms. Mercury concentrations in biota from Mill Lake appeared to be similar to those recorded for macrophytic plants (0.009 mg/kg) and some of the herbivorous zoobenthos (0.077 mg/kg), but somewhat higher than that recorded for carnivorous zoobenthos (0.083 mg/kg) in freshwater lakes (Mason, 1991). Metals levels in the aerial portion of emergent plants

from Mill Lake appeared to be at the lower end of the range reported for *Spartina* from estuarine locations (Kennish, 1997). Copper levels in algae were close to average for algae from a range of marine sites but lower than recorded from highly contaminated sites (Kennish, 1997). Generally, copper and zinc levels in the biota from Mill Lake appeared to be considerably lower than recorded elsewhere for benthic seaweed, and detritivorous and carnivorous crustaceans from contaminated estuarine sites (Kennish, 1997; Lockwood, 1976; Luoma et al., 2008). However, copper in the detritivore was similar to that recorded in amphipods from contaminated estuarine sites (Luoma et al., 2008).

Aluminium and zinc were high in both species of secondary consumer. Aluminium, arsenic, chromium, copper, nickel and tin were all higher in the detritivore than in the carnivore; whereas cadmium, mercury, lead were all higher in the carnivore. With the exception of copper, mercury and zinc, the highest levels of metals were found in the roots of macrophytes. Highest levels of zinc occurred in the algae. Several metals were higher in the detritivore and the carnivore relative to macrophyte rhizomes suggesting metals specific food chain bioaccumulation and potential for impacts at higher levels of the food web.

3.6. *Stormwater –Wetland Interactions*

Within the limitations of the methodology used in the present study, Mill Lake appeared to effect significant reduction of *E. coli* of raw stormwater. Accumulation of high concentrations of PAHs and some metals in Mill Lake relative to the concentrations in Main Lake indicate that the lake system is acting as a sedimentation basin and facilitating some removal of these contaminants from the water column. Moderate reduction of Total Nitrogen occurred on 3 occasions. Despite this reduction of key water quality parameters, nutrient levels in the outlet from Mill Lake remained high. Mill Lake appeared to significantly increase conductivity,

suspended solids, and turbidity of stormwater. The increase in conductivity was most likely due to evaporation, groundwater input, and the additional input of saline water to the lake system when water from the Merri River estuary is allowed to back-flow into the lakes on high tides. The increase in suspended solids and turbidity appeared to be related to wind mixing and re-suspension of fine sediments in the shallow lake. This process may provide a mechanism for return of some PAH and heavy metals from the sediments to the water column and may occasionally result in the lake sediments acting as a source of contaminants to the Merri River and associated marine sanctuary.

Table 11. Metal burdens in biota from Mill Lake sampled early April, 2008. All concentrations in mg/kg (dry weight).

Sample	Al	As	Cd	Cr	Cu	Pb	Hg	Ni	Sn	Zn
<i>Paratya</i>	97	3	<0.01	0.28	100	0.75	0.08	0.82	0.04	53
<i>Corixidae</i>	88	0.62	0.04	0.23	9.9	0.96	0.14	0.52	<0.02	77
<i>Typha</i> leaf	15	<0.05	<0.01	0.21	2.6	0.07	<0.01	0.46	0.03	10
<i>Typha</i> rhizome	32	0.1	<0.01	0.15	2.8	0.33	<0.01	0.7	<0.02	17
<i>Typha</i> roots	950	4.7	0.05	3	10	24	0.02	6.5	0.23	39
<i>Phragmites</i> stem	9.5	<0.05	<0.01	0.08	0.83	0.13	<0.01	0.25	<0.02	8.7
<i>Phragmites</i> rhizome	30	0.07	<0.01	0.3	3.3	0.23	<0.01	0.73	<0.02	19
<i>Phragmites</i> roots	830	3.6	0.09	2.9	12	13	<0.01	4.3	0.18	53

algae	-	-	-	-	13- 26	-	-	-	-	85 -95
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Intra-lake differences in basic water quality parameters in Mill Lake (Bird Hide versus Mill Outlet) were generally small, with the exception of suspended solids and turbidity which were much higher at the lake outlet on those occasions when wind mixing was pronounced. The fetch of Mill Lake is north-south and the outlet is situated on the southern shore line. When the prevailing wind is from the north or the north-west fine sediments are resuspended from the lake bottom and driven into the outlet.

The water at the outlet of Main Lake had a significantly higher conductivity than Mill Lake reflecting the input from brackish/saline sources. Mill Lake generally had higher conductivity than stormwater entering the system and this may impact on the solubility of heavy metals entering with the stormwater. It is known that exposure to lime (González-Alcaraz et al., 2013) and salt (Tromp et al., 2012) can increase the solubility of some metals in wetlands, potentially meaning that groundwater inputs to this lake system may increase the bioavailability or export of some metals. Suspended solids and turbidity were significantly higher in the water at the outlet of Mill Lake on those occasions when wind mixing resuspended fine sediments. Total Phosphorus concentration tended to be higher in the water at the outlet of Mill Lake but there were no consistent differences in Total Nitrogen or soluble nutrient (Nitrogen or Phosphorus) concentrations between the two lake outlets. There were no consistent differences in *E. coli* levels in the water at the outlet of the two lakes, and on two occasions *E. coli* levels were the same for both lakes. *E. coli* levels were below the trigger values for standards for water used for secondary contact (ANZECC, 2000). It can be concluded that Mill Lake performs reasonably well in terms of treatment of stormwater and that water quality at its outlet during the present study was not significantly poorer than that in Main Lake. Detailed study of heavy metal and PAH concentrations in the water entering and exiting the lake

system is warranted to fully understand the relative importance of adsorption and sedimentation processes in retaining these contaminants, versus the potential for export from the system during periods of higher turbulence as it is documented that retention of pollutants can be quite variable (Revitt et al., 2004). The management of this system for both recreational uses and stormwater treatment may be diminishing the effectiveness of the lake in protecting the Merri Marine Sanctuary from contaminant inputs. Increased growth of emergent macrophytes and reduced open water area would reduce the amount of sediment re-suspension during periods of high wind and would also slow water movement through the system, improving contaminant retention. These changes would improve the functioning of the wetland for contaminant processing but impact on the recreational uses. Topping up the lake system using bore water is increasing the concentration of nitrogen and dissolved salts and may be influencing metal retention and bioavailability. If the recreational uses of adapted wetland systems, are considered higher priority than the stormwater processing functions, then decisions may need to be made regarding stormwater pre-treatment (e.g. settling ponds (Shutes, 2001), or systems to divert the first flush of stormwater (Kim et al., 2007), or periodic sediment removal to reduce the accumulation of contaminants in the lakes. Vegetation harvesting would also assist with nutrient and trace metal removal; however the current practice of lowering the water level to desiccate problematic vegetation will simply be recycling nutrients and other contaminants back into the water column. Diverting the stormwater through larger grassed or macrophyte planted areas with periodic harvesting may also assist with limiting the input of hydrocarbons into the wetlands.

4. Conclusions

The results of this study suggest that the use of this recreational wetland system as a stormwater receival basin is effective in reducing the export of PAHs and some heavy metals to the nearby Merri Marine Sanctuary. Improvements in *E. coli* loads and

some basic water quality parameters were observed although nutrient export from the system is still higher than is desired, although further study is required with a larger sample set. Reduced open water area and increased plant biomass may help to improve contaminant retention; however the management of this wetland for recreational activities works against these objectives.

Groundwater inputs to the lake are complicating the polishing of stormwater, acting as a source of nitrogen, salts and metals. Water quality parameters were highly variable between stormwater drains and rain events. Suspended solids rapidly settle along open drains and shortly after entering the lake.

Stormwater inputs to the lake are likely to be having a detrimental impact on the biota residing in the wetland system, with a number of heavy metals and PAHs exceeding ANZECC (2000) guideline levels and some evidence for bioaccumulation of specific metals within the food chain.

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Supplementary Data,

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Table S1: Assessment of sampling variability.

<u>Parameter(units)</u>	<u>JapA/1</u>	<u>JapA/2</u>	<u>MilO/1</u>	<u>MilO/2</u>	<u>GiID/1</u>	<u>GiID/2</u>	<u>MilO/1</u>	<u>MilO/2</u>
	<u>30.8.08</u>	<u>30.8.08</u>	<u>30.8.08</u>	<u>30.8.08</u>	<u>22.9.08</u>	<u>22.9.08</u>	<u>22.9.08</u>	<u>22.9.08</u>
<u>pH</u>	<u>7.2</u>	<u>7.3</u>	<u>8.3</u>	<u>8.3</u>	<u>7.4</u>	<u>7.5</u>	<u>7.6</u>	<u>7.3</u>
<u>Conduct.(μS/cm)</u>	<u>630</u>	<u>630</u>	<u>660</u>	<u>660</u>	<u>150</u>	<u>150</u>	<u>690</u>	<u>700</u>
<u>Susp solids (mg/l)</u>	<u>9.1</u>	<u>8.3</u>	<u>170</u>	<u>170</u>	<u>25</u>	<u>24</u>	<u>4.6</u>	<u>5.0</u>
<u>B.O.D. (mg/l)</u>	<u>4</u>	<u>4</u>	<u>9</u>	<u>9</u>	<u>9</u>	<u>8</u>	<u>3</u>	<u>4</u>
<u>Total P (μg/l)</u>	<u>160</u>	<u>150</u>	<u>230</u>	<u>220</u>	<u>130</u>	<u>130</u>	<u>50</u>	<u>40</u>
<u>Sol Phosphate</u>	<u>30</u>	<u>30</u>	<u><10</u>	<u><10</u>	<u>50</u>	<u>50</u>	<u><10</u>	<u><10</u>

<u>(µg/l)</u>								
<u>Total N (µg/l)</u>	<u>1,200</u>	<u>1,200</u>	<u>1,600</u>	<u>1,600</u>	<u>890</u>	<u>930</u>	<u>590</u>	<u>520</u>
<u>Nitrate (µg/l)</u>	<u>300</u>	<u>270</u>	<u>30</u>	<u>40</u>	<u>390</u>	<u>280</u>	<u>10</u>	<u>10</u>
<u>Chloride (mg/l)</u>	<u>80</u>	<u>80</u>	<u>100</u>	<u>100</u>	<u>26</u>	<u>26</u>	<u>110</u>	<u>110</u>
<u>Sodium (mg/l)</u>	<u>50</u>	<u>50</u>	<u>65</u>	<u>60</u>	<u>16</u>	<u>17</u>	<u>80</u>	<u>75</u>
<u>Potassium (mg/l)</u>	<u>4.8</u>	<u>4.8</u>	<u>6.0</u>	<u>5.9</u>	<u>1.8</u>	<u>1.8</u>	<u>6.1</u>	<u>5.5</u>
<u>Calcium (mg/l)</u>	<u>55</u>	<u>55</u>	<u>65</u>	<u>60</u>	<u>6.2</u>	<u>5.7</u>	<u>43</u>	<u>48</u>
<u>Magnesium</u>	<u>14</u>	<u>14</u>	<u>18</u>	<u>17</u>	<u>2.2</u>	<u>2.2</u>	<u>18</u>	<u>17</u>
<u>(mg/l)</u>								
<u>Total Iron (mg/l)</u>	<u>0.68</u>	<u>0.65</u>	<u>1.2</u>	<u>1.3</u>	<u>0.11</u>	<u>0.12</u>	<u>0.11</u>	<u>0.12</u>
<u>Turbidity (NTU)</u>	<u>9.5</u>	<u>10</u>	<u>90</u>	<u>85</u>	<u>26</u>	<u>26</u>	<u>8.6</u>	<u>7.6</u>
<u>Fluoride (mg/l)</u>	<u>0.19</u>	<u>0.15</u>	<u>0.19</u>	<u>0.18</u>	<u><0.05</u>	<u><0.05</u>	<u>0.22</u>	<u>0.19</u>
<u>Sulfate (mg/l)</u>	<u>27</u>	<u>25</u>	<u>22</u>	<u>15</u>	<u>1.7</u>	<u>1.4</u>	<u>26</u>	<u>34</u>

<u>E. coli</u>	<u>60</u>	<u>40</u>	<u>160</u>	<u>160</u>	<u>1,300</u>	=	<u><10</u>	=
<u>(orgs/100ml)</u>								

Table S2 Water pH across all sampling dates

<u>Site</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>	<u>12/12/2008</u>	<u>14/03/2009</u>
<u>Japan A</u>	<u>6.9</u>	<u>7.2-7.3</u>	<u>7</u>	<u>6.6</u>	<u>7.1</u>
<u>Japan B</u>	<u>7.2</u>	<u>7.4</u>	<u>7.1</u>	<u>7.3</u>	<u>7</u>
<u>Gillies</u>		<u>7.9</u>	<u>7.4-7.5</u>	<u>7.6</u>	<u>7.5</u>
<u>Gillies Japan</u>		<u>7.4</u>	<u>7.3</u>	<u>7.3</u>	<u>7</u>
<u>Fairy</u>		<u>7.1</u>	<u>7.4</u>	<u>7.6</u>	<u>7.6</u>
<u>Mill Lake</u>	<u>9.5</u>				
<u>Mill Lake</u>					
<u>Outlet</u>	<u>9.5</u>	<u>8.3</u>	<u>7.3-7.6</u>	<u>7.9</u>	<u>7.9</u>
<u>Main Lake</u>	<u>9.4</u>		<u>7.5</u>	<u>8.7</u>	<u>8.7</u>
<u>Mill Bird</u>		<u>8.7</u>	<u>8</u>	<u>7.8</u>	<u>8</u>

Table S3 Sodium concentrations (mg/L)

	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u>19</u>	<u>21</u>	<u>4.8-50</u>	<u>43</u>
<u>Japan B</u>	<u>37</u>	<u>35</u>	<u>60</u>	<u>48</u>
<u>Gillies</u>	<u>19</u>		<u>16</u>	<u>16-17</u>

<u>Gillies Japan</u>	<u>42</u>	<u>55</u>	<u>22</u>
<u>Fairy</u>	<u>17</u>	<u>15</u>	<u>16</u>
<u>Mill Lake</u>	<u>130</u>		
<u>Mill Lake</u>	<u>130</u>	<u>65-60</u>	<u>80-75</u>
<u>Outlet</u>			
<u>Main Lake</u>	<u>550</u>		<u>280</u>
<u>Bore 1</u>	<u>280</u>		
<u>Mill Bird</u>		<u>60</u>	<u>95</u>

Table S4 Calcium concentrations (mg/L)

<u>Site</u>	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u><2</u>	<u>8.5</u>	<u>55</u>	<u>50</u>
<u>Japan B</u>	<u>50</u>	<u>25</u>	<u>90</u>	<u>70</u>
<u>Gillies</u>	<u><2</u>		<u>13</u>	<u>6.2-5.7</u>
<u>Gillies Japan</u>	<u>65</u>		<u>85</u>	<u>10</u>
<u>Fairy</u>	<u>5.9</u>		<u>12</u>	<u>7.3</u>
<u>Mill Lake</u>		<u>38</u>		
<u>Mill Lake</u>		<u>38</u>	<u>65-60</u>	<u>43-48</u>
<u>Outlet</u>				
<u>Main Lake</u>		<u>34</u>		<u>50</u>
<u>Bore 1</u>		<u>90</u>		
<u>Mill Bird</u>			<u>45</u>	<u>40</u>

Table S5 Potassium concentrations (mg/L)

<u>Site</u>	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u>4.4</u>	<u>1.9</u>	<u>3.8-50</u>	<u>4.5</u>
<u>Japan B</u>	<u>7.9</u>	<u>3.2</u>	<u>6.4</u>	<u>5.8</u>

<u>Gillies</u>	<u>3.4</u>		<u>3.3</u>	<u>1.8</u>
<u>Gillies Japan</u>	<u>8</u>		<u>6.1</u>	<u>2.2</u>
<u>Fairy</u>	<u>3.9</u>		<u>2.2</u>	<u>2.2</u>
<u>Mill Lake</u>		<u>8.9</u>		
<u>Mill Lake</u>				
<u>Outlet</u>		<u>9.1</u>	<u>5.9-6.0</u>	<u>5.5-6.1</u>
<u>Main Lake</u>		<u>26</u>		<u>21</u>
<u>Bore 1</u>		<u>20</u>		
<u>Mill Bird</u>			<u>5.5</u>	<u>6.6</u>

Table S6 Magnesium concentrations (mg/L)

<u>Site</u>	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u>2.5</u>	<u>2.3</u>	<u>14</u>	<u>13</u>
<u>Japan B</u>	<u>8.8</u>	<u>6</u>	<u>18</u>	<u>14</u>
<u>Gillies</u>	<u>2.4</u>		<u>4.6</u>	<u>2.2</u>
<u>Gillies Japan</u>	<u>10</u>		<u>17</u>	<u>3</u>
<u>Fairy</u>	<u>2.6</u>		<u>2.5</u>	<u>2.3</u>
<u>Mill Lake</u>		<u>21</u>		

<u>Mill Lake</u>	<u>21</u>	<u>18-17</u>	<u>18-17</u>
<u>Outlet</u>			
<u>Main Lake</u>	<u>85</u>		<u>50</u>
<u>Bore 1</u>	<u>50</u>		
<u>Mill Bird</u>		<u>16</u>	<u>22</u>

Table S7 Iron concentrations (mg/L)

<u>Site</u>	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u><0.20</u>	<u>0.29</u>	<u>0.68-0.65</u>	<u>0.62</u>
<u>Japan B</u>	<u>0.78</u>	<u><0.20</u>	<u>0.11</u>	<u>0.14</u>
<u>Gillies</u>	<u><0.20</u>		<u>0.2</u>	<u>0.11-0.12</u>
<u>Gillies Japan</u>	<u>0.67</u>		<u>0.12</u>	<u>0.14</u>
<u>Fairy</u>	<u><0.20</u>		<u>0.4</u>	<u>0.17</u>
<u>Mill Lake</u>		<u><0.20</u>		
<u>Mill Lake</u>		<u>0.21</u>	<u>1.2-1.3</u>	<u>0.11-0.12</u>
<u>Outlet</u>				
<u>Main Lake</u>		<u><0.20</u>		<u>0.04</u>
<u>Bore 1</u>		<u><0.20</u>		

<u>Mill Bird</u>	<u>0.18</u>	<u>0.58</u>
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Table S8 Chloride concentrations (mg/L)

<u>Site</u>	<u>20/02/2008</u>	<u>21/02/2008</u>	<u>30/08/2008</u>	<u>22/09/2008</u>
<u>Japan A</u>	<u>26</u>	<u>26</u>	<u>80</u>	<u>65</u>
<u>Japan B</u>	<u>65</u>	<u>44</u>	<u>85</u>	<u>80</u>
<u>Gillies</u>	<u>30</u>		<u>20</u>	<u>26</u>
<u>Gillies Japan</u>	<u>65</u>		<u>85</u>	<u>36</u>
<u>Fairy</u>	<u>24</u>		<u>20</u>	<u>24</u>
<u>Mill Lake</u>		<u>220</u>		
<u>Mill Lake</u>		<u>220</u>	<u>100</u>	<u>110</u>
<u>Outlet</u>				
<u>Main Lake</u>		<u>960</u>		<u>500</u>
<u>Bore 1</u>		<u>490</u>		
<u>Mill Bird</u>			<u>100</u>	<u>150</u>

Table S9 Fluoride concentrations (mg/L)

<u>Site</u>	<u>20/02/08</u>	<u>21/02/08</u>	<u>30/08/08</u>	<u>22/09/08</u>	<u>12/12/08</u>	<u>14/03/09</u>
<u>Japan A</u>	<u><0.05</u>	<u>0.09</u>	<u>0.15-</u> <u>0.19</u>	<u>0.17</u>	<u>0.09</u>	<u>0.33</u>
<u>Japan B</u>	<u>0.09</u>	<u>0.1</u>	<u>0.21</u>	<u>0.16</u>	<u>0.14</u>	<u>0.62</u>
<u>Gillies</u>	<u><0.05</u>		<u>0.06</u>	<u><0.05</u>	<u>0.05</u>	<u>0.23</u>
<u>Gillies Japan</u>	<u>0.09</u>		<u>0.17</u>	<u><0.05</u>	<u>0.1</u>	<u>0.3</u>
<u>Fairy</u>	<u><0.05</u>		<u>0.29</u>	<u><0.05</u>	<u><0.05</u>	<u>0.27</u>
<u>Mill Lake</u>		<u>0.27</u>				
<u>Mill Lake</u>		<u>0.27</u>	<u>0.18-</u>	<u>0.19-</u>	<u>0.21</u>	<u>0.33</u>
<u>Outlet</u>			<u>0.19</u>	<u>0.22</u>		
<u>Main Lake</u>		<u>0.34</u>		<u>0.27</u>	<u>0.26</u>	<u>0.31</u>
<u>Bore 1</u>		<u>0.29</u>				
<u>Mill bird</u>			<u>0.15</u>	<u>0.22</u>	<u>0.2</u>	<u>0.36</u>

Table S10 Sulfate concentrations (mg/L)

<u>Site</u>	<u>20/02/08</u>	<u>21/02/08</u>	<u>30/08/08</u>	<u>22/09/08</u>	<u>12/12/08</u>	<u>14/03/09</u>
<u>Japan A</u>	<u>6.3</u>	<u>3.7</u>	<u>25-27</u>	<u>9.3</u>	<u>1.6</u>	<u>11</u>
<u>Japan B</u>	<u>6.3</u>	<u>4.2</u>	<u>28</u>	<u>25</u>	<u>28</u>	<u>5.7</u>

<u>Gillies</u>	<u>4</u>	<u>2.7</u>	<u>1.4-1.7</u>	<u>2.6</u>	<u>2.3</u>
<u>Gillies Japan</u>	<u>5.8</u>	<u>27</u>	<u>1.5</u>	<u>9.6</u>	<u>4.2</u>
<u>Fairy</u>	<u>5.8</u>	<u>2.2</u>	<u>17</u>	<u>1.5</u>	<u>4.1</u>
<u>Mill Lake</u>	<u>2.9</u>				
<u>Mill Lake</u>	<u>2.9</u>	<u>15-22</u>	<u>26-34</u>	<u>42</u>	<u>105</u>
<u>Outlet</u>					
<u>Main Lake</u>	<u>170</u>		<u>130</u>	<u>115</u>	<u>235</u>
<u>Bore 1</u>	<u>100</u>				
<u>Mill Bird</u>		<u>21</u>	<u>2.4</u>	<u>46</u>	<u>100</u>

Table S11 Turbidity (NTU)

<u>Site</u>	<u>20/02/08</u>	<u>21/02/08</u>	<u>30/08/08</u>	<u>22/09/08</u>	<u>12/12/08</u>	<u>14/03/09</u>
<u>Japan A</u>	<u>27</u>	<u>10</u>	<u>9.5-10</u>	<u>8.8</u>	<u>12</u>	<u>7</u>
<u>Japan B</u>	<u>13</u>	<u>7.9</u>	<u>3.1</u>	<u>7.1</u>	<u>2.8</u>	<u>7.5</u>
<u>Gillies</u>	<u>19</u>		<u>90</u>	<u>26</u>	<u>39</u>	<u>10</u>
<u>Gillies Japan</u>	<u>11</u>		<u>4.2</u>	<u>26</u>	<u>16</u>	<u>5.5</u>
<u>Fairy</u>	<u>35</u>		<u>85</u>	<u>37</u>	<u>3.8</u>	<u>19</u>
<u>Mill Lake</u>		<u>11</u>				

<u>Mill Lake</u>	<u>13</u>	<u>85-90</u>	<u>7.6-8.6</u>	<u>27</u>	<u>140</u>
<u>Outlet</u>					
<u>Main Lake</u>	<u>5</u>		<u>6</u>	<u>37</u>	<u>2.9</u>
<u>Mill Bird</u>		<u>17</u>	<u>6.2</u>	<u>22</u>	<u>21</u>

Table S12 Total Phosphorus (mg/L)

<u>Site</u>	<u>20/02/08</u>	<u>21/02/08</u>	<u>30/08/08</u>	<u>22/09/08</u>	<u>12/12/08</u>	<u>14/03/09</u>
<u>Japan A</u>	<u>0.42</u>	<u>0.22</u>	<u>0.15-</u> <u>0.16</u>	<u>0.09</u>	<u>0.24</u>	<u>0.3</u>
<u>Japan B</u>	<u>0.52</u>	<u>0.14</u>	<u>0.04</u>	<u>0.05</u>	<u>0.05</u>	<u>0.31</u>
<u>Gillies</u>	<u>0.18</u>		<u>0.31</u>	<u>0.13</u>	<u>0.24</u>	<u>0.19</u>
<u>Gillies Japan</u>	<u>0.31</u>		<u>0.05</u>	<u>0.12</u>	<u>0.15</u>	<u>0.25</u>
<u>Fairy</u>	<u>0.22</u>		<u>0.99</u>	<u>0.19</u>	<u>0.22</u>	<u>0.25</u>
<u>Mill Lake</u>		<u>0.26</u>				
<u>Mill Lake</u>		<u>0.34</u>	<u>0.23</u>	<u>0.05</u>	<u>0.1</u>	<u>0.25</u>
<u>Outlet</u>						
<u>Main Lake</u>		<u>0.04</u>		<u>0.04</u>	<u>0.04</u>	<u>0.19</u>
<u>Bore 1</u>		<u>0.06</u>				

<u>Mill Bird</u>	<u>0.1</u>	<u>0.07</u>	<u>0.09</u>	<u>0.14</u>
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Table S13 Filterable Reactive Phosphorus (mg/L)

<u>Site</u>	<u>20/02/08</u>	<u>21/02/08</u>	<u>30/08/08</u>	<u>22/09/08</u>	<u>12/12/08</u>	<u>14/03/09</u>
<u>Japan A</u>	<u>0.16</u>	<u>0.07</u>	<u>0.03</u>	<u><0.01</u>	<u>0.04</u>	<u>0.05</u>
<u>Japan B</u>	<u>0.15</u>	<u>0.03</u>	<u><0.01</u>	<u>0.01</u>	<u><0.01</u>	<u>0.04</u>
<u>Gillies</u>	<u>0.07</u>		<u><0.05</u>	<u>0.05</u>	<u>0.08</u>	<u>0.04</u>
<u>Gillies Japan</u>	<u>0.04</u>		<u><0.01</u>	<u>0.05</u>	<u>0.06</u>	<u>0.04</u>
<u>Fairy</u>	<u>0.1</u>		<u>0.61</u>	<u>0.07</u>	<u>0.07</u>	<u>0.09</u>
<u>Mill Lake</u>		<u>0.11</u>				
<u>Mill Lake</u>		<u>0.11</u>	<u><0.01</u>	<u><0.01</u>	<u>0.02</u>	<u><0.01</u>
<u>Outlet</u>						
<u>Main Lake</u>		<u><0.01</u>		<u><0.01</u>	<u><0.01</u>	<u><0.01</u>
<u>Bore 1</u>		<u><0.01</u>				
<u>Mill Bird</u>			<u><0.01</u>	<u>0.01</u>	<u>0.02</u>	<u>0.01</u>